The Sizewell C Project

6.14 Environmental Statement Addendum
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Chapter 2 Main Development Site
Appendix 2.17.A Marine Ecology and Fisheries

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APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

Summary of supplementary information on fish assessments provided for the Sizewell C DCO application

Ten reports associated with the potential impacts of entrainment and impingement by Sizewell C are provided herein. These comprise new reports that have been prepared since the original DCO application and reports that's have been updated since the Application. The new and revised assessments are the results of on-going consultation with the Environment Agency, Marine Management Organisation, Natural England and the Eastern Inshore Fisheries Conservation Authority.

The reports are summarised in **Table 1** which details the report number (including revision number and whether the report is a new, previously unsubmitted report or an updated version of a report cited in the original DCO submission), report title, the reason for the updated or new assessment and a summary of whether the new/updated report alters the original assessment significantly).

Table 1: Summary of additional fish entrainment and impingement reports

Report No.	Title	Purpose	Effects on DCO conclusions
TR339 (Rev 03) (updated)	Sizewell Comprehensive Impingement Monitoring Programme 2009 - 2017	Updated impingement assessment methodology and results based upon suggestions provided by the Environment Agency and resulting from an independent statistical review.	Small changes to expected impingement numbers that do not change the Sizewell C assessment conclusions of no adverse effects on fish and shellfish from cooling water abstraction. For abundant fish species the changes were a few percent, for less common species changes were in the tens of percent with a few showing greater effects. Results are now much more statistically robust, in particular the confidence intervals can now be relied on. Updating this report forced updates to TR520.
TR406 (Rev 07) (updated)	TR406 Impingement predictions based upon specific cooling water system design	Updated report on SZC fish and shellfish abstraction effects. Changes are due to assimilating the changes in TR339 and all of the SPPs listed in this table.	Many textual changes incorporating the new material from the other reports described in this table in response to stakeholder comments. However, the conclusions of the assessment on the significance of the abstraction effects of SZC are unchanged. In particular a new analysis has provided the information that stakeholders have requested on the very local effects of SZC on local WFD and HRA receptors.
TR520 (Rev 01) (new report)	Sizewell C Water quality effects of the fish recovery and return system	Updated report that had not been released at the time of the DCO application but whose conclusions were embedded in other documents (Environmental	Small numerical changes to discharge source terms. No changes to the previously reached conclusions of no adverse effects of the Fish Recovery and Return (FRR) discharge on local receptors.

		Statement (ES), WFD & HRA assessments, WDA permit application). Update forced by updates to TR339.	
SPP099 (Rev 04) (updated)	Predicted performance of the SZC LVSE intake heads compared with the SZB intakes	Modelled hydrodynamic parameters replaced with measured values from SZC intake locations. Extended sensitivity analyses incorporated.	The ratio of tidal intercept for Sizewell C compared with Sizewell B is slightly reduced (from 0.383 to 0.357) but with increased confidence. This means that Sizewell C is predicted to entrap fewer fish per cubic metre of water abstracted tnan previously estimated (now 35.7% of the Sizewell B catch compared with 38.3% reported for the DCO). This makes no significant difference to the conclusions reported in TR406 (Rev 06) provided in the original DCO Application.
SPP100 (Rev 01) (new report)	Estimates of European populations of twaite shad and cucumber smelt of relevance to Sizewell.	Report that had not been released at the time of the DCO application but whose conclusions were embedded in other documents (ES, WFD & HRA assessments, WDA permit application). Report issue had been delayed to allow for the completion of genetic origin studies but these have been delayed due to COVID-19.	Shad and smelt population numbers updated after further analyses but these changes made no significant difference to the conclusions reported in TR406 (Rev 06) provided in the original DCO Application.
SPP101 (Rev 03) (new report)	Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell.	To respond to questions raised by the Environment Agency on whether SZC could affect smelt populations in the WFD Alde and Ore transitional water body	The analyses demonstrate that Sizewell C would not affect the Alde Ore water body classification
SPP102 (Rev 01) (new report)	Use of Spawning Production Foregone EAVs for impingement assessment.	To respond to suggestions by the Environment Agency that Spawning Production Foregone EAVs were more appropriate to use than the EAV methodology proposed in TR383.	This report demonstrates that the use of the Spawning Production Foregone (SPF) EAV does not enhance the quality or reliability of the Sizewell C fish abstraction assessment. As described the SPF EAV would provide incorrect assessment conclusions and would increase assessment uncertainties.
SPP103 (Rev 03)	Consideration of potential effects on	To respond to questions raised by	New method that quantifies the very small size of the local effects on fish

(new report)	selected fish stocks at Sizewell.	the Environment Agency and subsequently by the Eastern IFCA, the Marine Management Organisation and Natural England.	populations caused by SZC cooling water abstraction. Conclusions provided in tabular and at the suggestion of the EIFCA, in heat map format This work complements the other analysis in TR406 that demonstrates the negligible effects on fish populations at the ICES stock level.
SPP104 (Rev 03) (new report)	Worst case glass eel entrainment assessment for Sizewell C.	To respond to questions raised by the Environment Agency on the lack of a quantitative assessment of glass eel entrainment in TR318. Worst case numerical assessment developed using existing data sources.	All Environment Agency questions answered. Overall conclusion unchanged that entrainment in SZC would have a negligible effect on the sustainability on local eel populations
SPP108 (Rev 03) (new report)	Sensitivity of the Alde Ore Transitional Fish Classification Index (TFCI) to changes in smelt, *Osmerus eperlanus*, abundance.	To respond to a series of questions raised by the Environment Agency on whether the status of the Alde Ore water body would be changed due to fish abstraction at Sizewell C	A comprehensive assessment was undertaken including quality assurance by one of the original developers of the transitional fish classification index. All the analyses demonstrated that Sizewell C would not adversely affect the WFD classification of the Alde Ore water body.

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. TR339

CONTRACTOR DOCUMENT FRONT SHEET

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Sizewell Comprehensive Impingement Monitoring Programme 2009 - 2017

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Executive summary

EDF Energy plans to build a new nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites on the Suffolk coast. As part of the application for the building and operation of the new station, EDF is required to evaluate the effects that the abstraction of water for cooling the condenser steam may have on organisms in the marine environment. Although the intake structure is protected by coarse screens to prevent the abstraction of large animals or debris, entrapment of fish, invertebrates, seaweed and litter into the system would occur. Large organisms and trash are removed prior to entering the cooling water systems by capture (impingement) on rotating drum and band screens and returned to the sea via a Fish Recovery and Return (FRR) system. However, not all will survive and EDF Energy is required to evaluate these losses.

Between February 2009 and March 2013, impingement sampling (known as the Comprehensive Impingement Monitoring Programme, CIMP), was carried out at SZB. A total of 128 samples were collected on the number and weight of fish, invertebrates and other material passing through the station cooling water system. A further period of sampling took place between April 2014 and October 2017 (77 visits), with gaps in sampling due to station outages. During each visit, 6 x 1 hourly and one 18-hour overnight sample was taken. Each sample was raised to the full pumping capacity of the station (4 pumps) and summed to provide an estimate of the number and weight of fish, invertebrates and other material impinged over that 24-h period.

Data from the two sampling programmes were combined to provide a dataset comprising 205 samples over a nine-year period. Data were used to provide, for each species, an annual mean estimate of the numbers and weights of finfish and invertebrates impinged at the SZB station, with estimates of variability derived using the statistical method of bootstrapping. The calculation approach, assumptions and limitations are discussed.

Ninety-one finfish taxa were recorded during the CIMP. The eight most abundant taxa by number were sprat *Sprattus sprattus*, herring *Clupea harrengus*, whiting *Merlangius merlangus*, seabass *Dicentrarchus labrax*, sand gobies *Pomatoschistus* spp., Dover sole *Solea solea*, European anchovy *Engraulis encrasicolus*, and Dab *Limanda limanda.* Together these eight species contributed 95.3% of the total numbers of fish impinged averaged over the period 2009 - 2017. An estimated mean weight of 203.2 t of fish were impinged by SZB annually between 2009-2017. Seven species contributed to the top 95.6% of total finfish impingement by weight, these were herring, whiting, seabass, sprat, Atlantic cod *Gadus morhua,* thin-lipped grey mullet *Liza ramada*, and Dover sole.

Many fish species showed distinct seasonal patterns of abundance, with some species predominating impinged in either the winter or summer. The majority of sprat (86%) and thin-lipped grey mullet (73.0%) are captured in quarter 1 (Q1; January - March) each year. In the case of seabass, 99.5% are impinged in Q1 (79.9%) and Q4 (19.6%; October – December) each year. Herring and whiting are also predominantly winter species, caught in Q1 and Q4 each year, whereas anchovy are impinged in the greatest numbers in Q2 (81.5%; April - June) and Q3 (18.4%; July - September).

Seventy-five invertebrate and macroalgae taxa were recorded. The three most abundant invertebrate taxa by number were ctenophores, brown shrimp *Crangon crangon* and pink shrimp *Pandalus montagui*. These three taxa contributed 95.9 % by number to the total invertebrate count, with ctenophores alone contributing 86.2 %. The annual estimated weight of invertebrates was 184 t, of which six taxa contributed to the top 95.7% of weight. These included ctenophores, jellyfish, brown shrimp, white weed, common prawn *Palaemon serratus*, and unidentified hydroids. Gelatinous zooplankton (ctenophores and jellyfish) were predicted to contribute approximately 156 t annually. An estimated 35 t of seaweed is impinged annually; brown seaweed (Phaeophyceae), red seaweed (Rhodophyceae), and *Ulva lactuca* contribute to a cumulative 98.1% of the seaweed impinged.

Seasonal patterns of brown shrimp, ctenophores and jellyfish were similar to those previously reported for 2009 - 2013 with ctenophores being most abundant in Q2, jellyfish in Q3 and brown shrimps in Q3 (61.8%) and Q4 (23.3%).

Updates in Version 2:

Estimates of impingement in version 1 of this report were derived from model outputs for each species that provided a mean impingement estimate and estimated upper and lower boundaries. The lower and upper estimates applied in the model represented an interval of modelled daily values for each species multiplied by the number of days in the month and then limits for all months were added. This assumes that either the minimum monthly limit or maximum monthly limit was impinged on every day of the year. This is considered highly unlikely, and results in values that will substantially overestimate the variability in impingement. As such the upper and lower values were not true 95th percentiles and Cefas urged caution in their application in BEEMS Technical Report TR406.

During stakeholder consultation on BEEMS Technical Report TR406 version 6, the Environment Agency expressed concern regarding the status and size of the lower and upper estimates of impingement predictions for SZC. The Environment Agency also asked if it was possible, given the gaps in the dataset, to derive impingement estimates using the bootstrapping method used at Hinkley Point (BEEMS Technical Report TR456). As a result of these comments, Cefas undertook a statistical review of the model and potential alternative methodologies.

This review concluded that the modelling approach used to derive the impingement estimates in TR406 v6 was not the most appropriate. Version 2 of this report presents a bootstrapping approach, akin to the methods employed at Hinkley Point, with an additional step to account for period when sampling was not possible due to station outages described in Section 3.6.

The SZB CIMP dataset is more extensive than for HPB, whereby 205 samples have been collected over nine years from 2009-2017 at Sizewell compared with the 40 samples from one year at Hinkley Point. However, the modelling approach used at Hinkley Point could not directly be replicated for Sizewell. This is because it incorporates multiple years of data and the distribution of samples over the sampling period is uneven, with no sampling in some quarters largely due to station outages. The bootstrapping method applied here randomly reallocates samples from adjacent years to account for sampling gaps.

The bootstrap outputs for each species provide the mean numbers impinged by quarter, which was converted to percentage impingement by quarter to show the seasonal patterns and the mean annual impingement with 95th percentile confidence intervals based on 5000 bootstrap iterations.

The bootstrapping approach is considered the most robust for calculating mean annual impingement estimates, allowing 95th percentile confidence intervals to be clearly defined. A consistent approach can be applied across all species for both numbers and weights impinged. The results of this report will be applied to Sizewell C impingement predictions in an updated version of BEEMS Technical Report TR406.

The development of the model and bootstrap approaches along with the advantages and limitations is provided in Section 5.1.

A comparison of the model and bootstrap approaches and implications for Sizewell C impingement predictions is provided in Section 5.1.1.

Updates in Version 3:

Version 3 of this report responds to stakeholder comments on version 2 regarding scaling methodologies for sub-sampling during hourly and bulk monitoring and well as the effect of station pumping capacity on impingement estimates (Sections 2 and 3).

The unattended 18-hour bulk samples at Sizewell have proved more prone to sample loss than the attended hourly samples but due to sampling logistics on an operational power station it has not been possible to reduce the frequency of such sample losses or to increase the volume of the sampling net. Impingement rates from bulk and hourly samples are compared to determine the presence of any diurnal bias in fish impingement in four key species herring *Clupea harrengus*, seabass *Dicentrarchus labrax*, smelt *Osmerus eperlanus* and sand gobies *Pomatoschistus spp*. (Appendix F). Should such bias in diurnal behaviour occur, the potential for over- or underestimation of 24-hour impingement exists when bulk samples are invalid. The results from 22 monitoring visits with successful hourly and bulk samples between 2014 and 2017 do not indicate diurnal bias.

During QA of raw impingement monitoring data files an error was identified form one monitoring visit in early 2009, whereby a sub-sample was not scaled up correctly. This error has been corrected throughout the resulting impingement calculations and tables presented in this report. The net effect is annual fin-fish impingement weight increases by approximately 1% from 201.3t to 203.2t per annum. All species impinged during that monitoring visit are subject to minor changes in impingement predictions; the greatest changes are observed in seabass and cod.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station will be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of water may have on organisms in the marine environment. Although the cooling water intakes will be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, a significant number of organisms (small fish and crustaceans, and plankton) will inevitably be entrapped in the cooling water intake. The larger entrapped organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh screens (10 mm at the existing SZB and 10mm proposed for SZC) which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the screens are entrained and pass through the power station cooling system without causing significant blockages. Impinged organisms will be returned to the sea via a Fish Recovery and Return (FRR) system. Not all will survive this process and EDF Energy is required to evaluate the effect of the loss of these organisms on marine communities and also of the potentially polluting effects of dead biota discharged from the FRR system on local water quality and marine ecology. The Centre for Environment, Fisheries and Aquaculture science (Cefas) has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

The basis for predictions of impingement by SZC is data collected at the SZB station. Between February 2009 and March 2013, impingement sampling (known as the Comprehensive Impingement Monitoring Programme, CIMP), was carried out at SZB by Pisces Conservation Ltd, as part of the BEEMS programme. A total of 128 24-h samples were collected on the number and weight of fish, invertebrates and other material passing through the station cooling water systems (BEEMS Technical Reports TR120; TR196; TR215; TR270).

A total of 77 fish taxa and 26 invertebrate taxa were recorded. Fish catches were dominated by sprat *Sprattus sprattus*, herring *Clupea harrengus*, whiting *Merlangius merlangus*, seabass *Dicentrarchus labrax*, sand gobies *Pomatoschistus sp*., and Dover sole *Solea solea*. Invertebrate catches were dominated by the brown shrimp *Crangon crangon*, the pink shrimp *Pandalus montagui*, the Atlantic prawn *Palaemon serratus*, common swimming crab *Liocarcinus holsatus* and the shrimp *C. allmanni*. In addition, several fin-fish species of conservation concern were recorded, including the European eel *Anguilla anguilla*, twaite shad *Alosa fallax*, and river lamprey *Lampetra fluviatilis* (BEEMS Technical Reports TR120; TR196; TR215; TR270).

The first CIMP programme ran for 4 years from 2009 and after an initial evaluation of the data, the programme was resumed in April 2014. This report considers data collected up until October 2017. During which period 77 impingement monitoring visits occurred, with gaps in sampling due to station outages.

This report describes the taxa, catch composition and estimated numbers and weights of organisms impinged during the period 2009-2017, and annual estimates for impingement at SZB.

2 Sampling 2009 – 2013

A full description of the sampling methodologies used and the raw data obtained during the original CIMP programme (2009-2013) are given in the four annual reports presented by Pisces Conservation Ltd (BEEMS Technical Reports TR120; TR196; TR215; TR270). The final sampling programme consisted of 36 × 24-h periods in 2009-10, 31 in 2010-11, 28 in 2011-12, 24 in 2012-13 and 7 in 2013. The 24-h sampling approach was used at SZB to eliminate the risk of diurnal and short-term (~13-h cycle) tidal bias, with longer-term tidal bias eliminated by randomising the sampling dates. It was not possible to select sampling dates completely at random because of power station operational constraints e.g. sampling could only take place on week days. However, selection was not constrained by biological factors.

2.1 Sampling method

Each 24-h sampling period was covered by seven samples - six samples of one hour each during daylight, and one 18-h bulk sample. On each site visit, Pisces staff started their sampling at 12:30, collecting hourly samples from one trash basket at 13:30, 14:30 and 15:30. The screen wash flow was then diverted into the other trash basket fitted with the bulk sample net, to collect the overnight sample. At 09:30 the following day, the sample was lifted out and placed on the ground on a large tarpaulin for sorting. At the same time, collection of the hourly samples was resumed, with the final 1-h sample being completed at 12:30. This pattern allowed 24-h coverage and analysis of smaller samples by day that would not have been damaged by crushing and the subsequent loss of smaller, damaged fish and crustaceans through the net.

Sub-sampling was sometimes necessary because of the large volumes of material collected. If possible, the whole sample was sorted for fish, and sub-sampling carried out only for invertebrates and debris. However, if the sample was particularly large and held many fish (especially the 18-h bulk sample), the entire sample was sub-sampled. When sub-sampling was carried out, the unsorted portion was weighed. The number and the weight of species or debris in the sorted portion were then raised by the sub-sample to whole sample weights ratio to provide a representation of the entire sample.

Temporal sub-sampling was occasionally necessary when the volumes of impinged material became too large for the nets. This happened particularly in spring and summer because of the large volumes of ctenophores and jellyfish present. When necessary, impinged material was collected in the nets for a proportion of the sample time, then for the remaining time the sample was diverted out to sea, as during normal operations at the station. Numbers and weights were multiplied up according to the proportion of time sampled to make a full 24-h sample. At least some part of each sample (the bulk and the 6 × 1-h samples) was always taken to ensure representation across the entire 24-h period.

2.2 Treatment of the bulk sample

The 18-h bulk sample was often very large and, once it was emptied from the net, assessment of the need to subsample was made before sorting. Generally, large fish (all individuals >30 cm long) were picked from the sample and the remainder sub-sampled. Invertebrates and debris were generally sub-sampled further. Fish were sorted to species and, for each species, at least 100 lengths (if available) per day were recorded to the nearest mm. The total numbers and weights of each species were recorded.

After removing the fish, any particularly large invertebrates (e.g. edible crabs) or pieces of debris were removed because they might not have been represented in a small sub-sample. The remaining invertebrates and debris were then often sub-sampled − usually about one-eighth of the sample weight was sorted. This subsample was picked through carefully; all invertebrates were identified to species, counted and weighed, and all debris was placed in categories and weighed. Any fish found in the sub-samples were identified to species, measured and weighed.

Where possible, the whole sample was measured. When samples were too large a representative sub-sample was measured, and the weight of the measured sub-sample was recorded to allow the sample to be raised accordingly to estimate numbers in the total sample.

Due to restricted access at operational power stations it is not possible to staff the bulk sample collection overnight. In some cases, this resulted in samples overflowing e.g. due to large numbers of ctenophores or pelagic fish. Overflowed bulk samples had to be discarded. High biomass periods are accounted for by hourly samples, however, the potential for diurnal patters of impingement rates leading to over/underestimates of impingement of fish with a bias in daily hourly samples or 18-hour bulk samples was investigated. Estimated daily impingement from bulk and hourly samples was examined for four key species, herring *Clupea harrengus*, seabass *Dicentrarchus labrax*, smelt *Osmerus eperlanus* and sand gobies *Pomatoschistus spp* (Appendix F).

2.3 Treatment of the hourly samples

While one part of the sampling team was processing the bulk sample, two other members of the team were assigned to work on the hourly samples. The hourly sample net was normally deployed for the full period of 60 min. However, if significant quantities of fish, ctenophores or jellyfish were being captured, so that it would have been impossible to sort, measure and weigh the sample in a reasonable time, or such that the net would be filled to overflowing, then the sampling period was reduced to 30, 15 or even 10 min, as appropriate. For the remaining time, the sample was diverted to sea. When samples were large and the sample time was reduced, the weight of the measured sub-sample was recorded to allow the sample to be raised accordingly to the duration of sampling to estimate numbers in a total hourly sample.

As with the bulk sample, fish were sorted to species, and for each species, at least 100 standard lengths (if available) per day were recorded, as well as the total numbers and weights of each species. Ctenophores were weighed, then macro-crustaceans were sorted to species and the total weight of each species recorded. The wet weight of the remaining debris, divided into categories, was recorded.

2.4 Fish length measurements

For most fish species, standard length (*SL*) was recorded. Standard length refers to the length of a fish measured from the most anterior tip of the body to the posterior end of the vertebral column. This measurement does not include the length of the caudal fin and was used because fish can suffer damage to the caudal fin in power station intake screens. For fish with no or more-robust caudal fins, e.g. lampreys and elasmobranchs (sharks and rays) and eels, total length was used.

2.5 Data adjustment

As explained above, sub-sampling was routinely used when processing fish, invertebrates and debris. When sub-samples were taken, data were adjusted to provide a representation of a complete 24-h sample. When no bulk sample was obtained for logistical reasons, the 24-h estimate was drawn from the average of the hourly samples multiplied up to cover the full 24-h period.

Power station operational variables were also recorded for each sampling day; generally, the number of pumps running and the number of screens in use. The station does not always run at full capacity and can have two or three of the four pumps and screens functioning. Using the appropriate multiplication factor, these data were adjusted to represent a 24-h sample at full (4-screen) capacity. Stakeholders have asked whether reduced pumping capacity could lead to underestimates of impingement, however, impingement rates are scaled to account for full capacity (assuming linear relationships between pumping capacity and impingement). Annual impingement rates are calculated based on full capacity operations 365 days a year, and therefore the effect of Sizewell B impingement is overestimated.

The full dataset consisted of 128 samples, each representing the numbers and weight of fish, invertebrates and other material impinged over 24-h if the station was pumping at full capacity (4 pumps in operation).

3 Sampling 2014 – 2017

Sampling was carried out in April - September 2014, April 2015 - March 2016 and June 2016 - October 2017 with a sampling target of 7 sampling visits per quarter. As with the original programme, sampling dates were distributed as randomly as possible, within the limitations of station access. SZB station has four rotating drum screens fitted with a 10 mm square mesh (Figure 1). Fish, invertebrates and debris are washed from the screen and returned to the sea via the cooling water seal pit and cooling-water discharge. During sampling, the water flow was diverted into a trash pit. For hourly sampling, a metal framed net was erected around the discharge channel (Figure 2 and Figure 3). Unless catches were high (and sampling was stopped), the flow of water remained diverted throughout the sampling period. At the end of each hour, the catch was shaken down into the cod end (Figure 4), emptied into a fish basket, and lifted from the trash pit (Figure 5). For the bulk sample the flow emptied directly into the trash basket, which had previously been lined with the bulk net. The following morning, the flow was diverted to its original course, the bulk net was drained of excess water, removed using a forklift truck and emptied onto a tarpaulin.

Figure 1 The four drum screens at Sizewell B

Figure 2 Placement of the hourly net in the trash bin, showing the position of the bulk net in the bottom

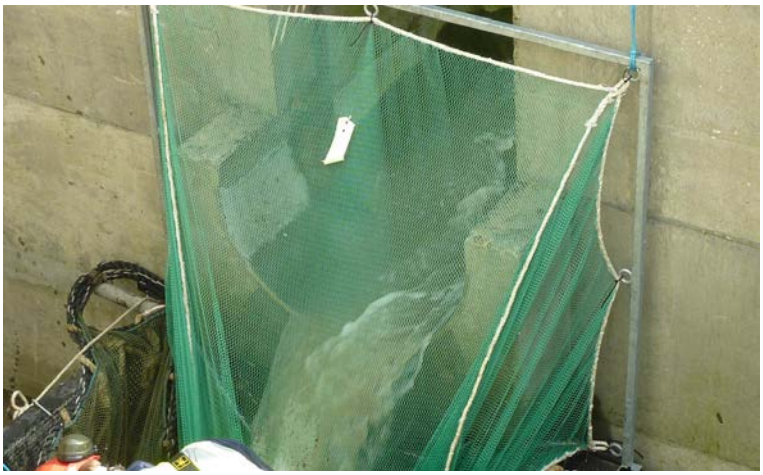


Figure 3 Position of the hourly net over the mouth of the screen wash divert culvert

Figure 4 Shaking the catch to the cod end of the net

Figure 5 Lifting the catch using the yoke and rope arrangement

3.1 Method

As with the 2009-2013 programme, each 24-h sampling period comprised six x 1-h samples and one x 18-h bulk sample. In contrast to the original sampling, all the hourly samples were collected on the first sampling day, and the bulk sample was processed on the second day. This was possible because the proximity of Cefas to SZB allowed staff to be on site by 08:30 on the morning of the first day, and to remain for longer on site on the second day. The benefit of this approach was that the whole sampling team (of 4 persons) was able to process each sample (as opposed to splitting the team in half to process the bulk and remaining 3 hourly samples on the second day). A summary of the sampling regime used by Cefas is given in Table 1.

Sub-sampling was used only if necessary. In general, the whole sample was sorted for fish first, before the invertebrates, weed, and litter portion was sorted. If this portion was small, it was fully sorted to the lowest taxon possible. However, if the volume of invertebrates/weed/litter was large a sub-sample was taken and sorted. When sub-sampling was carried out, the unsorted portion was weighed. The number and the weight of species or debris in the sorted portion were then raised by the sub-sample to whole sample weights to provide a representation of the entire sample. As in 2009-2013, temporal sub-sampling was occasionally necessary when the volumes of impinged material were large, in which case the sample was handled as described in Section 2.1 above.

Table 1 Standard sampling pattern for Cefas impingement sampling

Day	Time	Sample
Day 1	09:00 – 10:00	Hourly 1
Day 1	10:00 – 11:00	Hourly 2
Day 1	11:00 – 12:00	Hourly 3
Day 1	12:00 – 13:00	Hourly 4
Day 1	13:00 – 14:00	Hourly 5
Day 1	14:00 – 15:00	Hourly 6
Day 2	15:00 – 09:00	Bulk

3.2 Treatment of the bulk sample

If possible, the whole of the bulk sample was sorted, and generally samples of up to ~180 kg were fully processed. On the occasions where the bulk sample was too large to fully process, all large fish (>30 cm long) were picked from the sample and the remainder sub-sampled. Usually a minimum of about one-fifth of the sample was processed. All fish were sorted to species and each species was weighed and measured (WM). Where possible, the whole sample was measured, but if there were multiple baskets of a species, then a representative sub-sample was measured, and the weight of the measured sub-sample was recorded to allow the sample to be raised accordingly to estimate numbers in the total sample. Lengths were recorded to the nearest half cm below, as per standard BEEMS offshore sampling protocols (*i.e.* a fish measuring between 3.0 cm – 3.4 cm would be recorded as 3 cm, and a fish measuring 3.5 cm – 3.9 cm would be recorded as 3.5 cm).

After removing the fish, any particularly large invertebrates (e.g. edible crabs) or pieces of debris were removed from the invertebrate/weed/litter mix. Usually, sub-sampling was employed. All invertebrates were identified to species where possible and were weighed and counted (WC). If there was a particularly large number of a given species (e.g. shrimps or prawns), a portion of the total were counted, and the weight of the portion was recorded, so that the total number could be calculated. All edible crabs *Cancer pagurus* and velvet swimming crabs *Necora puber* were measured to the nearest mm, across the carapace.

Where possible, jellyfish and ctenophores were identified to species and weighed and counted, but if there were many broken pieces, the species was weighed only (WO). All weed, and trash was also weighed only. During 2014, litter was sorted on site into separate categories and each category was weighed separately. In 2015, following a request from EDF Energy for additional information to be collected on the litter component, the litter was simply weighed on site, placed in a labelled bag and returned to the Cefas laboratory for processing. Litter processing does not form part of the impingement assessment herein.

Due to restricted access at operational power stations it is not possible to staff the bulk samples overnight. In some cases, this resulted in samples overflowing. Overflowing bulk samples had to be discarded. High biomass periods are accounted for by hourly samples, however, the potential for diurnal patterns of impingement rates leading to over/underestimates of impingement of fish with a bias in daily hourly sample or overnight bulk samples was investigated. Estimated daily impingement from bulk and hourly samples was examined for four key species, herring *Clupea harrengus*, seabass *Dicentrarchus labrax*, smelt *Osmerus eperlanus* and sand gobies *Pomatoschistus spp* (Appendix F).

3.3 Treatment of the hourly samples

The hourly samples were treated in the same way as the bulk sample, and in general all fish in the hourly samples were measured. The remaining invertebrate/weed/litter mix was fully sorted wherever possible. In the summer months when the samples consisted almost entirely of ctenophores, the weight of the ctenophores was recorded, along with the weight of 200 counted individuals, to allow the total number of individuals in the catch to be calculated.

The hourly sample net was normally deployed for the full period of 60 min. If significant quantities of fish, ctenophores or jellyfish were being captured, so that it would have been impossible to sort, measure and weigh the sample in a reasonable time, or such that the net would be filled to overflowing, then the sampling period was reduced to 30, 15 or even 10 minutes, as appropriate. For the remaining time, the sample was diverted to sea. When samples were large and the sample time was reduced, the weight of the measured sub-sample was recorded to allow the sample to be raised accordingly to the duration of sampling to estimate numbers in a total hourly sample.

3.4 Fish length measurements

All fish were measured to Total Length (*TL*). *TL* is defined as the length from the most anterior point of the body to the tip of the longer lobe of the caudal fin, usually measured with the lobes compressed along the midline. This is a straight-line measurement and does not measure the curve of the body. *TL* is the length routinely used in all Cefas' biological sampling programmes (e.g. research vessel surveys; sampling of commercially landed fish on fish markets), and in the BEEMS offshore sampling.

Initial evaluations of the fish on site indicated that the caudal fins, even of delicate species such as herring or sprat, were not badly damaged on removal from the net (even the bulk net), and that almost all individuals were intact. Therefore, there appeared to be no merit in using the *SL* measure, which could create the risk of samplers unconsciously using the more familiar *TL*, which would erroneously be assumed to be *SL*. The use of a single measure also meant there would be no confusion between which species should be measured to *SL* and which to *TL*.

The use of the measurement of *TL* also meant that lengths need not be converted from *SL*. Most length-weight conversions and estimates of length at age or length at maturity are given in relation to *TL*, so a conversion must first be applied to the standard length data. To carry out this conversion, a sample of each species is measured at both lengths and a conversion factor established.

3.5 Data capture

All data were captured in Cefas' FSS (Fishing Survey Systems) database. This database is the central repository of all Cefas' research vessel and charter vessel survey data and is also where the BEEMS offshore survey data are stored. In 2014, all data were recorded to paper data sheets, and the data were manually input to the FSS on return to the laboratory. All data were cross-checked for quality. From April 2015, on-site data were captured using a portable Electronic Data Capture (EDC) system. The benefit of using this system is that it minimises the need to manually key in and check data on return to the laboratory.

The EDC consists of a mini computer, power pack and fan housed inside the bottom of a pelicase, beneath an aluminium plate (which prevents the ingress of water). Waterproof communications ports in the plate allow a monitor, keyboard and measuring wand to be attached, a USB data stick to be inserted to allow data downloading, and for the power pack to be plugged in to charge the computer. The monitor is housed in the lid of the case, and once the wand and keyboard are removed, the case can simply be shut and the whole unit easily transported. All power is internal, negating the need for electrical input on site, and each machine is fitted with a thermometer display and a fan to monitor and prevent internal overheating.

The measuring system uses a measuring board which has RFID pit tags embedded along its length at one cm and half cm intervals. Each tag is individually coded for the length that it represents. Additional tags are embedded in the measuring board and coded for the letters of the alphabet and numbers, along with other functions, such as 'enter' or 'complete'. The user tells the EDC unit, via the measuring board the sample number and the 3-letter code of the fish species being measured and whether the measure represents the complete sample or a sub-sample. In the latter case, the measurer inputs the sub-sample weight to allow for the calculation and application of the appropriate raising factors.

Two EDC units were generally used (both were loaded with the same software so that they could be interchangeable). The first unit used the 'EDC deckmaster', software which captured information on the sample (sample number, the sample type – hourly 1, hourly 2) and the total fish weights by species, along with the number and weight of all invertebrate species and weed and mud. All sample metadata were recorded on to a hard copy sheet, for subsequent input to the FSS system in the laboratory.

The second unit used the 'EDC measure' software to record all the fish lengths. Each fish was measured by placing it on the measuring board and moving the wand over the relevant pit tag to record the length. A light flashed on the wand to confirm the value had been recorded and a visual cue on the monitor showed the fish length being added. Once all the fish were measured, a communication cable was used to connect the 'measure' machine to the 'deckmaster' and the lengths and sub-sample weights were downloaded. During this process the 'deckmaster' machine conducts checks to ensure that all species that were weighed have been measured. Thus, if the incorrect species code is used there will be a mismatch between the two units, and this can be corrected before uploading to the database. Once back in the lab, the sample data was transferred to the FSS via a USB stick.

3.6 Data-raising and calculation of the number and weight of fish impinged

On the FSS, the sample metadata were input for each sample (in principle 7 samples were taken on each visit), from the paper sampling sheets. This data included the visit designation, sampling date, sampling time (start and end), sample type (hourly 1, hourly 2, bulk), and the number of pumps and screens in operation at SZB. All manually input metadata were independently checked to ensure accurate inputting. The data from the sampling visit were then uploaded directly into the FSS from the USB stick. Again, the system performs several checks against the data to ensure that the correct sample process codes (WM/WC/WO) were used to capture that data. Once the data are uploaded, the FSS system automatically calculates any raising factors for sub-sampled WC species, to raise up the sampled numbers to total numbers, and in the case of sub-sampled fish, the system automatically calculates the numbers at length raised to the total weight of fish present, using the measured numbers at length, the total weight and sub-sample weight.

3.6.1 Extraction of the data

For analysis, all data were extracted from FSS to an Access database, which was interrogated using a series of queries to produce 1) the complete set of sample data, 2) the number, weight and lengths of all weighed and measured (WM) fish and crabs, 3) the number and weight of all weighed and counted (WC) invertebrates, and 4) the weight of any weighed only (WO) components. These were combined to a single dataset which consisted of, for each sample, the sample details and the weight and number of all the catch components. Having the data in a single dataset ensured that any raising would be uniformly applied to all the components of a given sample.

3.6.2 Raising to the full sample duration

As described in the sampling methods, samples were of either 1-hr or 18-h duration, but in some cases, the sampling time was reduced because of high volumes of material entering the net. Therefore, each sample was raised to its intended sampling duration by multiplying by either (60/sampling duration) for hourly samples, or (1080/sampling duration) for bulk samples, where sampling duration was the number of minutes sampled based on the start and end times.

3.6.3 Raising to full pumping capacity

The SZB cooling water system uses 4 drum screens and 4 pumps, but these are not always operational. The number of pumps in operation was noted for each sample taken. All catch components were raised to full pumping capacity by multiplying by (4/number pumps in operation). Stakeholder have asked whether reduced pumping capacity could lead to underestimates of impingement, however, impingement rates are scaled to account for full capacity (assuming linear relationships between pumping capacity and impingement). Annual impingement rates are calculated based on full capacity operations 365 days a year, and therefore the Sizewell B impingement is an overestimate.

3.6.4 Combination of samples to give 24-h sample

Seven samples comprised each 24-h sample (6 * hourly, plus 1 * bulk). For visits where all 7 samples were available, the catch components (raised to their full duration and full pumping capacity) were simply summed to give the number and weight of organisms impinged during those 24 hours. For visits where the first hourly sample was not obtained, the remaining samples were summed (Hourly 2 – Hourly 6, plus the bulk), and the summed totals were raised to 24 hours by multiplying by (24/23). For visits where the bulk sample was deemed to be invalid, the 6 hourly samples were summed, and the summed total numbers and weights were raised to 24-hours by multiplying by (24/6).

The 2014-2017 dataset comprised 77 samples representing the number and weight of fish, invertebrates and other material impinged over 24-hours with the station pumping at full capacity, as well as the raised length distributions for all fish and measured crab species.

3.7 Calculation of annual impingement

3.7.1 Calculation approach

Estimates of impingement for Hinkley Point C (HPC) were based on a CIMP conducted at Hinkley Point B (HPB) in 2009 – 2010. For that dataset, 40 sampling visits were conducted. Estimates of annual impingement were calculated by first summing all samples from each quarter of the year and raising the quarterly total by the ratio between the number of days in the quarter and the number of sampling visits. The quarterly totals were summed to give an annual impingement estimate. Variability was estimated by bootstrapping the dataset – randomly selecting a set of visits from each quarter and repeatedly estimating the total annual impingement to give 95 % confidence estimates around the mean value.

The SZB CIMP dataset is more extensive than for HPC, spanning 9 years and 205 samples (Table 2 shows the number of sampling visits). For the SZB CIMP dataset, the bootstrap approach was extended to account for the multiple years of sampling and the presence of quarters without sampling. Minimal sampling

occurred in 2013 (sampling was suspended in March 2013) and quarter 1 in 2014, and there were no samples in quarter 4 in 2014 and quarter 1 in 2015). This was largely due to outages when no sampling could take place. It is important to account for sampling gaps, as raising a simple average to the whole year will bias species that show strong seasonal patterns in abundance, leading to under or over-estimates of impingement. For example, due to the lack of samples in quarter 1 in 2014 and 2015, the numbers of seabass and thin-lipped grey mullet recorded was small. Simply raising these years to an annual estimate would lead to underestimates of these two species in those years, which could be incorrectly interpreted as a decrease in abundance for those years.

3.7.2 Bootstrap methods

Impingement estimates were made by applying a bootstrap process (developed in R software version 3.6.1) to the impingement data. Each species was assessed separately and there is no aggregated result by habitat or trait. Sampling was carried out on 205 different days between 4th Feb 2009 and 5th October 2017, and if a species was not encountered on a sampling day it was marked as an absence (0) for that day. For some species this resulted in a timeseries of mainly zeros. The bootstrap method was non-parametric, meaning that it resampled directly from the observed samples, and therefore could be applied in the same way to each species.

Figure 6 describes the calculation and bootstrap process which was carried out for 5000 iterations for each species. The bootstrap strata were the combinations of quarter and year to match the sampling design of the programme. As previously noted, there was a gap in the sample programme in 2013, so estimates were not calculated for 2013. From the 32 remaining quarters, three had no samples (Table 2a) and one had a single sample. For these four cases, bootstrap strata were pooled with the same quarter from an adjacent year (Table 2b) except for the additional 31 invertebrate species where counts started in 2014. In this case, quarter 1 of 2014-2016 was a stratum (Table 2c). The invertebrate species where enumeration began in 2014 are shown in Appendix B. This approach retained the seasonal patterns known to exist for some species. Bootstrap samples from the pooled strata were then reassigned to quarters so there were at least two samples in each quarter. For example, within each bootstrap iteration the seven samples in quarter 4 of 2015 were randomly assigned with two to quarter 4 of 2014 and five to quarter 4 of 2015. This method ensured the total number of bootstrap samples was equal to the observed number of samples. Once the bootstrap strata and assignment rules were defined, all resampling and assignment of samples in each iteration was carried out automatically within the code.

Table 2 The number of 24-h samples in the analysis data (a); with the bootstrap strata (shown by cell borders) and number of resampled observations assigned to each quarter in each bootstrap iteration for species recorded 2009 to 2017 (b) and 31 invertebrate species where counts were recorded from 2014 to 2017 (c). Values in italics relate to pooled strata.

(a)	Quarter			
Year	1	2	3	4
2009	7	9	9	11
2010	8	6	7	10
2011	9	6	5	8
2012	5	7	7	7
2013	7	0	0	0
2014	0	7	6	0
2015	0	6	7	7
2016	7	2	7	7
2017	7	6	7	1

(b)	Quarter			
Year	1	2	3	4
2009	7	9	9	11
2010	8	6	7	10
2011	9	6	5	8
2012	5	7	7	7
2013	*0*	0	0	0
2014	*7*	7	6	*2*
2015	*2*	6	7	*5*
2016	*5*	2	7	*6*
2017	7	6	7	*2*

(c)	Quarter			
Year	1	2	3	4
2009				
2010				
2011				
2012				
2013				
2014	*2*	7	6	*2*
2015	*2*	6	7	*5*
2016	*3*	2	7	*6*
2017	7	6	7	*2*

The final bootstrap outputs provided for each species were: 1) the mean numbers impinged by quarter, which was converted to percentage impingement by quarter to show the seasonal patterns and 2) the mean annual impingement with 95th percentile confidence intervals based on 5000 bootstrap iterations.

The weights of finfish were recorded during each sampling visit providing data on the average observed weight of each fish species. Predicted impingement weight for each finfish species was established from observed data of average weights. The total weight of each species impinged was divided by the number impinged during monitoring to obtain an average weight. Average weights for each species were multiplied by the predicted mean annual impingement numbers and 95th percentile confidence intervals.

In the case of invertebrate taxa and seaweed, some species were weighed but not enumerated during impingement monitoring and individual weights for those that were enumerated are prone to reduced precision. Therefore, invertebrate and seaweed weight was estimated by applying the bootstrapping method to calculate weight impinged and 95th percentile confidence intervals.

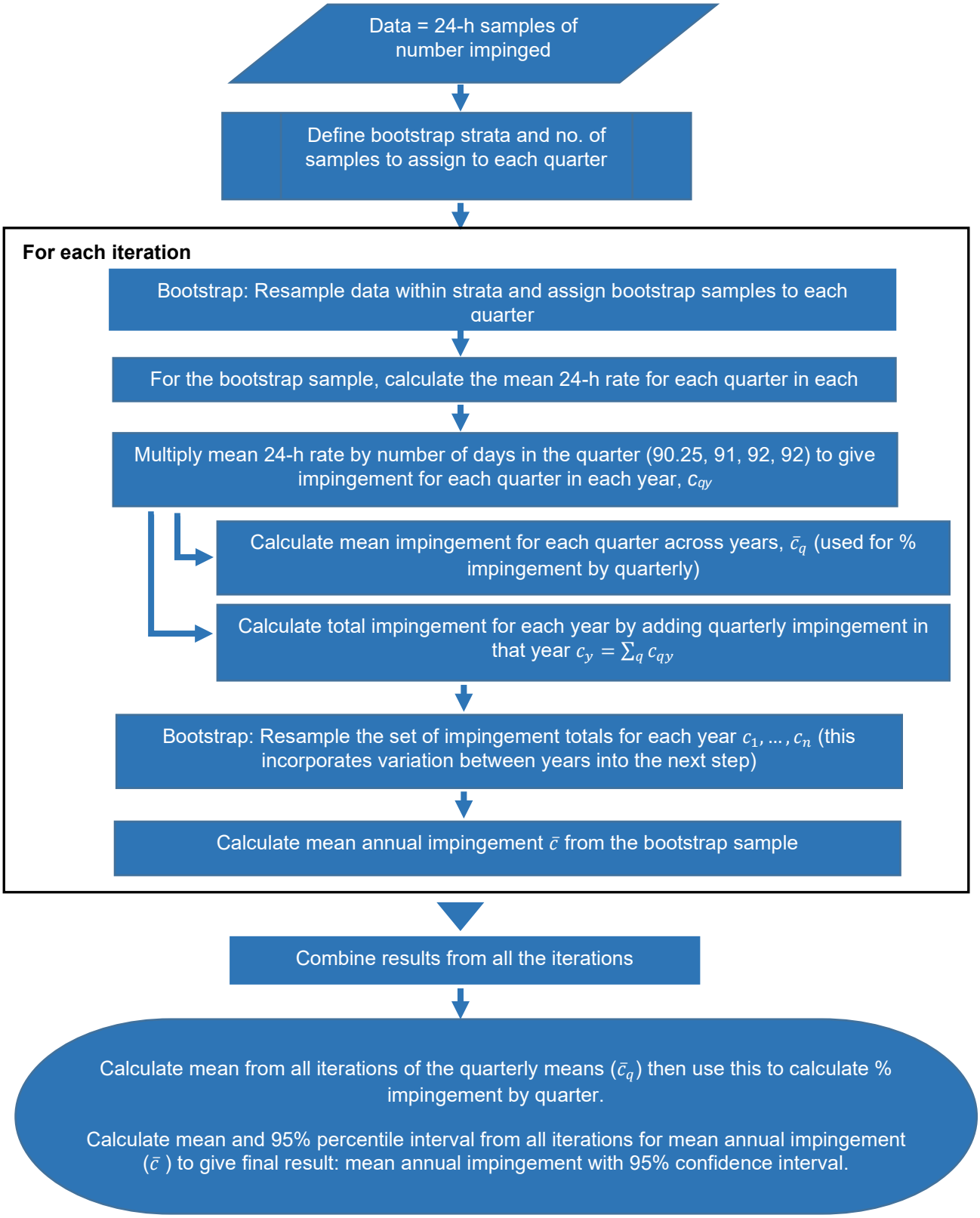

Data = 24-h samples of number impinged
Define bootstrap strata and no. of samples to assign to each quarter
For each iteration
Bootstrap: Resample data within strata and assign bootstrap samples to each quarter
For the bootstrap sample, calculate the mean 24-h rate for each quarter in each
Multiply mean 24-h rate by number of days in the quarter (90.25, 91, 92, 92) to give impingement for each quarter in each year, c_{qy}
Calculate mean impingement for each quarter across years, $\bar{c}_q$ (used for % impingement by quarterly)
Calculate total impingement for each year by adding quarterly impingement in that year $c_y = \sum_q c_{qy}$
Bootstrap: Resample the set of impingement totals for each year $c_1, \ldots, c_n$ (this incorporates variation between years into the next step)
Calculate mean annual impingement $\bar{c}$ from the bootstrap sample
Combine results from all the iterations
Calculate mean from all iterations of the quarterly means ($\bar{c}_q$) then use this to calculate % impingement by quarter.
Calculate mean and 95% percentile interval from all iterations for mean annual impingement ($\bar{c}$) to give final result: mean annual impingement with 95% confidence interval.

Figure 6 The bootstrap process applied to the Sizewell Comprehensive Impingement Monitoring Programme data

3.8 Individual species catches and size distributions

Finally, the individual catches (per 24-h) and annual length distributions were plotted for the most abundant finfish and invertebrate species (Appendix E). The length distributions data are required to calculate the Equivalent Adult Value (EAV) of the fish impinged (BEEMS Technical Report TR383) allowing losses to be contextualised against the relevant Spawning Stock Biomass (SSB) or fisheries landing statistics for the species in question (BEEMS Technical Report TR406).

4 Results

4.1 Summary of sampling completed 2014 – 2017

The first CIMP ran from February 2009 to March 2013. During the second CIMP, sampling was carried out between:

- 1 & 2 April and 3 & 4 September 2014;
- 28 & 29 April 2015 and 14 & 15 March 2016; and
- between 21 & 22 June 2016 and 5 & 6 October 2017.

Thirteen sampling visits were completed in 2014, 20 in 2015, 23 in 2016 and 21 in 2017 – a total of 77 samples. Only 17 complete 24-h samples were obtained. On four occasions, the sampling team were not able to collect the Hourly 1 sample – the first occasion was the first day that the team was on site, and on the other three occasions, the team were delayed with getting permission from station staff to divert the water flow. On 49 occasions, the bulk sample overflowed. On some occasions, the flow was stopped overnight by station staff, but only after the trash bin had already overflowed. On the remaining occasions, the sampling team arrived to find that the flow was still going and that the net was overflowing. On 6 occasions, the bulk sample was not started as the flow of material through the system indicated that it would almost certainly overflow, and the sample would be invalid. Data from all these samples were excluded from the analysis, as it was impossible to determine the lost portion of the sample. A summary of the number of visits completed each month are given in Table 3, and full details of the sampling visits and sampling dates are given in Appendix A.

Table 3 Summary of the number of sampling visits completed between 2009 – 2017

	First CIMP						Second CIMP					
	2009	2010	2011	2012	2013	Total	2014	2015	2016	2017	Total	Total
January		2	5	1	2	10			2	3	5	15
February	3	4	2	3	3	15			3	2	5	20
March	4	2	2	1	2	11			2	2	4	15
Q1 total	**7**	**8**	**9**	**5**	**7**	**36**	**0**	**0**	**7**	**7**	**14**	**50**
April	3	3	2	2		10	3	1		2	6	16
May	3	3	2	3		11	2	3		1	6	17
June	3		2	2		7	2	2	2	3	9	16
Q2 total	**9**	**6**	**6**	**7**	**0**	**28**	**7**	**6**	**2**	**6**	**21**	**49**
July	4	2	2	2		10	3	3	2	3	11	21
August	4	2	3	3		12	2	2	2	3	9	21
September	1	3		2		6	1	2	3	1	7	13
Q3 total	**9**	**7**	**5**	**7**	**0**	**28**	**6**	**7**	**7**	**7**	**27**	**55**
October		4	3	3		10		3	2	1	6	16
November	7	3	3	2		15		2	3		5	20
December	4	3	2	2		11		2	2		4	15
Q4 total	**11**	**10**	**8**	**7**	**0**	**36**	**0**	**7**	**7**	**1**	**15**	**51**
Total	**36**	**31**	**28**	**26**	**7**	**128**	**13**	**20**	**23**	**21**	**77**	**205**

4.2 SZB estimated numbers of fish and invertebrates impinged

Ninety-one fish taxa were recorded, and of these, 40 species were recorded in <5 % of the samples (10 samples or less), and 18 species were present in >50 % of the samples available (103 samples or more). The eight most abundant taxa by number were sprat *Sprattus sprattus*, herring *Clupea harrengus*, whiting *Merlangius merlangus*, seabass *Dicentrarchus labrax*, sand gobies *Pomatoschistus* spp., Dover sole *Solea solea*, European anchovy *Engraulis encrasicolus*, and Dab *Limanda limanda.* A mean estimated 5.3 million finfish are impinged by the SZB station annually, with the top eight species contributing 95.3 % to the total abundance (Table 4).

Sixty-two invertebrate taxa were enumerated. The three most abundant invertebrate taxa by number were ctenophores, brown shrimp *Crangon crangon* and pink shrimp *Pandalus montagui*. These three taxa contributed 95.9 % by number to the total invertebrate count, with ctenophores alone contributing 86.2 % (Table 4). A mean estimated 73.9 million invertebrates were impinged by the SZB station annually between 2009-2017, including 63.7 million ctenophores.

The estimated mean number of fish and invertebrates impinged by SZB annually for all taxa is given in Appendix B. The numbers presented represent total impingement, they do not attempt to account for impingement mortality or survival. Fish mortality contextualised against relevant SSB or fisheries landings is provided in BEEMS Technical Report TR406.

Table 4 Number of finfish and invertebrates contributing to the top 95% of mean annual estimated SZB impingement.

Common name	Scientific name	Number impinged SZB	% of total	Cumulative total (%)
Finfish				
Sprat	*Sprattus sprattus*	2,703,033	50.9	50.9
Herring	*Clupea harengus*	1,018,211	19.2	70.1
Whiting	*Merlangius merlangus*	650,479	12.3	82.3
European seabass	*Dicentrarchus labrax*	276,943	5.2	87.6
Sand gobies	*Pomatoschistus* spp.	196,545	3.7	91.3
Dover sole	*Solea solea*	91,370	1.7	93.0
European anchovy	*Engraulis encrasicolus*	69,058	1.3	94.3
Dab	*Limanda limanda*	56,054	1.1	95.3
Invertebrates				
Ctenophores		63,722,358	86.2	86.2
Brown shrimp	*Crangon crangon*	6,022,385	8.1	94.3
Pink shrimp	*Pandalus montagui*	1,173,611	1.6	95.9

4.3 SZB estimated weight of fish, invertebrates and weed impinged

Seven species contributed to the top 95.6% of total finfish impingement by weight, these were herring, whiting, seabass, sprat, thin-lipped grey mullet *Liza ramada,* Atlantic cod *Gadus morhua,* and Dover sole. An estimated mean weight of 203.2 t of fish was impinged by SZB, with the top 7 species contributing 95.6% to the total by weight (Table 5).

Sixty-seven invertebrate taxa and eight macroalgae taxa were recorded for weight (i.e. some taxa could not be enumerated but were weighed). The annual estimated weight of invertebrates was 184 t, of which six taxa contributed to the top 95.7% of weight. These included ctenophores, jellyfish, brown shrimp, white weed, common prawn *Palaemon serratus*, and unidentified hydroids (Table 5). Gelatinous zooplankton, ctenophores and jellyfish were predicted to contribute approximately 156 t annually.

An estimated 35 t of weed was predicted to be impinged annually at SZB between 2009-2017, with the majority (53.6%) being brown seaweed taxa (Phaeophyceae). Together with red seaweed (Rhodophyceae), and *Ulva lactuca,* brown seaweeds contribute to a cumulative 98.1% of total annual impingement by weight.

The estimated mean weight of finfish, invertebrates and weed impinged annually by SZB is given in Appendix C.

Table 5 Weight (kg) of finfish and invertebrates contributing to the top 95% of the mean annual estimated SZB impingement.

Common name	Scientific name	Weight impinged (kg)	Mean observed individual weight kg (2009-17)	% of total	Cumulative total (%)
Finfish					
Herring	*Clupea harengus*	93,724	0.0920	46.1	46.1
Whiting	*Merlangius merlangus*	36,783	0.0565	18.1	64.2
European seabass	*Dicentrarchus labrax*	30,998	0.1119	15.2	79.4
Sprat	*Sprattus sprattus*	24,622	0.0091	12.1	91.6
Atlantic cod	*Gadus morhua*	3,308	0.5156	1.6	93.2
Thin-lipped grey mullet	*Liza ramada*	2,936	0.0590	1.4	94.6
Dover sole	*Solea solea*	2,075	0.0227	1.0	95.6
Invertebrates					
Ctenophores		107,872		58.6	58.6
Jellyfish	Cnidaria	48,138		26.2	84.8
Common (brown) shrimp	*Crangon crangon*	7,540		4.1	88.9
White weed		5,859		3.2	92.0
Common prawn	*Palaemon serratus*	3,410		1.9	93.9
Unidentified hydroids	Hydroida	3,374		1.8	95.7

4.4 Monthly patterns of impingement for key species

Many of the species identified as key in the Greater Sizewell Bay area show seasonal patterns of abundance (Table 6; Appendix D). The majority of sprat (86.1%) and thin-lipped grey mullet (73.0%) are captured in quarter 1 (Q1; January - March) each year. In the case of seabass, 99.5% are impinged in Q1 (79.9%) and Q4 (19.6%; October – December). Herring and whiting are also predominantly autumn and winter species, caught in Q1 and Q4 each year, whereas anchovy are impinged in the greatest numbers in Q2 (81.5%; April - June) and Q3 (18.4%; July - September). Dover sole are also more abundant in Q2 and Q3, during which time 72.6% of impingement occurs. For other species, such as plaice, impingement occurred throughout the year with less pronounced peaks in abundance.

Seasonal patterns of brown shrimp, ctenophores and jellyfish were similar to those reported in 2009 - 2013 (BEEMS Technical Report TR120; TR196; TR215; TR270), with ctenophores being most abundant in Q2 (44.1%) and Q3 (28.9%), jellyfish in Q3 (85.8%) and brown shrimps in Q3 (61.8%) and Q4 (23.2%).

The observed number of key finfish and shellfish impinged during each sampling visit (raised to 24-h and full pumping capacity) is given in Appendix D.

Table 6 Proportion (%) of key finfish and invertebrate taxa (ctenophores and jellyfish included for information) impinged annually at SZB, by month and species. Cells have been colour-coded to highlight seasons of greatest impingement; dark green = 50 – 100 % of total number impinged annually, light green = 20 - <50 %, yellow = 5 - <20 %, pink = 1 - <5 %, no colour = < 1 %.

Common name	Number impinged per annum at SZB	Proportion of annual number of fish impinged (%)			
		Q1	Q2	Q3	Q4
Finfish					
Sprat	2,703,033	86.1	2.9	2.2	8.8
Herring	1,018,211	57.6	13.0	8.0	21.3
Whiting	650,479	49.9	12.2	10.1	27.8
European seabass	276,943	79.9	0.4	0.1	19.6
Sand goby	196,545	8.2	1.6	37.9	52.4
Dover sole	91,370	15.1	48.8	23.8	12.3
Anchovy	69,058	0.0	81.5	18.4	0.1
Dab	56,054	33.0	5.4	31.8	29.8
Thin-lipped grey mullet	49,751	73.0	0.3	0.0	26.7
Flounder	14,569	39.8	28.4	15.4	16.4
Plaice	9,653	27.6	23.1	20.1	29.1
Smelt	8,122	8.5	21.9	40.9	28.7
Cod	6,415	69.9	7.3	2.0	20.7
Thornback ray	2,913	21.6	51.9	9.8	16.8
Eel	1,184	16.7	16.6	31.3	35.5
Twaite shad	1,168	13.9	45.5	37.9	2.7
River lamprey	1,144	12.8	26.7	26.3	34.2
Horse mackerel	695	2.4	1.7	40.3	55.6
Mackerel	185	14.9	70.2	8.5	6.4
Tope	27	0.0	49.9	50.1	0.0
Sea trout	4	0.0	100.0	0.0	0.0
Sea lamprey	2	0.0	100.0	0.0	0.0
Allis shad	1	0.0	100.0	0.0	0.0
Salmon					
Invertebrates					
Ctenophores	63,722,358	17.9	44.1	28.9	9.1
Brown shrimp	6,022,385	6.7	8.3	61.8	23.2
Jellyfish	232,726	0.5	13.6	85.8	0.0
Brown crab	53,188	2.4	28.0	57.9	11.7
Lobster	15	0	0	41.6	58.4

4.5 Size distribution of impinged key species

The estimated numbers at length (24-h, all 4 pumps in operation) of key finfish or invertebrate species are given in Appendix E.

5 Discussion

EDF Energy is required to evaluate the potential impact that the abstraction of large volumes of seawater will have on fish and invertebrate populations.

The basis of impingement predictions for SZC are data collected at the operational SZB station. These data are used to provide an estimate of mean annual impingement by the SZB station along with estimates of variability that can be raised to represent the proposed capacity of the new build site. SZB data are considered appropriate to use as the basis for evaluation, as offshore data collected during the BEEMS programme has indicated that the Greater Sizewell Bay area is largely homogenous, with few distinct fish community patterns observed (BEEMS Technical Report TR345). Note the distribution of over wintering juvenile seabass was the major exception to this statement with 95% of fish being found inshore of the Sizewell-Dunwich Bank. The implications of this finding are discussed further in BEEMS Technical Report TR406. This report deals with SZB impingement sampling and annual estimates of SZB impingement. Raising of the data to SZC and completion of the impingement predictions for the proposed new build is provided in BEEMS Technical Report TR406.

Impingement sampling was undertaken at SZB between 2009 and 2017. Each sampling visit consisted of sampling for a 24-h period, using a combination of hourly and overnight sampling to provide the necessary sampling duration. Fish, invertebrates and other material impinged on the drum screens was diverted from its normal return flow to the sea into sampling nets to allow collection of the material. If large volumes of material entered the net, sampling duration was reduced (temporal subsampling) to allow sufficient time for sampling/recording to take place before the next sample was ready. Fish were sorted to species, weighed and measured (or subsampled if the number of fish was too great). The invertebrate catch was sorted to species and the separate taxa were either weighed and counted or weighed only.

If temporal subsampling was required, the sample was raised to its intended duration (either one hour or 18 hours for overnight samples), before the samples were summed to give the total number and weight of each species impinged over the 24-h period. Finally, the catch was adjusted to account for the number of pumps in operation at the time of sampling versus the maximum number of pumps (four). The final data set consisted of 205 samples, each representing the number and weight of individuals passing through the SZB station, if it was pumping at full capacity (4 pumps in operation), over the 24-h period.

Ninety-one finfish taxa were recorded in the 2009-2017 CIMP. Of these twenty-three were identified in the Greater Sizewell Bay area (BEEMS Technical Report TR345), the final key finish taxa, salmon, was absent from the impingement dataset (no salmon were impinged).

The eight most abundant taxa by number were sprat *Sprattus sprattus*, herring *Clupea harrengus*, whiting *Merlangius merlangus*, seabass *Dicentrarchus labrax*, sand gobies *Pomatoschistus* spp., Dover sole *Solea solea*, European anchovy *Engraulis encrasicolus*, and Dab *Limanda limanda.* Together these eight species contributed 95.3% of the total numbers of fish impinged averaged over the period 2009 - 2017. Seven species contributed to the top 95.6% of total finfish impingement by weight, these were herring, whiting, seabass, sprat, Atlantic cod *Gadus morhua*, thin-lipped grey mullet *Liza ramada,* and Dover sole..

Many fish species showed distinct seasonal patterns of abundance, with some species predominating in either the winter or summer. The majority of sprat (86.1%) and thin-lipped grey mullet (73.0%) are captured in quarter 1 (Q1; January - March) each year. In the case of seabass, 99.5% are impinged in Q1 (79.9%) and Q4 (19.6%; October – December) each year. Herring and whiting are also predominantly winter species, caught in Q1 and Q4 each year, whereas anchovy are impinged in the greatest numbers in Q2 (81.5%; April - June) and Q3 (18.4%; July - September).

Seventy-five invertebrate and macroalgae taxa were recorded. The three most abundant invertebrate taxa by number were ctenophores, brown shrimp *Crangon crangon* and pink shrimp *Pandalus montagui*. These

three taxa contributed 95.9% by number to the total invertebrate count, with ctenophores alone contributing 86.2 %. Three taxa contributed 88.9% of the total invertebrate weight and included ctenophores, jellyfish, and brown shrimp. Ctenophores and jellyfish were responsible for 84.8% of the cumulative invertebrate weight. An estimated 35t of seaweed is impinged annually; brown seaweed (Phaeophyceae), red seaweed (Rhodophyceae), and *Ulva lactuca* contribute to a cumulative 98.1% of the seaweed impinged.

Seasonal patterns of brown shrimp, ctenophores and jellyfish were similar to those previously reported for 2009 - 2013 with ctenophores being most abundant in Q2, jellyfish in Q3 and brown shrimps in Q3 (61.8%) and Q4 (23.3%).

5.1 Impingement model development

Estimates of impingement for Hinkley Point C (HPC) were based on a CIMP conducted at Hinkley Point B (HPB) in 2009 – 2010, during which time 40 sampling visits were completed. For that dataset, estimates of annual impingement by HPB were obtained by grouping samples quarterly and raising the mean number and weight of each species in the quarter to the ratio between the number of days in the quarter and the number of samples available. Estimates of variability were obtained by bootstrapping the data in each quarter – a sample of the data is selected and raised as described to provide an annual estimate. This process is repeated, allowing mean and 95th percentile confidence intervals to be calculated. The SZB CIMP dataset is more extensive than for HPB, whereby 205 samples have been collected over nine years from 2009-2017. However, the approach used at Hinkley Point was not directly replicated for Sizewell. This is because the distribution of samples over the sampling period is uneven (Table 2), with no sampling in some quarters largely due to station outages. This patchiness, coupled with the highly seasonal impingement rates of many species (Table 6) would lead to over-or underestimation of the impingement for those missing quarters.

To address this perceived issue, version 1 of this report used modelling approaches to determine impingement estimates. The results from this report were then scaled to predict impingement estimates for Sizewell C using volumetric scaling between the two stations (BEEMS Technical Report TR406). The model chosen to predict impingement rates at SZB for each species was different based on the number of occasions individuals of that species were present. Where possible, a model was applied which estimated how the number of organisms impinged varied from month-to-month and between half-year periods (February to July and then August to January in each year) (Zero-Inflated Negative Binomial model (ZINB)). It then recombined these effects such that the estimate in a specific month and year was the month effect multiplied by the half-year effect. For the ZINB model to be effective a relatively large (~10% or greater) number of presences was needed, and the presences needed to be distributed amongst the various years, which was not always the case. If the ZINB model with month could not be used, the temporal resolution was reduced. Instead of assessing each month as an effect, each two months (one sixth of a year) was used. If there was still too little data for the model to converge, three months (one quarter of a year) was used as an effect. If the model still did not converge (as was the case for any species where there were less than 22 out of the 205 sampling occasions), a simpler model was used that only estimated the effect of month, with no estimate of interannual variability (Negative Binomial model (NB)). Version 1 of the report noted the resulting NB model estimate should be less accurate than for the ZINB model.

Therefore, within the ZINB models, there are 3 different levels of resolution depending on how much data were available: every month, every bimester, and every quarter. Negative Binomial models had monthly resolution but did not assess the influence of year. Each species was evaluated separately for abundance by number and by weight, and for each species the most complex model available was fitted. The resulting outputs were the daily modelled mean abundance (by number or weight) impinged, by month, along with confidence intervals (lower and upper estimates), for each species. Monthly impingement was calculated by multiplying the daily mean by the number of days in that month, and the mean annual impingement of SZB was calculated by summing the 12 monthly values. Lower and upper values were calculated in the same manner.

A limitation of the modelling approach was that a different model was applied to each species depending on the frequency of its occurrence. By fitting the most complex model that would converge, there was a risk of

overfitting the model for some species making the intervals sensitive to extreme observations. The second, most recognisable limitation was the estimation of variability. The lower and upper estimates represent an interval of modelled daily values for each species. Upper and lower estimates are then derived by multiplying the minimum/maximum value by the number of days in the month. This assumes that either the minimum monthly limit or maximum monthly limit was impinged on every day of the year. This is highly unlikely, and results in values that will overestimate the variability in impingement numbers/weights. As such the upper and lower values are not true 95th percentiles and Cefas urged caution in their application.

During stakeholder consultation on BEEMS Technical Report TR406, the Environment Agency expressed concerns regarding the variation in lower and upper estimates of impingement predictions for SZC. As a result, and as an independent process, Cefas completed a statistical review of the model and alternative methods.

Following statistical review, a simplified approach to estimating impingement numbers was applied in versions 2 onwards of this report. The approach is similar to the bootstrapping methods applied for the CIMP data at Hinkley Point, however, it incorporates multiple years of data and incorporates an additional step specifically to resolve the issue on missing sampling opportunities during station outages. For example, minimal sampling occurred in 2013 (sampling was suspended in March 2013) therefore no estimate was produced for 2013. There were also no samples in quarter 1 in 2014, quarter 4 in 2014 and quarter 1 in 2015 and one sample in quarter 4 of 2017. Gaps in sampling were addressed by randomly reallocating samples from the same quarter in adjacent years for each bootstrap iteration, ensuring a minimum of two samples per quarter. Whilst this approach retains the total number of samples and seasonal patterns, it is possible that the unobserved impingement in the quarter without samples differed from that reallocated, which may result in under or overestimates. There may also be effects on the confidence intervals. Variation within a quarter could be less than estimated as fewer samples were reallocated than would have be sampled, conversely there may be additional variation not included due to sharing information across adjacent years in these cases.

All statistical methods rely on the data being representative and this is also true for the bootstrap approach, where confidence intervals are based on resampling the data with no assumptions made about the statistical distribution of the values. If the observed data misses the periods of highest or lowest impingement this will affect the estimates and intervals. However, sampling coverage throughout the year was achieved for the majority of years within the sampling programme.

The main advantage of the bootstrapping approach is that the confidence intervals can be clearly defined as 95th percentiles and reflect variation in the observations without making distributional assumptions. Furthermore, the same routine is applied for each species rather than a range of models reducing in complexity depending on the frequency of occurrence across samples. The approach is also akin to that applied at Hinkley Point.

A limitation of the bootstrapping approach is the temporal resolution is reduced to quarter year outputs, rather than a monthly resolution for the most data rich species in the previous model. However, the bootstrap does estimate for each quarter in each year, similar to adding an interaction term between year and quarter into the model.

The results in this report provide a single estimate of annual impingement over the years sampled. This was built up from the quarterly estimates. However, the approach does not explicitly attempt to model time trends within the sampling period (impingement can be prone to large daily and annual variations) nor forecast future trends. Impingement sampling at SZB is currently ongoing to further increase data collection and provide a longer time series that will further increase knowledge on temporal variability in impingement at this site, while results are contextualised against relevant time periods of spawning stock biomass or fisheries landings in BEEMS Technical Report TR406.

The implications for changes in impingement estimates in BEEMS Technical Report TR406, resulting from applying the bootstrap method, are considered in the following section.

5.1.1 Comparison between approaches

The results for the two approaches to predict Sizewell B impingement are compared in this section. Impingement predictions for Sizewell C, presented in BEEMS Technical Report TR406 v6, are based on volumetric scaling of Sizewell B impingement in the first instance, prior to additional mitigation measures being incorporated into the assessment. As such, differences in the annual mean and upper/lower confidence intervals have implications for Sizewell C impingement estimates. An updated version of BEEMS Technical Report TR406 (v7) will be produced in November 2020.

The most notable difference between the impingement predictions is the reduction in the upper confidence intervals when applying the bootstrapping approach. For example, in the case of sprat the upper confidence interval reduces from 7.11 million fish per annum in the ZINB model to 4.55 million in the bootstrapped method, whilst the mean changed minimally between the two approaches (-2.9%). The same is observed in herring where the mean changes -2% between the ZINB and the bootstrap approach but the upper estimate of 9.10 million predicted in the ZINB model reduced to 1.55 million when the bootstrap 95th percentile is derived (Table 7).

The mean estimates for the two approaches would be expected to converge the closest when the frequency a species occurs in impingement sampling is sufficiently high to smooth sporadic impingement events. For the eight species that account of over 95% of the estimated abundance (Section 4.2), modelling was able to apply the most complex ZINB with monthly resolution for each year, with the exception of anchovy. The two approaches converged closely for most species frequently impinged, with less than 4% differences in predicted annual numbers for sprat, herring, dab, flounder, cod and plaice. Dover sole, whiting, and smelt impingement predictions also match relatively closely between the two approaches with the bootstrapping method predicting small decreases in impingement, from 6.4% for Dover sole to 12.9% for smelt (Table 7).

In the case of seabass and sand gobies the bootstrapping method predicts increases in impingement of 23.3% and 31.9%, respectively in comparison to the ZINB model. Impingement of these species is seasonal and prone to large daily and annual variations in observed impingement numbers (Appendix D). The model fit aimed to estimate average monthly effects across years. For seabass, peak impingement values included five observations greater than 9000 fish per 24 hours during 2010 quarter 1. The bootstrap results were influenced more directly by these as they are quarter specific, producing a higher estimate. For sand gobies, peak impingement values were 16,804 on 29/9/2016 and 9,651 on 8/11/2016, Here, these individual values influenced bootstrap estimates across two quarters, increasing their effect relative to the model that was fitted with the effect of months. Whilst large variations in the data are also true for sprat and herring, the number of impingement events for these two most abundantly impinged species and relative regularity in impingement patterns allows the two approaches to converge.

The largest difference between the bootstrapping approach and the model for the key finfish taxa was for anchovy (139%) and thin-lipped mullet (88%). Both species were modelled using the ZINB with the time scale reduced to using bimonthly data. Both species are highly seasonal and the variation in numbers observed in impingement estimates each quarter relative to the total impingement (see Appendix D) is likely the cause of the divergence between the two model approaches.

The bootstrapping approach predicts lower impingement rates for less common species such as eel (-32.9%), river lamprey (-56.4%), twaite shad (-17.0%) and thornback ray (-30.9%). These species are impinged at relatively low numbers and do not have large seasonal fluctuations in impingement (Table 6). The reductions in impingement predictions are the result of the bootstrapping method being less susceptible to seasonal variations in abundance of species impinged at low frequency and the model predictions not fully reflecting the zero catches. For the bootstrap, seasonal peaks from one year do not influence estimates in other years, so specific quarters of zero or very low catch in one year will be directly estimated with the mean for that quarter and year.

The bootstrapping method with randomly reallocating samples to account for sampling gaps, is considered the most robust approach for calculating mean annual impingement estimates with 95th percentile confidence

intervals clearly defined. This method will be applied to Sizewell C impingement predictions in BEEMS Technical Report TR406 v7.

Table 7 Comparison of the bootstrapping approach applied in version 3 of this report and the model used to predict impingement rates in version 1 with key finfish and invertebrate taxa (including gelatinous zooplankton for information). The colour code for the model is explained in the key below.

Common name	Bootstrap predictions (TR339 version 3)			Model prediction (TR339 version 1)			Difference in mean (% change)
	Mean	Lower	Upper	Mean	Lower	Upper	
Sprat	2,703,033	1,427,342	4,549,040	2,782,934	1,089,329	7,110,821	-79,901 (-2.9%)
Herring	1,018,211	598,405	1,550,795	998,201	109,735	9,081,550	20,010 (2.0%)
Whiting	650,479	452,300	888,210	728,597	178,790	2,969,271	-78,118 (-10.7%)
European seabass	276,943	124,683	500,639	224,719	61,132	826,468	52,224 (23.2%)
Sand goby	196,545	81,220	383,493	149,045	40,410	549,860	47,500 (31.9%)
Dover sole	91,370	63,510	125,792	97,665	39,070	244,218	-6,295 (-6.4%)
Anchovy	69,058	18,123	166,131	28,849	4,627	180,746	40,209 (139.4%)
Dab	56,054	33,869	93,069	58,163	8,659	390,758	-2,109 (-3.6%)
Thin-lipped grey mullet	49,751	13,394	97,783	26,435	578	1,209,172	23,316 (88.2%)
Flounder	14,569	10,943	19,251	14,912	6,298	35,319	-343 (-2.3%)
Plaice	9,653	6,328	14,355	9,877	3,306	29,540	-224 (-2.3%)
Smelt	8,122	5,405	11,295	9,320	3,216	27,101	-1,198 (-12.9%)

Common name	Bootstrap predictions (TR339 version 3)			Model prediction (TR339 version 1)			Difference in mean (% change)
	Mean	Lower	Upper	Mean	Lower	Upper	
Cod	6,415	2,440	11,433	6,579	1,678	26,056	-164 (-2.5%)
Thornback ray	2,913	1,808	4,292	4,219	1,185	15,492	-1,306 (-30.9%)
Eel	1,184	762	1,719	1,764	714	5,015	-580 (-32.9%)
Twaite shad	1,168	574	2,036	1,407	224	9,275	-239 (-17.0%)
River lamprey	1,144	630	1,932	2,624	929	7,935	-1,480 (-56.4%)
Horse mackerel	695	220	1,715	1,592	312	11,890	-897 (-56.4%)
Mackerel	185	17	502	245	180	352	-60 (-24.6%)
Tope	27	0	106	25	12	51	2 (8.0%)
Sea trout	4	0	23	4	1	15	0 (0.0%)
Sea lamprey	2	0	11	2	0	13	0 (0.0%)
Allis shad	1	0	8	2	0	13	-1 (-50.0%)
Salmon							
Ctenophores	63,722,358	27,658,442	113,953,359	50,900,460	16,324,772	162,596,067	12,821,898 (25.2%)
Brown shrimp	6,022,385	4,293,876	8,357,183	6,277,209	3,296,740	11,966,573	-256,313 (-4.1%)

Common name	Bootstrap predictions (TR339 version 3)			Model prediction (TR339 version 1)			Difference in mean (% change)
	Mean	Lower	Upper	Mean	Lower	Upper	
Jellyfish	232,726	1,856	931,667	69,056	10	560,031,984	163,670 (237.0%)
Brown crab	53,188	28,452	82,122	40,730	8,556	193,949	12,435 (30.6%)
Lobster	15	0	51	17	6	58	-2 (-11.8%)

Model Code (in version 1)
Zero-Inflated Negative Binomial with two factors for Month and Year (Feb-Jul; Aug – Jan)
Zero-Inflated Negative Binomial with two factors for Sixth (2 months) and Year
Zero-Inflated Negative Binomial with two factors for Quarter and Year
Negative Binomial with one factor for Month

References

BEEMS Technical Report TR120. SZ Comprehensive Impingement Monitoring Programme 2009/10: Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR196. SZ Comprehensive Impingement Monitoring Programme Year II; Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR215. Comprehensive Impingement Monitoring Programme, Year III – Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR270. Comprehensive Impingement Monitoring Programme 2012-2013 at Sizewell B: Annual report. Cefas, Lowestoft.

BEEMS Technical Report TR345. Sizewell characterisation report - fish. Cefas, Lowestoft.

BEEMS Technical Report TR383. Sizewell - Equivalent Adult Value (EAV) metrics. Cefas, Lowestoft.

BEEMS Technical Report TR406. Sizewell C - impingement predictions based upon specific cooling water system design. Cefas, Lowestoft.

BEEMS Technical Report TR456. Ed2. v10. Revised predictions of impingement at Hinkley Point C - 2018. Cefas, Lowestoft.

Appendix A Sampling summary

Summary of sampling carried out during Cefas impingement sampling between April 2014 and October 2017. Each sample is designated by the sampling date, given as YYYY-MM-DD, where Day is the first day of the two-day sampling trip. The number of pumps in operation and the sampling duration (minutes) for each sample is given. For the bulk samples, where the trash bin overflowed before the sampling was stopped, the sample was deemed as invalid and was not used in the raising calculations and the visit is highlighted in yellow. On four occasions (highlighted in blue), the first hourly sample could not be completed, due to issues with getting on-site permission to start work and on one occasion (highlighted in pink) sampling was abandoned after two hours due to severe weather conditions. All times GMT.

		Sampling duration (minutes)						
Sampling visit	**Number of pumps**	**Hourly 1 09:00-10:00**	**Hourly 2 10:00-11:00**	**Hourly 3 11:00-12:00**	**Hourly 4 12:00-13:00**	**Hourly 5 13:00-14:00**	**Hourly 6 14:00-15:00**	**Bulk 15:00-09:00**
2014								
2014-04-01	3		35	60	60	60	60	1100
2014-04-15	3	41	60	60	60	60	60	1096
2014-04-30	3	60	60	60	60	60	60	1080
2014-05-15	4	60	60	60	60	60	60	*560*
2014-05-29	4	30	15	15	15	15	30	*525*
2014-06-09	4	20	30	25	30	10	10	*1065*
2014-06-23	4	20	20	20	15	15	15	50
2014-07-02	4	15	15	15	20	10	15	*1050*
2014-07-16	4	30	20	30	20	20	20	*1050*
2014-07-30	4	60	60	60	60	60	60	*1050*
2014-08-12	4	60	60	60	60	60	60	*814*
2014-08-27	4	45	30	30	30	30	30	*1050*
2014-09-03	4	35	23	60	60	60	60	1071
2015								
2015-04-28	3		45	60	60	60	60	1080
2015-05-06	4	60	60	60	60	60	60	1080
2015-05-14	4	60	60	60	60	60	60	*1110*
2015-05-28	4	60	60	60	60	60	60	*1080*
2015-06-15	4	30	60	60	60	60	60	*1035*
2015-06-25	4	30	10	10	10	15	15	*720*
2015-07-02	4	14	15	15	15	15	15	*570*
2015-07-15	4	20	20	15	20	20	20	*1020*
2015-07-28	4	23	30	45	40	15	15	*1030*
2015-08-03	4	25	40	50	38	20	35	*1030*
2015-08-18	4		60	60	60	60	60	1080
2015-09-08	4	60	60	60	60	60	60	1080
2015-09-29	4	60	60	60	60	60	60	1070
2015-10-08	4	60	60	60	60	60	60	1080
2015-10-15	4	50	45	60	45	45	45	*1035*
2015-10-29	4	45	60	60	60	60	60	1080
2015-11-03	4	60	60	60	60	60	60	1080
2015-11-18	3	60	45	30	45	60	60	*940*

		Sampling duration (minutes)						
Sampling visit	Number of pumps	Hourly 1 09:00-10:00	Hourly 2 10:00-11:00	Hourly 3 11:00-12:00	Hourly 4 12:00-13:00	Hourly 5 13:00-14:00	Hourly 6 14:00-15:00	Bulk 15:00-09:00
2015-12-01	3	45	30	30	30	60	60	*1020*
2015-12-15	3	45	60	60	60	60	60	*1050*
2016								
2016-01-07	3	60	60	45	30	30	20	*1125*
2016-01-19	3	15	30	45	40	60	60	*1078*
2016-02-01	3	35	30	30	60	60	60	*791*
2016-02-11	3	60	60	60	60	60	60	*666*
2016-02-23	3	60	60	60	60	30	45	*660*
2016-03-03	3	60	60	60	60	60	60	*1065*
2016-03-14	3	60	60	20	30	35	15	790
2016-06-21	4	120	120	120	60	60	60	-
2016-06-30	4	60	60	60	60	60	60	*1050*
2016-07-07	4	60	60	60	60	52	30	*1080*
2016-07-14	4	10	15	45	57	60	60	*1050*
2016-08-02	4	20	20	15	15	15	15	*1050*
2016-08-17	4	25	15	10	15	10	12	*240*
2016-09-06	4	60	60	60	60	18	25	60
2016-09-20	4	60	45	30	30	15	15	*995*
2016-09-29	4	15	20	30	15	15	15	*1080*
2016-10-11	4	60	60	60	60	60	60	1080
2016-10-31	4	60	60	60	60	60	60	1080
2016-11-08	4	60	60	60	60	60	60	*1050*
2016-11-15	3	60	60	60	60	60	60	640
2016-11-24	3	25	20	20	20	20	30	*295*
2016-12-07	3	60	55	45	45	45	40	*540*
2016-12-19	3	60	45	45	55	60	60	*1050*
2017								
2017-01-10	3	15	15	30	30	30	30	-
2017-01-18	3	45	45	30	30	45	40	*1080*
2017-01-26	3	30	30	20				-
2017-02-07	3	25	50	30	15	15	30	*720*
2017-02-23	3	90	80	45	30	30	30	-
2017-03-06	3	60	60	60	60	45	45	840
2017-03-22	3	60	60	60	60	60	60	*920*
2017-04-17	3	60	60	60	60	60	60	-
2017-04-26	3		60	60	60	60	60	1080
2017-05-11	3	60	60	60	60	60	60	1070
2017-06-01	4	45	60	60	60	60	60	*1050*
2017-06-15	3	30	20	30	45	45	60	-
2017-06-26	4	60	60	60	60	65	30	*690*
2017-07-02	3	30	45	60	60	60	45	1070
2017-07-13	4	60	60	60	58	40	35	*1050*
2017-07-26	3	45	60	60	60	50	35	*680*
2017-08-07	4	55	15	15	15	10	10	-

		Sampling duration (minutes)						
Sampling visit	**Number of pumps**	**Hourly 1 09:00-10:00**	**Hourly 2 10:00-11:00**	**Hourly 3 11:00-12:00**	**Hourly 4 12:00-13:00**	**Hourly 5 13:00-14:00**	**Hourly 6 14:00-15:00**	**Bulk 15:00-09:00**
2017-08-21	4	60	40	45	45	45	30	*1050*
2017-08-30	4	40	40	40	60	60	60	*1050*
2017-09-20	4	30	60	60	60	45	60	*1065*
2017-10-05	4	60	60	60	60	60	60	*1050*

Appendix B Estimated number of finfish and invertebrates impinged annually at SZB

Annual predicted numbers of individuals that are estimated to be impinged by SZB, based on data from 2009-2017. Colour-coding indicates the 23 key finfish (no salmon impinged) species in the Greater Sizewell Bay (BEEMS Technical Report TR345). The 31 invertebrates species marked with an asterisk (*) were only enumerated from 2014 onwards, all other species were enumerated from 2009 onwards.

The numbers presented represent total impingement, they do not attempt to account for impingement mortality or survival. Fish mortality contextualised against relevant SSB or fisheries landings is provided in BEEMS Technical Report TR406.

Key finfish and shellfish species

Common Name	Scientific Name	Mean	Lower	Upper
Finfish				
Sprat	*Sprattus sprattus*	2,703,033	1,427,342	4,549,040
Herring	*Clupea harengus*	1,018,211	598,405	1,550,795
Whiting	*Merlangius merlangus*	650,479	452,300	888,210
European seabass	*Dicentrarchus labrax*	276,943	124,683	500,639
Sand gobies	*Pomatoschistus spp*	196,545	81,220	383,493
Dover sole	*Solea solea*	91,370	63,510	125,792
European anchovy	*Engraulis encrasicolus*	69,058	18,123	166,131
Dab	*Limanda limanda*	56,054	33,869	93,069
Thin-lipped grey mullet	*Liza ramada*	49,751	13,394	97,783
Transparent goby	*Aphia minuta*	36,461	12,857	79,828
Bib	*Trisopterus luscus*	24,975	12,650	40,450
Lesser weever fish	*Echiichthys (trachinus) vipera*	20,222	12,839	30,149
Flounder	*Platichthys flesus*	14,569	10,943	19,251
Nilsson's pipefish	*Syngnathus rostellatus*	14,247	2,315	29,139
Pogge (hooknose)	*Agonus cataphractus*	10,410	7,185	14,587
European plaice	*Pleuronectes platessa*	9,653	6,328	14,355
Five-bearded rockling	*Ciliata mustela*	8,941	5,773	13,071
Cucumber smelt	*Osmerus eperlanus*	8,122	5,405	11,295
Atlantic cod	*Gadus morhua*	6,415	2,440	11,433
Lesser spotted dogfish	*Scyliorhinus canicula*	5,359	2,543	9,415
Great pipefish	*Syngnathus acus*	4,672	2,132	7,734
Grey mullets	*Mugilidae*	4,081	0	17,752
Thornback ray	*Raja clavata*	2,913	1,808	4,292
Common sea snail	*Liparis liparis*	2,902	1,514	4,584
Tub gurnard	*Trigla lucerna*	2,259	1,379	3,497
Pilchard	*Sardina pilchardus*	1,769	374	5,736

Common Name	Scientific Name	Mean	Lower	Upper
Unidentified herrings	*Clupeidae*	1,695	0	7,410
Common dragonet	*Callionymus lyra*	1,614	690	2,993
Poor cod	*Trisopterus minutus*	1,444	298	3,114
Starry smooth-hound	*Mustelus asterias*	1,379	683	2,202
European eel	*Anguilla anguilla*	1,184	762	1,719
Twaite shad	*Alosa fallax*	1,168	574	2,036
River lamprey	*Lampetra fluviatilis*	1,144	630	1,932
Three-spined stickleback	*Gasterosteus aculeatus*	1,121	314	3,012
Common sandeel	*Ammodytes tobianus*	1,021	377	2,179
Bullrout	*Myoxocephalus scorpius*	972	447	1,694
Scald fish	*Arnoglossus laterna*	803	416	1,323
Horse-mackerel	*Trachurus trachurus*	695	220	1,715
Witch	*Glyptocephalus cynoglossus*	664	4	2,043
Black goby	*Gobius niger*	576	72	1,699
Great sandeel	*Hyperoplus lanceolatus*	565	201	1,001
Brill	*Scophthalmus rhombus*	445	148	950
Rock goby	*Gobius paganellus*	405	12	1,511
Snake pipefish	*Entelurus aequoreus*	362	3	1,638
Lemon sole	*Microstomus kitt*	333	125	652
Tompot blenny	*Parablennius gattorugine*	247	26	802
Montague's seasnail	*Liparis montagui*	231	9	752
Solenette	*Buglossidium luteum*	221	68	446
Sand Smelt	*Atherina boyeri*	188	8	462
Mackerel	*Scomber scombrus*	185	17	502
Red mullet	*Mullus surmuletus*	145	41	309
Butter fish	*Pholis gunnellus*	138	47	257
Sea scorpion	*Taurulus bubalis*	129	14	324
Jeffrey's goby	*Buenia jeffreysii*	110	0	606
Sandeels	*Ammodytidae*	95	0	296
Baillons wrasse	*Symphodus balloni*	83	0	259
Grey gurnard	*Eutrigla gurnardus*	82	15	197
Garfish	*Belone belone*	71	13	165
Sand smelt	*Atherina presbyter*	70	0	292
Eelpout (Viviparous blenny)	*Zoarces viviparus*	68	0	211
Turbot	*Scophthalmus maximus*	64	6	155
Lesser forkbeard (tadpolefish)	*Raniceps raninus*	60	2	174
Frie's goby	*Lesueurigobius friesii*	55	0	168
Corkwing wrasse	*Crenilabrus melops*	40	0	118
Northern rockling	*Ciliata septentrionalis*	40	0	128
John dory	*Zeus faber*	35	0	96

Common Name	Scientific Name	Mean	Lower	Upper
Sandeel	*Ammodytes marinus*	27	0	110
Tope	*Galeorhinus galeus*	27	0	106
Lumpsucker	*Cyclopterus lumpus*	24	0	68
Four-bearded rockling	*Enchelyopus cimbrius*	23	0	61
Ballan wrasse	*Labrus bergylta*	22	0	86
Spotted ray	*Raja montagui*	21	0	67
Saithe	*Pollachius virens*	20	0	117
Crystal goby	*Crystallogobius linearis*	16	0	91
Thick-lipped grey mullet	*Crenimugil labrosus*	12	0	68
Norway bullhead	*Micrenophrys lilljeborgii*	9	0	45
Black seabream	*Spondyliosoma cantharus*	9	0	42
Cuckoo wrasse	*Labrus mixtus*	9	0	50
Snake blenny	*Lumpenus lampretaeformis*	8	0	46
Goldsinny	*Ctenolabrus rupestris*	8	0	46
Bigeye rockling	*Antonogadus macropthalmus*	6	0	20
Deep-snouted pipefish	*Syngnathus typhle*	5	0	31
Sea Trout	*Salmo trutta*	4	0	23
Norway pout	*Trisopterus esmarkii*	3	0	19
Shore rockling	*Gaidropsarus mediterraneus*	3	0	15
Red gurnard	*Aspitrigla cuculus*	3	0	17
Sea lamprey	*Petromyzon marinus*	2	0	11
Spotted dragonet	*Callionymus maculatus*	2	0	10
Sand sole	*Pegusa lascaris*	2	0	10
Pollack	*Pollachius pollachius*	2	0	10
Allis shad	*Alosa alosa*	1	0	8
Invertebrates				
Ctenophores*		63,722,358	27,658,442	113,953,359
Brown shrimp	*Crangon crangon*	6,022,385	4,293,876	8,357,183
Pink shrimp	*Pandalus montagui*	1,173,611	798,148	1,660,660
Common prawn	*Palaemon serratus*	1,130,928	603,033	1,846,382
Common swimming crab	*Liocarcinus holsatus*	591,467	316,403	1,018,651
	Crangon allmanni	391,498	180,585	668,201
Jellyfish*		232,726	1,856	931,667
Plumose anemone*	*Metridium senile*	225,690	150,164	325,554
Isopod*	*Idoteidae*	108,544	18,886	282,587
Dahlia anemone*	*Urticina (tealia) felina*	82,123	56,554	111,967
Brown crab	*Cancer pagurus*	53,188	28,452	82,122
Little cuttlefish	*Sepiola atlantica*	28,028	19,254	40,383
Velvet swimming crab	*Necora puber*	26,707	8,220	55,431

Common Name	Scientific Name	Mean	Lower	Upper
Edible mussel*	*Mytilus edulis*	20,796	0	55,958
Green shore crab	*Carcinus maenas*	16,703	7,837	29,840
Unidentified spider crab*	*Macropodia spp*	14,609	920	42,144
Common starfish	*Asterias rubens*	12,098	2,694	24,294
European common squid	*Loligo (alloteuthis) subulata*	9,511	5,872	14,634
Opossum shrimps*	*Mysidacea*	9,417	0	33,445
Ragworms*	*Nereis spp.*	9,286	4,269	15,291
Long-leg spider crab	*Macropodia rostrata*	7,100	91	27,939
Scaleworms*		5,260	2,120	9,283
Anemone unidentified*	*Anemone unidentified*	4,748	84	12,486
Crangonid (brown) shrimps*	*Crangonidae*	3,564	0	14,727
Sea slugs*	*Nudibranchia*	3,249	232	8,957
Hairy crab	*Pilumnus hirtellus*	2,980	386	7,589
Ghost shrimp	*Pasiphaea sivado*	1,848	0	10,189
Rissos crab*	*Xantho pilipes*	1,755	0	6,782
*	*Bolocera tuediae*	1,486	0	3,808
Long clawed porcelain crab*	*Pisidia longgicornis*	1,330	0	6,869
Lug-worm*	*Arenicola marina*	1,325	285	2,997
Swimming crab	*Liocarcinus depurator*	1,041	0	6,256
Slender spider crab*	*Macropodia tenuirostris*	792	8	2,276
Beadlet anemone*	*Actinia equina*	680	0	2,462
Contracted crab	*Hyas coarctatus*	518	13	1,635
Hermit crab	*Eupagurus bernhardus*	466	0	2,581
*	*Psammechinus miliaris*	438	15	1,464
	Processa canaliculata	427	31	1,184
Scorpion spider crab*	*Inachus dorsettensis*	412	0	2,275
Spider crabs*	*Majidae*	405	0	2,103
Hairy crab	*Pilumnus spinifer*	402	0	2,445
Unidentified sea urchin		374	0	1,493
Unidentified brittle star		321	0	1,599
Great spider crab	*Hyas araneus*	169	0	614
Dwarf-swimming crab	*Liocarcinus pusillus*	130	0	789
Marbled swimming crab	*Liocarcinus marmoreus*	93	0	489
Common cuttlefish	*Sepia officinalis*	68	0	361
European squid	*Loligo vulgaris*	59	0	349
Northern squid	*Loligo forbesi*	43	0	206
Xanthid crab*	*Xanthidae*	43	0	140
Necklace shell*	*Euspira (polinices) catena*	43	0	245
	Processidae	29	0	137
Bristle worms*	*Polychaeta*	25	0	106

Common Name	Scientific Name	Mean	Lower	Upper
Purple heart urchin*	*Spatangus purpureus*	16	0	92
Lobster	*Homarus gammarus*	15	0	51
*	*Upogebia deltaura*	13	0	78
*	*Gammaridae*	13	0	64
Brittle star*	*Ophiura ophiura*	13	0	79
Cuttle-fish*	*Sepia elegans*	6	0	27
Sea cucumbers*	*Holothuroidea*	4	0	27
Unidentified swimming crab	*Liocarcinus spp.*	3	0	15
	Axius stirhynchus	2	0	11

* Species enumeration began in 2014.

Appendix C Estimated weight (kg) of finfish, invertebrates and weed impinged annually at SZB

Annual predicted weight (kg) of finfish that are estimated to be impinged by SZB, based on data from 2009-2017. Weight is estimated from the observed average weight of individual fish scaled to the predicted numbers impinged in Appendix B. Colour-coding indicates the 23 key finfish (no salmon impinged) species in the Greater Sizewell Bay (BEEMS Technical Report TR345).

Invertebrate and weed weight is estimated based on bootstrapping methods as applied to calculate numbers.

Key species

Common Name	Scientific Name	Mean	Lower	Upper	Mean observed individual weight (2009-17)
Finfish					
Herring	Clupea harengus	93,724	55,082	142,747	0.0920
Whiting	Merlangius merlangus	36,783	25,576	50,225	0.0565
European seabass	Dicentrarchus labrax	30,998	13,956	56,036	0.1119
Sprat	Sprattus sprattus	24,622	13,002	41,437	0.0091
Atlantic cod	Gadus morhua	3,308	1,258	5,895	0.5156
Thin-lipped grey mullet	Liza ramada	2,936	790	5,770	0.0590
Dover sole	Solea solea	2,075	1,442	2,856	0.0227
Dab	Limanda limanda	1,997	1,207	3,316	0.0356
Flounder	Platichthys flesus	1,387	1,042	1,833	0.0952
European anchovy	Engraulis encrasicolus	936	246	2,252	0.0136
Bib	Trisopterus luscus	578	293	936	0.0232
Twaite shad	Alosa fallax	528	260	921	0.4525
Lesser spotted dogfish	Scyliorhinus canicula	404	192	710	0.0754
Sand gobies	Pomatoschistus spp	365	151	713	0.0019
Lesser weever fish	Echiichthys (trachinus) vipera	330	210	492	0.0163
European plaice	Pleuronectes platessa	319	209	475	0.0331
Thornback ray	Raja clavata	275	170	405	0.0942
Grey mullets	Mugilidae	236	0	1,026	0.0578
European eel	Anguilla anguilla	215	139	313	0.1818
Five-bearded rockling	Ciliata mustela	191	123	280	0.0214
Starry smooth-hound	Mustelus asterias	183	91	292	0.1327
Pilchard	Sardina pilchardus	162	34	527	0.0919
Cucumber smelt	Osmerus eperlanus	101	68	141	0.0125

Common Name	Scientific Name	Mean	Lower	Upper	Mean observed individual weight (2009-17)
Bullrout	*Myoxocephalus scorpius*	90	42	157	0.0929
River lamprey	*Lampetra fluviatilis*	61	34	103	0.0535
Tub gurnard	*Trigla lucerna*	58	35	90	0.0257
Horse-mackerel	*Trachurus trachurus*	56	18	137	0.0801
Mackerel	*Scomber scombrus*	43	4	117	0.2337
Pogge (hooknose)	*Agonus cataphractus*	36	25	51	0.0035
Transparent goby	*Aphia minuta*	36	13	78	0.0010
Turbot	*Scophthalmus maximus*	32	3	78	0.5016
Lumpsucker	*Cyclopterus lumpus*	27	0	77	1.1270
Great pipefish	*Syngnathus acus*	22	10	36	0.0047
Lemon sole	*Microstomus kitt*	17	6	33	0.0501
Common sea snail	*Liparis liparis*	16	8	25	0.0054
Tope	*Galeorhinus galeus*	15	0	60	0.5656
Brill	*Scophthalmus rhombus*	14	5	29	0.0308
Red mullet	*Mullus surmuletus*	11	3	24	0.0786
Common sandeel	*Ammodytes tobianus*	11	4	22	0.0103
Common dragonet	*Callionymus lyra*	9	4	17	0.0056
Scald fish	*Arnoglossus laterna*	9	5	15	0.0111
Poor cod	*Trisopterus minutus*	8	2	17	0.0054
Nilsson's pipefish	*Syngnathus rostellatus*	7	1	14	0.0005
Great sandeel	*Hyperoplus lanceolatus*	6	2	11	0.0114
Sea Trout	*Salmo trutta*	6	0	37	1.6000
Witch	*Glyptocephalus cynoglossus*	6	0	18	0.0087
Tompot blenny	*Parablennius gattorugine*	3	0	9	0.0109
Black goby	*Gobius niger*	3	0	7	0.0044
Sea scorpion	*Taurulus bubalis*	2	0	6	0.0188
Sea lamprey	*Petromyzon marinus*	2	0	13	1.2120
Grey gurnard	*Eutrigla gurnardus*	2	0	6	0.0289
Unidentified herrings	*Clupeidae*	2	0	10	0.0013
Saithe	*Pollachius virens*	2	0	12	0.1040
Three-spined stickleback	*Gasterosteus aculeatus*	2	1	5	0.0018
Allis shad	*Alosa alosa*	2	0	15	1.8585
Garfish	*Belone belone*	2	0	4	0.0256
Butter fish	*Pholis gunnellus*	2	1	3	0.0131
Rock goby	*Gobius paganellus*	2	0	6	0.0043
John dory	*Zeus faber*	2	0	4	0.0434
Solenette	*Buglossidium luteum*	1	0	3	0.0065

Common Name	Scientific Name	Mean	Lower	Upper	Mean observed individual weight (2009-17)
Sand Smelt	*Atherina boyeri*	1	0	3	0.0070
Eelpout (Viviparous blenny)	*Zoarces viviparus*	1	0	4	0.0179
Lesser forkbeard (tadpolefish)	*Raniceps raninus*	1	0	3	0.0181
Four-bearded rockling	*Enchelyopus cimbrius*	1	0	2	0.0400
Snake pipefish	*Entelurus aequoreus*	1	0	3	0.0021
Thick-lipped grey mullet	*Crenimugil labrosus*	1	0	4	0.0608
Montague's seasnail	*Liparis montagui*	1	0	2	0.0026
Corkwing wrasse	*Crenilabrus melops*	1	0	2	0.0140
Sand smelt	*Atherina presbyter*	0	0	2	0.0063
Jeffrey's goby	*Buenia jeffreysii*	0	0	2	0.0034
Baillons wrasse	*Symphodus balloni*	0	0	1	0.0044
Frie's goby	*Lesueurigobius friesii*	0	0	1	0.0063
Sandeel	*Ammodytes marinus*	0	0	1	0.0123
Ballan wrasse	*Labrus bergylta*	0	0	1	0.0149
Northern rockling	*Ciliata septentrionalis*	0	0	1	0.0080
Red gurnard	*Aspitrigla cuculus*	0	0	2	0.1000
Spotted ray	*Raja montagui*	0	0	1	0.0084
Snake blenny	*Lumpenus lampretaeformis*	0	0	1	0.0220
Shore rockling	*Gaidropsarus mediterraneus*	0	0	1	0.0558
Sandeels	*Ammodytidae*	0	0	0	0.0016
Sand sole	*Pegusa lascaris*	0	0	1	0.0526
Cuckoo wrasse	*Labrus mixtus*	0	0	0	0.0081
Goldsinny	*Ctenolabrus rupestris*	0	0	0	0.0080
Bigeye rockling	*Antonogadus macropthalmus*	0	0	0	0.0092
Deep-snouted pipefish	*Syngnathus typhle*	0	0	0	0.0041
Black seabream	*Spondyliosoma cantharus*	0	0	0	0.0018
Norway bullhead	*Micrenophrys lilljeborgii*	0	0	0	0.0015
Pollack	*Pollachius pollachius*	0	0	0	0.0057
Spotted dragonet	*Callionymus maculatus*	0	0	0	0.0040
Crystal goby	*Crystallogobius linearis*	0	0	0	0.0004
Norway pout	*Trisopterus esmarkii*	0	0	0	NA*

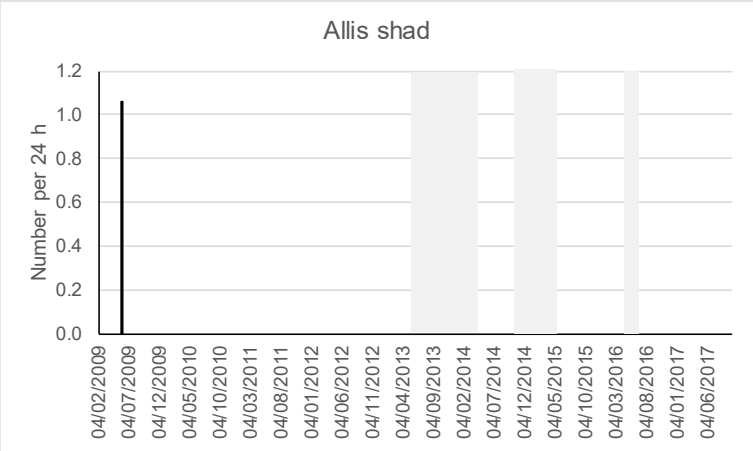
* Three Norway pout were impinged in monitoring samples, but weight data is not available.

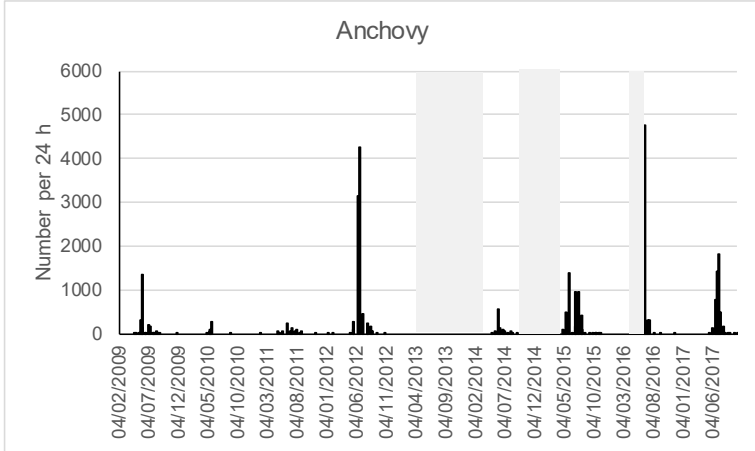
CommonName.TR339	ScientificName.TR339	Mean	Lower	Upper
Invertebrates				
Ctenophores		107,872	45,734	202,560
Jellyfish	Cnidaria	48,138	16,331	102,958
Brown shrimp	*Crangon crangon*	7,540	5,153	11,068
White weed		5,859	581	20,945
Common prawn	*Palaemon serratus*	3,410	2,059	5,269
Unidentified hydroids	*Hydroida*	3,374	1,575	5,570
Common swimming crab	*Liocarcinus holsatus*	2,540	1,468	4,008
Pink shrimp	*Pandalus montagui*	1,225	821	1,765
Plumose anemone	*Metridium senile*	639	433	959
Hornwrack	*Flustra foliacea*	581	88	1,512
Brown crab	*Cancer pagurus*	550	341	780
Dahlia anemone	*Urticina felina*	448	258	742
Velvet swimming crab	*Necora puber*	338	157	592
	Crangon allmanni	303	143	523
Hydroid	*Tubularia spp*	297	70	597
Curly weed	*Alcyonidium diaphanum*	196	15	588
Common starfish	*Asterias rubens*	190	59	379
Edible mussel	*Mytilus edulis*	146	0	565
Isopod	Idoteidae	85	21	200
Atlantic bobtail squid	*Sepiola atlantica*	65	45	91
Green shore crab	*Carcinus maenas*	45	20	78
European common squid	*Alloteuthis subulata*	41	26	59
Northern squid	*Loligo forbesi*	21	0	91
Ragworm	*Nereis spp*	15	7	25
Common cuttlefish	*Sepia officinalis*	13	0	57
Spider crab	*Macropodia spp*	11	1	29
Unidentified anemone		11	0	28
Whelk eggs		10	0	22
Sea slug	Nudibranchia	10	1	27
Green sea urchin	*Psammechinus miliaris*	6	0	18
Brown shrimps	Crangonidae	6	2	10
Scale worms	Polynoinae	6	0	33
Rissos crab	*Xantho pilipes*	5	0	20
Hairy crab	*Pilumnus hirtellus*	5	1	11
Sickle hydroid	*Hydrallmania falcata*	5	1	11
Lug-worm	*Arenicola marina*	5	0	19
Contracted spider crab	*Hyas coarctatus*	4	0	15
Swimming crab	*Liocarcinus depurator*	4	0	23
Lobster	*Homarus gammarus*	4	0	16

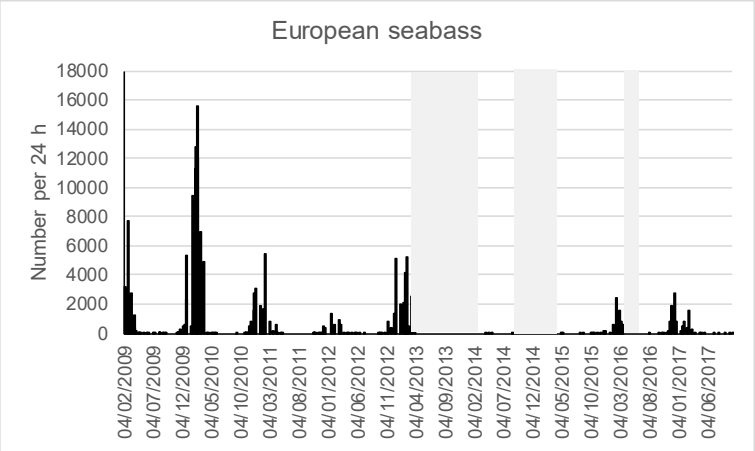
Long-leg spider crab	*Macropodia rostrata*	3	0	14
European squid	*Loligo vulgaris*	3	0	11
Opossum shrimps	Mysidacea	2	0	9
Anemone	*Bolocera tuediae*	2	0	6
Ghost shrimp	*Pasiphaea sivado*	2	0	11
Beadlet anemone	*Actinia equina*	2	0	5
Slender spider crab	*Macropodia tenuirostris*	1	0	4
Long clawed porcelain crab	*Pisidia longicornis*	1	0	5
Great spider crab	*Hyas araneus*	1	0	7
Hermit crab	*Eupagurus bernhardus*	1	0	5
Unidentified sea urchin		1	0	4
Sponges	Porifera	1	0	3
Scorpion spider crab	*Inachus dorsettensis*	1	0	3
	Processa canaliculata	1	0	1
Unidentified spider crabs	Majidae	0	0	2
White furrow shell	*Abra alba*	0	0	2
Dwarf-swimming crab	*Liocarcinus pusillus*	0	0	2
Purple heart urchin	*Spatangus purpureus*	0	0	1
Sponge	*Suberites spp.*	0	0	1
	Processidae	0	0	1
Marbled swimming crab	*Liocarcinus marmoreus*	0	0	1
Brittle star	Amphiuridae	0	0	1
Unidentified Xanthid crab	Xanthidae	0	0	0
Necklace shell	*Euspira catena*	0	0	0
Elegant cuttlefish	*Sepia elegans*	0	0	0
Thalassinoid shrimp	*Upogebia deltaura*	0	0	0
	Ophiura ophiura	0	0	0
Amphipod	Gammaridae	0	0	0
Polychaete worm	Polychaeta	0	0	0
Squid eggs		0	0	0
Sea cucumber	Holothuroidea	0	0	0
Seaweed				
Brown seaweeds	Phaeophyceae	18,912	816	56,847
Red seaweeds	Rhodophyceae	14,543	4,890	27,523
Sea lettuce	*Ulva lactuca*	972	50	4,756
Green seaweed	Chlorophyceae	344	149	650
Unidentified wrack	*Fucus spp.*	138	5	525
Bladder wrack	*Fucus vesiculosus*	101	0	500
Toothed wrack	*Fucus serratus*	61	0	373
Kelp	*Laminaria spp.*	6	0	27

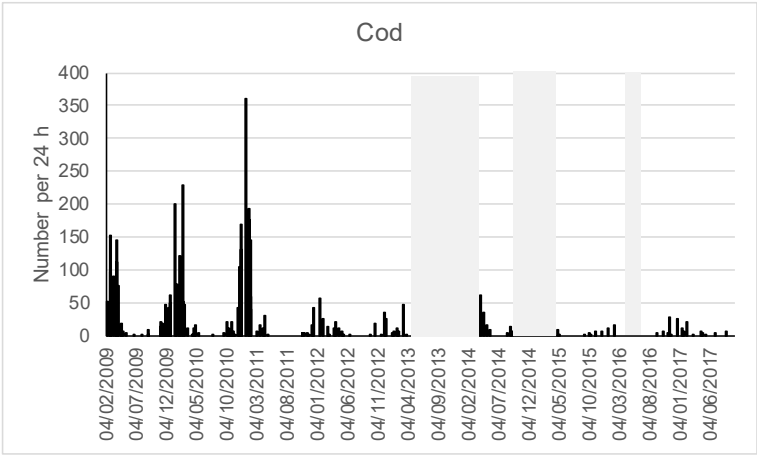
Appendix D Daily (24-h full capacity) impingement of key species

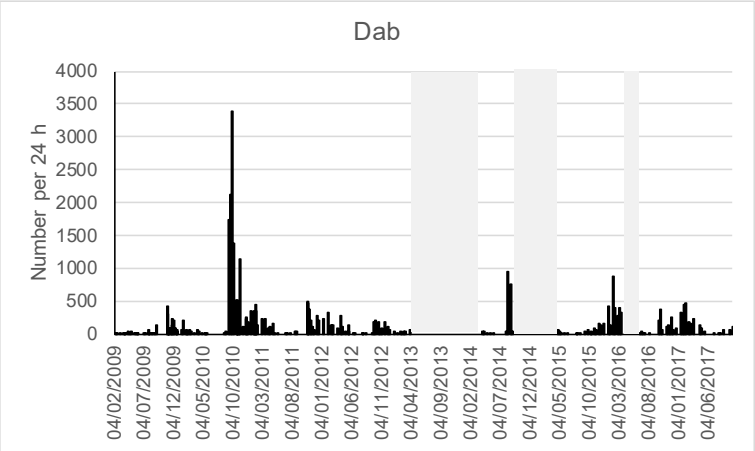
The observed number of key finfish and shellfish species impinged during each sampling visit (raised to 24-h and full pumping capacity). Grey shaded areas represent prolonged periods when sampling could not take place due to outages or other issues.

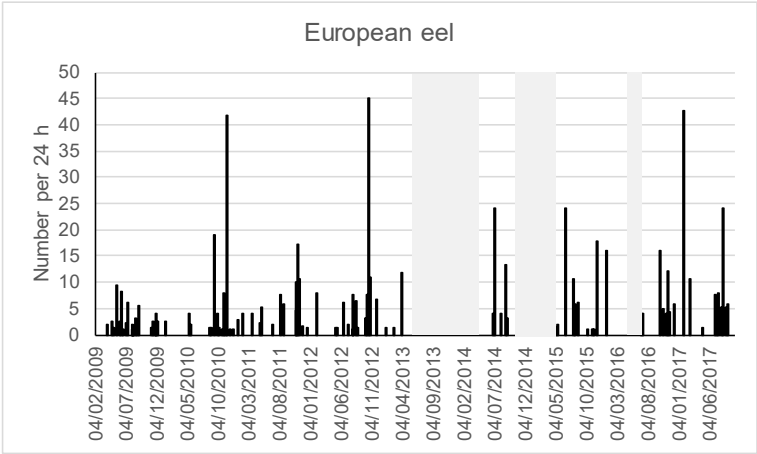


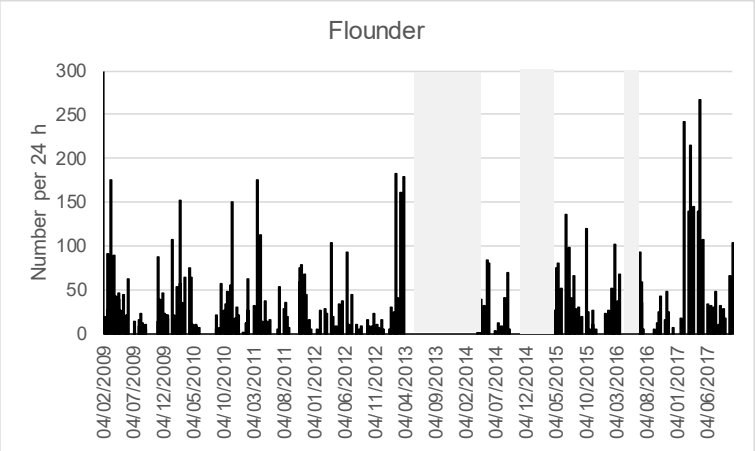


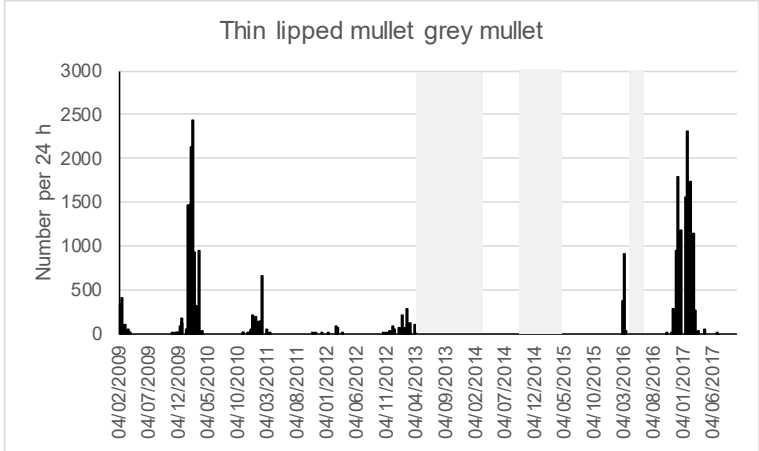

European seabass
Number per 24 h
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

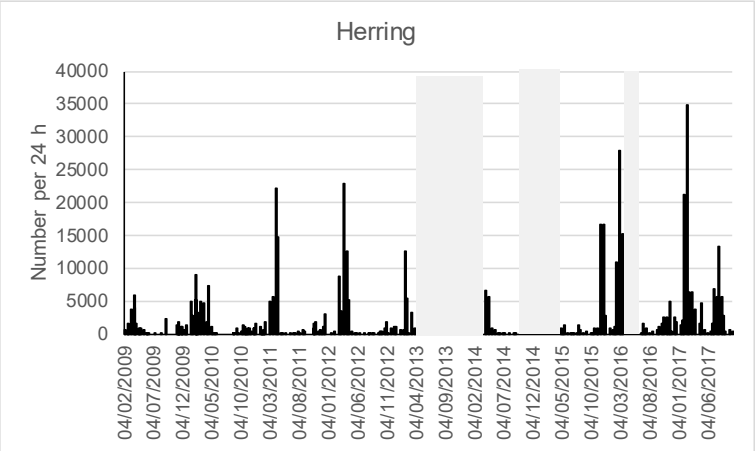

Cod
Number per 24 h
400
350
300
250
200
150
100
50
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

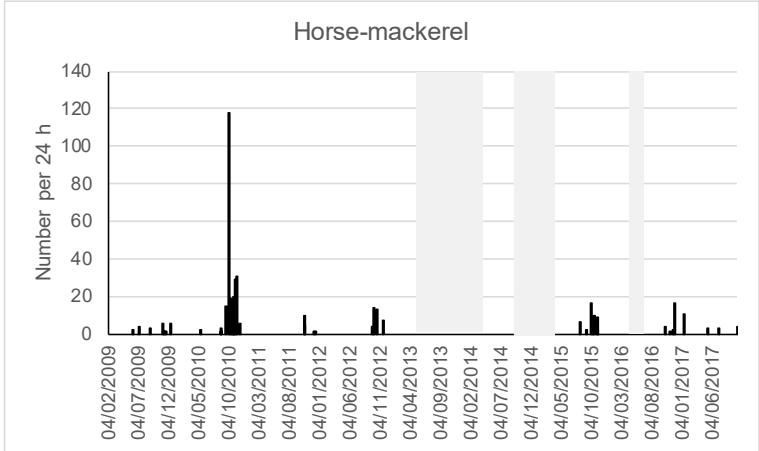

Dab
Number per 24 h
4000
3500
3000
2500
2000
1500
1000
500
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


European eel
Number per 24 h
50
45
40
35
30
25
20
15
10
5
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Flounder
Number per 24 h
300
250
200
150
100
50
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Thin lipped mullet grey mullet
Number per 24 h
3000
2500
2000
1500
1000
500
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Herring
Number per 24 h
40000
35000
30000
25000
20000
15000
10000
5000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Horse-mackerel
Number per 24 h
140
120
100
80
60
40
20
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
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04/09/2013
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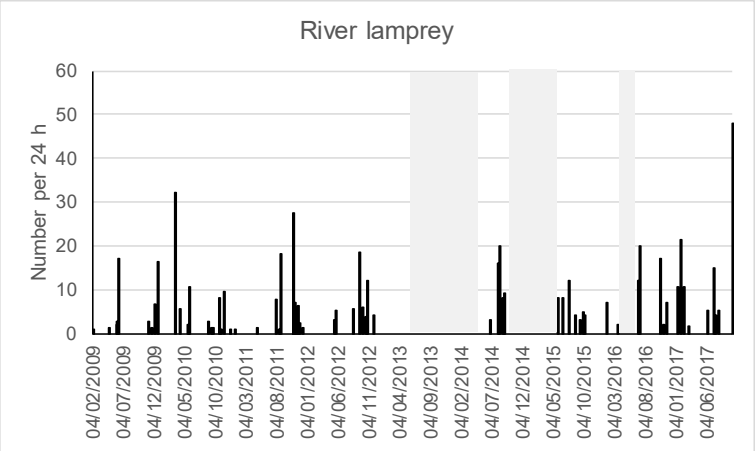
04/08/2016
04/01/2017
04/06/2017


River lamprey
Number per 24 h
0
10
20
30
40
50
60
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
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04/12/2014
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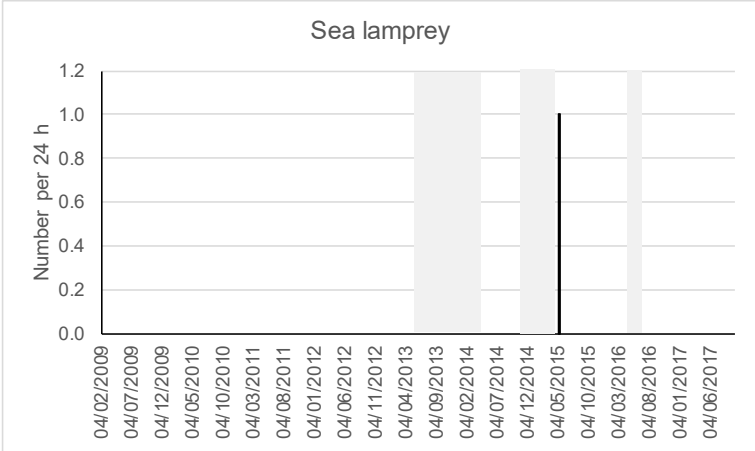
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Sea lamprey
Number per 24 h
0.0
0.2
0.4
0.6
0.8
1.0
1.2
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
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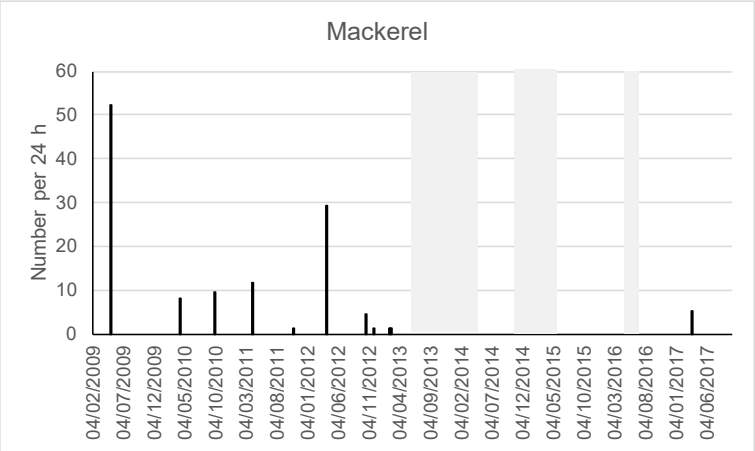
04/06/2017


Mackerel
Number per 24 h
0
10
20
30
40
50
60
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
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04/04/2013
04/09/2013
04/02/2014
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04/10/2015
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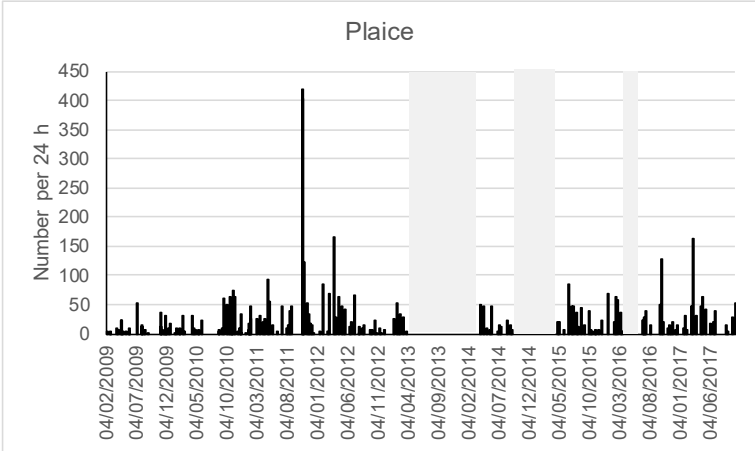
04/08/2016
04/01/2017
04/06/2017


Plaice
Number per 24 h
0
50
100
150
200
250
300
350
400
450
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
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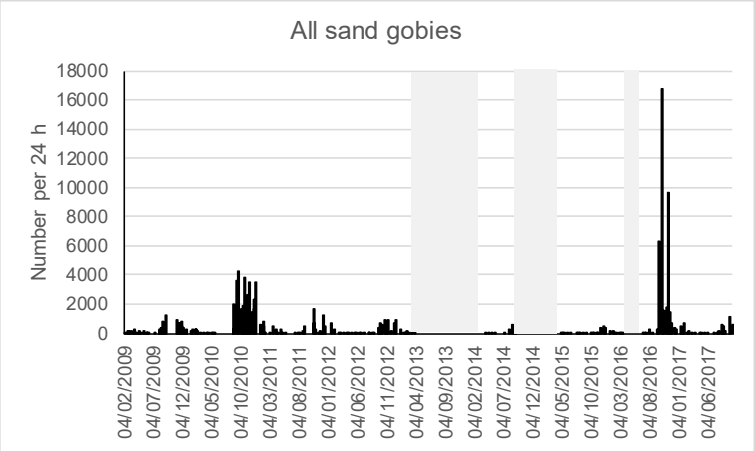
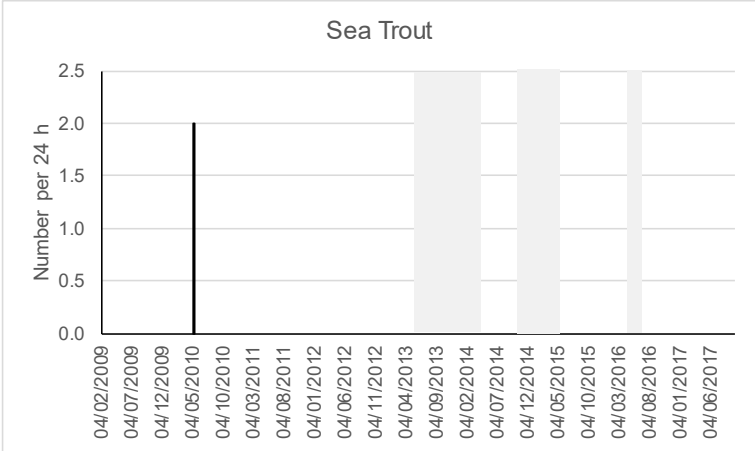
04/06/2017


All sand gobies
Number per 24 h
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Sea Trout
Number per 24 h
2.5
2.0
1.5
1.0
0.5
0.0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
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04/12/2014
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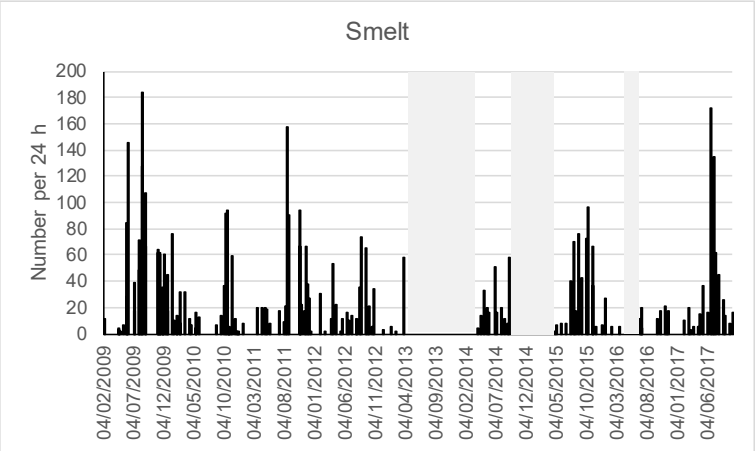
04/06/2017


Smelt
Number per 24 h
200
180
160
140
120
100
80
60
40
20
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
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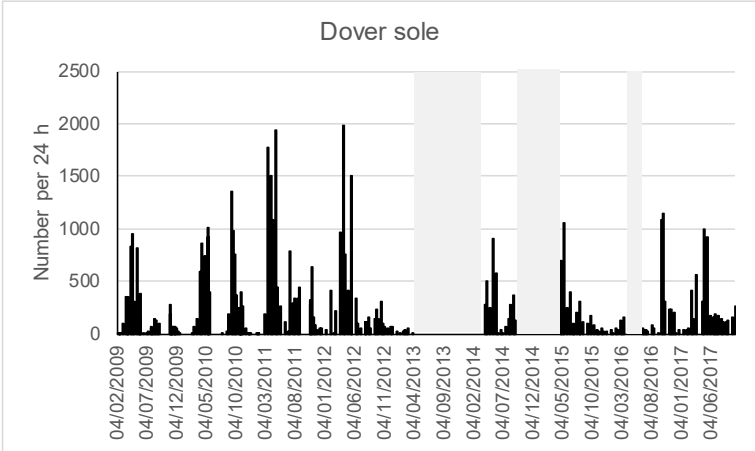
Dover sole
Number per 24 h
2500
2000
1500
1000
500
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
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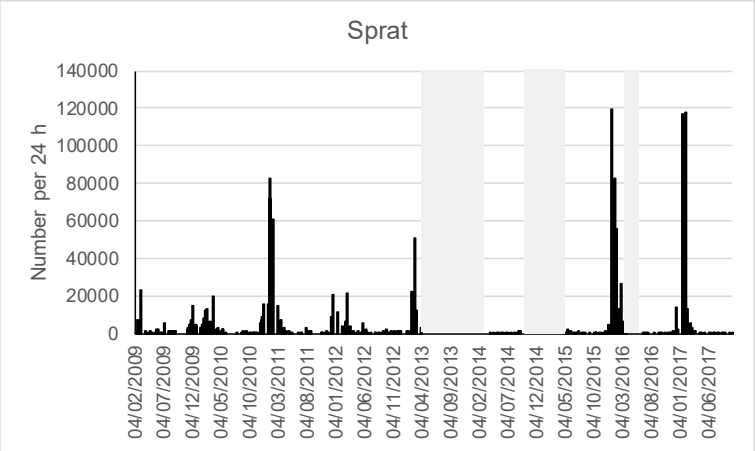
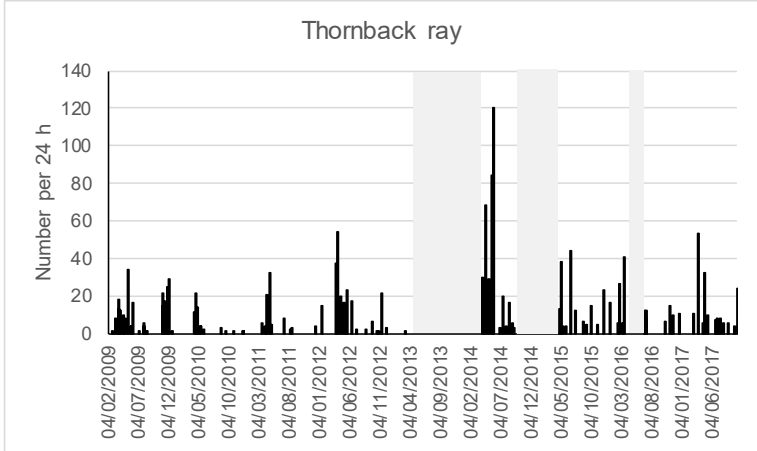
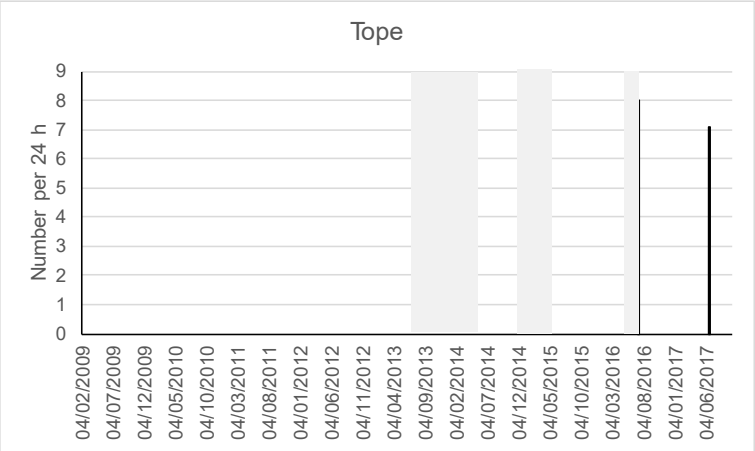
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Sprat
Number per 24 h
140000
120000
100000
80000
60000
40000
20000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Thornback ray
Number per 24 h
140
120
100
80
60
40
20
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017


Tope
Number per 24 h
9
8
7
6
5
4
3
2
1
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
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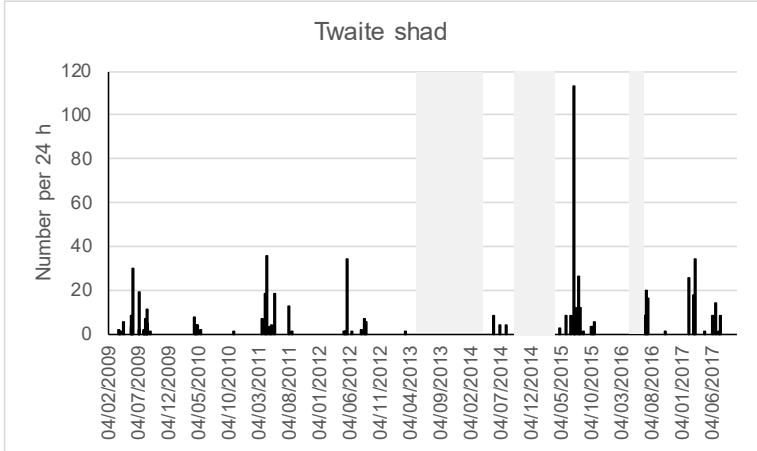
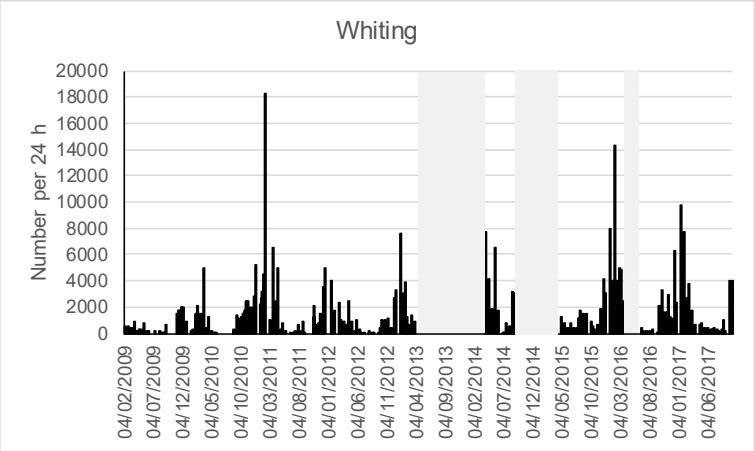
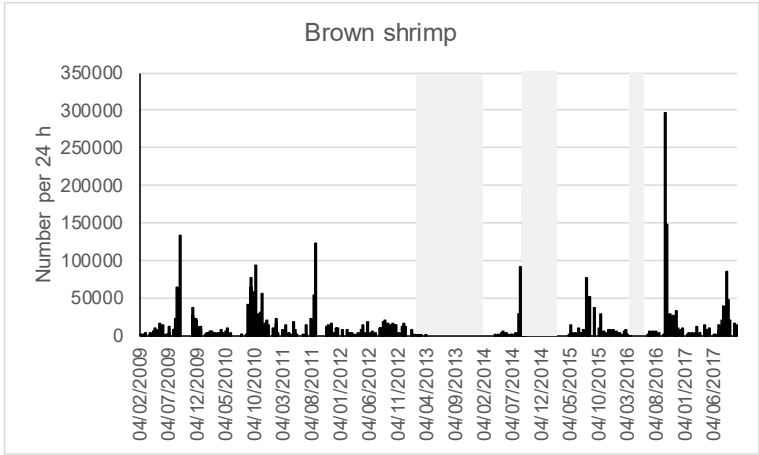
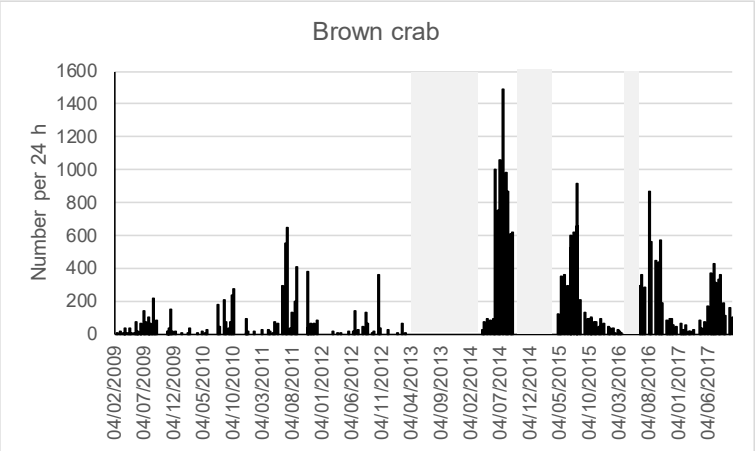
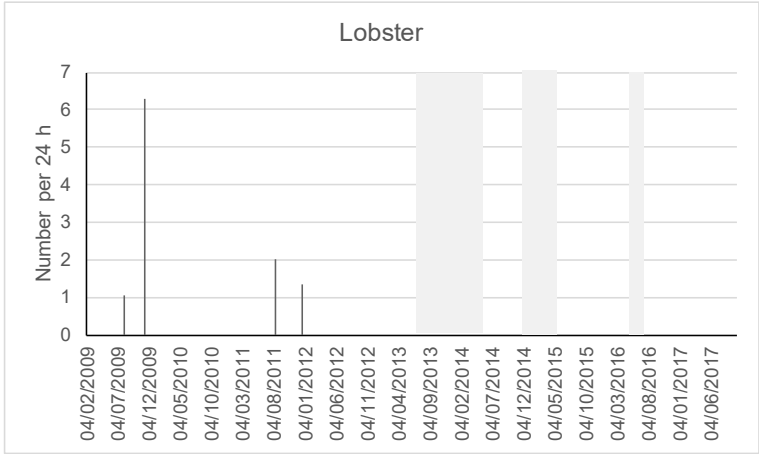
04/06/2017


Twaite shad
Number per 24 h
120
100
80
60
40
20
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

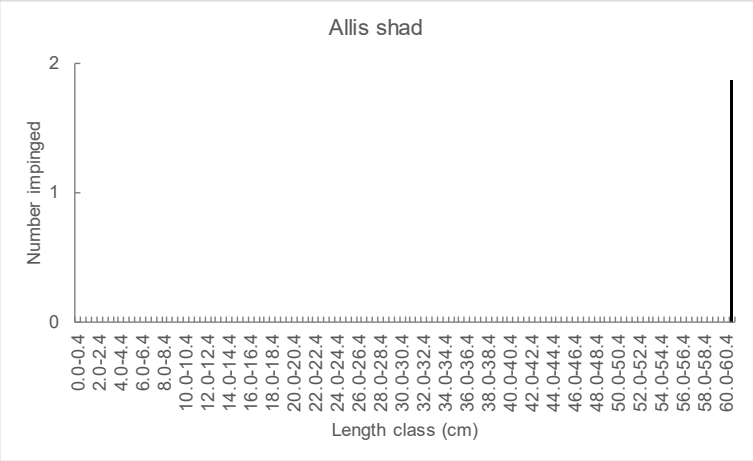

Whiting
Number per 24 h
20000
18000
16000
14000
12000
10000
8000
6000
4000
2000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

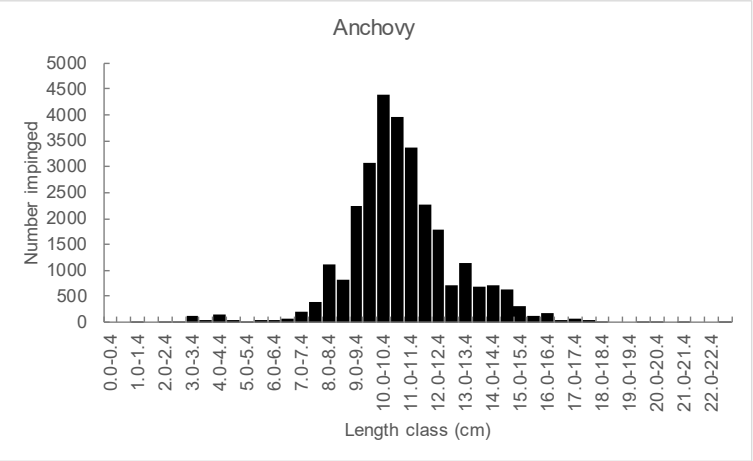

Brown shrimp
Number per 24 h
350000
300000
250000
200000
150000
100000
50000
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

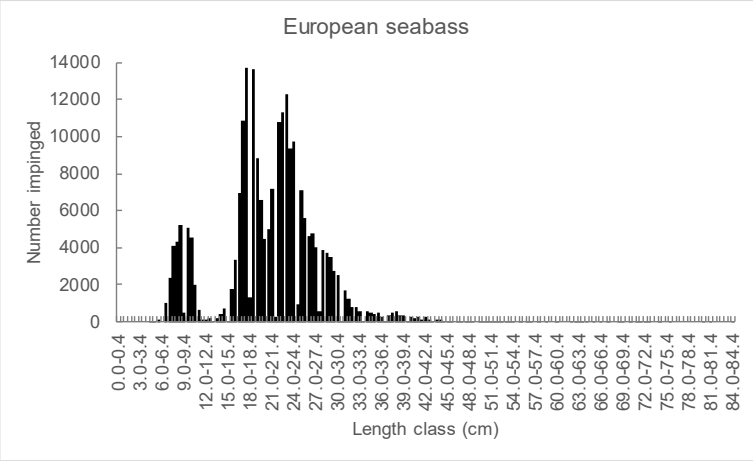

Brown crab
Number per 24 h
1600
1400
1200
1000
800
600
400
200
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

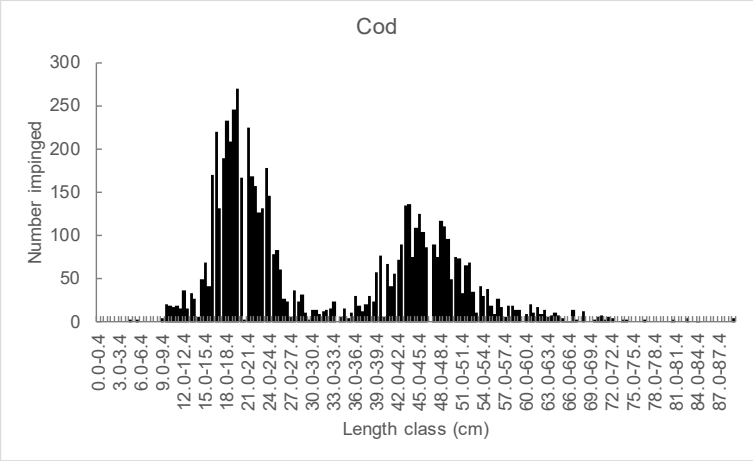

Lobster
Number per 24 h
7
6
5
4
3
2
1
0
04/02/2009
04/07/2009
04/12/2009
04/05/2010
04/10/2010
04/03/2011
04/08/2011
04/01/2012
04/06/2012
04/11/2012
04/04/2013
04/09/2013
04/02/2014
04/07/2014
04/12/2014
04/05/2015
04/10/2015
04/03/2016
04/08/2016
04/01/2017
04/06/2017

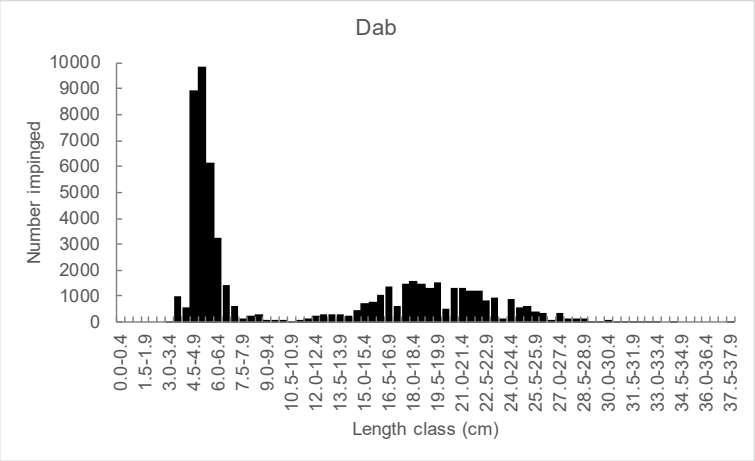
Appendix E Size distributions of key species impinged at SZB

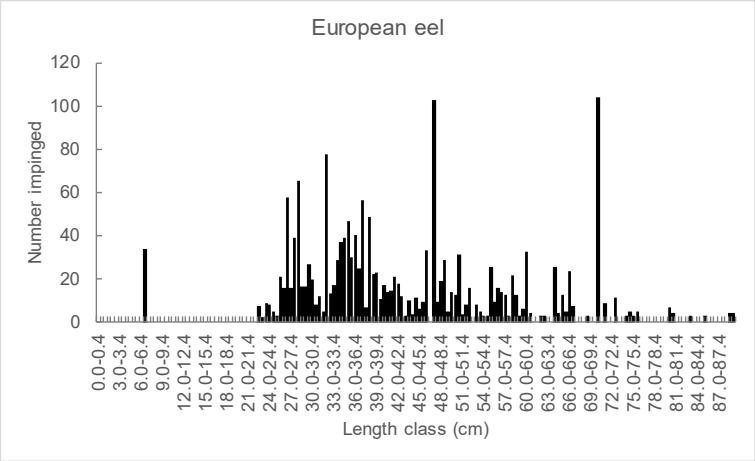


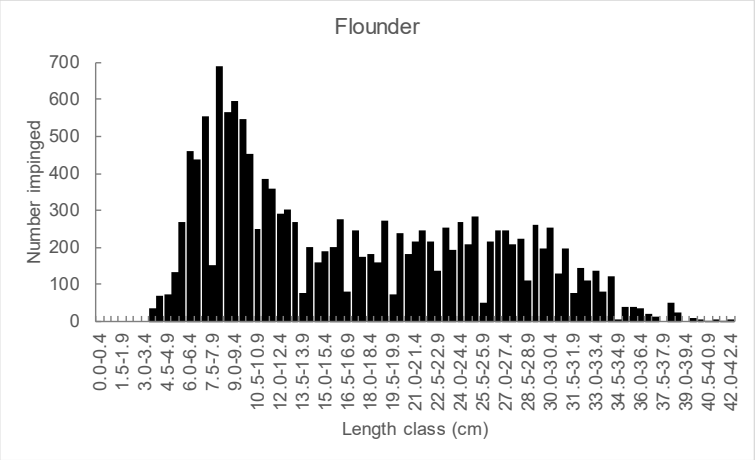


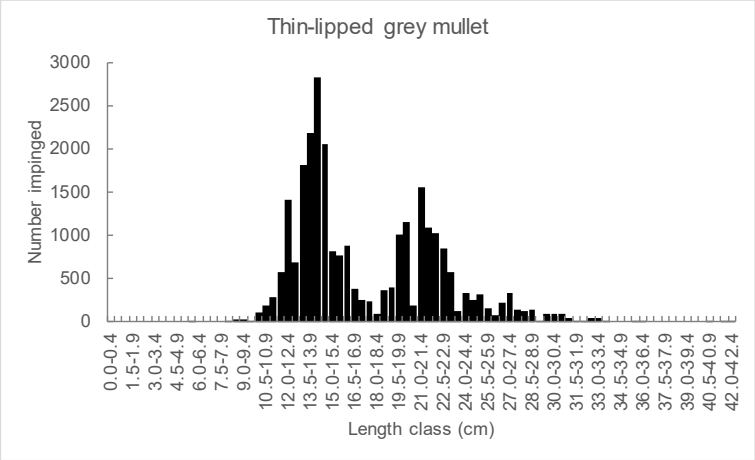


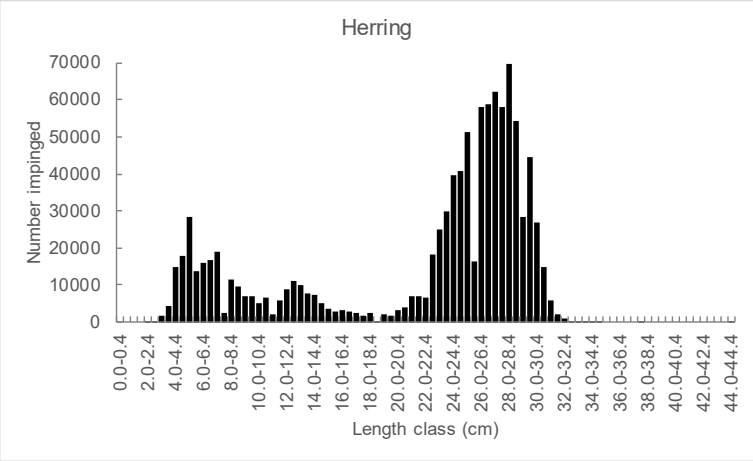


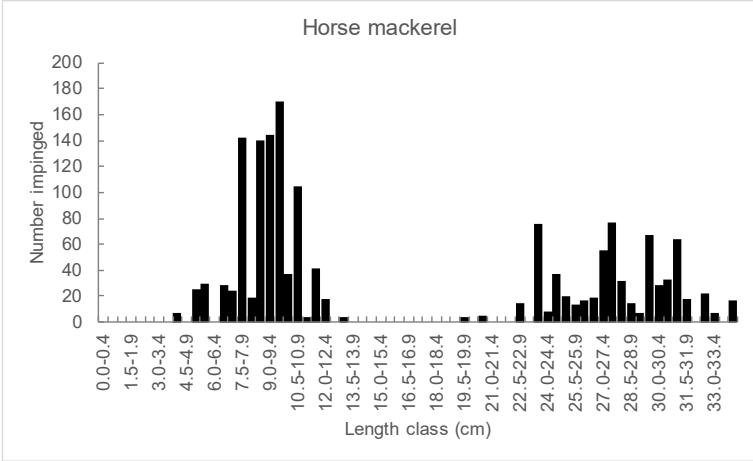

Dab
Number impinged
Length class (cm)

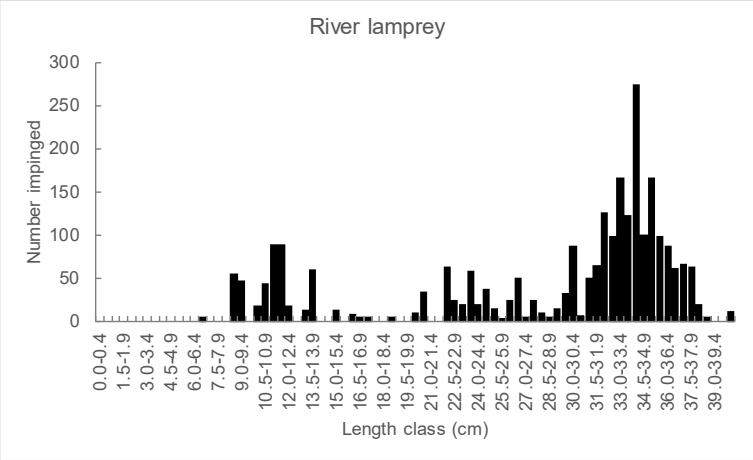

European eel
Number impinged
Length class (cm)

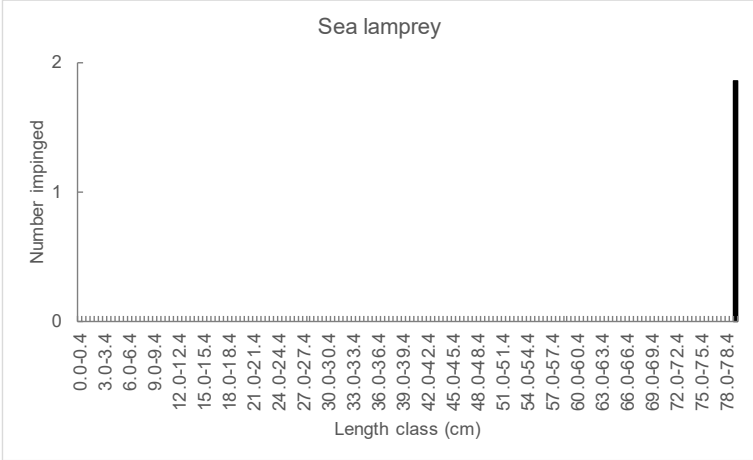

Flounder
Number impinged
Length class (cm)

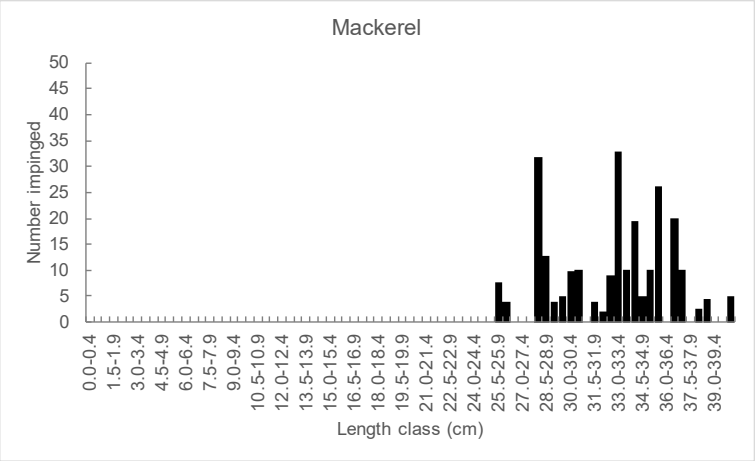

Thin-lipped grey mullet
Number impinged
Length class (cm)

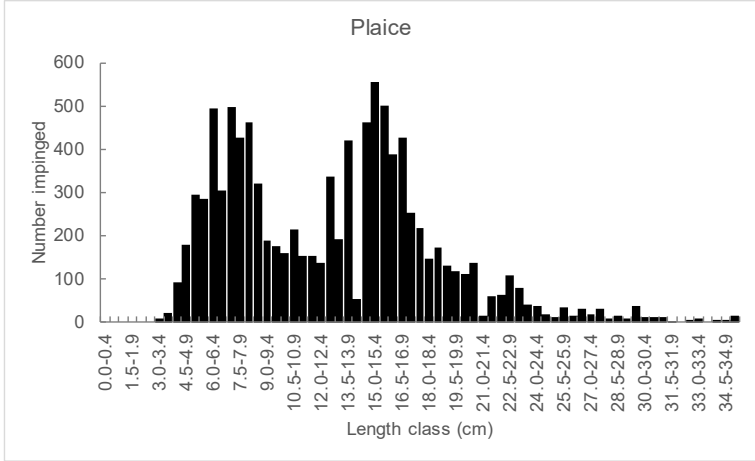

Herring
Number impinged
70000
60000
50000
40000
30000
20000
10000
0
0.0-0.4
2.0-2.4
4.0-4.4
6.0-6.4
8.0-8.4
10.0-10.4
12.0-12.4
14.0-14.4
16.0-16.4
18.0-18.4
20.0-20.4
22.0-22.4
24.0-24.4
26.0-26.4
28.0-28.4
30.0-30.4
32.0-32.4
34.0-34.4
36.0-36.4
38.0-38.4
40.0-40.4
42.0-42.4
44.0-44.4
Length class (cm)

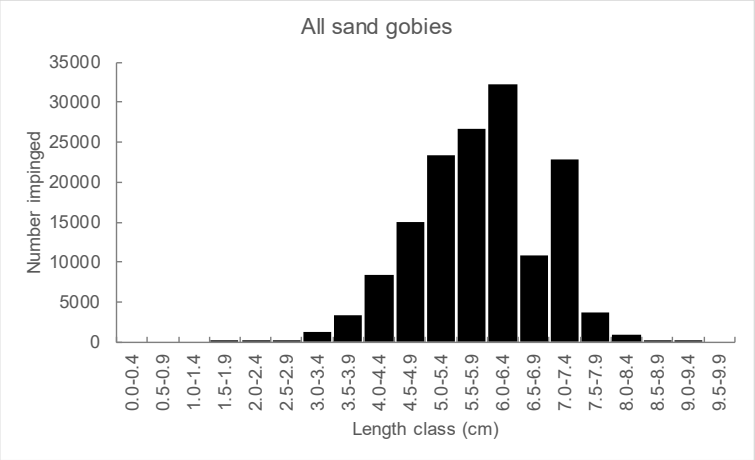

Horse mackerel
Number impinged
200
180
160
140
120
100
80
60
40
20
0
0.0-0.4
1.5-1.9
3.0-3.4
4.5-4.9
6.0-6.4
7.5-7.9
9.0-9.4
10.5-10.9
12.0-12.4
13.5-13.9
15.0-15.4
16.5-16.9
18.0-18.4
19.5-19.9
21.0-21.4
22.5-22.9
24.0-24.4
25.5-25.9
27.0-27.4
28.5-28.9
30.0-30.4
31.5-31.9
33.0-33.4
Length class (cm)

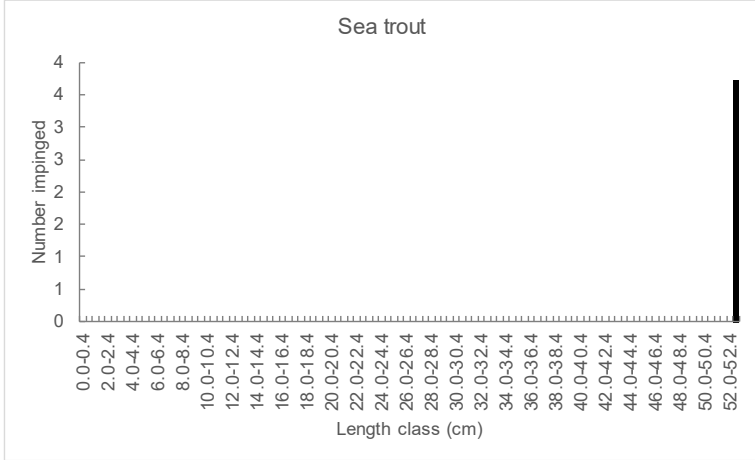

River lamprey
Number impinged
300
250
200
150
100
50
0
0.0-0.4
1.5-1.9
3.0-3.4
4.5-4.9
6.0-6.4
7.5-7.9
9.0-9.4
10.5-10.9
12.0-12.4
13.5-13.9
15.0-15.4
16.5-16.9
18.0-18.4
19.5-19.9
21.0-21.4
22.5-22.9
24.0-24.4
25.5-25.9
27.0-27.4
28.5-28.9
30.0-30.4
31.5-31.9
33.0-33.4
34.5-34.9
36.0-36.4
37.5-37.9
39.0-39.4
Length class (cm)

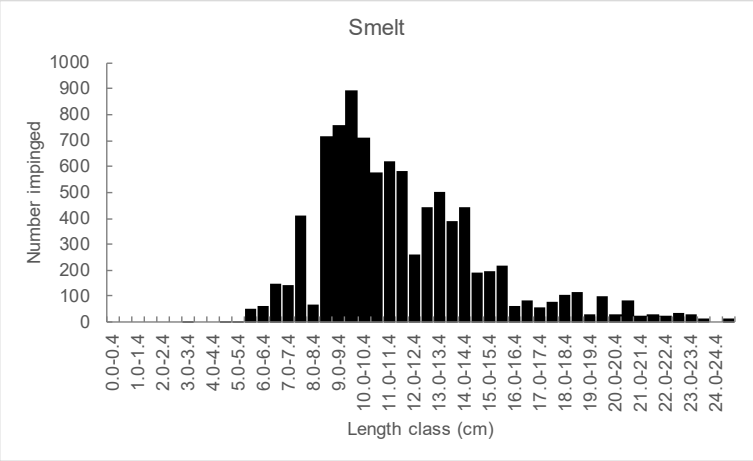

Sea lamprey
Number impinged
2
1
0
0.0-0.4
3.0-3.4
6.0-6.4
9.0-9.4
12.0-12.4
15.0-15.4
18.0-18.4
21.0-21.4
24.0-24.4
27.0-27.4
30.0-30.4
33.0-33.4
36.0-36.4
39.0-39.4
42.0-42.4
45.0-45.4
48.0-48.4
51.0-51.4
54.0-54.4
57.0-57.4
60.0-60.4
63.0-63.4
66.0-66.4
69.0-69.4
72.0-72.4
75.0-75.4
78.0-78.4
Length class (cm)

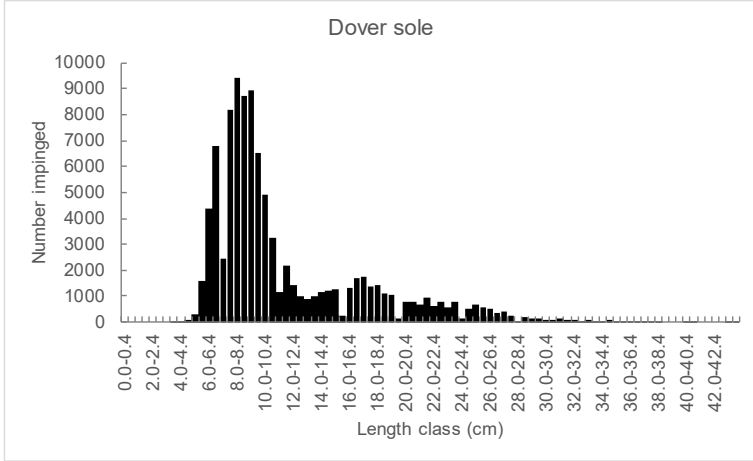

Mackerel
Number impinged
0
5
10
15
20
25
30
35
40
45
50
0.0-0.4
1.5-1.9
3.0-3.4
4.5-4.9
6.0-6.4
7.5-7.9
9.0-9.4
10.5-10.9
12.0-12.4
13.5-13.9
15.0-15.4
16.5-16.9
18.0-18.4
19.5-19.9
21.0-21.4
22.5-22.9
24.0-24.4
25.5-25.9
27.0-27.4
28.5-28.9
30.0-30.4
31.5-31.9
33.0-33.4
34.5-34.9
36.0-36.4
37.5-37.9
39.0-39.4
Length class (cm)

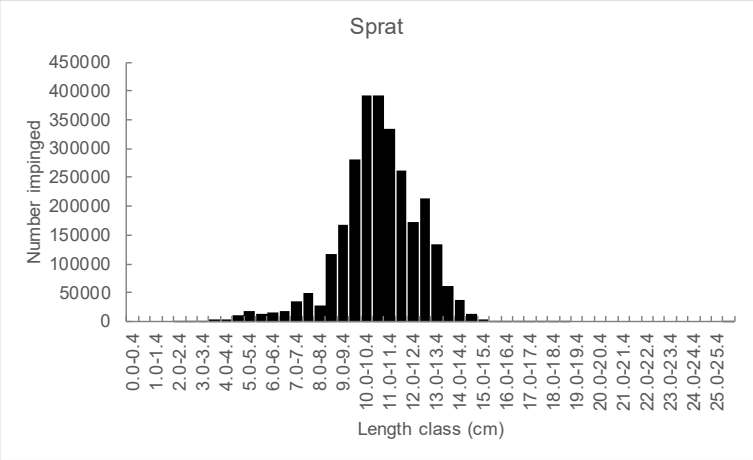

Plaice
Number impinged
0
100
200
300
400
500
600
0.0-0.4
1.5-1.9
3.0-3.4
4.5-4.9
6.0-6.4
7.5-7.9
9.0-9.4
10.5-10.9
12.0-12.4
13.5-13.9
15.0-15.4
16.5-16.9
18.0-18.4
19.5-19.9
21.0-21.4
22.5-22.9
24.0-24.4
25.5-25.9
27.0-27.4
28.5-28.9
30.0-30.4
31.5-31.9
33.0-33.4
34.5-34.9
Length class (cm)

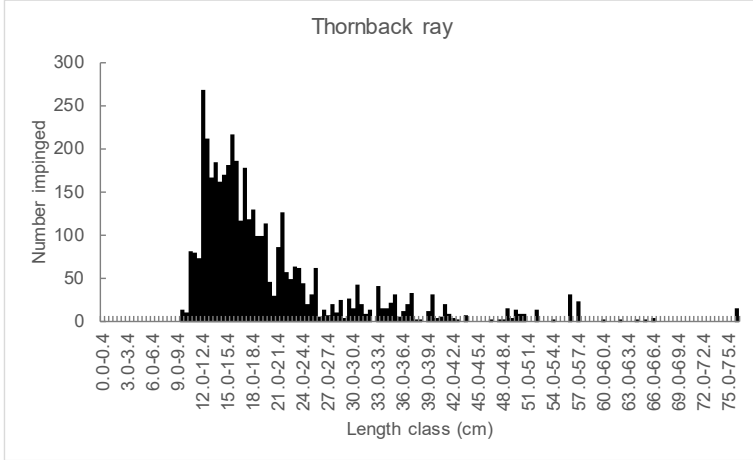

All sand gobies
Number impinged
0
5000
10000
15000
20000
25000
30000
35000
0.0-0.4
0.5-0.9
1.0-1.4
1.5-1.9
2.0-2.4
2.5-2.9
3.0-3.4
3.5-3.9
4.0-4.4
4.5-4.9
5.0-5.4
5.5-5.9
6.0-6.4
6.5-6.9
7.0-7.4
7.5-7.9
8.0-8.4
8.5-8.9
9.0-9.4
9.5-9.9
Length class (cm)

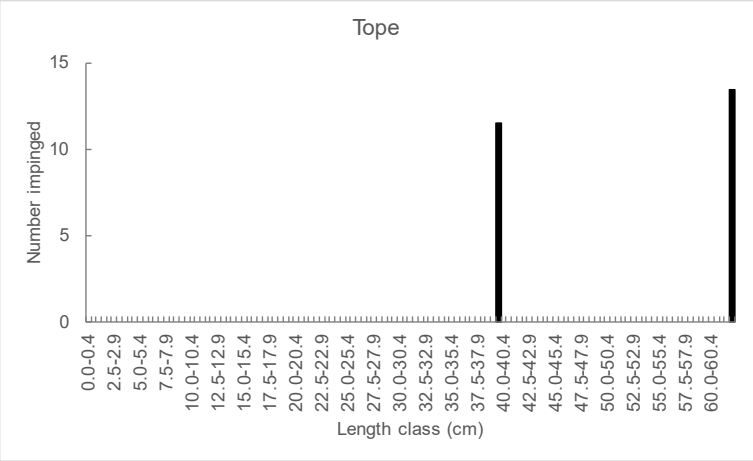

Sea trout
Number impinged
0
1
1
2
2
3
3
4
4
0.0-0.4
2.0-2.4
4.0-4.4
6.0-6.4
8.0-8.4
10.0-10.4
12.0-12.4
14.0-14.4
16.0-16.4
18.0-18.4
20.0-20.4
22.0-22.4
24.0-24.4
26.0-26.4
28.0-28.4
30.0-30.4
32.0-32.4
34.0-34.4
36.0-36.4
38.0-38.4
40.0-40.4
42.0-42.4
44.0-44.4
46.0-46.4
48.0-48.4
50.0-50.4
52.0-52.4
Length class (cm)

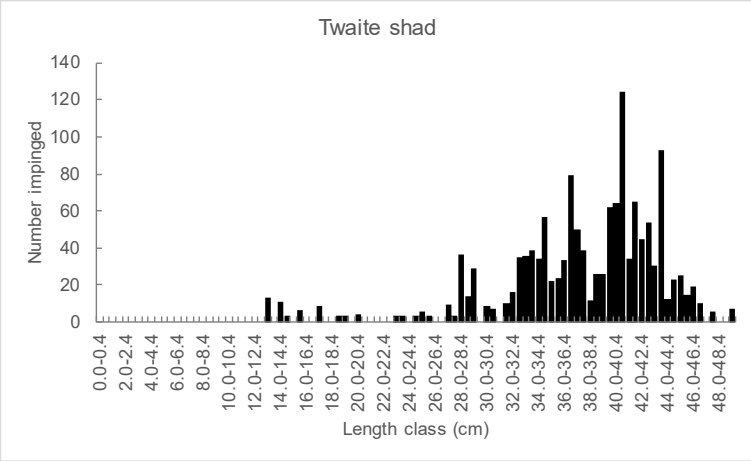

Smelt
Number impinged
1000
900
800
700
600
500
400
300
200
100
0
0.0-0.4
1.0-1.4
2.0-2.4
3.0-3.4
4.0-4.4
5.0-5.4
6.0-6.4
7.0-7.4
8.0-8.4
9.0-9.4
10.0-10.4
11.0-11.4
12.0-12.4
13.0-13.4
14.0-14.4
15.0-15.4
16.0-16.4
17.0-17.4
18.0-18.4
19.0-19.4
20.0-20.4
21.0-21.4
22.0-22.4
23.0-23.4
24.0-24.4
Length class (cm)

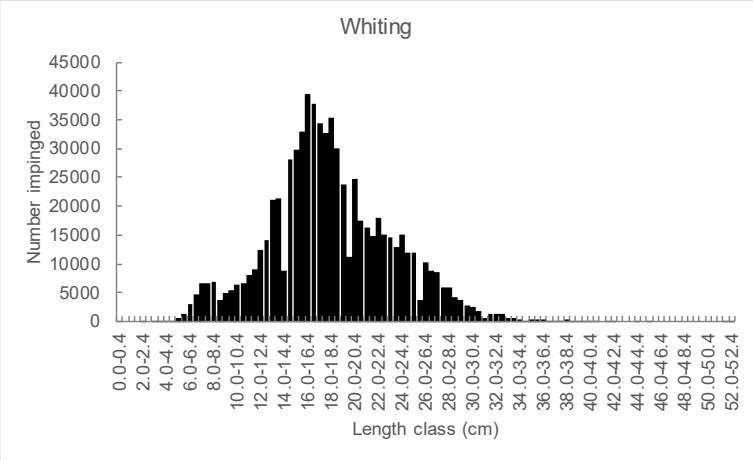

Dover sole
Number impinged
10000
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
0.0-0.4
2.0-2.4
4.0-4.4
6.0-6.4
8.0-8.4
10.0-10.4
12.0-12.4
14.0-14.4
16.0-16.4
18.0-18.4
20.0-20.4
22.0-22.4
24.0-24.4
26.0-26.4
28.0-28.4
30.0-30.4
32.0-32.4
34.0-34.4
36.0-36.4
38.0-38.4
40.0-40.4
42.0-42.4
Length class (cm)

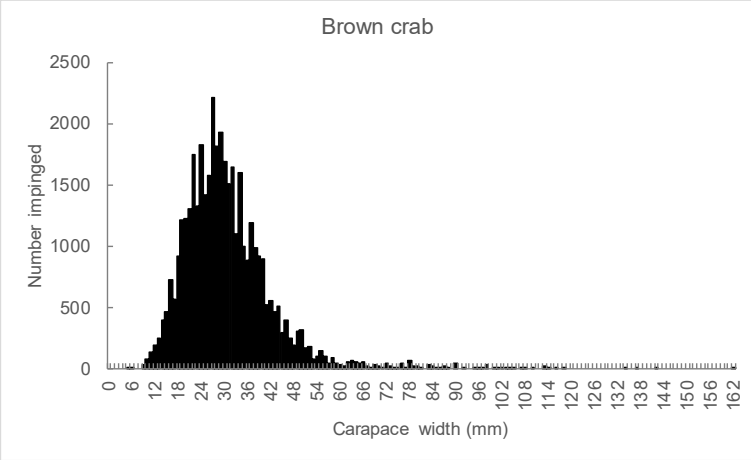

Sprat
Number impinged
450000
400000
350000
300000
250000
200000
150000
100000
50000
0
0.0-0.4
1.0-1.4
2.0-2.4
3.0-3.4
4.0-4.4
5.0-5.4
6.0-6.4
7.0-7.4
8.0-8.4
9.0-9.4
10.0-10.4
11.0-11.4
12.0-12.4
13.0-13.4
14.0-14.4
15.0-15.4
16.0-16.4
17.0-17.4
18.0-18.4
19.0-19.4
20.0-20.4
21.0-21.4
22.0-22.4
23.0-23.4
24.0-24.4
25.0-25.4
Length class (cm)


Thornback ray
Number impinged
300
250
200
150
100
50
0
0.0-0.4
3.0-3.4
6.0-6.4
9.0-9.4
12.0-12.4
15.0-15.4
18.0-18.4
21.0-21.4
24.0-24.4
27.0-27.4
30.0-30.4
33.0-33.4
36.0-36.4
39.0-39.4
42.0-42.4
45.0-45.4
48.0-48.4
51.0-51.4
54.0-54.4
57.0-57.4
60.0-60.4
63.0-63.4
66.0-66.4
69.0-69.4
72.0-72.4
75.0-75.4
Length class (cm)


Tope
Number impinged
Length class (cm)


Twaite shad
Number impinged
Length class (cm)


Whiting
Number impinged
Length class (cm)


Brown crab
Number impinged
Carapace width (mm)

Appendix F Comparison of bulk and hourly samples and the implications for data raising

Restricted access to operational power stations means it is not possible to staff the bulk samples overnight. Consequently, bulk sampling is prone to overflow and subsequent sample invalidation. This section compares impingement rates between bulk and hourly samples to determine the potential to mis-represent impingement by not accounting for the potential influence of diurnal patterns in fish behaviour. Should such bias in diurnal behaviour occur, the potential to over- or underestimation of 24-hour impingement exists when bulk samples are invalid. To investigate diurnal bias, bulk vs hourly catch was examined for four key species in the period 2014 – 2017. The key species included herring *Clupea harrengus*, seabass *Dicentrarchus labrax*, smelt *Osmerus eperlanus* and sand gobies *Pomatoschistus spp*.

Sampling visits with full 6 hourly samples and a valid bulk sample in the period 2014 – 2017 were considered in this investigation resulting in 22 sample dates. The average hourly impingement estimate for each species was calculated for the hourly and bulk samples. A non-parametric paired test (Wilcoxon signed rank test with continuity correction) was used to test differences between the 22 samples in order to see if the counts were significantly different between hourly and bulk samples.

No significant differences were observed between hourly and bulk impingement rates throughout the year for the species tested (Table 8). Based on these results there does not appear to be any consistent sampling bias between the hourly and bulk samples, leading to significant misrepresentation of impingement rates.

Table 8 Impingement rates expressed as fish per hour for four key species during all months compared between bulk and hourly impingement monitoring visits from 2014-2017.

Species		*n*	Mean	stdev	Min	Max	*p*
Seabass	Hourly	22	5.89	18.10	0	77	0.636
	Bulk	22	4.93	12.79	0	54	
Smelt	Hourly	22	1.63	2.69	0	13	0.281
	Bulk	22	1.60	1.72	0	6	
Herring	Hourly	22	274.5	909.20	0	4312	0.404
	Bulk	22	116.2	128.09	0	345	
Sand goby	Hourly	22	10.86	19.09	0	76	0.135
	Bulk	22	12.52	29.37	0	98	

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. TR406

NOT PROTECTIVELY MARKED

TR406: Impingement predictions based upon specific cooling water system design

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Please note that the red line boundary was amended after this document was finalised, therefore figures in this document do not reflect the boundaries in respect of which development consent has been sought in this application. However, amendments to the red line boundary does not have any impact on the findings set out in this document and all other information remains correct.

Executive summary

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture science (Cefas) has been contracted by NNB GenCo (SZC) Ltd to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

This report is structured to address the primary issues relating to abstraction of fish and shellfish by the proposed Sizewell C station. The sections are summarized below:

1. **Background** to the assessment including selection criteria for key taxa and conservation species.

2. **Selection of impingement mitigation technology for Sizewell C:** This section briefly reviews the viability of biota mitigation options, biofouling and blockage risks and mitigation optioneering.

3. **Relevant site features** of the Greater Sizewell Bay in relation to the proposed development and existing Sizewell B station are summarised.

4. **Impingement Assessment Methodology:** Provides a detailed step by step process of the impingement assessment process, which at the most fundamental level is based on scaling impingement estimates collected from nine years of impingement monitoring at Sizewell B.

5. **Assessing the significance of impingement effects:** There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by impingement in coastal power stations. This section provides the justification for appropriate screening thresholds selected such that impingement losses lower than the threshold would have negligible effects on the year to year sustainability of fish stocks.

6. **Finfish Impingement predictions for Sizewell C:** This section provides the results for the primary assessment of the effects of the station on fish stocks relative to spawning stock biomass (SSB) and fisheries landings (detailed in Section 5). To demonstrate the predicted efficiency of each mitigation measure and to provide full transparency, predictions apply each mitigation sequentially from unmitigated to fully mitigated.

   Appendices: Detailed appendices are provided to support impingement predictions allowing each step of the process to be evaluated. Appendices provide mean, upper and lower 95th percentile impingement estimates.

7. **Evaluation of uncertainty in impingement predictions:** The confidence of impingement predictions is determined by the underlying assumptions used to parameterise the assessment and the appropriateness of the comparator. This section determines the sensitivity of the assessment to uncertainties in the key parameters.

8. **Consideration of local-level effects on the fish assemblage at Sizewell:** This section provides a new additional assessment independent to the effects on the sustainability of fish stocks to determine the potential for local-level effects. Effects are considered in relation to migratory routes, spawning and nursery areas, and the availability of prey for designated species. The section also draws on experience from other operational sites, such as Hinkley Point to determine if evidence for local depletion of fish is apparent.

9. **Effect of SZC entrapment on the Water Framework Directive (WFD) status of local water bodies:** This section considers the potential for impingement to influence the status of transitional water bodies, in proximity to the proposed development, where the Environment Agency completes Transitional Fish Classification Index (TFCI) monitoring.

10. **Shellfish Impingement predictions for Sizewell C:** This section provides the results for assessment of the effects of the station on shellfish stocks. To demonstrate the predicted efficiency of each mitigation measure and to provide full transparency, predictions apply each mitigation sequentially from unmitigated to fully mitigated.

11. **Considerations of climate change effects:** This section considers how impingement rates may change in the context of future climate change.

12. **Conclusion:** The final conclusions of the assessment are provided.

51.5 cumecs for the existing Sizewell B (SZB). SZB is the most recent nuclear power station to be constructed in the UK (commissioned in 1995) and is fitted with two measures to reduce the losses of impinged fish and crustacea; specially designed capped intake head and an early example of a Fish Recovery and Return (FRR) system. The cooling water intakes for SZC would be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, but a significant number of small organisms (small fish and crustaceans, and plankton) will inevitably enter the cooling water intakes. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh (10 mm at SZB, also proposed for SZC) drum screens that protect the main cooling water supply to the station condensers, and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages.

A separate report considers the significance of entrainment impacts on marine organisms (BEEMS Technical Report TR318). The results of which are summarised in this report to produce a combined entrapment (entrainment + impingement) assessment for SZC (Section 6.7).

Between 2009 and 2017, a total of 205 impingement monitoring visits were completed at Sizewell B. Impingement visits consisted of six hourly samples and an 18-hour bulk overnight sample. During this period 91 finfish taxa were recorded. Of these 24 species have been selected as being representative of the fish assemblage based on selection criteria pertaining to commercially, ecological and conservation importance of the species.

Impingement monitoring has enumerated 62 invertebrate taxa, of which four shellfish species were selected for impingement assessments on the basis of commercial and ecological importance.

Impinged organisms will be returned to the sea via a Fish Recovery and Return system (FRR). Not all survive the process and separate assessments have been made to:

1. evaluate the effect of returned dead fish on local ecology and water quality (BEEMS Technical Report TR520).
2. determine whether any beached fish would constitute a nuisance on local beaches (BEEMS Technical Report TR511).

Selection of impingement mitigation technology for Sizewell C

The Environment Agency have issued guidelines for the types of measures that could be adopted at new direct cooled power stations to reduce the predicted environmental impacts of impingement and indirectly the potential for pollution by discharges of dead fish back into the marine environment (Environment Agency

2005, 2010). As explained in the Environment Agency guidelines, in practice the selection of potential impingement mitigation measures involves a complex consideration of the likely effectiveness of each measure in the marine environment at the station location, engineering feasibility and operational safety for staff and the plant.

The range of options is much larger in freshwater and some brackish environments that do not present a high biofouling risk to station plant and for low volume abstractions (e.g. of a few cumecs). Many of these options are infeasible for a coastal direct cooled power station (Environment Agency 2005). SZC's intakes would abstract 132 cumecs and would be mounted on the seabed in a highly turbid, coastal environment with high wave exposure offshore of the Sizewell-Dunwich Bank. The site is at high risk of biofouling.

A detailed consideration of the effectiveness and feasibility of the available impingement mitigation options has been conducted for Sizewell C and is summarised in Section 2. Mitigation options include both biota exclusion technologies, aimed to reduce the initial abstraction of fish and crustaceans, and biota recovery systems, aimed at returning impinged biota to the environment. Two measures were both assessed as feasible and likely to reliably deliver reductions in the predicted losses of fish and crustacea at SZC:

i. Low velocity side entry (LVSE) intake heads
ii. Fish Recovery and Return (FRR) systems with proposed 10mm mesh filtration and an anti-biofouling control policy that results in chlorination not being applied to the FRR system.

Both of these technologies are proposed for the SZC station and their use is predicted to reduce impingement mortality at SZC by the factors shown in Table 1 compared with an unmitigated SZC.

Table 1 Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared with an unmitigated SZC.

Group	Example species	Reduction in impingement mortality at SZC
Pelagic fish	sprat, herring, anchovy, shads	64%
Demersal fish	bass, cod, whiting, grey mullet	78-80%
Epibenthic fish	eel, lampreys, sole, sand goby	93%

Assessment of the significance of SZC impingement effects

There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by impingement in coastal power stations and therefore any assessment must be based on expert judgment.

For the purposes of this assessment we have adopted two screening thresholds that have been selected such that impingement losses lower than the appropriate threshold will have negligible effects on the year to year sustainability of a fish population. Effects above the appropriate threshold would not necessarily indicate a significant adverse effect but require further investigation to determine whether significant effects were, in fact, present.

The thresholds have been selected based upon internationally accepted scientific practice for the sustainability of fish stocks under anthropogenic pressures:

a. For commercially exploited stocks and conservation species (which includes stocks that are not currently exploited): 1% of the Spawning Stock Biomass (SSB) or, as a highly conservative proxy, 1% of international landings of the stock.
b. For unexploited stocks: 10% of the SSB or, as a highly conservative proxy, 10% of international landings of the stock.

The scientific rationale for the selection of these screening thresholds is detailed in Section 5.

For eel, twaite shad, allis shad, cucumber smelt and river lamprey a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. In particular, a highly precautionary approach was adopted for European eel whereby the Anglian River Basin District (RBD) escapement biomass was used as the stock reference due to the uncertainties surrounding both the current eel stock status and its stock dynamics (Sections 5.1.6 and 6.6.1). This is equivalent to adopting a highly precautionary threshold of approximately 0.005% SSB for the eel stock. In practice the predicted abstraction loss from SZC is much less than 1% of the RBD escapement biomass.

Derivation of Spawning Stock Biomass estimates

Fish stocks in the Northeast Atlantic are managed partly through the EU Common Fisheries Policy (CFP), whose objective is to maintain or rebuild fish stocks to levels that can produce their maximum sustainable yield (MSY). The International Council for the Exploration of the Sea (ICES) advises public authorities with competence for marine management including the European Commission (EC).

ICES' advice is produced through a process which is set up to ensure that the advice is based on the best available science and data, is considered legitimate by both authorities and stakeholders and is relevant and operational in relation to the policy in question.

The basis for the advice is the compilation of relevant data and analysis by experts in the field, normally through an expert group which includes core researchers in the field. This analysis is peer reviewed by scientists who have not been involved in the expert group and have no direct interest in the matter.

To support the stock by stock management system, ICES provides advice on fishing opportunities and stock status for individual stocks including estimates of Spawning Stock Biomass (SSB). To undertake their stock assessments ICES' scientists have identified biological stock areas that describe the distribution of a stock. These may be different from the areas defined by the EU, for example, for the management of fishing quotas and technical measures. Identification of appropriate stock boundaries has been a central theme of ICES' coordinated effort since its formation in 1902 and major advances in understanding have, and continue to be, made.

Are ICES stock units appropriate for assessing the effects of SZC on fish populations?

The appropriateness of using some of the existing ICES stock units, particularly for bass which has one of the largest stock units of the key fish species included in the SZC effects assessment needs to be considered. In particular, whether the stock areas being used for the assessment of impacts to certain species consider the impact to local sub-populations given numerous papers (including papers produced by ICES) provide evidence of sub-populations and more complex heterogeneous population structures.

Section 4.10.1 describes how ICES determines stock identity for fisheries management purposes, in particular how it uses evidence from ongoing research. The status of the bass stock unit and the direction of ongoing research is addressed and found not to alter the decision that ICES' current stock definition is scientifically the most appropriate. The section concludes that ICES' stock boundaries are compromises but they are based on a mature weighing of the best scientific evidence available and they are relied upon by governments to manage fish populations in the waters of all EU member states. Given the negligible predicted SZC impacts compared to those of fishing, and the precautionary nature of ICES' estimates of SSBs, no justification is found not to use the ICES' stock definitions to assess SZC effects on fish.

There are no isolated fish populations at Sizewell, fish live and move in an unconstrained coastal environment with most species undertaking wide spatial migrations throughout the year. As such, ICES stock units represent the best available evidence for assessing the impacts of the proposed development in relation to stock sustainability. However, it is recognised that assessments of effects on fish must consider aspects beyond stock sustainability and further information on the potential for localised effects of the station is provided in Section 8.

Impingement predictions for Sizewell C

Predictions of impingement have been provided for SZC with mitigation, with mitigation that separately includes LVSE intake heads and FRR systems fitted and for the station fitted with both of the two mitigation technologies (Section 2).

The mean and 95th percentile upper impingement estimate for the key species was predicted to be below the appropriate 1% threshold for SSB or fisheries landings, for all species except bass and European eel. For example, sprat, herring, and whiting are numerically the highest impinged species; mean losses for these species equate to 0.02%, 0.02% and 0.11% of SSB, respectively. Mitigation measures would reduce effects further by the factors described in Table 1.

Of the 24 key species at Sizewell, impingement from Sizewell C is predicted to result is the loss of 3.93 million fish per annum with an equivalent adult weight of 191.5t (Table 2). Sprat, herring and whiting are the three species with the highest rates of impingement contributing 93% of the weight of the impinged biomass.

In the case of bass, a combination of the LVSE intakes, FRR mitigation and applying knowledge of the distribution of bass within the GSB is anticipated to reduce effects to well below 1%. Depending on the efficiency of the mitigation measures effects of between 0.03 up to a maximum of 0.3% of SSB may be predicted (Section 7).

In the absence of mitigation, losses of eel due to impingement equates to 1 t per annum or 1.27% of the Anglian RDB SSB. Eel is a robust species, and at least 80% survival is anticipated through well designed FRR systems. The efficiency of LVSE intakes for poor swimming species such as European eel has been questioned but the benefit from LVSE intakes are considered to be mostly due to the reduction in cross sectional area due being mounted orthogonal to the tidal current and their short range zone of influence on local hydrodynamics. Assuming no LVSE benefit at all, effects from the FRR alone equates to 0.13% of SSB if the more appropriate yellow eel EAV is applied (Section 7).

The individual entrainment and impingement impacts are such that when combined into a single entrapment estimate, there is very little difference to the overall conclusions that are reached when each is viewed separately. With the proposed impingement mitigation fitted entrapment estimates (Table 18) show the only species that remains above the 1 % threshold is sand goby (entrainment = 1.40 % of abundance; impingement = 0.02 %, i.e. entrapment = 1.42 %). Sand gobies are a short-lived very abundant species that is ubiquitous in European coastal areas to at least a depth of 20 m. The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over the southern North Sea. Given that the species is not commercially-exploited, the appropriate negligible effects threshold is 10% of SSB (as discussed in Section 5.1). However, because of their short lifespan and early age of maturity, sand gobies have a sustainable harvesting rate of greater than 50% SSB (Section 5.1.1). Therefore, losses of 1.42 % of total abundance by SZC are regarded as negligible.

Conclusions on the predicted effects of SZC entrapment

a. Of the 24 key fish and 4 key shellfish species, no species exceeded the 1% impingement screening threshold for negligible effects with the proposed LVSE intake heads and FRR systems fitted when compared against stock estimates or, in the absence of these, international landings.

b. For European eel, twaite shad, allis shad, cucumber smelt and river lamprey assessments a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. For eel this was equivalent to adopting a highly precautionary screening threshold of approximately 0.005% SSB.

c. The predicted impingement effects of SZC on all of the key taxa were below the effect threshold (Table 2).

d. The predicted effects of SZC entrapment (i.e. impingement plus entrainment) with the proposed embedded impingement mitigation systems fitted were also negligible (Table 18).

Consideration of local-level effects on the fish assemblage at Sizewell

Section 8 considered the potential for localised depletion to have consequences on prey availability for designated Annex II species of relevance to the HRA and species of importance to the WFD Transitional Fish Classification Index (TFCI). Local scale depletion due to Sizewell C and Sizewell B acting in-combination is predicted to be less than 2.5% for demersal species, 1.5% for pelagic species and less than 1% of epibenthic species during periods of seasonal presence at the scale of the Greater Sizewell Bay and tidal excursion. Outside of this area depletion effects are predicted to be negligible.

An assessment of potential localised effects of SZC entrapment was undertaken and found no likely significant adverse effects, concluding that:

i. The proposed development would not have significant effects on local spawning or nursery areas in the vicinity of Sizewell.
ii. In the open coastal waters, Sizewell C represents a low risk to migratory species.
iii. Local depletion due to impingement is orders of magnitude below natural variability in abundance to which predator-prey relationships are adapted to. Therefore, impingement from Sizewell B and Sizewell C in-combination would not have adverse food-web effects on designated features of HRA sites.
iv. Local effects are not predicted to cause a deterioration in the WFD fish status of the Alde & Ore water body as local depletion is well below the sensitivity of the TFCI.

Impingement monitoring from Hinkley Point B showed no discernible effect after Hinkley Point A ceased abstraction. The unmitigated Hinkley Point A was comparable or lower than the relative effects predicted Sizewell C indicating the system is unlikely to be sensitive to such changes.

Table 2 Annual mean SZC impingement predictions considering the effect of the LVSE intake heads and FRR systems fitted and the corrections to thin-lipped grey mullet, bass, and eel assessments, detailed in Section 6.5. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Numbers in red font are either estimates of the population numbers (e.g. sand goby, twaite shad, allis shad) or reported catch numbers (salmon & sea trout).

Species	Mean SZC prediction	SZC prediction after intake head adjustment	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	2,470,731	2,470,731	1,856,146	19.50	220,757	0.01	151,322	0.01
Herring	2,607,016	930,705	930,705	665,658	125.59	2,198,449	0.01	400,244	0.03
Whiting	1,665,479	594,576	327,290	116,541	33.31	151,881	0.02	17,570	0.19
Bass (incl. distribution)[1]	70,907	25,314	13,937	3,121	4.78	14,897	0.03	3,051	1.16
Bass (even distribution)[1]	*709,082*	*253,142*	*139,372*	*31,214*	*47.78*	*14,897*	*0.32*	*3,051*	*1.57*
Sand goby	503,232	179,654	37,009	37,009	0.07	205,882,353	0.02	NA	NA
Sole	233,942	83,517	17,205	3,663	0.78	43,770	0.00	12,800	0.01
Anchovy	176,816	63,123	63,123	61,489	1.28	NA	NA	1,625	0.08
Dab	143,519	51,236	27,591	12,267	0.50	NA	NA	5,309	0.01
Thin-lipped grey mullet	127,382	45,475	25,038	2,087	1.09	600	0.18	120	0.90
Flounder	37,303	13,317	3,075	1,420	0.12	NA	NA	2,321	0.01
Plaice	24,716	8,824	1,818	628	0.15	690,912	0.00	80,367	0.00
Smelt (UK stock)[1]	20,795	7,424	7,424	5,653	0.09	50	0.19	8	1.11
Smelt (R. Elbe population)[2]	*20,795*	*7,424*	*7,424*	*5,653*	*0.09*	*23,861,520*	*0.02*	*8*	*1.11*
Cod	16,426	5,864	3,531	1,268	3.30	103,025	0.00	34,701	0.01
Thornback ray	7,460	2,663	549	106	0.34	NA	NA	695	0.05
European Eel [3]	3,031	1,082	223	112	0.037	79	0.047	14	0.53
Twaite shad	2,989	1,067	1,067	1,067	0.33	5,161,183	0.02	1	25.07
River lamprey	2,929	1,046	215	215	0.02	62	0.03	1	1.53
Horse mackerel	1,779	635	635	635	0.09	NA	NA	20,798	0.00
Mackerel	473	169	169	169	0.05	3,888,854	0.00	1,026,828	0.00
Tope	70	25	5	5	0.04	NA	NA	498	0.01
Sea trout	10	4	4	4	0.01	NA	NA	39,795	0.01
Sea lamprey	5	2	0.36	0.36	0.00	NA	NA	NA	NA
Allis shad	3	1	1	1	0.00	27,397	0.00	0	0.47
Salmon	0	0	0	0	0.00	NA	NA	38,456	0.00

[1] The density of bass is 20 times greater inshore of the Sizewell-Dunwich Bank in the vicinity of the SZB thermal plume than offshore of the Bank where the SZC intakes would be. Assuming 90% of bass would remain inshore of the Bank (rather than the measured 95%) the expected bass impingement for SZC falls 10-fold. Both the assessment including distribution and assuming even distribution is provided (Section 6.5.1).

[2] Considering only UK smelt populations is expected to substantially underestimate of the stock size. This is particularly the case considering the likelihood of smelt off Sizewell being part of a wider southern North Sea population embracing Belgian and German sub-populations. Comparisons have therefore also been made against estimates of population size for available data for the River Elbe (Section 6.6.1).

[3] Based on the recalculated EAV considering eels impinged at SZB are all yellow eels (Section 6.5.3).

Version History

A detailed version History have been provided in Appendix F.

V1 dated 03/06/2019

Version 1 of this report was released (03/06/2019). The report superseded the preliminary impingement assessment reports produced to allow consultation and iterative development of the methodology.

This report is the primary report for impingement assessments.

V7 report dated 06/11/2020

Version 6 of this report of 27/02/2020 was part of the DCO submission. Version 7 of this report has been modified following stakeholder consultation, Marine Technical Forum (MTF) meetings and initial responses to Relevant Representations provided on the 30th September 2020.

The Environment Agency produced a differences document 'EA Sizewell C - Marine Ecology Gap Analysis' based on version 6 of the report in February 2020 which preceded a Level 4 Marine Ecology meeting on 19th March 2020. On 23rd September 2020 a MTF was hosted including the Marine Management Organisation, Environment Agency, Natural England and Eastern IFCA in relation to key issues pertaining to fish impingement assessments provided in the DCO.

Version 7 of this report captures the main discussion and action points from the meetings and written comments from statutory stakeholders. Furthermore is provides additional evidence based on a series of supplementary reports to support the evidence base for impingement assessments.

The most totable changes to this report include:

1. **An updated LVSE factor.** Following Environment Agency consultation on BEEMS Scientific Position Paper SPP099, modelling was refined to capture the full range of tidal directions. Modelling indicates that the SZC station will abstract 0.357 per cumec of the fish abstracted by SZB (rather than 0.383 reported in version 6), because of the intake head design. The methodology used to derive this impingement reduction factor is described in BEEMS Scientific Position Paper SPP099 v4 and is based upon the methodology proposed by the Environment Agency for impingement studies at Hinkley Point C. Further information is provided in Section 4.6.

2. **Updated Sizewell B impingement predictions:** Following statistical review, a bootstrapping approach, akin to the methods employed at Hinkley Point, with an additional step to account for periods when sampling was not possible due to station outages, has been applied. The bootstrapping method randomly reallocates samples from adjacent years to account for sampling gaps. The bootstrap outputs for each species provides the mean numbers impinged by quarter for Sizewell B, which is converted to percentage impingement by quarter to show the seasonal patterns. The bootstrapping approach has been applied to calculate annual impingement numbers and 95th percentile confidence intervals for Sizewell B these numbers form the basis for scaled Sizewell C predictions. Further information is provided in Section 4.4.

   The development of the model and bootstrap approaches, the advantages, and limitations along with differences in impingement predictions between the approaches for Sizewell B is provided in BEEMS Technical Report TR339 v3.

3. **Evaluating uncertainty in impingement parameters:** The confidence of impingement predictions is determined by the underlying assumptions used to parameterise the assessment and the appropriateness of the comparator. **:** Section 7 provides an assessment of the sensitivity of impingement predictions and effects to uncertainties in the defining parameters including LVSE efficiency, FRR mortality, EAV values and changes in distribution of fish.

4. **Local effects assessment:** To address specific concerns relating to the potential for local effects Section 9 provides an additional local-level assessment.

Version 7 of this report has been updated further following ongoing evidence gathering, technical review and specific stakeholder concerns. A series of new Scientific Position Papers provides the evidence base that underpins edits to this report, these include:

- SPP101 - Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell.
- SPP102 - Use of Spawning Production Foregone EAVs for impingement assessment. Section 4.8.1.
- SPP103 - Consideration of potential effects on selected fish stocks at Sizewell. Section 4.10.2 and Section 8.
- SPP104 - Worst case glass eel entrainment assessment for Sizewell C. Section 6.6.2.3.
- SPP108 - Sensitivity of the Alde Ore Transitional Fish Classification Index (TFCI) to changes in smelt, *Osmerus eperlanus*, abundance. Section 9.

Whilst the edits to the report have in some cases resulted in changes to absolute impingement predictions, in terms of numbers and biomass, the scale of effects have not altered, and the conclusions remain unchanged.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station will be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF is required to evaluate the effects that the abstraction of water may have on organisms in the marine environment. Although the cooling water intakes will be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, a significant number of small organisms (small fish and crustaceans, and plankton) will inevitably enter the cooling water intake. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh (10 mm at SZB, also proposed for SZC) drum screens which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages. Impinged organisms will be returned to the sea via a Fish Recovery and Return (FRR) system. Not all will survive this process and EDF Energy is required to evaluate the effect of the loss of these organisms on marine communities and also of the potentially polluting effects of dead biota discharged from the FRR system on local water quality and marine ecology.

As was the case for Hinkley Point C (HPC), the impingement assessment process for SZC makes use of extensive impingement data collected at the adjacent power station (Section 4). At HPC, assessment of impingement losses was made using data collected at the Hinkley Point B (HPB) station (BEEMS Technical Report TR456). The HPC assessment used data collected in 2009 – 2010 from a sampling programme, known as the Cefas Comprehensive Impingement Monitoring Programme (CIMP), which comprised 40 * 24-h samples to estimate annual impingement of the station. The predictions were supported using information from the lower-resolution Routine Impingement Sampling Programme, which has sampled for over 35 years at the HPB site, but at a lower sampling effort.

Prior to the BEEMS programme, there was no regular impingement sampling at the SZB site. To fill this data gap, a CIMP programme was initiated in 2009 to provide the necessary data for predictions of impingement for the proposed SZC site. The CIMP was designed to provide 24-h sampling of the fish, invertebrates and other material passing through the SZB cooling water systems on 28 to 40 occasions per year. Between February 2009 and March 2013, 128 sampling visits were completed. Following a break in sampling, a further 77 visits were completed between April 2014 and December 2017, giving rise to a dataset comprising 205 samples. A description of the sampling undertaken between 2009 and 2013 can be found in BEEMS Technical Reports TR120, TR196, TR215, and TR270. Details of the full 2009 – 2017 sampling and impingement predictions for Sizewell B can be found in BEEMS Technical Report TR339. The dataset was used to provide an annual estimate of the numbers and weights of fish and invertebrates impinged at the SZC station for the 9-year period. A total of 91 finfish and 62 invertebrate taxa were enumerated.

1.1 Selection of key taxa for SZC impingement assessment

The impingement assessment process for SZC is described in Section 4.

For the purposes of the Sizewell marine ecology impact assessments, taxa are key in the ecosystem if they meet at least one of the following criteria:

- Socio-economic value: Species that contribute to the first 95 % of the first sale value of commercially -landed finfish in the area off the east Anglian coast and contributes to the first 95 % of total abundance in at least one of the available datasets (2 m beam trawl, otter trawl, BEEMS eel survey, annual impingement). Commercial landings are recorded using statistical rectangles that divide the southern North Sea into areas of 30 minutes latitude by 1 degree longitude and covering

approximately 900 nautical miles[2]. For the purposes of describing local commercial fisheries, 6 rectangles have been considered, that extend from north Norfolk to the Thames estuary and eastwards to the middle of the North Sea (BEEMS Technical Report TR123). Socio-economic value was calculated using data supplied by the Marine Management Organisation (MMO) and which was used in BEEMS Technical Report TR123. Six taxa (**herring, bass, sole, cod, plaice, thornback ray**).

- Conservation importance: The "S41 Priority Species" spreadsheet given by Natural England (http://publications.naturalengland.org.uk/publication/4958719460769792) was used to assess the conservation status of the fishes recorded in the Greater Sizewell Bay. This spreadsheet was built based on the legislation in Section 41 of the Natural Environment and Rural Communities (NERC) Act 2006. It is worth noting that measures in place to provide protection for the named species apply to the adult stock rather than the eggs or larvae, and focus on halting the decline of the spawning stock biomass mainly via restriction on exploiting recruited species. The resulting list contains one species which has not been detected in the extensive BEEMS sampling programmes (Atlantic salmon). 16 taxa (**allis shad, twaite shad, European eel, herring, Atlantic cod, whiting, plaice, sole, salmon, sea trout, cucumber smelt, river lamprey, sea lamprey, tope, mackerel, horse mackerel).**

- Ecological importance: If a taxon is present in at least 30 % of samples and contributes to the first 95 % of total abundance in at least one of the available datasets (2 m beam trawl, otter trawl, eel surveys, annual impingement), we consider it to be common and/or abundant enough to play a key trophic role within the ecosystem. 13 taxa (**sprat, herring, whiting, bass, sand goby, sole, dab, anchovy, thin lipped grey mullet, flounder, cod, plaice, thornback ray)**.

There are 24 key fish taxa in the Greater Sizewell Bay in total based on either their commercial value, their ecological importance, or their conservation status. Several taxa fall under more than one criterion and four taxa are important with respect to all three (Dover sole, herring, cod and plaice) (BEEMS Technical Report TR345).

The 24 species are representative of the fish assemblage at Sizewell because:

a. they represent an average of 94.6% of the total fish impingement numbers during the CIMP programme from 2009-2013;

c. they contain examples from all functional guilds with the exception of freshwater species which, as would be expected, are rarely found at Sizewell;

d. they contain examples from all the feeding guilds and habitat groups;

e. they contain all of the indicator species found in the vicinity of Sizewell that are assessed in the WFD “fish” biological quality element in transitional waters; and

f. they contain the key prey species that supports the food web at Sizewell (including for HRA protected marine birds.

1.1.1 Conservation species impingement data

In the 9 years of extensive impingement sampling, catches of 4 conservation species were extremely rare or non-existent:

- Atlantic salmon 0 fish
- Allis shad 1 fish in 2009
- Sea lamprey 1 fish in 2015
- Sea trout 1 fish in 2010

None of these species were caught in any of the BEEMS fishing surveys. Salmon is, therefore, not expected to be impinged at SZC.

The lone Allis shad is considered to be a straggler from the Garonne population (BEEMS Scientific Position Paper SPP071/s) because this is the largest self-sustaining population that is closest to Sizewell. (There is no evidence that the Allis shad found in the Tamar are part of a self-sustaining population). Making no allowances for the fact that only one fish was caught, the predicted unmitigated impingement at SZC is three fish per annum, which decreases to only one fish per annum with the effect of the intake head design. It is not statistically valid to extrapolate this data point to future years and a more meaningful assessment would be to compare the scaled-up impingement in 2009 with the stock estimate for that year of 27,397 adults (BEEMS Technical Report TR456). This is the stock comparator that has been used in this report.

The predicted impingement for sea lamprey is also five fish but with the effect of the LVSE intake head design, this drops to only two fish impinged. Actual impingement losses considering survival through the fitted FRR mitigation are < 1 fish per annum (again using a statistically invalid extrapolation technique from the one fish caught in the CIMP programme). Allowing for the fact that only one fish was caught in the eight years when sampling took place, the predicted impingement is < 1 fish (0.36) per annum which is ecologically insignificant. No attempt has been made to put such a negligible impact into an adult stock context as the effect is insignificant.

The lone sea trout caught in the CIMP programme is considered to be from the UK North East coast population. A highly conservative assessment has been made in this report for sea trout based upon scaling up the 1 sea trout caught in May 2010 to a model output of 10 fish per annum for SZC without impingement mitigation. After adjustment for the impingement reduction by the LVSE intakes, the annual impingement loss estimate for SZC is approximately 4 fish per annum (Table 15). This estimate has then been compared with the annual mean net catch for sea trout in the sampling period to produce an estimated SZC effect of 0.01% of annual landings. The use of landings statistics overestimates the effect on the stock as landings are much less than the SSB. However, this simplistic assessment is not appropriate for such rare impingement events as it does not take into account that no other sea trout were detected in the period 2009 -2017. Allowing for the fact that only one fish was caught in eight years when sampling took place, the predicted SZC impingement is approximately 0.5 fish per annum which is ecologically insignificant.

1.2 Aims of this report

The purpose of this report is to provide predictions of SZC impingement, based on the CIMP dataset collected between 2009 and 2017.

Aims:

- Provide predictions for all key finfish and shellfish species, based on the proposed design of the station with and without the selected impingement mitigation technology fitted.
- Place the predicted losses of these key species into the context of the most relevant stock unit area.
- Determine whether SZC impingement represents a significant effect for any of the key species
- Determine whether SZC entrapment (impingement predictions from this report +entrainment predictions from BEEMS Technical Report TR318) represents a significant effect for any of the key species
- The impingement predictions from this report are used to provide the source term for modelling the impacts of dead fish discharged from the SZC FRR system upon the local marine environment.
- Determine whether SZC, acting in-combination with SZB, has the potential to cause local-level effects.

2 Selection of impingement mitigation technology for Sizewell C

The Environment Agency have issued guidelines for the types of measures that could be adopted at new direct cooled power stations to reduce the predicted environmental impacts of impingement and indirectly the potential for pollution by discharges of dead fish back into the marine environment (Environment Agency 2005,2010). As explained in these guidelines, in practice the selection of potential impingement mitigation measures involves a complex consideration of the likely effectiveness of each measure in the marine environment at the station location, engineering feasibility and operational safety for staff and the plant. The range of feasible mitigation options is much larger in a freshwater and some brackish environments that do not present a high biofouling risk to station plant and for small abstractions of a few cumecs but many of these options are infeasible for a coastal direct cooled power stations (Environment Agency, 2005). SZC's intakes would require an abstraction of 132 cumecs and would be mounted on the seabed in a highly turbid, coastal environment with high wave exposure offshore of the Sizewell-Dunwich Bank. The site is classified as being at high risk of biofouling.

2.1 Biofouling and other blockage risks

Design decisions on the choice of impingement mitigation options must be taken in light of the environmental risks to the plant from the Sizewell marine environment. The cooling water system for SZC would be nuclear safety classified and it is therefore extremely important that the system is designed to prevent blockages of critical plant. The blockage hazards at Sizewell include:

- marine debris (discarded nets and ropes and marine litter – this risk also includes potential impact damage from large and heavy items) and large clogging organisms (e.g. in the southern North Sea: sprat shoals and ctenophore blooms)
- colonisation by biota (biofouling) that could cause blockages and subsequent reductions in cooling water flow in the system (e.g. shellfish, barnacles, reef forming organisms)
- siltation due to high suspended sediments (risks designed out by elimination of low velocity regions in high risk zones of the cooling water system e.g. intake heads and forebays

Established power station design practice is to progressively reduce the risks from marine debris and large clogging organisms by the use of robust, coarsely spaced intake bars at the intakes followed by two tiers of filtration within the plant (trash racks and then drum or band screens). Colonising organisms are deterred by chlorination of the cooling water system (Note chlorination is designed to deter settlement rather than to kill organisms).

EDF Energy's policy for its existing UK fleet is that stations exposed to a high biofouling risk should have the capability of maintaining a default regime of continuous, year-round chlorination to obtain 0.2 mg l^{-1} Total Residual Oxidant (TRO) in the discharge water from plant vulnerable to biofouling. Sizewell B is currently assessed as subject to a high risk of biofouling and operational practice is, therefore, to maintain the default regime. The detailed application of the EDF Energy policy, for example whether the entire cooling water (CW) system is dosed continuously or just critical plant, is dependent upon site specific issues such as the flexibility of the chlorination plant. At SZB the current policy is to dose the entire CW system, including the inlet tunnels, throughout the year.

Based upon the known risk of biofouling at Sizewell, it would be necessary to dose critical plant at Sizewell C (the condensers and essential cooling water systems) during the growing season when seawater temperatures exceed 10°C and also to have the flexibility to dose those systems at other times of the year based upon operational need. The chlorination policy for the other parts of the CW system has to be effective against any biofouling risk that would threaten the operation of the station whilst minimising toxicological effects on non-target species. In particular, Sizewell C will be fitted with a Fish Recovery and

Return (FRR) system (Section 2.4.1) to reduce the mortality of impinged fish. The Environment Agency best practice screening guidelines are that, wherever possible, chlorination should be avoided before the FRR so as to minimise any loss of fitness for those fish returned to the marine environment. This is a larger environmental issue for Sizewell C (SZC) than Sizewell B (SZB) due to the length of the SZC CW tunnels and the potential significant increase in TRO exposure time, dependent upon chlorination system design, for organisms abstracted into the CW system.

2.2 Impingement Mitigation Optioneering

EDF Energy have carefully considered each of the available options in the Environment Agency guidelines. The initial stage is to determine the location of the intake and outfall locations at a site based upon environmental and engineering considerations. In practice given practical engineering constraints on intake siting and intake tunnel length, the available options may not significantly change the predicted abstraction of organisms in a well-mixed environment. For most species this is the case at Sizewell (Section 4.1.1). There are measures to reduce the mortality of impinged fish and crustacea but for planktonic organisms (e.g. zooplankton, fish eggs and fish larvae) due to their small size (typically less than a few mm in length or diameter) little can be done to reduce the numbers and mortality of entrained organisms. However, due their number, spatially ubiquitous nature and the high natural mortality of the majority of planktonic organisms, entrainment impacts from coastal power stations are rarely significant and that is the case predicted for SZC (Section 6.7). There are two complementary mitigation technological approaches to minimise impingement losses using:

a. Biota exclusion technology - measures to minimise the number of organisms abstracted into the station intakes; and

b. Biota recovery technology - measures taken inside the cooling water system to filter organisms out of the cooling water stream and safely return as many as possible of them alive to sea.

2.3 Biota exclusion technology

Several techniques are available, with variable effectiveness, to reduce the number of fish and crustacea being abstracted with the seawater and impinged on the cooling water fine filtration systems (drum or band screens). These measures are located at or close to the intakes which for SZC would be in the open sea at more than 3km offshore to the east of the Sizewell-Dunwich sandbank and mounted on the seabed. From a design perspective the five key requirements of such technology for SZC are:

- Compatible with nuclear safety requirements for an uninterrupted supply of cooling water for the 60-year operational life of the station. This implies the use of systems that are highly resistant to damage or blockage and that are readily maintainable in all weather conditions, all year round.

- Operation and maintenance compatible with the EDF Energy’s zero harm safety policy for staff and contractors. i.e. no requirement for activities judged hazardous to human life.

- Proven operational experience in a similar environment that demonstrates reliable delivery of effective environmental mitigation.

- Due to the offshore environment any system should preferably use entirely passive technology e.g. requiring no power, chemical supplies or compressed air systems that could compromise reliability and hence nuclear safety and environmental effectiveness.

- System operational maintenance requirements must be compatible with high power plant availability.

The potential biota exclusion techniques include:

i. Physical barriers (e.g. wedge-wire screens and bubble curtains);

ii. Auditory or visual behavioural deterrents that aim to deter fish from a trajectory likely to cause their abstraction (e.g. acoustic fish deterrents (AFDs) and strobe lights; respectively).

iii. Design of the intake heads to minimise the risk of biota abstraction e.g. by use of velocity-capped intake heads, limiting intake velocities, minimising abstraction cross sectional area by mounting the intake orthogonal to the tidal flow. Low velocity side entry (LVSE) intake heads combine all of these attributes and represent the state of the art for such an approach and are included in the approved design for HPC.

Several of these measures have been deployed in riverine locations. However, for existing coastal power stations only limited intake head design improvements (e.g. use of velocity caps, and sizing intakes to reduce intake velocities) or more recently AFDs have been deployed. AFDs have only been deployed at stations with onshore or very nearshore intake locations and none have been deployed at far offshore locations (e.g. the 3km+ offshore locations proposed for SZC).

Each of the biota exclusion options are considered in turn below.

2.3.1 Passive wedge-wire cylinder screens (PWWC screens)

In principle, the best form of impingement mitigation could be to prevent abstraction of fish and crustacea by the use of very fine screens with gaps of a few millimetres. Environment Agency (2005) considers that "Passive wedge-wire cylinder (PWWC) screens are a tried and tested solution and are generally regarded in Britain as the best available technology for juvenile and larval fish protection". Wedge-wire' refers to the cross section of the welded wires that are wound helically to form a cylindrical screen surface.

PWWC screens are commonly used to exclude fish from small, riverine abstraction intakes and there are also some small industrial applications in brackish and saline waters. However, because of the very small gaps between the wires the screens are at high risk of becoming blocked or damaged by floating debris, weed and litter even in riverine environments. In the marine environment the risks are much greater due to the potential hazards from ctenophore blooms, pelagic fish shoals and marine debris and also particularly from biofouling by colonising organisms. To reduce the blockage risk in high risk environments, PWWC systems have complex maintenance requirements with frequent, active cleaning required (for example with rotating mechanical brushes or by the injection or high-pressure air). The maintenance advantage of the passive screen then largely disappears as the active cleaning systems present significant reliability risks. Practical PWWC systems are complex systems, that must be recovered for regular maintenance including to repair damage to the fine screens. To do this such filters are usually track mounted on large motorised platforms attached to the shore.

It is instructive to consider the theoretical sizing of a PWWC system for SZC. The abstraction capacity of PWWC filters depends upon the wire spacing. To eliminate most impingement issues (and to comply with the Environment Agency screening guidelines for glass eels) it would be necessary to employ a 2 mm wedge wire filter. The largest commercially available 2 mm PWWC filters would permit an abstraction of 2.7 cumecs with an 8m long, 2.5m diameter cylindrical filter i.e. to create the required 4 intake head system for SZC with a total abstraction of 132 cumecs would need a minimum of 13 PWWC filters per head, probably15+ allowing for redundancy for cleaning and repair. i.e. a total of up to 60+ filters for SZC with each head length being greater than 130m in length, all track mounted on 4 powered offshore platforms complete with compressed air cleaning and antibiofouling chemical supply. The lifetime for such complex systems are unknown but certainly are not the required 60 years for SZC. The required recovery frequency for the filters is also unknown but in the growing season could be monthly. The filters would require frequent replacement using heavy lifting equipment and the track mounting would also require replacement, probably on at least a decadal frequency. No such system has been deployed at any power station worldwide.

Conclusion: PWWC filtration is an unproven technology for direct cooled nuclear power stations and is not considered compatible with nuclear safety requirements for a constant supply of cooling water.

This is the only potential technology that could theoretically eliminate impingement and therefore any other mitigation measure would require technology to recover organisms from the station drum and band screen.

2.3.2 Bubble curtains

The use of a bubble curtain, whereby air is released along a section of seabed to create a 'curtain' of bubbles rising to the surface and thus creating a barrier to fish, is a potentially useful exclusion technique in still or slow-moving waters but is not suitable as a permanent exclusion system for waters where tidal currents ebb and flow at >1m/s and would break down the curtain of bubbles. Small sized bubble barrier systems to reduce underwater noise have been temporarily deployed at sea from jack up rigs during periods of low tidal velocities during windfarm piling activities but never around the required large intake heads of a power station at all states of the tide and in all weather conditions.

Conclusion: bubble curtains are an unproven technology for use around the offshore intakes of direct cooled nuclear power stations and are considered unlikely to deliver substantial impingement mitigation at Sizewell.

2.3.3 Behavioural deterrents – strobe lights

Strobe lighting, can be used to deter some fish (for example eels), however, their effectiveness in turbid coastal waters is unproven (turbidity relates to the amount of material; suspected in the water and thus restricting visibility). The surface water at Sizewell is classified as "intermediate turbidity" and experiences increased levels of turbidity in autumn to spring when storms and increased wave action stirs up sediment. Repeated surveys have demonstrated that it is extremely difficult to photograph any seabed features at Sizewell due to this limited visibility. The near-bed conditions at the Sizewell C intake locations have particularly high levels of suspended sediment with measured levels of greater than 2 g/l at the height of the proposed intake surfaces due to sediment transport around the Sizewell-Dunwich Bank and mean winter values of ca. 500 mg/l (BEEMS Technical Report TR498). Such high levels of suspended sediments dramatically reduce the penetration of light through the water and would prevent strobe lighting from having an effective deterrent function.

Conclusion: strobe lighting is an unproven technology for use around the offshore intakes of direct cooled nuclear power stations and is considered unlikely to deliver impingement reductions at Sizewell.

2.3.4 Behavioural deterrents - electric barriers

An electric fish barrier is a non-physical barrier that prevents fish passage from one location to another or induces fish movement from one area to another within a body of water using an electric current. Electric barriers pass an electrical current through the water, thus creating an electric field. As fish enter the electric field they become part of the electrical circuit and experience electric current flowing through their body. As the fish approaches the anode, the electric field intensifies, which causes the fish to generally turn around and swim away from the electric barrier. The set-up of an electric barrier requires a series of electrodes, alternating anodes and cathodes to span across a body of water. However, electric barriers are affected by water conductivity and are unsuitable for marine or brackish water environments, therefore, the use of an electric fish barrier is not feasible for the offshore Sizewell C intakes.

Conclusion: Not feasible

2.3.5 Behavioural deterrents - Acoustic fish deterrents (AFDs)

Well-designed AFD systems have been reported to reduce impingement of some fish species by creating high intensity sound fields of swept frequency pulses of sound around an intake thereby causing some fish to change direction and move away from the sound field. The available commercial systems are predicted to work well with sensitive pelagic species (e.g. sprat and herring), moderately for a range of demersal species (e.g. cod, whiting) and poorly for species with either low hearing ability or low responsiveness to the underwater frequencies used such as eels, lampreys, and some flatfish. All power station installations of AFDs are either on shore or very close to shore to facilitate system maintenance. There are no AFD systems operating in environments at multiple kilometres offshore as they would have to be at SZC.

An AFD system for SZC would necessarily be very similar to that evaluated for HPC and would require up to 288 underwater sound projectors located at the CW intakes approximately 3km offshore. The issue of system longevity is a particular constraint as the AFD sound projectors need to be recovered and serviced on shore currently at 12 monthly intervals, possibly at up to 18-month intervals with further research. Other key issues with installation of AFDs in offshore environments are the large number of electrical components required at each intake, the supply of reliable high levels of electrical power and control telemetry to the individual sound projectors and the required close proximity of the projectors to the intake heads themselves (without affecting the hydrodynamic performance or structural integrity of the head. The recovery of the projectors for maintenance would be a major issue. Four permanent offshore platforms with track mounted projectors could theoretically be used but the these would have to be very close to the intakes, affecting the hydrodynamic performance of the intake heads and the resultant sound field would be likely to have an unpredictable deterrent effect due to complex interference patterns caused by the structures themselves. More significantly, such a complex track mounted system with a profusion of electrical wiring would also have major issues with long term reliability for the 60-year lifetime of the station in the corrosive marine environment at the site. The use of such structures is not considered feasible. In the near zero underwater visibility at the SZC intakes the use of robotic servicing via remotely operated vehicles is not considered feasible and instead each of the projectors would have to be recovered and replaced using divers working by feel, operating from anchored support vessels for months during every year, for the 60-year lifetime of the station. The system would be extremely complex to construct and to maintain with offshore operations restricted to narrow tidal windows and subject to lengthy periods of weather downtime in the exposed location offshore of the Sizewell-Dunwich Bank. An assessment of the risks involved with such an operational system has concluded that the safety risks to maintenance staff would be unacceptable.

Conclusion: Logistical and safety considerations preclude the use of AFDs at Sizewell C.

2.3.6 Intake design - LVSE intakes

Low velocity side entry (LVSE) intake heads have already been designed and received regulatory approval for use at HPC. These very large intake structures are designed to minimise impingement by:

a. reducing vertical velocities which fish are ill equipped to resist by means of velocity caps on the intakes.

b. limiting the exposure of the intake surfaces to the tidal stream and in so doing reduce the risk of impingement for fish swimming with the tidal stream. i.e. to reduce the cross-sectional area of the intake to the prevailing tidal directions by mounting the head orthogonally to the tidal flow.

c. reducing intake velocities into the head to a target velocity of 0.3m/s over as much of the length of the intake surface which will maximise the possibility of most fish avoiding abstraction

LVSE intakes have the advantage of reducing impingement for all fish species at risk of abstraction. The HPC LVSE intakes will be the first deployment of this technology on an operational power station worldwide and represent a considerable advance in the design of intake heads. One option for SZC would have been to reuse the HPC design. However, Hinkley Point has a low biofouling risk whereas Sizewell has a high risk and the starting point for SZC would be an assumption of chlorinating the entire cooling water system starting at the intake heads. Studies demonstrated that the required chlorine dose combined with the exposure time in the 3km tunnels would significantly reduce the survival of many of the species that the FRR is designed to protect. After a consideration of risk and engineering feasibility by EDF Energy, based upon operational experience at Sizewell, the size of the proposed SZC intake tunnels and the use of a simple SZB style capped intake for SZC it was decided that it would be acceptable not to chlorinate the intakes and intake tunnels, thereby significantly reducing the exposure of abstracted organisms to TROs. This led to the initial recommended chlorination policy for SZC described in BEEMS Technical Report TR316:

a. Maintaining a TRO level of 0.2mg l^{-1} at the discharge of critical land-based plant (condensers and essential cooling water systems) throughout the year.

b. Intake heads, inlet tunnels and the forebays not to be chlorinated because of the impracticality of operational control of the chlorination dose in the intakes and inlet tunnels and so as not to compromise the FRR system effectiveness.

c. Velocity-capped intake heads of a similar design to Sizewell B to be employed at Sizewell C. Such intake heads are much more readily maintained and much less likely to biofoul than the low velocity side entry (LVSE) intake heads planned for Hinkley Point C.

d. In order to protect the drum screens and FRR system, chlorinate the drum screen wells but only in the growing season when seawater temperatures exceed 10°C.

e. Apply for a WDA discharge permit for TRO (measured before the discharge tunnel) of 0.2mg l^{-1} throughout the year.

Further studies demonstrated that even this revised chlorination policy would impair the effectiveness of the FRR system for demersal species such as the juvenile bass found at Sizewell. After a further careful engineering review the first dosing point in the cooling water system was moved to after the drum screens thereby removing chlorination from the FRR system.

The decision to use SZB style omnidirectional velocity-capped intakes was based upon a risk assessment that concluded that the unchlorinated LVSE intake heads designed for HPC would present an unacceptable biofouling risk at Sizewell due to the surface area of the baffles inside the intake head structure. If the HPC LVSE heads were fitted at SZC the entire CW system would have to be chlorinated, effectively eliminating the benefit of the FRR system for most species. However, the SZB style velocity-capped intake heads would offer no impingement rate improvement over the existing SZB intakes; in particular organisms swimming in the tidal stream would still experience high intake velocities in excess of 1m/s for a large part of the tidal cycle which would be too high to permit most organisms to avoid abstraction.

To reduce the biofouling risk, it was necessary to remove as many of the internal baffles in the HPC LVSE intake design as possible and to reduce areas of low velocity flow within the head. After extensive computational fluid dynamics (CFD) modelling studies of a range of LVSE designs it was determined that a modified version of the HPC LVSE intake heads could be designed to have low biofouling risk and achieve the same reduction in cross sectional area as the HPC intake heads. By improving flow dynamics within and around the heads the modified heads the variation in intake velocities across the intake surfaces could also be minimised and permit low intake velocities to be achieved over the whole tidal cycle (Section 4.2, BEEMS Scientific Position Paper SPP099).

Conclusion: LVSE intakes, modified from the HPC design to reduce biofouling risk would provide reliable and effective impingement reduction for SZC.

2.4 Biota Recovery Technology

2.4.1 Fish Recovery and Return System

A Fish Recovery and Return (FRR) system is designed to return robust species (particularly flatfish, eels, lampreys and crustacea and to a lesser extent demersal species such as bass, cod and whiting) that are impinged onto the station drum and band screens safely back to sea. A state-of-the-art fish recovery return system has been designed and received regulatory approval for Hinkley Point C. This system has been subject to intensive design scrutiny and complies with Environment Agency guidelines for such systems. EDF Energy policy for SZC is to replicate the design of HPC as far as possible and so it has been decided to incorporate the HPC design into SZC. The tidal range at Sizewell is less than at Hinkley Point and it has, therefore, been possible to improve the fish friendliness of the SZC FRR system by removing an Archimedes screw system that is essential to manage water levels in the HPC cooling water system.

The drum and band screen employ fine mesh filters to remove impinged organisms from the cooling water flow. The default mesh size for the EPR reactor is 5 mm square as opposed to the 10 mm mesh filters employed at SZB. After careful consideration of the risk of clogging by summer ctenophore blooms at Sizewell, it has been proposed to fit 10 mm mesh filtration for SZC which has been operationally proven not

to cause clogging at SZB. Trash racks are used to protect the drum and band screens from damage and over loading. With a 5 mm mesh the track rack has to have 50 mm vertical bar spacing, a 10 mm mesh size could allow the trash rack bar spacing to be increased to 75 mm. The environmental effects of the larger track rack bar spacing and the larger mesh size are beneficial and will reduce the predicted effects on fish stocks. The larger mesh size will also reduce discharges of dead biomass from the FRR outfall. There would be no adverse effect on the survival of any glass eel that may be abstracted as glass eels would pass through both 5mm or 10mm filtration.

Conclusion: FRR systems with proposed 10 mm mesh screens would prove reliable and effective impingement reduction for SZC.

2.5 Conclusions

Studies of impingement reduction technologies for SZC demonstrated that two measures were both feasible and likely to reliably deliver reductions in the predicted losses of fish and crustacea. These are LVSE intakes and an FRR and both of these technologies are planned for the proposed SZC station.

With these measures fitted at SZC, the predicted reduction in impingement mortality compared with an unmitigated station based upon the expected performance of the LVSE intakes (Section 4.6) and the FRR system (Table 6) are shown in Table 3.

Table 3. Predicted reduction in impingement mortality for SZC fitted with LVSE intakes and FRR system compared.

Group	Example species	Impingement reduction
Pelagic fish	sprat, herring, anchovy, shad	64%
Demersal fish	bass, cod, whiting, grey mullet	78-80%
Epibenthic fish	eel, lampreys, sole, sand goby	93%
Shellfish	Crab, brown shrimp	93%

3 Relevant site features

The Sizewell site hosts SZA and SZB. SZA ceased operation at the end of 2006 and is being decommissioned. SZB is a direct-cooled nuclear power station using a pressurised water reactor design to generate an electrical output of about 1195 MW. The SZB intake and outfall structures are located inshore of the Sizewell Bank system (Figure 1). For SZB the volume of water extracted is 51.5 m^3 s^{-1} (51.5 cumecs). The intake structure consists of two intake heads, each initially leading to its own tunnel, which then join to form a single tunnel. Each head is octagonal and ~11.5 m across and is omnidirectional. The structure sits ~1.5 m above the seabed and the intake aperture is 3 m high. The tidal flows in the region are highly rectilinear and peak at 1 m s^{-1} on spring tides. During peak tides, a tidal streamline the width of the intake enters the intake, whereas at slack water the water is drawn from a radius around the intake.

The SZB intake was designed to not include any large superstructure that might attract fish and has a cap to limit drawdown from the surface. There are no screen bars or other devices that are designed to reduce fish entrainment or impingement at the intakes. There is also no provision for maintenance or internal access. The intake tunnel consists of nine square precast concrete caissons, each 4.82 m wide (internal) and 702 m long. Flow velocities in the tunnel are approximately 2.5 m s^{-1}, giving a passage time through the tunnel of approximately 5 min. The capped intake design was intended to ensure that warm, surface water was not drawn into the station, as well as to reduce the likelihood of there being a surface vortex. However, the capped design, by eliminating vertical water movement, was also expected to help fish avoid entrapment into the intakes, because fish are ill-equipped to respond to vertical water movements. Velocity caps were expected to be especially protective of pelagic species such as sprat and herring. Studies undertaken in March/April 1994 concluded that the B station impinged significantly fewer fish than the A station, which was not fitted with a velocity cap (Turnpenny and Taylor, 2000). The SZB discharge is through two tunnels that point offshore and upward with the discharge in the nearshore region ~150 m from the low tide mark (Figure 1).

The proposed SZC intakes will be low velocity side entry (LVSE) structures designed to reduce fish impingement. A total of four LVSE heads will be fitted with two heads fitted on each of the two intake tunnels. The intake heads will located approximately 3km offshore, on the eastern side of the Sizewell-Dunwich Bank in approximately 15m (ODN) depth (Figure 1). The detailed design of the intake heads is not yet complete but the performance of the design with the worst hydrodynamic performance has been used for this impingement assessment in order to envelope worst case impingement effects. This design has intake surfaces of the same size as those planned for Hinkley Point C (HPC) with the bottom of the intake surfaces being at more than 1m off the seabed (as at HPC) in order to reduce abstraction of benthic organisms. The preferred outfall locations 09a and 09b marked are contingent on final engineering geotechnical assessments but will be there or slightly further offshore and in approximately 18m depth.

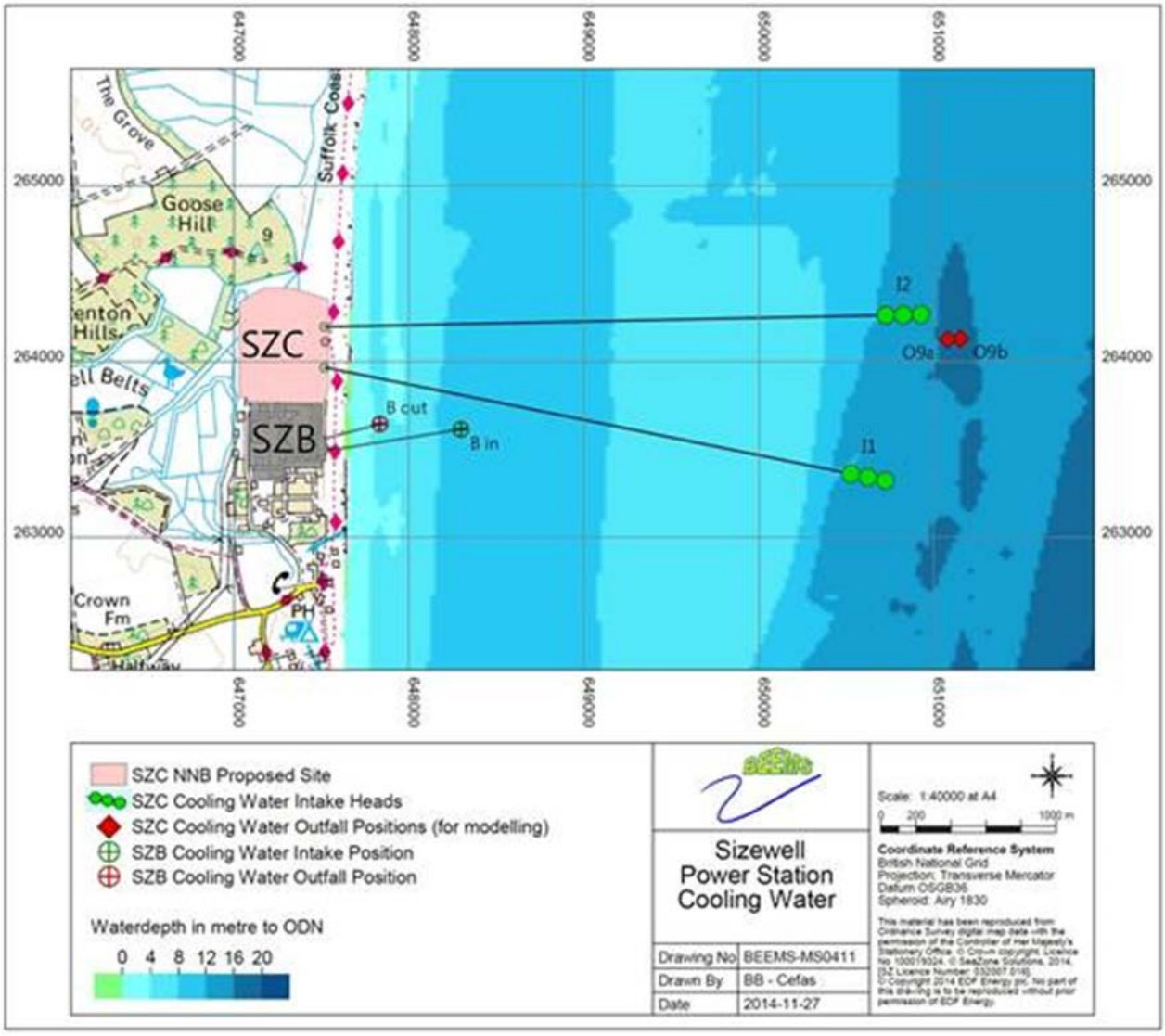


Figure 1 The coast at Sizewell, showing the locations of the intake and outfall for SZB and the proposed intakes and outfalls for SZC. The locations of three intake locations are shown for each SZC tunnel but only 2 heads will be fitted per tunnel with locations dependent upon geotechnical considerations.

4 Impingement Assessment Methodology

4.1 Introduction

To estimate the unmitigated impingement at SZC the assessment approach adopted in this report scales the measured impingement at SZB by the ratio of the cooling water volumes extracted by the two stations. The accuracy of the assessment depends upon whether:

- the fish community is the same at the location of the SZC intakes (approximately 3 km offshore) as at the SZB intakes (approximately 700 m offshore); and
- the SZC intakes will abstract the same amount of fish per cumec as SZB.

4.1.1 Differences in the abundance of fish at the SZB and SZC intake locations

The results of subtidal fishing surveys in the greater Sizewell Bay area are described in BEEMS Technical Report TR345). Ten demersal fishing surveys were carried out over a 4-year period. Sampling was conducted using two different fishing gears – a 2 m beam trawl and a commercial otter trawl. A coastal pelagic fish survey was also carried out in March and June 2015. These surveys found predominantly juvenile fish. Forty species were identified in the 2 m beam trawl catches, 25 in the commercial otter trawl catches. These fishing surveys found no significant spatial differences in the fish community nor the fish length distributions between the locations of the SZC and SZB intakes.

4.1.1.1 Bass distribution within Sizewell Bay

Bass impingement at SZB consists mostly of juvenile fish with few mature adults. From the CIMP survey from 2009- 2017 it is known that bass impingement at SZB takes place almost exclusively in winter (with 98.8% in the period November to March and 1.2% from April to October). In the period of lowest seawater temperatures (December to March) bass impingement is 96.1% of the annual total). It is well known that juvenile bass are attracted to the warm water outfalls of power stations in winter (Jennings *et al* 1991) and this has led to the creation of areas of restricted fishing around several UK power station outfalls. It was, therefore, expected that there would be differences in bass spatial density at Sizewell due to the SZB discharge but the extent of the area of attraction at Sizewell was unknown. To resolve this issue a highly targeted survey programme was undertaken to investigate bass distribution in the Greater Sizewell Bay (BEEMS Technical Report TR380). The aim of the programme was to quantify the impact of the SZB discharge upon the spatial distribution of bass and specifically to determine how the abundance of bass differed inside and outside of the SZB plume and whether there was a significant difference in abundance at the locations of the SZB and proposed SZC intakes.

In February 2016 a 5-day sampling programme was undertaken using an otter trawl known to be efficient at sampling bass. The high trawl opening allowed the sampling of fish from a larger part of the water column than a beam trawl (from just off the bottom to approximately 1m off the seabed) and the 4 mm mesh liner retained small individuals. February was chosen as this corresponds to the period of peak bass impingement at SZB (43% of the annual impingement total). Sampling was undertaken inside and outside of the Sizewell-Dunwich Bank, and close to and distant from the current and proposed intake/outfall locations of SZB and SZC, respectively. Forty-one tows were completed in the Greater Sizewell Bay. Three sampling sites were located outside the Sizewell-Dunwich Bank (“offshore”) in similar depth strata and habitats. One site was in the area of the proposed SZC intake/outfall structures and the other two were some distance north and south of this. Five sites were located inside the Bank (“inshore”); the first immediately north of the SZB outfall; two about 1 km north and south of this, but within the SZB thermal plume; and two several km north and south and outside the plume. During the survey, 110 bass were recorded, ranging between 15.5 and 45 cm TL, with the majority being 2 y old (70.9%) followed by 3 y olds (14.5%) and the remainder up to 6 years of age (No 0 or 1 group fish were caught). The survey found:

a. A statistically significant majority of bass at Sizewell (105 bass or 95% of the bass caught) were recorded inshore of the Sizewell-Dunwich Bank.

b. A statistically significant increase in bass abundance was found near the SZB plume inshore of the Bank compared with other inshore stations to the north and south of the SZB outfall that were effectively outside of the influence of the SZB plume.

c. A statistically significantly greater abundance of herring and sprat were observed inshore of the Bank than offshore. There was no significant difference in the distribution of whiting across the Bank.

The sampling programme therefore confirmed previous observations that bass are attracted to warm water in winter. In particular, the abundance of bass at Sizewell was significantly different inshore and offshore of the Bank which would be expected based upon the significant difference in water temperatures near to the seabed at the two locations. The inshore sampling also demonstrated that the abundance of bass was significantly greater inside of the SZB plume than outside. These results provide confidence that the observed bass distribution was strongly associated with the presence of the SZB discharge plume and this has implications for the expected impingement rates for SZC which are discussed in Section 6.5.

4.2 Effect of the proposed SZC intake heads upon the expected impingement rate

SZC would be fitted with four low-velocity side-entry (LVSE) intake heads, mounted on the seabed approximately 3 km offshore. The proposed intake heads would be capped structures with the intake surfaces orthogonal to the direction of the tidal flows. The intakes are specifically designed to reduce the cross-sectional area available to intercept any fish being transported in the tidal flows. The reduction in cross-sectional area combined with the low intake velocity is predicted to substantially reduce the number of fish abstracted per cumec of abstracted seawater compared with SZB, which is fitted with conventional omnidirectional intake heads with intake velocities that exceed the ability of most fish to avoid being abstracted.

Following Environment Agency consultation on BEEMS Scientific Position Paper SPP099, modelling was refined to capture the full range of tidal directions. Modelling indicates that the SZC station will abstract 0.357per cumec of the fish abstracted by SZB, because of the intake head design. The methodology used to derive this impingement reduction factor is described in BEEMS Scientific Position Paper SPP099 v4 and is based upon the methodology proposed by the Environment Agency for impingement studies at Hinkley Point C.

The effect of the intake head design on the abstraction of organisms, is assessed by multiplying all predicted values for SZC (unmitigated), by the factor of **0.357**. The full CIMP impingement assessment process is illustrated in Figure 2.

4.3 SZB impingement data collection and collation

Impingement sampling at SZB was initiated in 2009 to provide information that could be used to predict the losses of the proposed SZC station. The sampling scheme consisted of sampling for 6 * 1-h samples in the daylight in addition to one * 18-h sample that was collected overnight. In each sample, the impinged material was sorted to species where possible, weighed and the fish fauna were measured. If subsampling was required, the data were raised to the individual sample first, before all 7 samples (six hourly, one overnight) were summed to give an estimate of the 24 hours of sampling. A total of 128 sampling visits were completed between February 2009 and March 2013. Each sample represented the estimated number and weight of fish that would have been impinged during the 24-h period, if the station was working at full capacity (i.e. 4 pumps in operation, which is not always the case during the year). The impingement data for 2009-2013 are described in four separate annual reports (BEEMS Technical Report TR120; TR196; TR215; TR270).

Sampling resumed in April 2014 and is ongoing. Prior to April 2014, the sampling methods were reviewed, particularly the need for the overnight sampling, to ensure that sampling was still appropriate to the data requirements. In general, the same sampling methods were adopted when data collection re-commenced. The only differences to sampling methodology were a change to the order in which the hourly and overnight samples were collected, the use of an Electronic Data Capture system (EDC) to improve the efficiency and reliability of data capture, and a change from measuring fish using Standard Length, to using Total Length. Sampling has continued since April 2014. Between 2014 and 2017, a further 77 samples were obtained, and these were similarly raised to represent 24 hours with the station pumping at full capacity. Details of the

sampling review, revised sampling methodology and data handling are given in BEEMS Technical Report TR339 v3.

Both data sets were brought together to provide a single data set of 205 sampling visits, each providing information on the number and weight (and number at length for fish), of individuals impinged by the SZB station in 24-h at full capacity (BEEMS Technical Report TR339 v3).

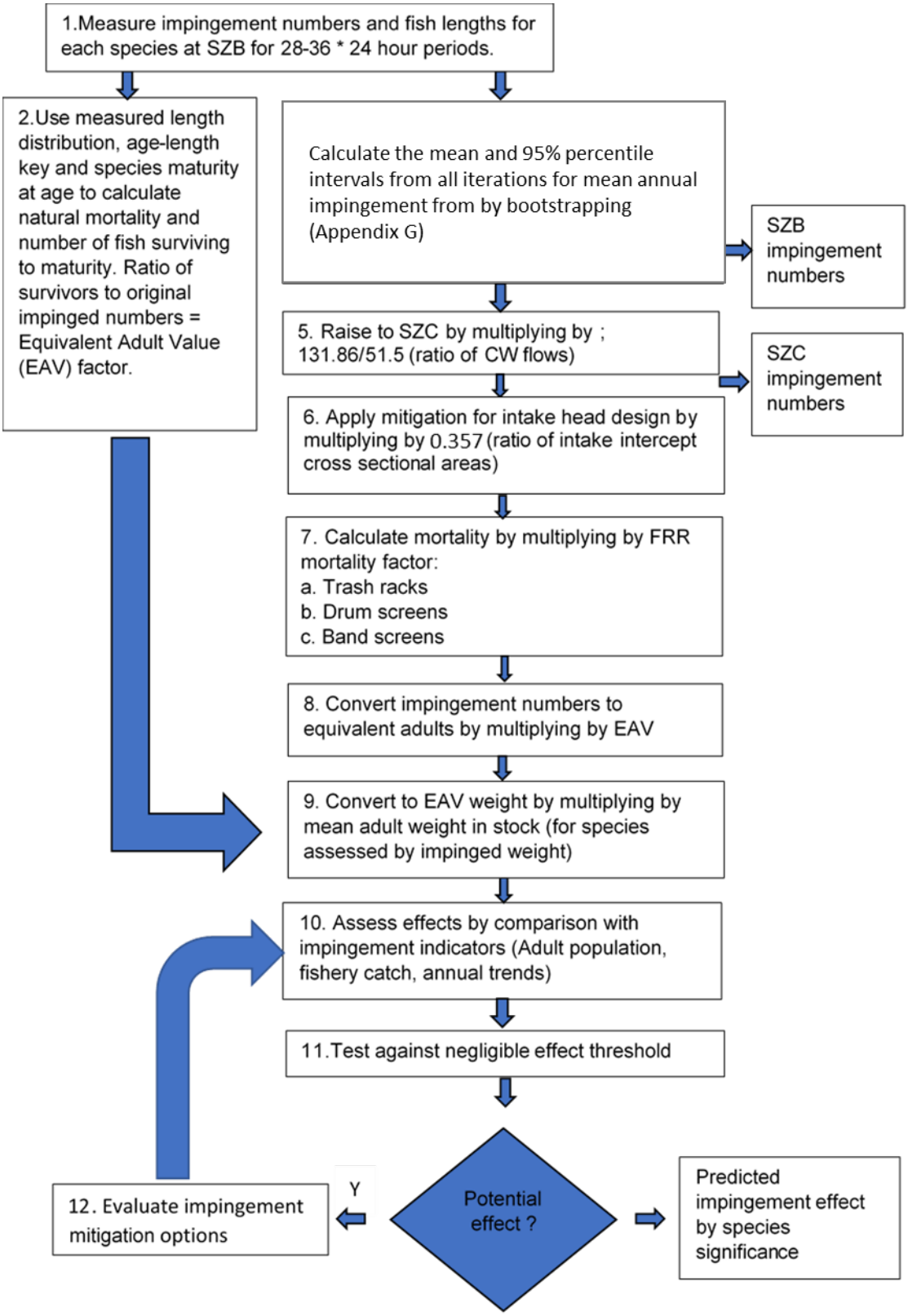


Figure 2 The SZC CIMP impingement assessment process.

4.4 Estimated annual mean, minimum and maximum losses by the SZB station

Estimates of impingement for Hinkley Point C were based on a CIMP conducted at HPB in 2009 – 2010. For that dataset, sampling was conducted over 40 visits throughout the year. Estimates of annual impingement were calculated by first summing all samples from each quarter of the year and raising the quarterly total by the ratio between the number of days in the quarter and the number of sampling visits. The quarterly totals were summed to give an annual impingement estimate. Variability was estimated by bootstrapping the dataset – randomly selecting a subset of visits from each quarter and repeatedly estimating the total annual impingement to give 95% confidence estimates around the mean value. This was possible because the number of samples available was evenly distributed thought the sampling year.

The SZB CIMP dataset is extensive, spanning 9 years and 205 samples. The modelling approach used at Hinkley Point could not directly be replicated for Sizewell. This is because the Sizewell data set incorporates multiple years of data and the distribution of samples over the sampling period is uneven, with no sampling in some quarters largely due to station outages. Hinkley Point B has dual reactors which means that even during outages there are rarely periods when the station is not abstracting seawater, and this allows a near continuous impingement record. Sizewell B has a single reactor and therefore during outages no seawater is abstracted and therefore the impingement record is discontinuous. In particular, very little sampling occurred in 2013 (sampling was suspended in March 2013), and there are large sampling gaps particularly in the later sampling years (e.g. quarter 1 in 2014 and 2015). This was largely due to outages when no sampling could take place (Table 6). Therefore, the Hinkley Point assessment methodology cannot be directly applied

Estimates of impingement in previous versions of this report were derived from model outputs for each species that provided a mean impingement estimate and estimated upper and lower boundaries. The lower and upper estimates applied in the model represented an interval of modelled daily values for each species multiplied by the number of days in the month and then limits for all months were added. This assumes that either the minimum monthly limit or maximum monthly limit was impinged on every day of the year. This is considered highly unlikely, and results in values that will substantially overestimate the variability in impingement. As such the upper and lower values were not true 95th percentiles and Cefas urged caution in their application. During stakeholder consultation on BEEMS Technical Report TR406 version 6, the Environment Agency expressed concern regarding the status and size of the lower and upper estimates of impingement predictions for SZC. The Environment Agency also asked if it was possible, given the gaps in the dataset, to derive impingement estimates using the bootstrapping method used at Hinkley Point (BEEMS Technical Report TR456). As a result of these comments, Cefas undertook a statistical review of the model and potential alternative methodologies.

Following statistical review, a bootstrapping approach, akin to the methods employed at Hinkley Point, with an additional step to account for periods when sampling was not possible due to station outages was proposed (BEEMS Technical Report TR339 version 3). The bootstrapping method randomly reallocates samples from adjacent years to account for sampling gaps. The bootstrap outputs for each species provide the mean numbers impinged by quarter for Sizewell B, which is converted to percentage impingement by quarter to show the seasonal patterns and the mean annual impingement with 95th percentile confidence intervals based on 5,000 bootstrap iterations.

The bootstrapping approach is considered the most robust for calculating mean annual impingement estimates and allowing 95th percentile confidence intervals to be clearly defined. The bootstrapping approach has been applied to calculate annual impingement numbers and 95th percentile confidence intervals for Sizewell B. The development of the model and bootstrap approaches, the advantages, and limitations along with differences in impingement predictions between the approaches for Sizewell B is provided in BEEMS Technical Report TR339 v3.

A flow chart detailing the methodology for calculating Sizewell B impingement is provided in Appendix G.

Table 4 Summary of the number of sampling visits to SZB completed between 2009 and 2017

Month	2009	2010	2011	2012	2013	2014	2015	2016	2017	Total
January		2	5	1	2			2	3	15
February	3	4	2	3	3			3	2	20
March	4	2	2	1	2			2	2	15
Q1 total	**7**	**8**	**9**	**5**	**7**	**0**	**0**	**7**	**7**	**50**
April	3	3	2	2		3	1		2	16
May	3	3	2	3		2	3		1	17
June	3		2	2		2	2	2	3	16
Q2 total	**9**	**6**	**6**	**7**	**0**	**7**	**6**	**2**	**6**	**49**
July	4	2	2	2		3	3	2	3	21
August	4	2	3	3		2	2	2	3	21
September	1	3		2		1	2	3	1	13
Q3 total	**9**	**7**	**5**	**7**	**0**	**6**	**7**	**7**	**7**	**55**
October		4	3	3			3	2	1	16
November	7	3	3	2			2	3		20
December	4	3	2	2			2	2		15
Q4 total	**11**	**10**	**8**	**7**	**0**	**0**	**7**	**7**	**1**	**51**
Total	**36**	**31**	**28**	**26**	**7**	**13**	**20**	**23**	**21**	**205**

4.5 Predicted annual mean, minimum and maximum losses by the SZC station

The predicted (unmitigated) losses that will be incurred by the SZC station were calculated by simply raising the mean, lower and upper 95th percentile predictions for SZB by the ratio of the two pumping capacities (i.e. 131.86/51.5) for each species.

4.6 Effect of the intake head design

The predicted effect of the proposed LVSE intake head design on the abstraction of organisms, was assessed by first multiplying all predicted values for SZC (unmitigated), by the factor of **0.357**. (Section 4.2)

4.7 FRR system mortality

The proposed drum screen mesh size for SZC is 10 mm allowing a direct comparison with the current mesh size employed at SZB. In the best practice guide for screening for intakes and outfalls Environment Agency (2005) recommend *"mesh size should be as small as is practical, and of no more than 6 mm aperture"*. However, Environment Agency (2010) acknowledge that at coastal sites a 6 mm mesh may lead to the risk of ctenophore blockage during summer months. Gelatinous ctenophores would more readily distort under drum screen conditions and squeeze through a 10 mm mesh screen (Environment Agency, 2010). After a consideration of clogging risk EDF Energy have proposed that SZC's fine filtration systems should have 10 mm mesh. The trash rack bar spacing might then be relaxed to 75 mm bar without exceeding design criteria for the drum screens.

4.7.1 Trash rack mortality

Located immediately before the drum and band screens, will be a series of trash racks, designed to protect the screens from debris but which will also prevent the passage of large fish. The racks, which will have a

bar spacing of 75 mm can be raised for cleaning and any material that cannot pass through the bars will be sent to the debris recovery building (HCB). The debris recovery building has another trash rack with 200 mm bar spacing and fish that pass through this secondary trash rack will be returned to sea via the FRR tunnel. Any that remain (i.e. cannot pass through 200 mm) will go to waste. It is assumed (subject to review of the SZC design), that all organisms that cannot pass through the primary trash rack (75 mm bar spacing) will suffer 100 % mortality, even if they would then pass the secondary trash.

Each of the 24 key species has a different body shape and maximum size, and therefore the size at which an individual will be able to pass will depend on its species. The proportion of the total number of that species that will not pass will depend on its size distribution in the cooling water systems.

4.7.1.1 Calculation of annual length distributions

The final impingement dataset also included the number at length of the key species, by sampling visit. These were used to provide an annually raised length distribution for each of the key species. The number of individuals in the annually raised length distribution equalled the number of individuals estimated to have been impinged by SZC per year.

Several steps were required to account for seasonal growth in the length distributions:

- First, the samples were grouped by month, and the numbers at length were summed. The total numbers at length were then divided by the number of samples in that month. This provided a standardised length distribution for each month separately.
- Next, the 12 monthly length distributions were summed, and their total numbers at length were divided by 12 to give a single standardised length distribution representing one year.
- Finally, the numbers at length were raised to an annual total by multiplying by (number of individuals impinged by SZC/number of individuals in the standardised length distribution).

(These annually raised length distributions were also used in the calculation of the EAVs - see BEEMS Technical Report TR383).

4.7.1.2 Calculation of the cut-off for passing through the 75 mm bar spacing.

For each species, its ability to pass through the 75 mm trash rack will depend on its width, which can be calculated from its length. However, HPB already has trash racks fitted with a 75 mm bar spacing, and length data show that for some species (e.g. cod and bass), even fish with a calculated width > 75 mm can pass through to the drum screens (BEEMS Technical Report TR456). This indicates that the passage of a fish through the trash racks may not simply be limited to its calculated width. This has been accounted for in calculations on the proportion of a species that will or will not pass the 75 mm bar spacing. Species were grouped depending on how their passage through the trash racks was assessed:

- Group 1. For some species (e.g. sand gobies), expert judgement was used to conclude that all individuals would pass, irrespective of size (i.e. proportion retained by the trash racks = 0).
- Group 2. For all other species, the width of the largest observed individual was calculated. Group 2 species are those whose calculated width of the largest observed individual was ≤ 75 mm (e.g. river lamprey *Lampetra fluviatilis*). It was assumed that no individuals will be retained by the trash racks.
- Group 3. For almost all the remaining species the calculated width of the largest SZB fish was ≥ 75 mm, indicating that a proportion of individuals will not pass. Most of these species were recorded during impingement sampling at HPB, which is already fitted with 75 mm trash racks. For this report, the length of the largest observed individual to pass through the 75 mm HPB trash rack was used as the maximum size that would also pass through the proposed 75 mm SZC trash rack. The proportion of a species that would be retained on the trash rack was calculated from the annually raised SZB length distributions of that species (Section 4.7.1.1).

- Group 4. For two species (sea trout and Allis shad), the calculated width of the largest SZB fish was ≥ 75 mm, indicating that a proportion of individuals will not pass through, but the species was not recorded at HPB. The approach used for Group 3 fish could not be used. In this case, the length of a fish with a width of 75 mm was calculated as the maximum size that would pass through the trash racks. Given that for many species, individuals of a calculated width were able to pass through the trash racks, this is likely a conservative approach.

For each species, the proportion of fish that would be retained by the 75 mm trash rack was applied to the number of fish that passed through the intake head, giving the number of fish lost to the trash racks (Table 5). Fish passing through the trash racks will go on to encounter the drum or band screens.

Table 5 Proportion of fish, by species that will not pass through the 75 mm wide trash racks, and the length size used for the cut-off

	Calculation type	Length of largest SZB fish (mm)	Calculated width of largest SZB fish (mm)	largest observed HPB fish	Calculated length at 75 mm width (cm)	Proportion not passing trash rack	Comment
Sprat	Group 2	260	53			0.000	All observed fish will pass
Herring	Group 3	445	94	224		0.715	Use largest observed HPB fish length (impingement datasets)
Whiting	Group 2	525	48			0.000	All observed fish will pass
Seabass	Group 3	850	232	657		0.0002	Use HPB largest observed individual (TR456)
Sand goby	Group 1	100	NA	74		0.000	All observed fish will pass
Dover sole	Group 3	440	124	449		0.000	Use HPB largest observed individual (TR456)
Dab	Group 3	380	160	139		0.419	Use largest observed HPB fish length (impingement datasets)
Anchovy	Group 1	230	NA	169		0.000	All observed fish will pass
Thin-lipped grey mullet	Group 3	425	91	524		0.000	Use largest observed HPB fish length (impingement datasets)
Flounder	Group 3	425	119	339		0.031	Use largest observed HPB fish length (impingement datasets)
Plaice	Group 3	355	150	382		0.000	Use HPB largest observed individual (TR456)
Smelt	Group 1	250	NA			0.000	All observed fish will pass
Cod	Group 3	895	133	709		0.006	Use HPB largest observed individual (TR456)
Thornback ray	Group 3	765	503	952		0.000	Use HPB largest observed individual (TR456)
River lamprey	Group 2	405	25			0.000	All observed fish will pass
European eel	Group 2	895	56			0.000	All observed fish will pass
Twaite shad	Group 3	490	103	299		0.883	Use largest observed HPB fish length (impingement datasets)
Horse mackerel	Group 2	355	71			0.000	All observed fish will pass
Mackerel	Group 2	405	69			0.000	All observed fish will pass
Tope	Group 2	625	63			0.000	All observed fish will pass
Sea trout	Group 4	530	121	not at HPB	33	1.000	Calculate length at 75 mm width
Allis shad	Group 4	610	128	not at HPB	36	1.000	Calculate length at 75 mm width
Sea lamprey	Group 2	795	50	807		0.000	all observed fish will pass

4.7.2 FRR survival

Well-designed FRR systems have been reported to achieve 80−100 % survival rates for robust species such as eel *Anguilla anguilla* , plaice *Pleuronectes platessa* and flounder *Platichthys flesus*, and moderate rates (~50−60%) for demersal species such as the robust gadoids (e.g. cod). However, survival rates for delicate pelagic species such as herring, sprat and shad (twaite shad *Allosa fallax* and Allis shad *A. alosa*) are usually low (<10%, Environment Agency, 2005). The proposed FRR system for SZC at both the band and drum screens has been designed to achieve high rates of survival for European eels (eel, *A. anguilla*) and lamprey (river lamprey *Lampetra fluviatilis* and sea lamprey *Petromyzon marinus*), and it is expected that survival rates for other epibenthic (flatfish including rays) and demersal species will also be higher than achieved in older designs. For the purpose of this study the conservative FRR recovery rates given in the Environment Agency science report (2005) are used as the basis for FRR survival rates.

However, the estimates of FRR survival have been modified to account for the SZC location and station design. The SZC FRR system will discharge inside the Sizewell-Dunwich Bank and there is potential for a few fish discharged from the SZC FRR to be subsequently taken up by the SZB intake. Modelling confirms that this risk is very minor (<1%) at the proposed SZC FRR outfall locations (BEEMS Technical Report TR333), but it is conservatively assumed that any such fish will suffer 100 % mortality during their passage through SZB. The SZC mortality is increased to compensate for this effect (Table 6).

As well as using drum screens for the main cooling water flow, SZC will be equipped with band screens to protect the essential and auxiliary cooling water systems. Due to their safety role, the band screens must be seismically qualified and capable of surviving an aircraft impact. The normal operating mode of such band screens is to be stationary and to only rotate intermittently at 6 hourly intervals unless significant clogging occurs. It is possible to fit an FRR system to the band screens, but this would have little to no purpose if the screens only rotated every 6 hours. It would, however, serve a purpose if the screens rotated continuously. The band screen manufacturer considers that the screens could be operated continuously at a ‘creep’ rotation speed of 0.5 metres per minute; any faster would have unacceptable implications for the operational life and maintenance of the safety-classified band screen motor and chains. For HPC, it was assumed that with the size of the band screens, at a rotation speed of 0.5 m min^{-1}, the fish retention time in the band screen fish buckets would be approximately 33 minutes at Mean Sea Level (MSL) and 50 minutes at the Lowest Astronomical Tide (LAT). As a conservative assumption, it was considered that demersal fish would not survive this time in the fish buckets. However, with a fish-friendly design ensuring they cannot fall out of the buckets during the predicted retention time, robust epibenthic species such as flatfish, eels and lamprey are expected to survive. The tidal range at Sizewell is lower than at Hinkley Point and the size of the SZC drum and band screens will therefore be smaller leading to reduced rotation time and reduced fish retention time in the fish buckets, which will enhance survival. In this assessment we have assumed, as a conservative estimate, the same FRR survival estimates that were applied to the HPC band screens; i.e. that the fish survival percentages for epibenthic species will be the same for drum screen and band screen FRR systems, but that the survival of demersal species in the band screens will be 0 % (Table 6). This conservative assumption may be revisited when further information is available on detailed station design.

To calculate drum screen and band screen losses, the number of fish remaining after passage through the trash racks was apportioned into those that will encounter the drum screens and those that will encounter the band screens based on the proportion of water flowing through each (drum screens = 91 %; band screens = 9 % of the cooling water). This is considered a reasonable approach as the approach velocity is identical for all of the SZC drum and band screens. The proportion of numbers of fish that would be retained by the drum and band screens was calculated for each species separately.

4.7.3 Other factors which could potentially affect FRR survival rates

In principle two issues that have not yet been discussed in this report could affect the survivability of impinged fish in the SZC FRR system:

a. Overloading of the SZC FRR system by dead fish that do not survive the recovery process (mostly pelagic fish such as sprat and herring) causing fatal oxygen depletion in parts of the system where fish densities are the highest (the drum screen buckets)

b. Clogging of the SZC FRR system by ctenophore blooms in summer

The SZC seawater filtration system is designed to protect the CW condensers and heat exchangers from blockage from marine organisms. The system was designed based upon operational experience at EDF Energy coastal power stations without impingement mitigation technology and as such there was no assumption in the design of the filtration system for reductions in fish impingement due to the SZC Low Velocity Side Entry (LVSE) intake heads. The SZC filtration system has been designed to have considerable capacity to respond adaptively to extreme fish densities at the drum and band screens by increasing the rotation rate of the screens such that organisms are returned to sea via the FRR system at a faster rate. For example, the drum screen rotation rate can be increased from the normal 2.5 m min^{-1} to 10 and then 20m min^{-1} in response to different screen loadings i.e. the system can provide an 8-fold increase in filtration capacity under extreme conditions. The band screens are even more adaptable and can provide a 20-fold increase in filtration capacity by increasing the rotation rate from 100 minutes per rotation to 20 and then 5 minutes.

The main clogging risk at Sizewell are from winter sprat inundations and particularly from summer ctenophore blooms. As noted in Section 4.7, after consideration of the clogging risk from ctenophores, EDF Energy is proposing to fit 10mm mesh filters at SZC. Operational experience at Sizewell B has shown that such a mesh size has avoided problems of clogging from either ctenophores or sprat. In terms of relative risk, SZC would be at much lower risk than SZB from sprat inundations due to the use of LVSE intake heads at SZC which it has been calculated will reduce sprat impingement per cumec by a factor of 0.357 (Section 4.2). A study for HPC (BEEMS Technical Report TR493) determined that there was a negligible risk to fish survival in the HPC FRR system due to dead impinged fish (overwhelmingly sprat). The rotation path lengths of the SZC drum and band screens will be smaller than those deployed at HPC due to the smaller tidal range at Sizewell and therefore fish residence times in the fish buckets will be shorter. The risk to fish survival from dead fish in the fish buckets is, therefore, expected to be lower at SZC than at HPC and therefore also negligible.

Ctenophores are present for most of the year at Sizewell but only occur in dense blooms in summer (typically in June and July). The proposal to use a 10mm mesh means that the majority of entrapped ctenophores will be entrained rather than impinged. There have not been any shutdowns at Sizewell B which also uses 10mm mesh filtration due to gelatinous species. With negligible clogging risk there remains the question of whether the loading of ctenophores would reduce fish survivability in the SZC FRR system. As can be seen from Figure 3 the weight of fish impinged during summer ctenophore blooms is extremely small. Fish weight in the FRR systems typically peaks during winter and spring whilst ctenophore biomass peaks in summer. Much smaller peaks in ctenophore abundance occur at other times of the year but the additional ctenophore biomass is much smaller than the peak fish biomass that the FRR system can handle with negligible risk to fish survival.

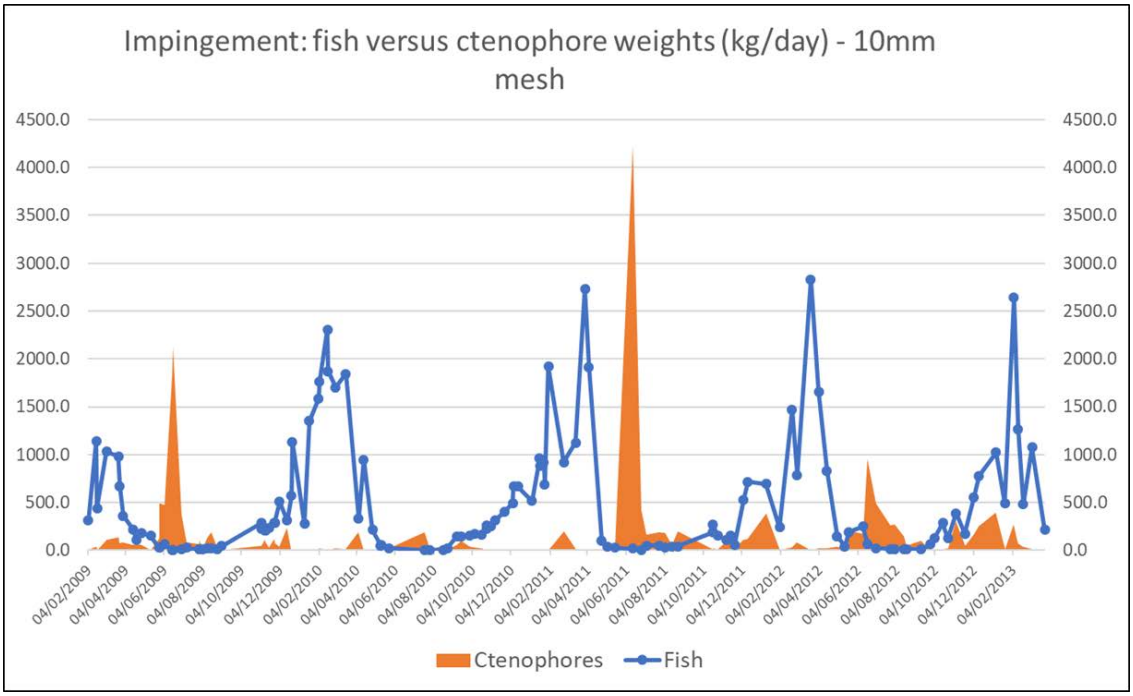


Figure 3 Comparison between estimated daily impingement weights for ctenophores and fish from SZB CIMP data.

Clogging of the SZC FRR system and the consequent risk to fish survival of recovered fish is, therefore, considered to be negligible.

4.7.4 Values used for FRR survival in the SZC impingement assessment

The predicted total mortality suffered by each species passing through the SZC cooling water systems, considering FRR survival, was calculated as:

Total mortality = Trash rack losses + band screen losses + drum screen losses

The results are shown in Table 6.

Table 6 Predicted FRR mortality by species through the SZC trash rack, drum and band screens and percentage of the total flow.

Species	Proportion lost			Species group
	Trash rack (100%)	Drum (0.91)	Band (0.09)	
Sprat	0.0000	1.000	1.000	pelagic
Herring	0.7150	1.000	1.000	pelagic
Whiting	0.0000	0.506	1.000	demersal
Bass	0.0002	0.506	1.000	demersal
Sand goby	0.0000	0.206	0.206	epibenthic
Sole	0.0000	0.206	0.206	epibenthic
Dab	0.4190	0.206	0.206	epibenthic
Anchovy	0.0000	1.000	1.000	pelagic
Thin-lipped grey mullet	0.0000	0.506	1.000	demersal
Flounder	0.0310	0.206	0.206	epibenthic
Plaice	0.0000	0.206	0.206	epibenthic
Smelt	0.0000	1.000	1.000	pelagic
Cod	0.0060	0.560	1.000	demersal
Thornback ray	0.0000	0.206	0.206	epibenthic
River lamprey	0.0000	0.206	0.206	epibenthic
Eel	0.0000	0.206	0.206	epibenthic
Twaite shad	0.8830	1.000	1.000	pelagic
Horse mackerel	0.0000	1.000	1.000	pelagic
Mackerel	0.0000	1.000	1.000	pelagic
Tope	0.0000	0.206	0.206	epibenthic
Sea trout	1.0000	*0.506*	*1.000*	demersal
Allis shad	1.0000	*1.000*	*1.000*	pelagic
Sea lamprey	0.0000	0.206	0.206	epibenthic

4.8 EAV conversion factors

Equivalent Adult Values (EAVs) are used to adjust the number of lost juveniles to a corresponding number of lost adults. This adjustment is required because juveniles suffer higher natural mortality when compared with adults of the same species, and the loss of one juvenile does not result in the loss of one adult. Conversion of the numbers impinged to the equivalent number of adults is a simple matter of multiplying the former (the total impingement mortality) with the appropriate EAV value for each species. The EAV values used were calculated using a method developed as part of the BEEMS programme (BEEMS Technical Report TR383). The reliability of the EAV calculation method has been extensively evaluated for HPC studies in BEEMS Technical Report TR456 and the calculation method used at Sizewell is the same as that used for HPC, in particular the natural mortality corrections applied are the same. These calculations use data on the size distribution of the impinged fish along with information on size at age and size at maturity. Since the model outputs giving rise to the SZB estimates and SZC predictions were based on the combined 2009 – 2017 data, only a single EAV could be calculated for each species (Table 7) that used the size distribution from the whole sampling period (see Section 4.7.1.1 for details on the calculation of the size distribution).

All methods for calculating EAVs are subject to biological uncertainties. The method used for this assessment and described in BEEMS Technical Report TR383 is based upon the best available science at the time of writing and uses peer reviewed input parameters appropriate to the local populations at Sizewell.

A key advantage of this method is that the approach represents a method for calculating EAV that can be widely applied to many species and requires data that are more readily available than the method described by Turnpenny (1989) that has been used for some previous power station effects analyses. Turnpenny (1989) defined the EAV as “the average lifetime fecundity of an adult that has just reached maturity which is required to replace that juvenile”. To calculate the EAV curve or construct the associated life table for a given species, a variety of parameters are required, such as the average fecundity of each female age class, the number of age classes in the population and the average fecundity of mature females of the final age class. Many of these parameters are difficult to reliably estimate, even to orders of magnitude. Consequently, Turnpenny (1989) only provided EAV life tables/curves for six commercial species (cod, whiting, plaice, Dover sole, dab and herring) and the accuracy of these values is uncertain.

4.8.1 Discussion of the Spawner Production Foregone method for calculating EAVs

For Hinkley Point C assessment purposes an alternative method for calculating EAVs has been suggested by the Environment Agency using the ‘Spawner Production Foregone’ (SPF) method of deriving equivalent adult value factors (Environment Agency 2019). Spawner Production Foregone is a branch of fisheries science that was developed to determine how much spawning capacity could be taken per year by fishing without affecting population sustainability. The science was adapted for use in the USA to formulate a compensation equation for power station entrainment losses. To the knowledge of the authors of this report, SPF EAVs have not been used in those studies to produce quantitative impingement effects calculations using SSBs or landings data.

The SPF approach calculates the total loss to the population as the sum of the equivalent adults lost from the spawning population in a year plus the future spawning potential of lost fish that would otherwise have reproduced over a series of years until mortality. The reason why this approach is not applied to the management of fishing is, therefore, clear as each fish caught would have to be revalued to account for its future production potential using an accounting procedure with uncertain and time varying input parameters. This inflated catch estimate would then have to be compared with an inflated version of the stock estimate as ICES stock estimates also do not account for potential future production. Managing fishing quotas whereby one landed fish became one real fish plus ‘x’ future fish (EAV > 1) in one season or another value in another season would be seen by fishers as an illogical management regime. Furthermore, only adjusting the catch and not the stock for future potential would lead to unreasonable and incorrect assumptions about the sustainability of the catch. Simply put it is incorrect to compare catches adjusted for SPF with SSB estimates calculated in the internationally agreed manner that do not include the effects of future spawning potential.

The SPF methodology assumes that long lived fish could spawn in multiple future years after they reached maturity and therefore some fish populations could have an EAV value of greater than 1, whereas the method described in BEEMS Technical Report TR383 has a maximum EAV value of 1 (i.e. a fish caught in SZC could never be treated as being more than 1 adult). A major assumption of the Environment Agency methodology is that many long-lived species would survive to spawn in multiple future years after reaching maturity as they would only experience natural mortality due to predation and disease. However, for many species this is an invalid assumption as after reaching maturity (often before reaching maturity), the fish become at risk of fishing mortality which is frequently greater than natural mortality such that few fish live to spawn on multiple occasions. This effect is particularly pronounced for species such as cod and bass. Since fishing mortality can and does occur before the age of maturity, the method described in BEEMS Technical Report TR383 for calculating EAVs could also overestimate the EAV for populations with large numbers of fish at or near maturity. However, the potential error in not accounting for fishing mortality is much less than with the SPF method as the TR383 method explicitly does not account for multiple spawning potential and secondly most populations at Sizewell are of juveniles for which the potential overestimation does not occur (but the overestimation does occur in the SPF methodology).

The most significant problem with the proposed SPF method is that ICES fisheries assessment models do not account for production foregone as they simply reflect the Spawning Stock Biomass (SSB) of fish alive at a point in time. So, the population effects derived using SPF EAVs are not comparable with ICES SSBs, particularly as the proposed SPF method does not account for fishing mortality (something that ICES stock

assessments fully account for). A the simplest level the TR383 method computes the value of a lost mature adult as 1 fish lost from the Spawning Stock Biomass, it does not try to account for the potential future production from that fish if it lived to spawn again. As such it is fully compatible with the methods used to assess the effects of fishing and the derivation of SSB by ICES.

All fisheries management processes use accounting techniques to evaluate effects on fish stocks. It is not the case that by not accounting for future spawning potential that standard fisheries management calculations are wrong, it is simply that the accounting framework does not require that input. The standard assessment methods could be recast to include future production potential to both losses and the stock but there is no benefit in doing so (the final result is essentially the same, but the additional computations introduce new uncertainties for no practical benefit).

Comparison with ICES stock assessments is the standard method of assessing the effect of fishing or power station entrapment and the ability to undertake reliable, quantitative comparisons of power station mortality against SSB is fundamental to the assessment of SZC effects. To lose the ability to use these comparators would make it virtually impossible to make assessments of power station abstraction effects.

BEEMS Scientific Position Paper SPP102 evaluates the SPF EAV methodology described in Environment Agency 2019 using seabass and an example. The report concludes that there is no benefit from the use of SPF EAVs in impingement assessment. If only the SZC catch is adjusted for future spawning potential, the effects assessment will produce erroneous conclusions that will overestimate the real level of SZC effects.

4.8.2 Potential detail change to the EAV calculation methodology in TR383

The effect on EAV results if the calculation stopped at 50% maturity and not the 100% maturity used in TR383 has been queried. The rationale for this question is that the Turnpenny (1989) method is based upon ages at 50% maturity as are some stock assessments, and a revised methodology could be more conservative.

It was considered that such a change probably would not materially affect the predicted SZC losses but Cefas examined whether this initial view was justified. The potential effect of such a change was checked for three different types of fish taxa with different life cycle parameters (bass, cod, herring, sprat) and calculated EAVs increased by 3.6%, 11.5%, 4.3%, and 3.3% resulting in consequential changes in predicted entrapment losses by the same percentage. For example, the EAV for bass would change from 0.224 to 0.232 resulting in an increase in unmitigated impingement losses from 1.63% SSB to 1.69% SSB and cod losses would increase from 0.0035% SSB to 0.0039% SSB. Such changes are considered immaterial to the SZC assessment and have not, therefore, been included in this assessment report.

Section 7 considers the sensitivity of the assessment to variation in the assessment parameters including EAVs.

4.9 Conversion from EAV numbers to equivalent weight

As most stock information (particularly for commercially-exploited species) is given as weight rather than number of fish, the EAV numbers of impingement losses were converted to EAV weights (t), using values of mean weight per species (Table 7). The preferred method of calculation would have been to calculate the mean weight of adults only for all species. However, the final method used depended on the data available for that species.

For some species where international stock assessments are carried out, the datasets contain the mean weight of fish in each age group, the catch numbers in that age group and the proportion of fish mature in each age group. In this case, the numbers of mature fish caught in each age group was calculated by multiplying the number caught in the age group by the proportion that were mature. Multiplication of the number mature in the age group by the mean weight of an individual in the age group provided the total

weight of fish in the age group. The weight of all fish in the age groups was summed and then divided by the total number of mature fish to give a mean weight of each mature fish.

However, this level of data is not available for all species, and alternative methods were used for these species. For salmon and sea trout for example, the Environment Agency catch statistics reports (Table 8) give the average weight of individuals of each species caught in the fishing year, based on catch returns, and these values were used. For lamprey, the mean weight of fish entering the Ouse was used, as it was assumed that these fish are entering the estuary to spawn and are therefore mature.

In each case, judgement was based on the data available.

Table 7 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights at SZC. See BEEMS Technical Report TR383 for full EAV calculations. Species where an EAV could not be calculated are highlighted in yellow and the impingement losses are overestimates

Species	EAV	Mean weight per individual (kg)	Data source for mean weight
Sprat	0.751	0.011	ICES catch weight at age data
Herring	0.715	0.189	ICES catch weight at age data
Whiting	0.356	0.286	ICES catch weight at age data
Seabass	0.224	1.531	ICES catch weight at age data
Sand goby	1.000	0.002	Mean size of gobies impinged at SZB
Sole	0.213	0.214	ICES catch weight at age data
Dab	0.445	0.041	ICES catch weight at age data
Anchovy	0.974	0.021	Mean weight of anchovy the Bay of Biscay catches
Thin-lipped grey mullet	0.083	0.520	Estimated from weight at 75 % maturity (age 3.5)
Flounder	0.462	0.082	Estimated from weight at 21 cm (fully mature - ICES)
Plaice	0.345	0.246	ICES catch weight at age data
Smelt	0.761	0.017	Estimated from length at maturity (knife-edge)
Cod	0.359	2.602	ICES catch weight at age data
Thornback ray	0.193	3.193	Mean length of market sampled fish in southern N. Sea
River lamprey	1.000	0.079	Mean weight of lampreys entering Ouse fishery in 2018 – regulator comment to Version 1 of this report
European eel	1.000	0.329	Mean of a mature female (568.9 g) and male (89.9 g) adult (Aprahamian, 1988), assuming a 50:50 sex ratio
Twaite shad	1.000	0.313	Calculated from adult size of 32 cm
Horse mackerel	1.000	0.140	Mean weight in the catch
Mackerel	1.000	0.319	ICES catch weight at age data
Tope	1.000	6.900	Modal length of fish in observer catches = 100 cm
Sea trout	1.000	1.734	from EA catch statistics
Allis shad	1.000	0.572	Calculated from adult size of 40 cm
Sea lamprey	1.000	1.212	Weight of single individual (length = 79 cm) impinged at SZB
Salmon	1.000	3.684	from EA catch statistics

Note: The use of an EAV of 1 results in an overestimate of impingement effects for most species e.g. it is known that the impinged eels at Sizewell were yellow eels that will suffer additional natural mortality before reaching the adult silver eel stage (Section 6.6.3). Similarly, not all of the impinged twaite shad were adults.

4.10 Evaluating the effect of SZC impingement losses – comparison with ICES stock estimates

Fish mortality due to impingement at SZC can be considered as a form of fish harvesting.

Fish stocks in the Northeast Atlantic are managed partly through the EU Common Fisheries Policy (CFP), whose objective is to maintain or rebuild fish stocks to levels that can produce their maximum sustainable yield (MSY). The International Council for the Exploration of the Sea (ICES) advises public authorities with competence for marine management including the European Commission (EC). ICES was established in 1902 and its advice integrates the work of approximately 5,000 scientists from over 700 marine institutes in the organisation's 20 member countries and beyond.

ICES' advice is produced through a process which is set up to ensure that the advice is based on the best available science and data, is considered legitimate by both authorities and stakeholders and is relevant and operational in relation to the policy in question.

The basis for the advice is the compilation of relevant data and analysis by experts in the field, normally through an expert group which includes core researchers in the field. This analysis is peer reviewed by scientists who have not been involved in the expert group and have no direct interest in the matter. To support the stock-by-stock management system, ICES provides advice on fishing opportunities and stock status for individual stocks.

For many species, annual analytical assessments are carried out that utilise information on life history, fishing effort and catches to assess the size of the stock, in particular the spawning stock biomass (SSB). To undertake an assessment, ICES scientists must evaluate information on the life history and fishery characteristics of a stock to determine the most scientifically appropriate geographical area in which to assess the stock (the stock unit). Stock assessments are carried out by Expert Groups (also known as Working Groups), each of which is responsible for a specified number of species and stocks. The outcomes of the assessments are released as official ICES advice.

Wherever possible, in cases where a full analytical assessment is available for a species impinged at Sizewell, and the SSB has been estimated, the SZC predicted impingement losses were compared with the ICES estimated SSB for the stock area as these estimates provide the most robust peer reviewed scientific evidence. As the annually raised impingement numbers were based on data collected in 2009 – 2017, comparisons were made with the average SSB values over this time.

This is the preferred measure for determining the effects of fish impingement (and entrainment) losses and this is how the much larger environmental effects of fishing are internationally assessed and managed. It must be emphasised that the comparison with the SSB in the assessment year(s) conducted in this report are not a full fisheries population assessment, and in stocks where the population biomass is heavily dependent upon new recruits which suffer a high rate of natural mortality this simple measure can provide a misleading overestimate of impingement effects. However, a full population assessment is both unnecessary and disproportionately difficult to undertake for species where impingement effects are negligible. If the predicted effects of impingement on a particular species were above the precautionary 1% negligible effects threshold used in this report (Section 5) a full population assessment is one of the steps that could be considered to reduce uncertainties and to determine if there was in fact any risk to the sustainability of the population.

4.10.1 Are ICES stock units appropriate for assessing the effects of SZC on fish populations?

To undertake their stock assessments ICES' scientists have identified biological stock areas that describe the distribution of a stock. These may be different from the areas defined by the EU, for example, for the

management of fishing quotas and technical measures. Identification of appropriate stock boundaries has been a central theme of ICES' coordinated effort since its formation in 1902 and major advances in understanding have, and continue to be, made.

SZC stakeholders have queried the appropriateness of some of the existing stock units, particularly for bass which has one of the largest stock units of the key fish species included in the SZC effects assessment. In particular, they have queried whether the stock areas being used for the assessment of impacts to certain species consider the impact to local sub-populations given that several papers (including papers produced by ICES) provide evidence of sub-populations and more complex heterogeneous population structures.

There is extensive literature focused on specific aspects of fish migratory behaviour and this information is periodically reviewed by the appropriate ICES Working Groups. The questions then is; do the specific behaviours described in the research papers change the weight of evidence used to assign stock units for management purposes? This section focuses on how ICES determines stock identity for fisheries management purposes.

As a result of decades of research, it is clear that the population structures of marine species fall along a continuum from panmictic (e.g. European eel, *Anguilla anguilla*) to numerous distinct sub-populations (e.g. North Sea herring, *Clupea harengus*) with the majority of species exhibiting complex structure. In the open sea, the sub-populations of many species mix to a considerable extent; especially during summer feeding and on nursery areas, with harvesting affecting multiple components of the overall population simultaneously. Spawning areas have been delineated for many fish species, but determination of the amount and timing of mixing is much more complex. Creation of appropriate stock units to manage harvesting must consider all these issues and it is recognised that in so doing harvesting may not always be optimal.

The current ICES stock boundaries are the result of decades of research and are inevitably compromises that try to embrace known uncertainties. Stock unit boundaries are not changed without careful weighing of the evidence for the need and the value to be gained from a change. However, stock boundaries are subject to periodic review and do change as a result of acquired evidence. For example,in the case of North Sea sandeel, where understanding of the vulnerability of largely sedentary post-settlement sub-populations led to a reorganisation of the original four stock units into seven units; with some stock units becoming larger and some smaller. For some species it is known that there is strong site fidelity during parts of the species lifecycle and/or seasonally, and some researchers cite this evidence to justify potential changes to stock units. However, it can be the case that there is considerable mixing of the sub-populations and an appraisal of all of the evidence leads to a conclusion that there is no evidence that a change could lead to better assessment and management advice. In such cases, the risks of realignment of stock boundaries are not warranted.

North Sea herring is one such example where the population is known to consist of several different sub-populations that have discrete well identified spawning areas but the population is managed as a single stock because of the degree of mixing that takes place. However, despite the use of a single stock unit, when the population collapsed in the 1970s the most at-risk spawning area was subsequently protected by limiting fishing effort in that location during the spawning season; i.e. adaptive management measures can be, and are, used to protect sub-stocks even within a single stock unit.

Bass is a species where appropriateness of ICES stock units has been queried, so management of this species is specifically described in Section 4.10.2.

4.10.2 Scientific status of the ICES Bass stock unit and the consequent SSB estimate

The ICES stock unit for bass is spatially very large and during consultation with stakeholders the accuracy of this spatial extent has been queried particularly in relation to some tagging studies that have recommended amendments to stock areas. This section provides a summary of the latest ICES' advice and the aim of current scientific studies.

Summarising the history of previous tagging studies, the ICES Benchmark Workshop on Sea Bass stated (ICES 2018e):

"*The sea bass inhabiting the Atlantic Ocean show a remarkable homogenous genetic structure although homing based on mark–recapture data suggests some level of population structure. Off the Strait of Gibraltar (9.a) there is evidence of introgression by the Mediterranean group. Sea bass inhabiting the areas Northern (4.b&c, 7.a,d–h) and Biscay (8.a&b) represent genetically one population unit. The current management in two stocks (Northern and Biscay) can be considered a conservative and correct measure*.

DSTs [data storage tags] *have been used to investigate individual behaviour patterns of adult sea bass between the North Sea and English Channel (Quayle et al., 2009) and between the Western Channel and Bay of Biscay. Movements were reconstructed using information about release and recapture locations alongside the temperature and depth of individual fish using hidden Markov models (e.g. Woillez et al., 2016). Five of 11 recaptured bass exhibited migrations of greater than 100 km between the central and southern North Sea and western English Channel, which supports previous studies (Thompson and Harrop, 1987; Pawson et al., 1987; 2007b) and provides evidence of migratory links between the North Sea and English Channel. Ifremer tagged sea bass with DSTs in the Iroise Sea from 2010–2012. Reconstructed tracks confirmed the highly migratory nature of bass, with three behavioural strategies: residency in the Iroise Sea; winter spawning migrations into the Bay of Biscay; and winter spawning migrations into the Western Channel/Celtic Sea. Site fidelity was found not only on summer feeding grounds as previously observed (Pawson et al., 2008), but also and for the first time, on winter spawning grounds. This indicated that sea bass populations may have a much finer spatial structure than supported by the genetics. However, sample sizes are very small, so it is difficult to generate robust estimates of the levels of migration between the current Northern and Biscay stocks*."

To explore the degree of isolation between Northern and Biscay stocks further, ICES launched two large tagging programmes. The first programme (C-Bass) is being led by the Cefas (Cefas, 2018) and has tagged almost 200 sea bass with electronic data storage tags (DSTs). The BARGIP study is being led by Ifremer (France) and has released 1,220 fish with DSTs at ten locations in the Channel and Bay of Biscay (ICES 2018e).

"Preliminary results of these studies indicated that "*adult bass can undertake very large-scale migrations. Fish tagged in 4.c exhibited two migration patterns: one that remained in the North Sea; and a second that involved long migrations as far as the Irish Sea. Fish that were tagged in May in 4.c were in spawning condition, suggesting the presence of spawning ground in the North Sea. Bass tagged in area 7.e appeared to remain in the English Channel, but individuals tagged in the Irish Sea could move between the Northern and Biscay stocks*" (ICES 2018e).

Sea bass movement between ICES stock units is plausible, as evidenced from some individuals tagged with DSTs, and fidelity to both spawning and feeding grounds may provide evidence of fine population structure that was identified from genetics. However, it is not possible to quantify the proportion of fish migrating between the stocks currently due to the small numbers of fish tracks analysed. As further DSTs are analysed, it may be possible to quantify exchange between stocks, so the next benchmark should consider how exchange between stocks should be incorporated into the assessment process" (ICES 2018e).

Therefore, ICES have a sound scientific background to consider seabass of the central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea representing the single stock ("The Northern stock") and respectively the single management unit (ICES 2019). There is a possibility of existence of sub-populations in Irish and Welsh waters (ICES, 2108e) but if those exist, they are still part of the same population and stock.

The following correspondence summarises the current position:

"Based upon the most up to date science, ICES considers the assessment stock unit for bass as Divisions 4b-c, 7a and 7d-h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic

Sea, Figure 4*). Previously, Pawson et al 2007 recommended amended stock units for assessment purposes based upon some UK tagging studies. However, scientific knowledge about bass has advanced since these 2000-2005 studies and ICES continues to recognise the 4b-c, 7a and 7d-h stock unit as the most appropriate for bass stock assessment purposes based upon all of the available scientific evidence.*

ICES recognises that there are probably separate bass sub populations based upon preferred residence areas at certain life stages and seasons in different areas (e.g. North Sea, Irish Sea) but studies have shown that because of a high degree of intermixing these 'sub populations' have the same stock dynamics which would not be the case if the fish were from separate stock units. The stock identity for bass is under regular review and was last assessed by ICES working group scientists in 2018 when it was concluded that the current stock unit continued to be appropriate for assessing the sustainability of the bass stock.

Scientific studies on bass stock identity including tagging programs, microchemistry and genetics are currently underway that are designed to provide more information on the movements of sea bass and the levels of mixing between stocks. The primary purpose of these studies is to determine whether the Bay of Biscay stock unit is indeed a separate stock or whether it should form part of the 4b-c, 7a and 7d-h stock". (pers. comm. Lisa Readdy Cefas, UK member of ICES Working Group for the Celtic Seas Ecoregion (WGCSE), 12 February 2020. This working group is responsible for the assessment of sea bass in Divisions 4b-c, 7.a and 7.d-h).

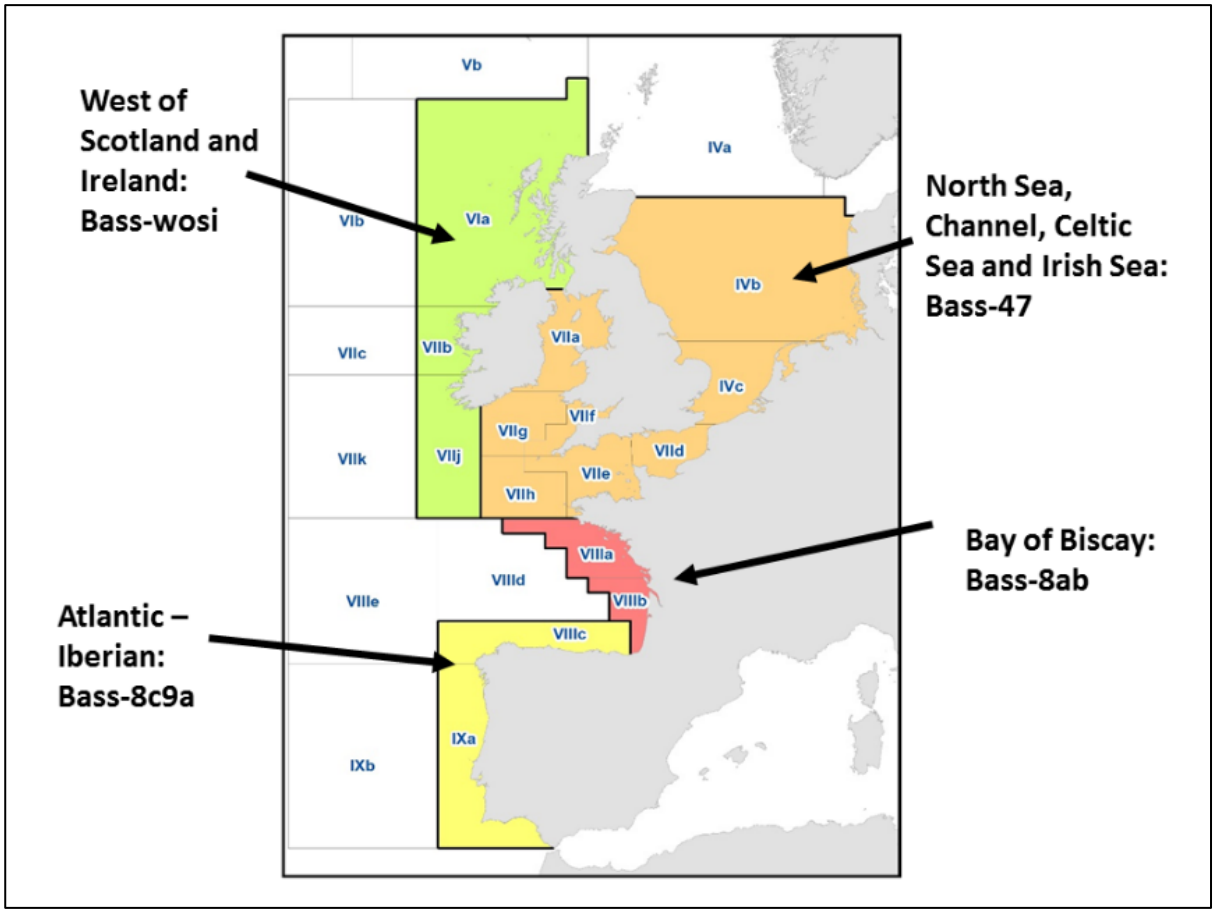


Figure 4 ICES stock units for bass (ICES WGCSE 2019)

In addition to questions about the separateness of the Bay of Biscay bass stock unit, there are similar questions about whether the bass in Irish coastal waters are indeed functionally separate (ICES WGCSE 2019). Stock identity studies are therefore currently focussed on the whether the stock unit size for bass

should be expanded and there are no suggestions within ICES of a reduction in the bass stock unit relevant to the Sizewell C assessments.

Finally, after overfishing was identified on the 4.b-c, 7.a and 7.d-h bass stock, ICES recommended a series of technical measures to reduce fishing mortality and to recover the size of the bass spawning stock biomass. These were implemented under the EU Common Fisheries Policy and the evidence to date is that the stock has partially recovered as predicted by the stock dynamics modelling that uses the 4.b-c, 7.a and 7.d-h stock identity.

ICES recommendations are science led and use the most up to date evidence available. ICES have recently reviewed the bass stock identities and found no compelling evidence to change the existing definitions. We therefore conclude there is strong confidence in the validity of the ICES' stock unit used as the basis for the SZC effects assessment on bass.

4.10.3 Conclusions on the validity of ICES stock units

Given the extensive body of ongoing research that supports the ICES' stock definitions, the questions that must be asked are whether it is appropriate, or proportionate, for EDF Energy to attempt to derive new stock assessment unit boundaries for the assessment of the effects of SZC and would any such new boundaries have any scientific credibility?

Specifically, would such an effort add anything to advance the sustainability of fish populations given that the predicted effects of SZC (Table 18) are orders of magnitude below those from commercial fishing which is managed using ICES' stock boundaries?

We have concluded that ICES' stock boundaries are compromises but they are based on a mature weighing of the best scientific evidence available and they are relied upon by governments to manage fish populations in the waters of all EU member states. Given the negligible predicted SZC impacts compared to those of fishing, and the precautionary nature of ICES' estimates of SSBs, we can find no justification not to use the ICES' stock definitions to assess SZC effects on fish stocks. The potential for local level effects is considered in Section 8.

4.10.4 How does ICES deal with non-fishing impacts on stocks?

In general, the ICES' approach to advice on fishing opportunities integrates ecosystem-based management with the objective of achieving maximum sustainable yield (MSY). Many of the models used by ICES to estimate MSY and associated parameters assume that factors not explicitly included in the models either remain constant or vary around an historical long-term average. Marine ecosystems are dynamic and fish stocks *may* change not only in response to the fisheries but in response to anthropogenic impacts and naturogenic effects as a consequence of bird and seal predation.

However, ICES do not include anthropogenic losses due to power stations in its stock assessments of commercially important species as such losses are considered as *de minimis*. This is supported by the predicted losses from SZC which show that the power station losses on fish populations would typically amount to less than 0.1% SSB or recorded international landings of commercially exploited species. (Pers. comm. Dr. C. O'Brien, Chief fisheries science advisor to Defra and vice President of ICES,18 February 2020).

4.11 Evaluating the effect of SZC impingement losses – Comparison with international landings data

For some species, although ICES collate all available fishery information, there are not enough data to carry out a full analytical assessment. While ICES may assess the status of the stock based on trends (e.g. trends in established surveys) and provide relative estimates of SSB, absolute SSB may not be estimated. In this case, we have compared impingement losses with the international landings for the stock area. However, such a comparison is unrealistically conservative as landings will be less than the stock size. For an unexploited stock, landings will typically be much less than 20% of the adult stock size and even for a heavily exploited stock, landings will rarely exceed 50% of the stock size.

Revisions in landings data in version 7.

Following consultation comments on version 6 of this report, stock units of various species were again reviewed based on the most contemporary evidence. Full details of the stock unit review are provided in BEEMS Scientific Position Paper SPP103. The fisheries landing data units for dab, flounder and thornback ray have been modified in Version 7 of this report. In the case of dab, the appropriate scale for impingement losses at Sizewell should apply annual landings from the North Sea Subarea 4 only as the comparator, rather than incorporating Division 3.a as previously done. Landings data for dab in Subarea 4 for the period 2009 to 2017, coinciding with impingement monitoring, ranged from 2,751 t (preliminary 2017 catch statistic) to 7,240 t with a mean of 5,309 t (ICES, 2019b). The estimated loss of dab is 0.5 t EAV relative to landings of 5,309 t equivalent to 0.01% of landings. The change in landings data comparator has no bearing on the outcome of the assessment. Furthermore, it should be noted that using landings as a comparator is not equivalent to the larger (undefined) SSB for the species, as such the assessment is precautionary.

Based on the latest ICES advice the previous assessment area for thornback ray of Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak, Kattegat & eastern English Channel) and may be too extensive. An alternative, more precautionary approach uses the assessment/TAC-allocation Subarea 4 and Division 2a (North Sea and Norway) as defined by ICES (ICES 2019c). During the period 2010-2017 landings of thornback ray in Division 2a were negligible (< 1 t. per annum), whereas landings in the North Sea (Subarea 4) were of 491-905, mean 695 t (ICES, 2020). Stock size indicator demonstrated a strong increase in ray abundance from around 2012 onwards, and the agreed TAC (798-1,404 t.) was never taken by fishermen (ICES, 2019c). Predicted losses from the proposed development are of 0.52 t or 0.07% of annual fisheries landings from the new stock areas. It should be noted that landings have consistently remained within the TAC set annually since 2012 (798-1,404t.). The predicted annual losses from the proposed development account for between 0.02 and 0.04% of the TAC (BEEMS Scientific Position Paper SPP103).

In the case of flounder, updated information on landings in the different sub-divisions of the North Sea (excluding Kattegat and Skagerrak) have been applied and the mean 2009-2017 landings are reported as 2,321.4t opposed to 2,309t in BEEMS Technical Report TR406.v6. The change in landings statistics has a nominal effect with the proposed development predicted to result in losses of 0.12 t of flounder per annum with full mitigation, representing 0.0051% of mean landings (BEEMS Scientific Position Paper SPP103).

4.12 Evaluating the effect of SZC impingement losses – Other comparative data sources

For other species alternative sources for population sizes, catches or landings were used:

a. Losses of thin-lipped grey mullet *Liza ramada* and were compared against ICES' Official Nominal Catches 2006 – 2017 (ICES, 2019), downloaded from the ICES website.

b. Environment Agency and Defra data were used to source catch data for salmon, sea trout, and eels (Table 8).

c. The silver eel SSB used for assessment purposes was from the Anglian RBD only.

d. For river lampreys, losses were compared against a spawning run size estimate for the Humber catchment made in 2018 by the Hull International Fisheries Institute (supplied by Dr J Masters, Environment Agency).

e. Data on shad are more difficult to source, as they are not part of a directed fishery and they may not be landed commercially in the UK. Some landings, of bycatch, are available in the ICES catch data, but these will substantially underestimate the size of the parent stocks. The twaite shad caught at Sizewell are considered to be part of a wider North Sea population that spawns in the rivers of Europe (predominantly in the Elbe but also in the Weser and the Scheldt) (BEEMS Scientific Position Paper SPP100), and the numbers of individuals impinged at SZC have been compared with abundance estimates from monitoring surveys conducted on the Elbe and the Scheldt.

f. Similarly, some landings data for smelt (*Osmerus eperlanus)* are available, but fishing in the UK for this species is limited by the number of fishing authorisations granted. The smelt caught at Sizewell are considered to be part of a wider North Sea population that spawns in at least the Rivers Elbe and Scheldt (BEEMS Scientific Position Paper SPP100). Numbers of individuals impinged at SZC have been compared with adult abundance estimates from monitoring surveys conducted in the River Elbe alone. The adult smelt stock has therefore been underestimated and the effect of SZC impingement has been overstated in this assessment. An assessment has also been made using relevant UK landings data from the 'Anglian' region as the existence of a southern North Sea population has yet to be proven.

g. Sand gobies are an unexploited fish stock and accordingly few abundance data are available. In Cefas Young Fish Surveys (YFS) of the east and south coasts of England, gobies were the dominant species throughout the survey area, with highest densities recorded in the area from Flamborough to Winterton (region 1), followed by the area between Winterton and North Foreland (region 2), and the lowest densities between North Foreland and Portland Bill (region 3) (Rogers and Millner, 1996). For region 2, estimated densities in September of each year were approximately 41 individuals/1000 m^2, which is comparable with the abundances observed in the June BEEMS offshore survey (BEEMS Technical Report TR345). Population estimates for *Pomatoschistus* spp. in region 2 ranged between 36 – 197 million individuals between 1973 and 1995 (mean = 94.7 million, st. dev = 41.3 million individuals). However, studies on the catching efficiency of 2 m beam trawls showed that the gear is only 46% efficient at catching *Pomatoschistus* spp. over coarse sand (Reiss et al., 2006). Taking the trawl efficiency data, this would suggest that only 46 % of the gobies present in the areas surveyed during the YFS were recorded, leading to an underestimation of their abundance. This would imply that the mean population abundance of 97.4 million individuals should be raised to 205.8 million individuals to account for the *Pomatoschistus* spp. not caught.

A summary of the sources for SSBs, landings, catches or population and an indication of whether analytical assessments exist for each species is given in Table 8.

Table 8 Data sources used to provide information on relevant stock unit, landings and SSB. Updates following stakeholder consultation, detailed in BEEMS Scientific Position Paper SPP103, are identified in bold.

Species	ICES Working Group	Stock unit	Assessment type	Impingement effect comparator	Reference
Sprat	HAWG	Subarea 4 (North Sea)	Analytical assessment	SSB	ICES, 2018
Herring	HAWG	Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak & Kattegat, Eastern Channel)	Analytical assessment	SSB	ICES, 2018
Whiting	WGNSSK	Subarea 4, Division 7.d (North Sea, Eastern Channel)	Analytical assessment	SSB	ICES, 2018b
Bass	WGCSE	Divisions 4.b-c, 7.a, & 7.d-h (Central & southern N Sea, Irish Sea, English Channel, Bristol Channel & Celtic Sea)	Analytical assessment	SSB	ICES, 2018c
Sand goby	-	Not defined	Not assessed	Population abundance	Rogers and Millner (1996)
Sole	WGNSSK	Subarea 4 (North Sea)	Analytical assessment	SSB	ICES, 2018b
Dab[1]	WGNSSK	**Updated: Revised stock area Subarea 4 (North Sea).**	Trends only	Landings	ICES, 2019b
Anchovy	WGHANSA	Given as 'Northerly anchovy'	Not assessed	Landings	ICES, 2018d
Thin-lipped grey mullet	-	Not defined	Not assessed	ICES Landings	ICES, 2019
Flounder	WGNSSK	Subarea 4 & 3.a (North Sea & Skagerrak and Kattegat). **Minor revision to updated landings data.**	Trends only	Landings	ICES, 2018b
Plaice	WGNSSK	Subarea 4 IV & Subdivision 20 (North Sea & Skagerrak)	Analytical assessment	SSB	ICES, 2018b
Cucumber Smelt	-	Not defined but includes the East Anglian coast and rivers on the European coast from the Elbe to the Scheldt. **Primary assessment is based on conservative UK landings and SSB.**	Estimated adult numbers migrating upriver	Elbe populations EA landings & ICES Landings	BEEMS Scientific Position Paper SPP100 EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c; ICES, 2019
Cod	WGNSSK	Subarea 4 & Subdivisions 7.d & 20 (North Sea, Eastern Channel, Skagerrak & Kattegat)	Analytical assessment	SSB	ICES, 2018b
Thornback ray[1]	WGEF	**Updated:** Subarea 4 and Division 2.a (North Sea and Norway).	Trends only	Landings	ICES, 2018e

Species	ICES Working Group	Stock unit	Assessment type	Impingement effect comparator	Reference
River lamprey	-	Humber catchment	Estimated run numbers	Numbers converted to weight	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c
Eel	WGEEL	Anglian River Basin District (RBD)	Biomass estimated	Estimated silver eel biomass	(Defra, 2018, 2015)
Twaite shad	-	Not defined but includes the River Elbe and Belgian river Scheldt. A separate spawning population on the river Weser has not been included in the assessment. **Revised population estimate.**	Estimated adult numbers migrating upriver	European populations in the Elbe. ICES Landings	BEEMS Scientific Position Paper SPP100 ICES, 2019
Horse mackerel	WGWIDE	Divisions 3.a, 4.b,c & 7.d (North Sea)	Trends only	Landings	ICES, 2018f
Mackerel	WGWIDE	Subareas 1–8 and 14, & Division 9.a (the Northeast Atlantic & adjacent waters)	Analytical assessment	SSB	ICES, 2018f
Tope	WGEF	North east Atlantic	Not assessed	Landings	ICES, 2018e
Sea trout	-	Not defined	Assessment based on CPUE	EA Catch numbers, UK	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c
Allis shad	-	Garonne	Analytical assessment	Adult stock in 2009, ICES Landings	BEEMS Scientific Position Paper SPP071/s) ICES, 2019
Sea lamprey	-	Not defined	Not assessed	-	
Salmon	WGNAS	North Atlantic	North Atlantic	EA Catch numbers, UK	EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c

[1] Following consultation comments on the stock units of various species, the stock unit of dab and thornback ray has been modified in version 7 of this report. Full details of the stock unit review are provided in BEEMS Scientific Position Paper SPP103. In both cases, the change in stock comparator has no bearing on the assessment outcome.

Working group acronyms:
HAWG - Herring Assessment Working Group for the Area South of 62°N
WGNSSK - Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak
WGCSE - Working Group on Celtic Seas Ecoregion
WGHANSA - Working Group on Southern Horse Mackerel, Anchovy and Sardine
WGEF - Working Group on Elasmobranch Fishes
WGEEL - Joint EIFAAC/ICES/GFCM Working Group on Eels
WGWIDE - Working Group on Widely Distributed Stocks
WGNAS - Working Group on North Atlantic Salmon

5 Assessing the significance of impingement effects

There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by impingement in coastal power stations and therefore any assessment must be based on expert judgment.

For the purposes of this assessment we have adopted two screening thresholds that have been selected such that impingement losses lower than the appropriate threshold will have negligible effects on the year to year sustainability of a fish population. Effects above the appropriate threshold would not necessarily indicate a significant adverse effect but require further investigation to determine whether significant effects were, in fact, present.

The thresholds have been selected based upon internationally accepted scientific practice for the sustainability of fish stocks under anthropogenic pressures:

c. For commercially exploited stocks and conservation species (which includes stocks that are not currently exploited): 1% of the SSB or, as a highly conservative proxy, 1% of international landings of the stock.
d. For unexploited stocks: 10% of the SSB or, as a highly conservative proxy, 10% of international landings of the stock.

In this section, scientific rationale for the selection of these screening thresholds are detailed.

Note that at the time of the HPC DCO the screening test that was applied and accepted for potentially significant environmental effects in the HPC Environmental Statement, shadow HRA and WFD was whether the predicted impingement of any of the assessed species was >1% of the SSB or fishery landings.

5.1 Screening thresholds for negligible effects in context

5.1.1 What is meant by a sustainable fish population?

Fishing is the selective removal (or harvesting) of fish. Impingement is therefore a form of fishing but of lower selectivity and much lower impact magnitude than fishing. Fish populations grow and replace themselves and they are therefore renewable resources. In the absence of harvesting, the population size of a stock does not increase indefinitely and stabilises around a maximum that a given habitat can support (the carrying capacity); i.e. it is under density control. The scientific basis for the sustainable use of a renewable marine resource evolved during the first half of the 20th century and is based upon a fundamental ecological principle of density dependent population regulation. As the abundance of a density regulated population is reduced by harvesting, per capita net production increases (by means of increased rates of growth, survival and reproduction), until the population cannot compensate for additional mortality after which point the productivity of the stock decreases and eventually becomes at risk of collapse. The production generated by this compensation (known as surplus production) can be harvested on a sustainable basis on a year on year basis (Rosenberg et al., 1993). Sustainability can therefore be framed as ensuring a sustainable harvest rate; i.e. where the rate of entrapment is less than or equal to the rate at which the population can regenerate itself. Determination of that rate for different fish stocks has been an internationally coordinated endeavour for more than 70 years and has led to well established stock assessment principles.

For well monitored stocks (data-rich stocks) quantitative stock assessment can be carried out which produces spawning stock biomass reference points below which a stock is either at risk of becoming unsustainable or is in an unsustainable condition, together with limits on the maximum harvest rate. However, fisheries scientists are frequently required to advise on harvesting rates (also known as exploitation rates) of many data-limited stocks where an alternative precautionary approach is required.

Several analytical approaches are applied in such circumstances which are largely determined by the availability and quality of the data. The different approaches are essentially based upon:

a. Limiting fishing mortality (F) to no greater than the natural mortality (M) of the species (determined by the life-history of the species).

b. For stocks where there is a record of fish landings, limiting F to the average fishing mortality (or index of fishing mortality) that did not lead to stock decline.

(Source: Food and Oceans Canada 2001)

[Note: The harvesting rate as a percentage of SSB is given by: $1-e^{-F}$ (where F is fishing mortality) and for many demersal and benthic species in UK latitudes the adult M is in the range of 0.1 to 0.2]

Approach a. above is an internationally adopted management approach:

> *'Escapement strategies are used to manage short-lived species and exploitation rates of up to 20% are advised by ICES'* (pers. comm. Chief Fisheries Science Advisor to Defra, 2018).
>
> *'Limiting the exploitation rate to 10-20% of the estimated spawning stock biomass will ensure that fishing does not cause the stock to decline to unsustainable levels'* (Giannini et al., 2010).
>
> *'…a constant harvest rate of 20% of the spawning population became coastwide management policy …'*. (Hall et al., 1988).

M ranges from approximately 0.1 for some benthic species to >0.5 for some pelagic species at Sizewell (BEEMS Technical Report TR383); i.e. sustainable harvest rates vary with the lowest values being for long-lived, late maturating species. The sustainable harvest rate calculated from approach a. above approximates to but is more precautionary than the maximum sustainable yield and as such is well above the biological reference point where the stock would be at risk of becoming unsustainable. For many unexploited species that occur at UK latitudes, this approach implies a sustainable harvesting rate of 10%-20% for such species. However, this formula is not conservative for very short lived (or tropical species) where the sustainable value of fishing mortality is less than M, typically 0.25 to 0.5 M (Caddy and Csirke, 1983). Gobies are such a short-lived species reproducing within the first year of life. They are a very abundant species that is ubiquitous in European coastal areas to at least a depth of 20 m. The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over large geographic areas (see BEEMS Scientific Position Paper SPP103). Sand gobies have an estimated M of 3.3 (Fishbase) implying a sustainable harvesting rate of greater than 50%.

On a precautionary basis a harvesting rate threshold of 10% SSB is considered appropriate as a screening threshold for potentially significant effects that may affect the sustainability of an unexploited fish stock.

5.1.2 Natural variability of fish stocks

Fish stocks are subject to considerable annual variability due to highly variable levels of recruitment, food availability and predation pressure. Individual populations and ecosystems are resilient to such high levels of variability. Impingement at SZB mirrors the variability of local fish populations as the power station is an efficient sampler with low interspecies bias unlike trawl or other net sampling techniques.

Table 9 shows the year to year variability in predicted impingement rates at Sizewell B based on monitoring data and applying the bootstrapping method presented in Section 4.4. The predicted variability for many species is substantially less than at the Hinkley Point estuarine location (BEEMS Technical Report TR456). This is as expected due to the composition of the fish assemblage at the Sizewell coastal location with substantially smaller numbers of 0 group fish whose numbers show the greatest year to year variability due to changes in annual recruitment.

Mean interannual variability over the period 2009-2017 ranges from 1.27 for eels to 5.02 for anchovy. In the case of sand gobies mean interannual variability is 4.20 whilst maximum interannual variability was estimate to be 15.32. Given the magnitude of such changes at Sizewell B a <1% change to SSB due to impingement is negligible. This is particularly pertinent to predator-prey relationships which are adapted to cope with the much greater natural variability.

Table 9 Year to year variations in annual predicted SZB impingement numbers (2009-2017) for selected species.

Species	The largest year-year changes in annual numbers from the Sizewell B CIMP dataset 2009-2017[1] (shown as the ratio of predicted impingement numbers in adjacent years)	
	Mean interannual variability (2009-2017)	Maximum interannual variability between any two adjacent years
Sprat	2.02	3.64
Herring	1.33	1.70
Whiting	1.56	2.04
Bass	2.36	6.11
Sand goby	4.20	15.32
Sole	1.53	2.26
Dab	2.40	7.88
Anchovy	5.02	8.85
Thin-lipped grey mullet	3.94	9.0
Flounder	1.35	1.74
Plaice	1.78	2.74
Smelt	2.16	3.99
Cod	1.94	4.02
Eel	1.27	1.62
River lamprey	1.68	3.32
Twaite shad	3.29	7.56
Thornback ray	1.75	2.92

[1] Data from the year 2013 is missing, interannual variability looks at changes from 2012 to 2014.

5.1.3 Comparison with sustainable levels of harvesting rate for data rich stocks

In Section 5.1.1 the internationally accepted precautionary harvest rate of 10-20% SSB was described for unexploited species where little monitoring data exists. It is useful to consider a 1% negligible effects threshold in the context of sustainable harvest rates for data rich stocks which in many cases are much greater than 20% (see Table 10).

ICES produce estimates of the precautionary levels of fishing mortality beyond which sustainability is at risk (F_{pa}). Other anthropogenic activities resulting in impact on fish stocks, including water abstraction for operational power stations, form part of the baseline against which SSB is established, however, such impacts are not assessed by ICES (Section 4.10.4). Examples from ICES stock assessments are shown in Table 10. Set against such numbers, an impingement mortality of less than 1% from SZC is negligible. An additional 1% mortality in addition to the effects of fishing is in the noise for practical stock assessments and

in practice such a level of effect is much smaller than that due to the uncertainty in the input parameters which are already assessed on a precautionary basis in the stock assessment.

Table 10 Sustainable fishing mortality values based upon a precautionary management approach for species relevant to Sizewell

Species	Sustainable fishing mortality reference values using precautionary approach (Fpa)	ICES Working Group Report	Coefficient of Variation of the SSB 1998-2017	Reference
Herring†	26%	HAWG	18%	ICES, 2018a
Whiting‡	28%	WGNSSK	12%	ICES, 2018b
Bass*	19%	WGCSE	23%	ICES, 2018c
Sole‡	36%	WGNSSK	30%	ICES, 2018b
Plaice‡	31%	WGNSSK	58%	ICES, 2018b

This point is further underlined by the actual predictions of total SZC entrapment losses in Table 18 of 0.01%, 0.02%, 0.03%, 0.00% and 0.00% of SSB for herring, whiting, bass, sole and plaice respectively.

5.1.4 An example of where screening thresholds for fish mortality have been applied for major infrastructure projects

A 10% SSB screening threshold has been previously adopted by the Thames Tideway Strategy Group. This group comprised representatives from the Environment Agency, Port of London Authority, Thames Water and others and developed water quality standards for the regulation of dissolved oxygen levels in the Thames Tideway to protect fish from mortality associated with storm discharges through combined sewer outfalls (Turnpenny et al., 2004). The efficacy of different standards was compared using an ecotoxicological model, the Tideway Fish Risk Model (TFRM). The Turnpenny report argued that commercial fishery exploitation rates could be sustainable at >50% SSB, depending on the population dynamics of the species. Based upon the Turnpenny report, the TFRM considered annual mortality rates of up to 10 % to be sustainable for all species not subject to fishing mortality (i.e. the integrity of the population would not be threatened), and up to 30 % for longer-lived species such as seabass and salmon. The 10% value was also considered to be the practical minimum change likely to be detectable through ongoing routine WFD Transitional and Coastal (TrAC) water fish surveys.

The subsequent DCO application for the Thames Tideway Tunnel contained a review of the robustness of assumptions made in the TFRM including the definition of fisheries sustainability used in the model. The results of an independent expert peer review of the fisheries work were also provided (Thames Tideway Tunnel, 2013). The review conclusions were that the TFRM remained fit for purpose.

5.1.5 The appropriateness of a 1% SSB screening threshold for impingement effects

To have a negligible impact on a fish stock the predicted total anthropogenic harvest rate must be less than the value whereby the stock can replace itself on a year to year basis. For unexploited data limited species, a precautionary level of 10%-20% SSB is considered sustainable in international fisheries management practice. ICES provides advice in the context of current management policy which is to manage all species within sustainable limits by 2020; and policy measures have been recommended to the European Commission, which is responsible for managing marine fisheries in Europe, and are now being implemented in order to meet this objective as soon as possible in relation to the 2020 target.

For species which are heavily exploited by fishing a lower effect threshold for impingement is considered appropriate and 1% negligible effect screening threshold for annual impingement for all species provides a

precautionary level which is negligible compared with fishing mortality on exploited stocks and would have no effect on their sustainability. A precautionary level of 1% is much less than the natural variability of any species at Sizewell which the ecosystem is adapted to and hence would have no significant effects on predator prey relationships. The use of a negligible effect threshold of 1% of SSB is, therefore, considered to be precautionary. At the request of stakeholders, the sustainability of this threshold and how it has been applied in the SZC assessment for the critically endangered European Eel stock is discussed in 5.1.6.

For non-exploited stocks, a 1% threshold is highly precautionary based on fish population dynamics and any observed decline in stock numbers would be due to other factors well beyond the influence of SZC impingement. For such species, a 10% screening threshold has been adopted in this assessment (e.g. for sand goby and thin-lipped grey mullet)

5.1.6 Appropriateness of a 1% SSB threshold for the endangered European eel

Recruitment indices for glass eel arriving in continental waters decreased continually from 1980 to a low point in 2011. In 2011 a change occurred, and the recruitment trend has been increasing in the period 2011–2019 with a rate significantly different from zero. The reasons for the decline are uncertain but may include overexploitation, pollution, non-native parasites, diseases, migratory barriers and other habitat loss, mortality during passage through turbines or pumps, and/or oceanic factors affecting migrations (WGEEL 2019). Whether the continental stock has declined as much as recruitment is unclear but the decline in recruitment was preceded by a decline in landings two or more decades earlier, indicating a decline of the continental stock (Dekker 2003).

The reasons for the increasing recruitment trend are also unclear but may be linked to a substantial reduction in fishing mortality, some of which has been driven by economics, some by regulatory restriction. Commercial landings of glass eels have reduced by 97% from 2,000 t in 1980 to less than 60 t in 2019. Commercial landings of yellow and silver eels have fallen by nearly 90% from 20,000 t in the 1950s to less than 2,700 t currently. Recreational fisheries have also reduced substantially in recent years although the slope of the trend is unclear due to under reporting (WGEEL 2019).

The International Union for the Conservation of Nature (IUCN) has assessed the European eel as ‘critically endangered’ and included it on its Red List in 2009. It renewed this listing in 2014 but recognised that: “*if the recently observed increase in recruitment continues, management actions relating to anthropogenic threats prove effective, and/or there are positive effects of natural influences on the various life stages of this species, a listing of Endangered would be achievable”* and therefore *“strongly recommend an update of the status in five years*.”

To date it has not been possible to determine the size of the eel stock SSB.

The current goal of the European eel management, set by the Eel Regulation (Council Regulation 1100/2007), is to “*reduce anthropogenic mortalities so as to permit with high probability the escapement to the sea of at least 40% of the silver eel biomass relative to the best estimate of escapement that would have existed if no anthropogenic influences had impacted the stock…. (The Eel Management Plans)… shall be prepared with the purpose of achieving this objective in the long term*.”

There are insufficient data to reliably determine the biological reference points used in ICES quantitative stock assessments and the purpose of EMPs is to gradually reduce total anthropogenic mortality to achieve a recovery in the adult stock. Given the 10 to 25 y+ taken for glass eels to reach maturation this will inevitably be a lengthy process. It is not clear that the recent (9 y long) trend in increasing recruitment is the early sign of a recovery or not.

For such a population it is reasonable to ask whether 1% of SSB represents a precautionary no effects threshold for SZC. There is no SSB estimate for the entire stock so the alternative measure would be 1% of landings (of 2,700 t) or 27 t. Given the lack of biological reference points and the uncertainty surrounding eel stock dynamics, in this assessment it was not considered that 27 t would be sufficiently precautionary and

instead the stance was taken of assessing the station effects against the Anglian RBD silver eel biomass of 78.6 t. with a 1% of SSB = 0.8 t (section 6.6.2.2). The Anglian RBD silver eel production is a small percentage of the entire stock SSB (on a crude measure of ratio of commercial landings it may only be 0.5% (13.9 t/ 2,700 t) of the entire SSB and therefore 1% of the RBD SSB is a highly precautionary measure that could equate to approximately 0.005% SSB.

In conclusion, the 1% of SSB was retained as a screening threshold for eel assessment but to be precautionary the SSB of just the Anglian RBD was used instead of population measures for the entire stock. The predicted losses are much less than 1%.

6 Finfish Impingement predictions for Sizewell C

This section presents impingement predictions for finfish at SZC. To demonstrate the predicted efficiency of each mitigation measure and to provide full transparency, predictions apply each mitigation sequentially from unmitigated to fully mitigated. The sensitivity of the assessment to uncertainty in parameterisation of mitigation measures is provided in Section 7.

It should be noted that impingement predictions and associated 95th percentile confidence limits are provided as exact numbers. Whilst it is clearly recognised that impingement methodologies do not have the inbuilt accuracy to predict impingement to the n^{th} fish, this level of precision is provided, firstly to allow an audit trail of the data and secondly, in the case of conservation species, discussions with statutory stakeholders consider effects at the individual level.

6.1 Predicted impingement without embedded mitigation measures

Annual estimates of all species impinged at SZB, and predictions for all species at SZC are given in Appendix B. These are raw unmitigated values, with no adjustment for the embedded station mitigations or conversion to EAV equivalents.

Ninety-one fish taxa were recorded between 2009 and 2017, of these, 40 species were recorded in <5 % of the samples (10 samples or less), and 18 species were present in >50 % of the samples available (103 samples or more). The eight most abundant taxa by number were sprat *Sprattus sprattus*, herring *Clupea harrengus*, whiting *Merlangius merlangus*, seabass *Dicentrarchus labrax*, sand gobies *Pomatoschistus* spp., Dover sole *Solea solea*, European anchovy *Engraulis encrasicolus*, and Dab *Limanda limanda.* An estimated mean of 5.3 million finfish are impinged by the SZB station annually, with the top eight species contributing 95.3 % to the total abundance (BEEMS Technical Report TR339.v3).

Sixty-two invertebrate taxa were enumerated. The four most abundant invertebrate taxa by number were ctenophores, brown shrimp *Crangon crangon*, pink shrimp *Pandalus montagui* and common prawn *Palaemon serratus*. These four taxa contributed 97.5 % by number to the total invertebrate count, with ctenophores alone contributing 86.2 %. A mean estimated 73.9 million invertebrates were impinged by the SZB station annually between 2009-2017, including 63.7 million ctenophores (BEEMS Technical Report TR339.v3). The predicted unmitigated SZC impingement effects for the 24 key species after adjusting to equivalent adults are given in Table 11. Of the 24 key finfish species only the following three species exceeded the 1% threshold, these included:

- Seabass (1.63 % of SSB)
- Thin-lipped grey mullet (4.6 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (1.27 % of the precautionary biomass estimate, Section 5.1.6)

For all other species the predicted impingement losses in the absence of mitigation are < 1 %. (For sea lamprey the predicted unmitigated impingement mortality of 5 fish per year is ecologically negligible and could have no effect on the sustainability of the stock).

The following sections consider the individual, and combined benefits from impingement mitigation. An assessment of the uncertainties in the mitigation parameters is provided in Section 7.

6.2 Predicted SZC impingement with LVSE intake heads fitted

With the fitting of LVSE intake heads (designed to reduce the numbers of fish and other organisms being abstracted), the impingement losses of those species that exceeded 1 % in the absence of mitigation was reduced to thin-lipped grey mullet only (Table 12):

- Seabass (0.58 % of SSB)
- Thin-lipped grey mullet (1.64 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.45 % of the precautionary biomass estimate)

Fitting of the LVSE intake heads alone therefore reduces the impingement losses of seabass and European eel to below the 1 % threshold, while thin-lipped grey mullet remains above this threshold.

6.3 Predicted SZC impingement with FRR systems fitted

With the inclusion of the FRR alone (no LVSE), designed to increase the survival of more robust species, impingement losses of those species that exceeded 1 % in the absence of mitigation was reduced to thin-lipped grey mullet only (Table 13):

- Seabass (0.90 % of SSB)
- Thin-lipped grey mullet (2.53 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.26 % of the precautionary biomass estimate)

Fitting of the FRR systems alone therefore reduces the impingement losses of seabass and European eel to below the 1 % threshold, while thin-lipped grey mullet remains above this threshold.

6.4 Predicted SZC impingement with the effect of the LVSE intake heads and FRR systems fitted

With the combined effect of the intake head design and the inclusion of the FRR, impingement losses fall below the 1 % threshold for all three species that exceed the threshold in the absence of mitigation (Table 14):

- Seabass (0.32 % of SSB)
- Thin-lipped grey mullet (0.90 % of landings) - as an unexploited stock, a 10% threshold is appropriate
- European eel (0.09 % of the precautionary biomass estimate)

All steps in the mitigation process, including the lower and upper estimates, are given in Appendix C. Given the mitigation of the intake head design and the fitting of the FRR system, no species exceed the 1 % negligible effects threshold.

Further consideration of the predicted SZC effects on thin lipped grey mullet, bass and conservation species is given in Sections 6.5 and 6.5.

Table 11 Annual mean SZC impingement predictions with **no impingement mitigation**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or catch numbers (salmon & sea trout).

Species	Mean SZC prediction	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	5,199,288	54.61	220,757	0.02	151,322	0.04
Herring	2,607,016	1,864,587	351.78	2,198,449	0.02	400,244	0.09
Whiting	1,665,479	593,040	169.50	151,881	0.11	17,570	0.96
Bass	709,082	158,807	243.10	14,897	1.63	3,051	7.97
Sand goby	503,232	503,232	0.96	205,882,353	0.24	NA	NA
Sole	233,942	49,802	10.66	43,770	0.02	12,800	0.08
Anchovy	176,816	172,238	3.57	NA	NA	1,625	0.22
Dab	143,519	63,810	2.60	NA	NA	5,309	0.05
Thin-lipped grey mullet	127,382	10,618	5.52	NA	NA	120	4.60
Flounder	37,303	17,226	1.41	NA	NA	2,321	0.06
Plaice	24,716	8,536	2.10	690,912	0.00	80,367	0.00
Smelt	20,795	15,834	0.26	50	0.52	8	3.11
Cod	16,426	5,899	15.35	103,025	0.01	34,701	0.04
Thornback ray	7,460	1,438	4.59	NA	NA	695	0.66
Eel	3,031	3,031	1.00	79	1.27	14	7.18
Twaite shad	2,989	2,989	0.94	5,161,183	0.06	1	70.22
River lamprey	2,929	2,929	0.23	62	0.37	1	20.77
Horse mackerel	1,779	1,779	0.25	NA	NA	20,798	0.00
Mackerel	473	473	0.15	3,888,854	0.00	1,026,828	0.00
Tope	70	70	0.48	NA	NA	498	0.10
Sea trout	10	10	0.02	NA	NA	39,795	0.02
Sea lamprey	5	5	0.01	NA	NA	NA	NA
Allis shad	3	3	0.00	27,397	0.01	0	1.31
Salmon	0	0	0.00	NA	NA	38,456	0.00

Table 12 Annual mean SZC impingement predictions considering the effect of the intake head design (LVSE). Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, smelt, twaite shad, allis shad) or reported catch numbers (salmon & sea trout).

Species	Mean SZC prediction	SZC prediction (adjusted)	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	2,470,731	1,856,146	19.50	220,757	0.01	151,322	0.01
Herring	2,607,016	930,705	665,658	125.59	2,198,449	0.01	400,244	0.03
Whiting	1,665,479	594,576	211,715	60.51	151,881	0.04	17,570	0.34
Bass	709,082	253,142	56,694	86.79	14,897	0.58	3,051	2.84
Sand goby	503,232	179,654	179,654	0.34	205,882,353	0.09	NA	NA
Sole	233,942	83,517	17,779	3.81	43,770	0.01	12,800	0.03
Anchovy	176,816	63,123	61,489	1.28	NA	NA	1,625	0.08
Dab	143,519	51,236	22,780	0.93	NA	NA	5,309	0.02
Thin-lipped grey mullet	127,382	45,475	3,791	1.97	NA	NA	120	1.64
Flounder	37,303	13,317	6,150	0.50	NA	NA	2,321	0.02
Plaice	24,716	8,824	3,047	0.75	690,912	0.00	80,367	0.00
Smelt	20,795	7,424	5,653	0.09	50	0.18	8	1.11
Cod	16,426	5,864	2,106	5.48	103,025	0.01	34,701	0.02
Thornback ray	7,460	2,663	513	1.64	NA	NA	695	0.24
Eel	3,031	1,082	1,082	0.36	79	0.45	14	2.56
Twaite shad	2,989	1,067	1,067	0.33	5,161,183	0.02	1	25.07
River lamprey	2,929	1,046	1,046	0.08	62	0.13	1	7.42
Horse mackerel	1,779	635	635	0.09	NA	NA	20,798	0.00
Mackerel	473	169	169	0.05	3,888,854	0.00	1,026,828	0.00
Tope	70	25	25	0.17	NA	NA	498	0.03
Sea trout	10	4	4	0.01	NA	NA	39,795	0.01
Sea lamprey	5	2	2	0.00	NA	NA	NA	NA
Allis shad	3	1	1	0.00	27,397	0.00	0	0.47
Salmon	0	0	0	0.00	NA	NA	38,456	0.00

Table 13 Annual mean SZC impingement predictions with FRR systems fitted (no adjustment for the intake head design). Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are estimates of the population numbers (e.g. sand goby, twaite shad, allis shad) or reported catch numbers (salmon & sea trout).

Species	SZC prediction	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	6,920,815	5,199,288	54.61	220,757	0.02	151,322	0.04
Herring	2,607,016	2,607,016	1,864,587	351.78	2,198,449	0.02	400,244	0.09
Whiting	1,665,479	916,779	326,445	93.30	151,881	0.06	17,570	0.53
Bass	709,082	390,396	87,434	133.85	14,897	0.90	3,051	4.39
Sand goby	503,232	103,666	103,666	0.20	205,882,353	0.05	NA	NA
Sole	233,942	48,192	10,259	2.20	43,770	0.01	12,800	0.02
Anchovy	176,816	176,816	172,238	3.57	NA	NA	1,625	0.22
Dab	143,519	77,287	34,362	1.40	NA	NA	5,309	0.03
Thin-lipped grey mullet	127,382	70,135	5,846	3.04	NA	NA	120	2.53
Flounder	37,303	8,613	3,978	0.33	NA	NA	2,321	0.01
Plaice	24,716	5,091	1,758	0.43	690,912	0.00	80,367	0.00
Smelt	20,795	20,795	15,834	0.26	50	0.52	8	3.11
Cod	16,426	9,890	3,552	9.24	103,025	0.01	34,701	0.03
Thornback ray	7,460	1,537	296	0.95	NA	NA	695	0.14
Eel	3,031	624	624	0.21	79	0.26	14	1.48
Twaite shad	2,989	2,989	2,989	0.94	5,161,183	0.06	1	70.22
River lamprey	2,929	603	603	0.05	62	0.08	1	4.28
Horse mackerel	1,779	1,779	1,779	0.25	NA	NA	20,798	0.00
Mackerel	473	473	473	0.15	3,888,854	0.00	1,026,828	0.00
Tope	70	14	14	0.10	NA	NA	498	0.02
Sea trout	10	10	10	0.02	NA	NA	39,795	0.02
Sea lamprey	5	1	1	0.00	NA	NA	NA	NA
Allis shad	3	3	3	0.00	27,397	0.01	0	1.31
Salmon	0	0	0	0.00	NA	NA	38,456	0.00

Table 14 Annual mean SZC impingement predictions considering **full mitigation** accounting for the effect of the intake head design and with FRR systems fitted. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby, twaite shad, allis shad) or reported catch numbers (salmon & sea trout).

Species	Mean SZC prediction	SZC prediction after intake head adjustment	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	2,470,731	2,470,731	1,856,146	19.50	220,757	0.01	151,322	0.01
Herring	2,607,016	930,705	930,705	665,658	125.59	2,198,449	0.01	400,244	0.03
Whiting	1,665,479	594,576	327,290	116,541	33.31	151,881	0.02	17,570	0.19
Bass	709,082	253,142	139,372	31,214	47.78	14,897	0.32	3,051	1.57
Sand goby	503,232	179,654	37,009	37,009	0.07	205,882,353	0.02	NA	NA
Sole	233,942	83,517	17,205	3,663	0.78	43,770	0.00	12,800	0.01
Anchovy	176,816	63,123	63,123	61,489	1.28	NA	NA	1,625	0.08
Dab	143,519	51,236	27,591	12,267	0.50	NA	NA	5,309	0.01
Thin-lipped grey mullet	127,382	45,475	25,038	2,087	1.09	NA	NA	120	0.90
Flounder	37,303	13,317	3,075	1,420	0.12	NA	NA	2,321	0.01
Plaice	24,716	8,824	1,818	628	0.15	690,912	0.00	80,367	0.00
Smelt	20,795	7,424	7,424	5,653	0.09	50	0.18	8	1.11
Cod	16,426	5,864	3,531	1,268	3.30	103,025	0.00	34,701	0.01
Thornback ray	7,460	2,663	549	106	0.34	NA	NA	695	0.05
Eel	3,031	1,082	223	223	0.07	79	0.09	14	0.53
Twaite shad	2,989	1,067	1,067	1,067	0.33	5,161,183	0.02	1	25.07
River lamprey	2,929	1,046	215	215	0.02	62	0.03	1	1.53
Horse mackerel	1,779	635	635	635	0.09	NA	NA	20,798	0.00
Mackerel	473	169	169	169	0.05	3,888,854	0.00	1,026,828	0.00
Tope	70	25	5	5	0.04	NA	NA	498	0.01
Sea trout	10	4	4	4	0.01	NA	NA	39,795	0.01
Sea lamprey	5	2	0.36	0.36	0.00	NA	NA	NA	NA
Allis shad	3	1	1	1	0.00	27,397	0.00	0	0.47
Salmon	0	0	0	0	0.00	NA	NA	38,456	0.00

6.5 Further consideration of impingement effects on eel, bass and thin-lipped grey mullet

6.5.1 Bass

As described in Section 4 of this report, the density of bass was 20 times greater inshore of the Sizewell-Dunwich Bank in the vicinity of the SZB thermal plume than offshore of the Bank. When SZC begins operation, it will generate a thermal plume but in the deeper water at the SZC outfalls there will be negligible warming at the seabed and the greatest thermal plume effects will be limited to the top 1 m of the sea surface. The SZC plume will have the effect of further warming the inshore waters inshore of the Bank. Bass is a demersal species, but it is known to feed at the surface at night and so could be attracted to the SZC surface plume at night. However, at the surface bass would be invulnerable to the impact of SZC abstraction by the seabed mounted intakes. At depth, the water inshore of the Bank would be appreciably warmer than at the SZC intakes and there is no reason to consider that the distribution of bass would materially change from what it is now. Making a precautionary assessment that 90% of bass would remain inshore of the Bank (rather than the measured 95%) the expected bass impingement for SZC is 0.032% SSB (Table 15) and not the 0.32% SSB described in Section 6.4.

6.5.2 Thin lipped grey mullet

There is not a directed commercial fishery for grey mullet in the southern North Sea and therefore the landings data (120 t) will substantially underestimate the SSB. The mean length in the commercial catch has been estimated to be in the range 36 to 42cm. At this size the natural mortality is in the range of 0.5 to 0.4 (BEEMS Technical Report TR383) and the calculated sustainable harvesting rate is approximately 33% - 39% SSB (Section 5.1.1). Mullet impingement numbers at SZB show no significant trend over the period 2009-2017 and provide no evidence that fishing on the stock is unsustainable. It is therefore considered unlikely that mortality on the stock is 33%+ in the southern North Sea and instead a conservative assumption has been made that landings represent 20% of SSB. Using this figure, the predicted impingement in Section 6.4 at 0.9 % of landings is equivalent to approximately 0.18% SSB i.e. below the screening threshold of 1 % of SSB. For such a species without a directed fishery, the use of a 1% threshold is itself overly precautionary and a 10% threshold is indicated by internationally accepted fisheries assessment methodology (Section 5.1), further reducing the significance of the predicted impingement effect.

The effects of changes on the predicted SZC impingement assessment based on further considerations for bass and thin lipped grey mullet is shown in Table 15.

6.5.3 European Eel

Although comparisons of eel mortalities due to impingement with population estimates for individual catchments are theoretically possible, there is uncertainty as to which are the relevant populations, and the European eel is considered to be a single reproductive stock throughout its distribution range. Given the small scale of the yellow and silver eel fisheries along the Suffolk coast, the most appropriate indicator of the perceived impact of the Sizewell power station on local eel stocks is considered to be a comparison between impingement data for eels by life stage, (raised to an equivalent silver eel biomass assuming 90 g for males, 570 g for females and a 1:1 sex ratio Dr. A. Walker, Cefas, pers. comm.) and, for fisheries, the combined mean yellow and silver eel catch for 2010-2017 (13.9 t). For the population comparator, the mean estimated silver eel production for the Anglian RBD is 78.6 t.

With full mitigation the total annual predicted impingement mortality of eel at a SZC, would be about 223 fish. Using a length-weight conversion factor of 0.329 kg per fish derived assuming a 50:50 sex ratio, that males mature at 89.9g and that females mature at 568.9g (Aprahamian, 1988), 223 eels equates to 0.073 t, equivalent to 0.093 % of the estimated RBD biomass.

This figure is an overestimate as due to the lack of necessary biological and population data, it has not been possible to date to derive an EAV for eel, so a worst-case value of 1 has been assumed. Based on the eye index (Beullens *et al*., 1997) of biologically sampled impinged individuals at SZB (n = 89), all eels impinged were yellow eels. No silver eels were impinged. The eels impinged would therefore have an EAV of < 1 due

to the natural mortality experienced by yellow eels in their multi-year growth period before maturation to silver eels. For example, the natural mortality of yellow eels is estimated to be 13% per annum (Dekker 2000). The yellow eels abstracted at SZB were in the age range 2 to >25 years and the majority would have remained at the yellow stage for many years before maturation. Assuming that the yellow eels would have spent an average of 5 years at Sizewell before maturation, the predicted SZC impingement effect would be reduced by 50% to 0.047% of the RBD silver eel biomass (Table 15).

A further assessment of the effects of entrapment by the station on all eel life-history stages is provided in Section 6.6.2.

Table 15 Annual mean SZC impingement predictions considering the effect of the LVSE intake heads and FRR systems fitted and the corrections to thin-lipped grey mullet, bass, and eel assessments, detailed in Section 6.5. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Numbers in red font are either estimates of the population numbers (e.g. sand goby, twaite shad, allis shad) or reported catch numbers (salmon & sea trout).

Species	Mean SZC prediction	SZC prediction after intake head adjustment	FRR mortality	EAV number	EAV weight (t)	mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	6,920,815	2,470,731	2,470,731	1,856,146	19.50	220,757	0.01	151,322	0.01
Herring	2,607,016	930,705	930,705	665,658	125.59	2,198,449	0.01	400,244	0.03
Whiting	1,665,479	594,576	327,290	116,541	33.31	151,881	0.02	17,570	0.19
Bass	70,907	25,314	13,937	3,121	4.78	14,897	0.03	3,051	1.16
Sand goby	503,232	179,654	37,009	37,009	0.07	205,882,353	0.02	NA	NA
Sole	233,942	83,517	17,205	3,663	0.78	43,770	0.00	12,800	0.01
Anchovy	176,816	63,123	63,123	61,489	1.28	NA	NA	1,625	0.08
Dab	143,519	51,236	27,591	12,267	0.50	NA	NA	5,309	0.01
Thin-lip grey mullet	127,382	45,475	25,038	2,087	1.09	600	0.18	120	0.90
Flounder	37,303	13,317	3,075	1,420	0.12	NA	NA	2,321	0.01
Plaice	24,716	8,824	1,818	628	0.15	690,912	0.00	80,367	0.00
Smelt	20,795	7,424	7,424	5,653	0.09	50	0.19	8	1.11
Cod	16,426	5,864	3,531	1,268	3.30	103,025	0.00	34,701	0.01
Thornback ray	7,460	2,663	549	106	0.34	NA	NA	695	0.05
Eel	3,031	1,082	223	112	0.037	79	0.047	14	0.53
Twaite shad	2,989	1,067	1,067	1,067	0.33	5,161,183	0.02	1	25.07
River lamprey	2,929	1,046	215	215	0.02	62	0.03	1	1.53
Horse mackerel	1,779	635	635	635	0.09	NA	NA	20,798	0.00
Mackerel	473	169	169	169	0.05	3,888,854	0.00	1,026,828	0.00
Tope	70	25	5	5	0.04	NA	NA	498	0.01
Sea trout	10	4	4	4	0.01	NA	NA	39,795	0.01
Sea lamprey	5	2	0.36	0.36	0.00	NA	NA	NA	NA
Allis shad	3	1	1	1	0.00	27,397	0.00	0	0.47
Salmon	0	0	0	0	0.00	NA	NA	38,456	0.00

6.6 Consideration of the impingement losses of finfish species of conservation concern

6.6.1 Cucumber Smelt

Smelt (*Osmerus eperlanus*) are found in coastal waters and estuaries around the western coast of Europe, from southern Norway to north-west Spain (Maitland, 2003a). Although there are several non-migratory populations in large freshwater lakes in Scandinavia, it is usually found in coastal waters and migrates into large clean rivers to spawn (Wheeler, 1969). Adults live in the marine environment, but migrate to estuarine or slightly brackish rivers in early spring (February to April) to spawn, after which the adults return to sea (Maitland, 2003a). Smelt shed their adhesive eggs onto the river bed in the brackish reaches of tidal rivers during March and April, where they hatch in about 3−4 weeks. Spawning appears to be determined by temperature and tides. In the River Thames, spawning takes place in the Wandsworth area of the estuary and 0+ fish first appear at 18mm at Greenwich in mid-May (Colclough et al., 2002).

The smelt was once common in Great Britain and supported commercial fisheries in the estuaries of most large rivers from the Clyde and Tay south. Maitland (2003a) reports that fisheries for smelt existed in the tidal reaches of all the Broads rivers in Norfolk until at least 2002; commercial fisheries 'yielding 3 to 6 t' per annum were still active in the River Waveney in 1991; smelt are occasionally taken in herring nets in the Orwell Estuary; and commercial fishermen were taking large catches – 190−250 kg per day in the Medway and the River Thames by 2002. Today, smelt occur in at least 36 water courses in England and Wales, with large populations in the rivers Thames, Humber and Dee, the Wash and Great Ouse, as well as in water courses of the Norfolk Broads. Smaller populations exist in the rivers Alde/Ore, Ribble and Conwy, and recovery of supposed extinct populations seems to be underway in the rivers Tyne and Mersey (Colclough and Coates, 2013). Smelt are listed as a Priority Species in Section 41 of the Natural Environment and Rural Communities (NERC) Act 2006.

There are commercial fisheries for smelt in the Rivers Waveney, Bure and Yare, predominantly for angling baits, although smelt are now sold to restaurants (Dr. A. Moore, Cefas, pers. comm.) and since 2011 there has been a requirement for commercial smelt fisheries to be authorised by the EA and to make annual catch returns. The annual catch of smelt in 2014 was 11,006 kg from 4 licence holders (EA, 2015). However, the report does not state the rivers that the licence holders exploited, but it is known that they are based in the Ouse (Yorkshire and Cambridge), Waveney/Yare and Thames (Dr. A. Moore, Cefas, pers. comm.). The estimated smelt fishery for the assessment period, against which impingement statistics are compared, is a mean annual landing of 8t (Table 8).

Smelt are found all along the Anglian coast, in the southern North Sea and on the European coast from the Channel to Denmark but there is no targeted fishery at sea. The nearest estuary to Sizewell with a known smelt population is the Alde/Ore, approximately 25 km to the south of Sizewell. Other than that, the nearest estuary to Sizewell is the Blyth at approximately 12 km to the north of Sizewell. Adult smelt have been sampled in the Blyth but there is no evidence of a breeding population. Surveying in April and May 2016 found no evidence of suitable spawning habitat, a barrier to upstream migration, no eggs nor any smelt in spawning condition at the time that other Anglian rivers contained spawning aggregations (BEEMS Technical Report TR382). This work concluded that it was highly unlikely that there was a spawning population in the Blyth primarily due to a lack of suitable spawning habitat and the presence of a barrier to upriver migration.

Information on smelt stocks is limited. Colclough and Coates (2013) concluded that the smelt found in the Wash are probably from a common stock which may access some or all of the tributaries that flow into the Wash, and Maitland (2003a) reported that it is likely that stocks in Suffolk belonged to a population associated with the Norfolk Broads and the estuarine and brackish waters around Great Yarmouth and Lowestoft. More recent genetic analysis of 215 smelt collected from the SZB CIMP programme and from the Thames, Waveney, Great Ouse and Tamar estuaries showed that East Anglian smelt are genetically homogeneous with no genetic structuring seen within the region. Smelt from the Tamar was clearly distinct from the East Anglian collections (BEEMS Technical Report TR423).

Given the genetic information on the smelt at Sizewell, it is probable that the smelt impinged are from multiple locations on the east coast of the UK and, based on the comparable distances, from European estuaries of at least the Scheldt (Belgium) and the Elbe in Germany (this hypothesis is considered reasonable but it is recognised that it has not yet been proven). As such, a precautionary approach was applied based on UK landings and an estimated SSB.

Smelt abundance at Sizewell as indicated by the SZB impingement data has no trend from 2009 to 2017 and has apparently not changed since 1981/82 (Section 11.1). Losses due to commercial fishing in the 35+ year period has not had any discernible effect on smelt numbers at Sizewell and fishing mortality must therefore have been low with SSB being much greater than UK landings. On a conservative basis the maximum sustainable harvesting rate on an 'Anglian' smelt stock would be 16% (using the precautionary assumptions in Section 5.1.1 for calculating the sustainable harvesting rate for short lived species (i.e. the use of natural mortality/4) and an assumed natural mortality of 1.0 for 100% mature 2-year old fish, BEEMS Technical Report TR383). Given the stability of the Sizewell population as indicated by impingement numbers, fishing and other anthropogenic mortality is most likely less than 16% of SSB. Therefore, SSB would be 6.25 times UK landings (mean 8t), possibly more given the very restrictive licensing policy operating in the UK (e.g. with only 4 licences issued in 2015). The indicated effect level of SZC entrapment in Table 18 of 1.11% of landings would equate to 0.19% SSB (Table 15).

Considering only UK populations and given the limited number of licences issued for commercial exploitation, the size of fishery landings will be a substantial underestimate of the stock size. This is particularly the case considering the likelihood of smelt off Sizewell being part of a southern North Sea population embracing Belgian and German sub populations. Comparisons have therefore also been made against estimates of population size for the River Elbe[1]. Between 2009 and 2017, an estimated annual average 23.9 million adult smelt passed through the River Elbe (BEEMS Scientific Position Paper SPP100). In the absence of mitigation, the losses of smelt by SZC equates to 15,834 adults representing 0.066% of the Elbe population numbers. With consideration of the LVSE intake design losses fall to 0.02 % of the Elbe population. It is therefore concluded that the losses of smelt are negligible.

6.6.2 European eel

The European eel (*Anguilla anguilla*) is distributed across the majority of coastal countries in Europe and North Africa, with its southern limit in Mauritania (30°N) and its northern limit situated in the Barents Sea (72°N) and spanning the entire Mediterranean basin (WGEEL 2019).

The European eel life history is complex, being a long-lived semelparous (mature adults die after spawning) and widely dispersed stock. The shared single stock is genetically panmictic and data indicate the spawning area is in the southwestern part of the Sargasso Sea. The newly hatched leptocephalus larvae drift for two to three years with the ocean currents for more than 5000 km to the continental shelf of Europe and North Africa before entering continental waters. There, they metamorphose into the post-larval transparent glass eel. At this stage, glass eels migrate across the continental shelf to the coast. After reaching the coast, glass eels enter estuaries (in the UK Severn starting from about January/February). Glass eels metamorphose into pigmented elvers which either remain and feed in coastal marine or estuarine waters or begin active upstream migration to freshwater. The growth stage, known as yellow eel, may take place in marine, brackish (transitional), or freshwaters. This stage may last typically from two to 25 years (and could exceed 50 years) prior to metamorphosis to the "silver eel" stage and maturation. At this stage silver eels migrate 5000 km+ back to their spawning grounds. Age-at-maturity varies according to temperature (latitude and longitude), ecosystem characteristics, and density-dependent processes. The European eel life cycle is

[1] Data on the abundant Scheldt population could not reliably be extracted from the available publications and would require clarifications from the Belgian authorities before their data could be used in the SZC assessment.

shorter for populations in the southern part of their range compared to the north. Age at maturity ranges from 10 to 20+ years in northern temperate waters (Vøllested, 1992), and is earlier for males than for females (WGEEL 2019, McCleave 1993, Tesch 2003, Harrison *et al* 2014).

The stock of the European eel is described as being outside safe biological limits, with urgent action required by European Union Member States to assist recovery of the panmictic stock (Harrison *et al* 2014). Historical publications indicate that the decline in stock abundance and/or fishing yield might have started as early as in the 1800s and might have been related to inadvertent side effects of anthropogenic actions (water management). The downward trend in yield has been acknowledged internationally since the late 1960s, but up to today, it is unclear what processes were causing the decline, which occurred even in times of high recruitment up to 1980 (Dekker and Beaulaton 2015).

6.6.2.1 Eel catch statistics (from WGEEL 2019)

Glass eels

Glass eel fisheries within the EU take place in France, UK, Spain, Portugal and Italy. Glass eel landings have declined sharply from 1980, when landings were larger than 2,000 tonnes, to 62.2 t in 2018, 58.6 t in 2019 (provisional figure), and a mean for the previous 5 years (2013–2017) of 56.5 t. The amount of glass eel arriving in continental waters declined dramatically in the early 1980s to a low point in 2011. The reasons for this decline are uncertain but may include commercial overexploitation, pollution, non-native parasites, diseases, migratory barriers and other habitat loss, mortality during passage through turbines or pumps, and/or oceanic factors affecting migrations. Statistical analyses of time-series from 1980–2019 show that there was a change in the trend of glass eel recruitment indices in 2011; the recruitment has stopped decreasing and has been increasing in the period 2011–2019 with a rate statistically significantly different from zero. The highest point during the period from 2011–2019 was in 2014. It is not yet clear whether this change indicates a sustained recovery in eel recruitment.

Yellow and silver eels

Total EU commercial landings of yellow and silver eels were estimated to be around 20,000 t in the 1950s to 2,000–3500 t around 2009, most recently being 2,393 t in 2017, 2,694 t in 2018 (provisional) and a mean of 2,729 t for the preceding 5 years (2012–2016).

Recreational catches and landings are poorly reported, and so values must be regarded as minima. Recreational landings for yellow and silver eel combined declined from 543 t for 2016 (from ten EU countries), 195 t for 2017 (eight countries reporting) and 148 t for 2018 (five countries reporting). Overall, the impact of recreational fisheries on the eel stock remains largely unquantified although landings are considered to be at a similar order of magnitude to those of commercial fisheries.

A rough estimate of eel loss to all non-fishery anthropogenic factors (largely hydropower and pumps in rivers) estimated from reported mortality indicators from approximately half of the countries that report eel statistics to the EU is 1,625 tonnes annually.

Summarising these statistics, the annual EU anthropogenic impact on the yellow and silver eel stock in 2016 was approximately 4,900 t (comprising 2,729 t fishing, 543 t recreational fishing and 1,625 t from pumps and hydropower in rivers) compared with approximately more than 22,000 tonnes in the 1950s when commercial fishing accounted for 20,000 tonnes

There is no internationally agreed SSB estimate for the entire eel stock.

6.6.2.2 Local eel population

There was once a considerable fishery for yellow and silver eels in the Anglian River Basin District (RBD), the catchments of rivers that drain to the North Sea along the east coast of England between the Humber and Thames (Defra, 2010). However, eel fishing is now mainly a subsistence activity for the remaining

fishers in Suffolk rivers and the adjacent coast, who use fyke nets for yellow eels between March and November and for silver eels from September to December.

Although yellow eel populations in freshwater catchments may be considered as largely separate, there is no evidence that silver eels migrating outwards or inward-recruiting glass eels comprise separate biological entities, i.e. the European eel is a single stock unit (Maes et al., 2006; Palm et al., 2009). ICES (ICES, 2013) demonstrated that the overall recruitment of European glass eels is now <10% of the levels observed in the 1960s and 1970s, but there is no assessment of the whole stock. In 2008 and again in 2013, the European eel was listed in the IUCN Red List as a critically endangered species (Jacoby and Gollock, 2014).

The most comprehensive assessment available for the status of the eel population in East Anglia is provided in the Eel Management Plan (EMP) for the Anglian River Basin District (RBD). Data on yellow eel populations for Essex and Suffolk catchments are derived from electric fishing surveys, carried out as part of the EA's routine monitoring programme. An assessment, based on combined data gathered from 2009 to 2011 and reported to the European Commission in June 2012 as part of the UK's EMP Progress Report, estimated a total output of 62.3 t of silver eels from the Anglian RDB each year (Defra, 2012). Estimates were updated and presented for individual years in 2015 and 2018 (Defra, 2018, 2015).

6.6.2.3 Potential risks of SZC eel entrapment

In principle, the following eel life stages could be at risk from entrapment in SZC:

- Glass eels migrating along the coast from the north or the south to find a suitable estuarine/freshwater habitat;
- yellow eels moving between different river systems along the East Anglian coast or living in coastal waters; and
- silver eels migrating to the Sargasso Sea that after leaving estuaries along the East Anglian coast may transit past Sizewell.

Glass eels

Glass eels enter estuaries all year round, with migration peaks depending on latitude and also the variability of oceanic factors. In southwest Spain, highest densities occur between late autumn and spring with two migration peaks observed, whereas peak glass eel migration in the UK is later, typically occurring from February to May.

Glass eels that contribute to UK populations first arrive in the Western Approaches and then transit with the tidal currents either through the English Channel into the southern North Sea or from the north, following currents that flow around Scotland and southwards into the southern North Sea. The time to reach the southern North Sea is dependent on met-ocean conditions over Northern Europe and the relative strength of the Gulf Stream and associated currents around the British Isles. However, little is known about the residence times of glass eels in the southern North Sea. It is considered that glass eels reach the coast and then seek a salinity or other chemical cue to commence migrations up estuaries and then, for a large proportion of their number, to freshwater. The time spent in the open North Sea will, therefore, be dependent on the tidal currents and when the glass eels sense estuarine cues. In the journey from the Western Approaches to the southern North Sea the density of glass eels in coastal waters will be reduced progressively and substantially as large proportions of the eels migrate up estuaries encountered on route. In particular, eels travelling through the Channel and then heading north will encounter the very large Thames freshwater signal followed by signals from Essex and Suffolk rivers before they reach the coast in the vicinity of Sizewell. Residual hydrodynamic flows will also tend to carry a proportion of eels passing through the Straits of Dover towards the Dutch Coast. Glass eels migrating from the north will also encounter freshwater signals at for example the Humber, the Wash, North Norfolk coast rivers and the Broads at Yarmouth. Thereafter residual flows will tend to carry eels towards continental Europe within the East Anglian plume. The net result of these hydrodynamic patterns and riverine cues is that the expected glass eel density in the vicinity of Sizewell would be expected to be amongst the lowest in the UK.

Given their morphology of typically 4 mm width (and up to 8mm for 130 mm elvers), it is likely that most glass eels will pass through the 10 mm mesh on the SZB and proposed SZC cooling water screens and only rarely appear in impingement samples. In the BEEMS CIMP programme from 2009 to 2017 two glass eels have been sampled; 1 in March 2013 and 1 in January 2017 with both of length of approximately 67.5mm. The BEEMS targeted glass eel surveys in April and May 2015 only detected 1 glass eel in 105 valid tows using a methodology which successfully sampled many glass eels in the Bristol Channel. No glass eels or elvers have been detected in water drawn from the SZB forebay during the 12 month BEEMS Comprehensive Entrainment Monitoring Programme at Sizewell in 2011 (BEEMS Technical Report TR318) nor in any of the extensive multi-annual plankton surveys conducted at Sizewell. The totality of data from this extensive sampling programme led to the conclusion in BEEMS Technical Report TR318 that whilst glass eels are present in Sizewell coastal waters, that their density was very low at this location. The potential impact of glass eel entrainment in SZC was therefore assessed as negligible, especially given the high measured survival in laboratory studies that mimic the physical and chemical conditions and time of exposure and that any entrained glass eels would encountered within the power station (BEEMS Technical Report TR318).

During stakeholder engagement on the effects of SZC on fish, stakeholders have questioned whether the BEEMS surveys would have adequately detected glass eels. In particular, it has been suggested that:

1. the glass eel specific surveys targeted the surface waters during daylight when glass eels would have been seeking refuge near the seabed.
2. all of any glass eels sampled in the entrainment monitoring programme could have crawled out of the sampling nets before the nets were emptied.

These points have been carefully considered. In relation to point 1, the eel behaviour described is from upper estuaries not in lower estuaries or coastal waters (Harrison et al., 2014). At Hinkley Point (i.e. in the lower estuary) large numbers of glass eels were successfully sampled using the same methodology employed at Sizewell of sampling in the upper part of the water column during the daytime on the flood tide. As expected virtually no glass eels were detected on the ebb tide which is the behaviour that would be expected from fish using selective tidal stream transport (STST) to migrate on all of the available flood tides. This confirms that glass eels do migrate during daylight hours on flood tides at such locations and not just at night. This is supported by Lambert et al., (2007) where 30% of glass eels were found to migrate on the flood by day in the lower section of the Gironde estuary. Glass eels have poor swimming abilities and STST is the most energy efficient means of transport. Based upon the relative timings of eel arrivals in UK estuaries it is considered likely that glass eels employ STST on flood tides in coastal waters during day and night, particularly where underwater light levels are low due to high suspended sediment concentrations such as on the UK east coast. The question of whether the BEEMS surveys were too late could also be posed. The sample timings were determined from known glass eel arrival times on the UK east coast. For example, Environment Agency monitoring at Beeleigh Cut on the Blackwater (i.e. to the south of Sizewell where glass eel densities would be expected to be higher than off Sizewell assuming their likely southern migration route around the UK) showed that peak glass eels numbers were detected in May with substantial numbers in April and June but very low numbers of arrivals in March which would imply that the BEEMS survey timing (in April and May) was appropriate. The year 2015 could also have been a year of anomalously low glass eel recruitment but Figure 5 shows that this was not the case and in fact 2015 was a year that reflected the increase in glass eel recruitment observed across Europe from 2011 onwards described in Section 6.6.2.1. The BEEMS targeted glass eel surveys are, therefore, not considered invalid and if substantial number of glass eels were present at Sizewell the surveys would have detected them.

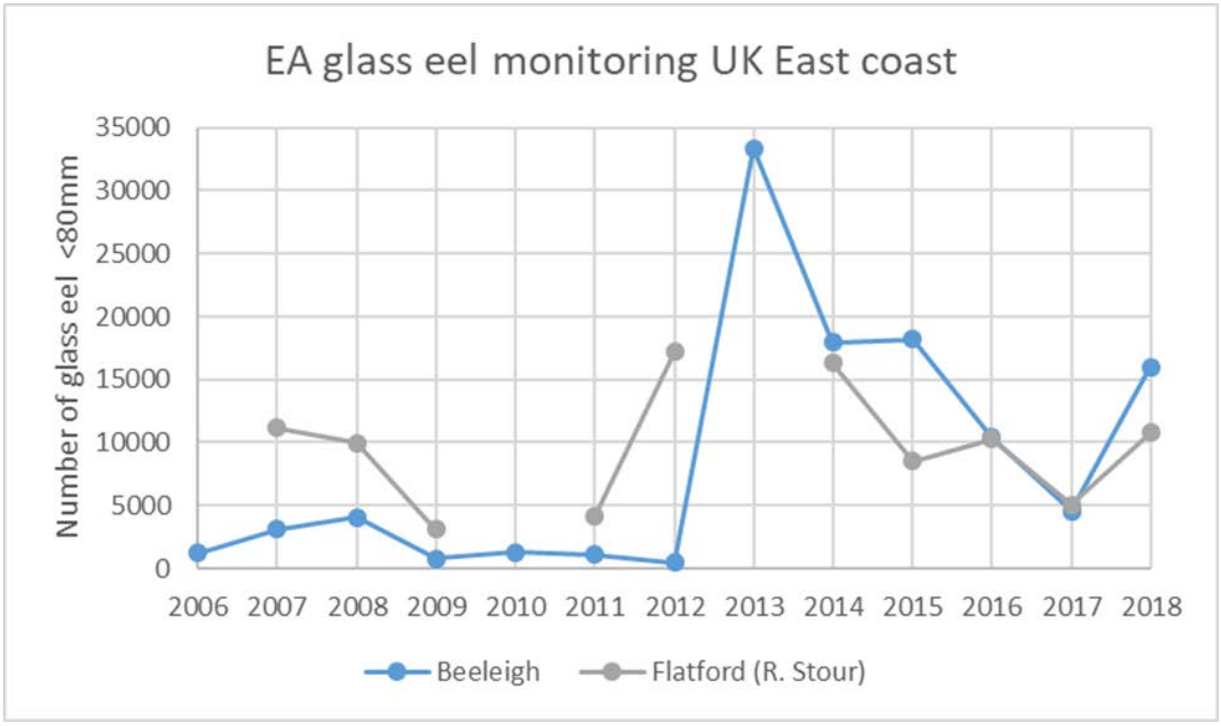


Figure 5 EA monitoring data for upriver glass eel migration from Beeleigh in the Blackwater Estuary in Essex and Flatford in the Suffolk Stour, both to the south of Sizewell.

In relation to the second point, glass eels are indeed known to crawl to overcome barriers but this is reported behaviour from upper estuaries:

"*Although STST is the primary mechanism facilitating migratory passage through estuaries, where tidal effects become weaker in upper estuarine zones, a behavioural shift to active swimming is necessitated to effect further dispersion upstream at the freshwater interface or more certainly from the point where they accumulate glass eels change their behavioural pattern and actively migrate counter current. Such an active migration is revealed in the 'crawling' behaviour that glass eels display on trapping ladders."* Harrison et al., 2014.

The crawling behaviour described by stakeholders is a behavioural change associated with the tidal interface not in the coastal zone. Even if glass eels in the marine environment can climb barriers, the net sides in the entrainment sampling tanks were very steep and Cefas considers it highly unlikely that, if eels had been caught, that all would have climbed out of all of the many nets deployed. In particular, the pump used to sample the SZB forebay for the entrainment sampling was not selected to ensure the survival of glass eels and the small impellor size would have delivered glass eels either moribund or dead into the sampling nets, substantially reducing the possibility of escapement from the sampling nets. In addition, the sampling pump was inspected after each of the 40 entrainment sampling events and no eels were found within the pump or its strainer. On a weight of evidence-based approach, we are therefore confident that any glass eels caught in the entrainment sampling system would have most likely been retained by the sampling system and not have escaped.

In conclusion, the glass eel migration pattern around the UK, the strength and direction of coastal currents and the large number of freshwater rivers that the eels would encounter on route would mean that glass eel densities at Sizewell would be expected to be very low and amongst the lowest on the UK coast (on the eel migration route). This low-density conclusion is supported by monitoring data. However, that monitoring data

does confirm that a few glass eels do transit past Sizewell whilst seeking freshwater signals. On energy efficiency grounds this migration is most likely to use a form of STST in near surface waters. When the tide is in the 'wrong' direction the evidence suggests that glass eels are stationary on, or even buried in the bottom sediments to avoid being carried away from their preferred migration course. Such a migration strategy will mean that there is a low risk of abstraction into power stations with near-bottom mounted intakes which do not abstract surface water except minimally at slack water. The deeper the intakes, the lower the risk of abstraction. It would therefore be expected that glass eel abstraction at SZB would be greater than at SZC due to substantially deeper water at the proposed SZC intakes. The abstraction risk zone for the SZC intake heads depends on the swimming ability of the species. Glass eels are weak swimmers and can sustain approximately 0.25 m/s for only 3 minutes before exhaustion and have a sustained swimming speed of no more than 0.05 m/s for long periods (McCleave 1980). Glass eels resting on the seabed would be unlikely to be abstracted as the SZC intake surfaces would be 1.5 to 3.5m above the bed. The only times that glass eels would be at risk is when they were settling towards or moving off the seabed and then only for those that were within a worst case 7 m of the intakes (entrapment risk zone where velocities exceed 0.05 m/s). This represents a very small volume of water at the SZC intakes compared to the potential volume that the eels could settle in within Sizewell Bay and the abstraction risk is, therefore, considered minimal. The same argument would apply at SZB (whilst recognising that the risk would probably be larger due to the shallower water at the SZB intakes). Low entrapment potential combined with the expected low glass eel densities at the site and their migration pattern in surface waters would provide a coherent explanation of the absence of glass eels in the SZB entrainment monitoring surveys.

The targeted glass eel surveys only detected 1 individual from which it is not possible to deduce anything about their spatial distribution in Sizewell Bay and, in particular, whether the expected density would be lower or higher at the SZC intakes than at the SZB intakes. However, it is known that glass eels are seeking a freshwater signal as a cue to migrate up estuary. Due to dilution and the effects of tidal advection, the probability of detecting such signals will reduce rapidly with distance from the coast especially given the very strong shore parallel tidal currents along the Suffolk coast and the presence of offshore sandbanks. Many of the freshwater discharges on the East Anglian coast are relatively small, especially in the vicinity of Sizewell e.g. the Blyth and discharges from the Minsmere sluice, and such small signals in combination with the effects of dilution and tidal advection would indicate that a close to shore migration strategy would be the most likely to be allow the eels to find estuaries. On that basis the working hypothesis is that glass eels migrating on the coast will preferentially swim close to the coast and that their density offshore at the location of the SZC intakes would be lower than at the SZB intakes. Some evidential support for this hypothesis is provided by glass eel behaviour in lower estuaries where it is known that they occur in the highest densities closest to the shore when migrating up the estuary (for example in the Severn Estuary, BEEMS Technical Report TR274).

Consultation on version 6 of this report raised further concerns about the adequacy of the target glass eel survey data and the potential entrainment effects on European eel. Scientific In response BEEMS Scientific Position Paper SPP104 was issued which calculates the limit of detection of the surveys, compares the results with the closest Environment Agency glass eel monitoring stations at Flatford in Suffolk and Beeleigh in Essex (Figure 5) and provides worst-case entrapment assessments together with associated sensitivity results. The conclusions of the report are as follows:

i. The BEEMS glass eel survey tows had a limit of detection of 0.000387 eels m^{-3} per tow. Assuming that eels are only at risk of capture during one tidal direction, the Sizewell survey detected a mean density of 7.44 10^{-6} eels m^{-3} in 52 flood tide tows. The same survey methodology and sampling equipment at Hinkley Point, detected mean densities of 0.0058 m^{-3} and a peak density of 0.0457 m^{-3}. From these data it is concluded that the density of glass eels at Sizewell is at least two orders of magnitude lower than at Hinkley Point. Based upon the predicted Sizewell glass eel density, the amount of effort required to increase the predicted Sizewell glass eel catch from 1 eel to 10 to 15 eels would be infeasible and would require impractically long working days and increasing the number of survey days from 11 to most days during the migration period. This amount of survey effort is disproportionate when a worst-case estimate can be derived from the data provided by the existing surveys.

ii. The BEEMS entrainment survey at Sizewell in 2011 would not have been sensitive enough to detect glass eels at the expected density in that year. The necessary increase in sampling to detect one eel would have been infeasible based upon the 2015 glass eel survey results.

iii. The calculated glass eel entrainment loss at SZC would be 0.0089% Anglian RBD SSB based upon 2015 survey data.

iv. Considering the cumulative effect of the sensitivity analyses, the SZC entrainment losses could have been underestimated by a factor of 2.66, assuming a hypothetical increase in night time density of a factor of 10 (noting that there are no measurements to support this figure). Using the calculated Gironde day: night ratio the SZC entrainment losses may have been overestimated by a factor of 1.25. On that basis the SZC entrainment losses are estimated to be between 0.0071% to 0.024% Anglia RBD SSB.

To put the SZC entrainment estimate into context, the estimated SZC mean silver eel impingement loss is 0.047% Anglian RBD silver eel SSB. i.e. the predicted SZC entrapment loss is in the range 0.054% to 0.071% Anglian RBD. One percent of the Anglian RBD SSB is equivalent to an estimated 0.005% of the European eel stock.

Considering the totality of the monitoring evidence and the implications of glass eel migration pattern around the UK, the risk of any significant entrainment effects on glass eel recruitment and at the regional eel stock level would be negligible.

Yellow eels

SZB impingement monitoring during the CIMP programme (with 10 mm mesh filtration) detected 2 glass eels (67.5 mm long) and a number of yellow eels ranging in length from 228 mm to 893 mm (

Figure 6 and Figure 7) i.e. with body widths from 14.25 mm to 55.8 mm (using morphological data reported in Environment Agency 2005). Ninety percent of the impinged eels were greater than 280 mm in length with a median length of approximately 400 mm. The length distribution of yellow eels is similar to that obtained from impingement sampling at Hinkley Point B. Studies to determine the age of the eels caught at Sizewell have not been undertaken but assuming a similar growth curve to that found at Oldbury in the Severn Estuary (Bird et al., 2008), the yellow eels impinged at Sizewell B ranged from 2 to >25 years old. Yellow eels were caught throughout the year with the peak period of impingement in October and November and lowest catches in February to April and in December (Figure 8). From the length data and the SZB 10 mm mesh size it can reasonably be hypothesised that small (i.e. very young) yellow eels and elvers were not present at Sizewell; if young yellow eels had been present the length distribution would have been expected to continue down to below 160 mm. Similarly, elvers were unlikely to have been present in any significant numbers as these larger fish would have been more likely to have been impinged than the two glass eels and would be unlikely to be present without young yellow eels. No glass eels or elvers were found in entrainment sampling. From these data it was concluded that yellow eels above 228 mm will be at risk of impingement at SZC with the majority of fish at risk having a length greater than 280 mm. All of the yellow eels are expected to be able to pass through the proposed 75 mm trash bar spacing at SZC.

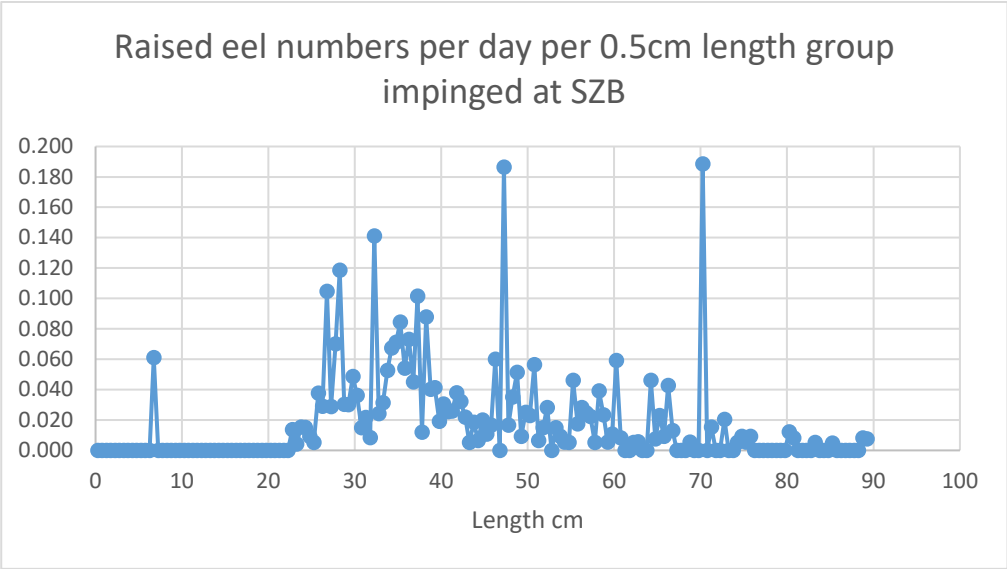


Figure 6 SZB impinged eel length frequency 2009-2017. The peak at 6.75cm corresponds to the 2 glass eels that were impinged in the period.

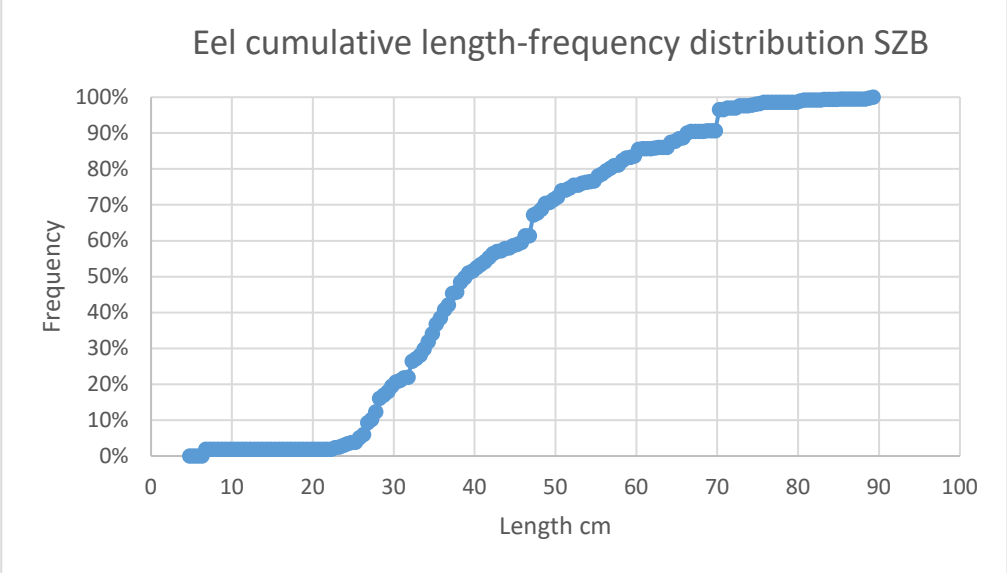


Figure 7 SZB impinged eel cumulative length frequency distribution 2009-2017.

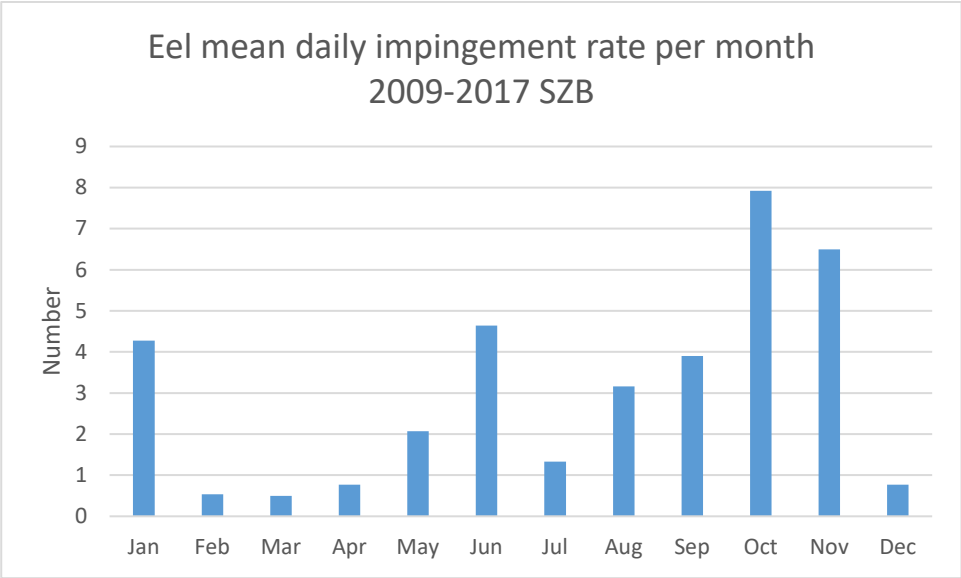


Figure 8 Variation in mean impingement rates by month at SZB.

Silver eels

No silver eels were caught in the SZB CIMP programme but this is not surprising as this life stage is known to migrate near to the surface at night and would be at a low risk of impingement at SZB's seabed mounted intakes and much less at SZC due to the deeper intakes.

6.6.2.4 Assessment of predicted effects of SZC

Section 6.5.3 considers the impact of SZC on the RBD silver eel biomass. Assuming that the yellow eels impinged at SZB would have spent an average of 5 years at Sizewell before maturation, the predicted SZC impingement effect would be 0.047% of the RBD silver eel biomass.

Another means of putting the SZC impingement estimate into context is to consider the predicted silver eel loss of 0.037 t per annum in the context of the estimated total EU anthropogenic impact on the stock of approximately 4,900 t per annum (Section 6.6.2.1). I.e. The SZC impact is equivalent to less than 0.001% of the total EU anthropogenic impact (from licenced fishing and secondly from hydropower and pumps operating largely in rivers).

As justified above and in BEEMS Technical Report TR318, the predicted effect of SZC entrainment on glass eel recruitment from SZC is considered negligible.

Compared to the other anthropogenic impacts on the stock, the predicted SZC entrapment effect on the EU component of the European eel stock is more than 4 orders of magnitude lower.

6.6.3 River lamprey and sea lamprey

The river lamprey (*Lampetra fluviatilis*) is found in coastal waters, estuaries and accessible rivers in western Europe, from southern Norway to the western Mediterranean. It is widespread in catchments throughout the UK, except in northwest Scotland and in industrial areas where water quality is poor or where obstacles prevent the upstream migration of adults prior to spawning (Maitland, 1972). The rivers of the Severn Estuary are thought to be the most important area in the UK for sea lamprey and possibly river lamprey too (Bird, 2008).

The biology and ecology of lampreys have been described in detail and reviewed by Maitland (2003b). Both sea and river lampreys spawn in coarse, well aerated riverbeds and juveniles, known as ammocoetes, spend several years living in aerobic silt beds filtering sediments, before transforming to migrants that move downstream to sea in the spring. After some years growing in the marine environment they move back into freshwater, where they migrate upstream to spawn. River lamprey move into freshwater during the previous summer, winter and spring before spawning in spring, whereas sea lamprey migrate into estuaries in the spring and then upriver to spawn in late spring to early summer (Hardisty, 1986). The fish are semelparous and die after spawning (Larsen, 1980).

Genetic studies suggest that sea lamprey (*Petromyzon marinus*) are a single, pan-European population (Almada et al., 2008) with widespread distribution. This is thought to be determined to a large extent by the movements of the fish hosts on which the lampreys feed and the fact that the adult lamprey do not display any apparent homing behaviour during spawning migrations (Berstedt and Seelye, 1995). River lamprey also display the same parasitic behaviour in the marine environment and are also not considered to home to natal rivers. As would be expected they show only low levels of genetic differentiation between local stocks across England (Bracken et al., 2015).

Fisheries for river lamprey exist in the Netherlands, Sweden, Finland, Estonia and in the UK. Sea and river lampreys are qualifying features of the Humber SAC. River lamprey are a primary reason for selecting the Derwent SAC, with sea lamprey a qualifying feature.

Despite a historical long-term decline in status, the distribution of river lamprey in waters of England and Wales appears to have increased in recent years, coincident with increased water quality in UK and European rivers. However, recording effort has also increased and JNCC, 2013 considered that it remained unclear whether the apparent increase in range is a consequence of increased levels of reporting or a true change in status. Results from impingement measurements at Sizewell showed no river lamprey caught in the years1981/82 at SZA but an estimated annual mean of 1,144 caught at SZB (Section 11.1). At Sizewell there has, therefore, been a substantial increase in river lamprey population sometime in the period 1982 to 2009. This is mirrored in multinational North Sea ecoregion monitoring surveys which found an almost complete absence of lampreys from 1977 to 1992 followed by a substantial increase in numbers thereafter (Heesen *et al* 2015).

Historically, commercial fisheries for river lamprey have taken place in several large rivers, including the Severn and Yorkshire Ouse. The Ouse bait fishery was re-established in 1995, with annual catches around 4,000 kg (Masters et al., 2006) which was estimated to have amounted to approximately 20% of the Ouse population but without showing any evidence of population decline. Since 2013, the Environment Agency has controlled the fishery by means of authorisations: 4 for 2015 and 2016 (EA, 2017a, 2017b). This fishery is subject to a catch limit (JNCC, 2013), and an overall agreed anthropogenic loss limit for river lampreys of 5 % of the population size estimated from mark-recapture experiments carried out in 2003 (Masters et al., 2006). There is also a limit on the season (open: 1 November to 10 December). Total allowable catch (TAC) has varied over the years and is often different for various rivers of the Humber catchment (Dr J Masters, Environment Agency, pers. Comm.). In 2017 and 2018 fishing was allowed on a non-consumptive basis only (catch and release), but for 2019 a quota of 898 kg has been set for the Humber catchment.

In 2018, the Hull International Fisheries Institute (HIFI) undertook mark-recapture experiments of lampreys in the commercial fishery at Naburn, the River Ouse. Almost 1,500 lampreys marked with PIT tags were used to estimate the run size of the lamprey population in the Humber catchment. Work was carried out during the fishing season only, but lampreys can migrate between October and February, so the estimated run size is likely an underestimation of the total spawning run. The Environment Agency used the HIFI data to create a population estimate for the Humber catchment of 783,043 individuals, equating to 61.86 t.

SZC with the proposed intake head design and FRR mitigation is predicted to take 215 individuals or 0.02 t of river lampreys, which equates to 0.03 % of the estimated 2018 lamprey run in the Humber catchment. The Southern North Sea population of river lamprey are probably one stock with spawning taking place in the Ouse in the UK but also in the Scheldt in the Netherlands where the adult population is estimated to be in the

100,000s (Jansen et al., 2007) and in other European rivers that drain into the North Sea (e.g. rivers Eider, Elbe, Weser and Ems). The SZC predicted impact on the river lamprey population is, therefore, considered negligible.

For sea lampreys, the estimated annual impingement loss with LVSE mitigation only is equivalent to 2 adult fish. This is considered negligible for a stock that is widespread throughout the North Sea particularly as the species is anticipated to have high survival rates in the FFR system. The expected annual impingement losses of sea lamprey at Sizewell C with the full LVSE and FRR mitigation is less than 1 fish (0.36), which equates to 1 fish every 2-3 years with a maximum of 1 individual per annum.

6.6.4 Twaite shad and allis shad

Allis shad (*Alosa alosa*) and twaite shad (*Alosa fallax*) both belong to the herring family and historically had a broad distribution along the Northeast Atlantic coast. Both species are anadromous; adults spend most of their lives in the marine environment but migrate through estuaries to spawn in freshwater. Populations of both species have declined, their distribution has diminished, and they are both classified as species of conservation concern. Both are listed in Appendix III of the Bern Convention and Annexes II and V of the Habitats Directive.

Alosa alosa was historically distributed along the eastern Atlantic seaboard from Norway to North Africa and in the western Mediterranean. It has declined significantly throughout its range and is now extinct in several former areas. Currently, populations of *A. alosa* exist along the north-eastern Atlantic coasts in some large rivers of France (Loire, Gironde–Garonne–Dordogne, and Adour) and Portugal (Minho and Lima) (BEEMS Scientific Position Paper SPP071/s). There are currently no known spawning sites for this species in the United Kingdom, and only two locations in the UK where individuals in breeding condition have been recorded: the river Tamar in SW England and the Solway Firth on the border between England and Scotland (Jolly *et al*., 2012). Immature adults are occasionally found in the Bristol Channel, the English Channel and the UK east coast. It is considered probable that British-caught specimens are from the Loire to Gironde populations (BEEMS Scientific Position Paper SPP071/s). *A. alosa* only spawns in any substantial numbers in France and Portugal (the species has recently been reintroduced into the Rhine but the number of recruits is still small). There is no international stock assessment for *A. alosa* but some assessments are performed on specific French watersheds. The Gironde–Garonne–Dordogne basin had a notable commercial fishery at the end of the 20th century. The adult population (age 4+) was estimated to be 710,000, 798,000, and 834,000 in 1994, 1995, and 1996, respectively, with a mean exploitation rate by the commercial fishery of 44%. It was reported that the commercial fishery in that basin caught approximately 500 t annually. However, in the first decade of the 21st century, there was a recruitment collapse probably due to over fishing and a fishery moratorium was imposed in the Gironde estuary from 2008 (BEEMS Scientific Position Paper SPP071/s). The estimated adult stock size in the basin was 27,397 in 2009. The Loire watershed also has a breeding population of *A. alosa* and a small commercial fishery. The count of *Alosa* was 2,557 in 2009 but the video counting system does not cover all the tributaries of the Loire and cannot distinguish between *A. alosa* and *A. fallax*. The counters are located relatively high in the river basin at ranges of 260 – 663km from the sea and are, therefore, probably counting mostly *A. alosa*. It is also known that a substantial amount of spawning takes place downstream of the counters thereby underestimating adult numbers (BEEMS Scientific Position Paper SPP071/s).

A. fallax is distributed along most of the west coast of Europe from the eastern Mediterranean Sea to southern Norway and in the lower reaches of large rivers along these coasts that are accessible to the fish (i.e. rivers that lack barriers to migration). The species has declined substantially across Europe and in the UK; it is now known to breed only in the Severn River Basin District (RBD – in the Severn, the Wye, the Usk and the Tywi) and in the Solway Firth. There are also non-breeding populations in the UK off the southern and eastern coasts, at Looe Bay, Hastings and Sizewell (Jolly et al., 2012). The decline of the *A. fallax* population has not been as severe as that of *A. alosa*, probably because of its ability to use spawning sites closer to the sea than those of *A. alosa*; sites that are not, therefore, subject to the barriers to migration that block *A. alosa* from accessing its traditional spawning sites.

Allis shad are protected under the Wildlife and Countryside Act (1981), and it is illegal to fish for the species in the UK.

Most of the twaite shad reported are taken as bycatch in coastal or estuarine net fisheries or by anglers fishing for other species (although several hundred are usually taken each year by anglers in the River Wye and other rivers into which twaite shad run to spawn).

Shad of both species migrate into rivers and spawn in flowing water over stones and gravel from mid-May to mid-July. Allis shad normally spawn only once, but twaite shad may spawn several times in their lives (Miran Aprahamian, Environment Agency, unpublished).

Young twaite shad remain in estuaries to feed on invertebrates, initially insect larvae and zooplankton and then increasingly on larger crustaceans such as shrimps and mysids and also small fish as they grow, reaching 10–15 cm after one year (Aprahamian, 1989). Water temperature during the months June−August seems to be an important influence in determining year-class strength in 0+ twaite shad.

Little is known about the movements of adult shad along the coast, but they have been studied when they enter estuaries and migrate upstream to spawn in freshwater, in particular in relation to barriers caused by obstacles such as waterfalls or man-made dams and weirs, and pollution (Aprahamian, 1982; Maitland, 1972). Adult shad of both species gather in the estuaries of suitable rivers in April and May, and the upstream migration from the estuary appears to be modified by temperature, with peak migratory activity at 10–14ºC and during relatively high discharge levels (Aprahamian, 1982; Claridge and Gardner, 1978).

On the North Sea coast of Europe adult twaite shad are relatively common from the Belgium to Denmark. Adult populations are increasing in the known German spawning rivers of the Elbe/Weser (Helcom, 2013; Magath and Thiel, 2013). Twaite shad populations are also increasing in the Baltic, particularly in Poland and Lithuania where the species is classified as in good condition with increasing populations (Helcom, 2013). Genetic analyses of twaite shad from Sizewell demonstrate that they do not originate from the Severn catchment (Jolly et al., 2012). Sabatino and Alexandrino (2012) identified a North Sea twaite shad population with low genetic diversity between fish sampled off Belgium (Scheldt) and Denmark and also the Solway Firth. These analyses identified separation between the Baltic and North Sea populations and the North Sea population would therefore appear to most likely originate from the German rivers of the Elbe/Weser and the Belgian river Scheldt. The twaite shad caught at Sizewell are >1 yr old juveniles to sexually mature adults that are part of widely dispersed feeding population in the North Sea before eventually returning, probably, to European rivers to reproduce.

Figure 9 shows a map of Natura 2000 sites designated for *A. fallax* in the southern North Sea and eastern Channel (https://eunis.eea.europa.eu/species/Alosa%20fallax downloaded November 2019). There are no known UK east coast spawning sites nor HRA designated sites for the species. European sites designated for the species include the estuarine and coastal areas in which the species either feeds as juveniles or transits on its way to southern North Sea feeding grounds as adults. (For example, the entire German North Sea coast is in one of more designated sites). More materially the designated sites include the European rivers where the species is known from monitoring data to successfully spawn. The Elbe in Germany has the largest breeding population with other breeding populations in the Scheldt (Belgium) and sporadically in the Weser (Germany). The other rivers shown in Figure 9 have negligible or no spawning currently.

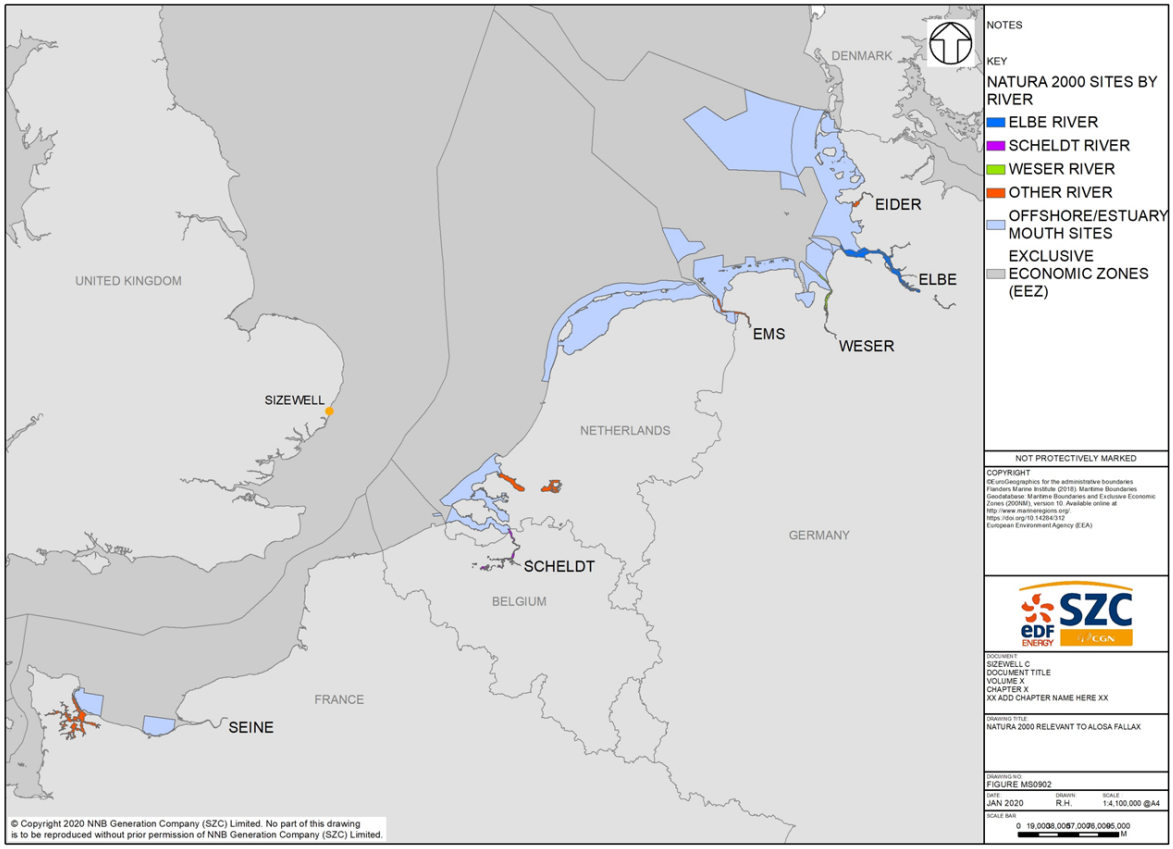


Figure 9 Southern North Sea and Channel Natura 2000 sites designated for Twaite shad

An estimated 2,989 twaite shad will be lost by the proposed SZC station in the absence of any mitigation. This number is reduced to 1,067 individuals with the effect of the LVSE intake heads (FRR mitigation does not increase shad survival as the group is classed as pelagic). There is currently no SSB for the North Sea twaite shad population and no directed fishery, so comparison with landings data does not provide a meaningful assessment.

Losses have therefore been compared with the population estimates available from spring monitoring surveys conducted on the Rivers Elbe and Scheldt. Between 2009 and 2017, an average estimated 5.2 million adult twaite shad pass through these two river systems (BEEMS Scientific Position Paper SPP100). In the absence of mitigation, the losses of twaite shad by SZC represent 0.06 % of these population numbers; with consideration of the LVSE intake head design losses fall to 0.02 % of the population. This represents a negligible effect on the species in these two river systems alone, which would have no effect on the sustainability of the population. It is therefore considered that there would be no significant transboundary effects on any European site designated for *A. fallax*.

6.6.4.1 Identification of 0 group twaite shad in SZB impingement samples

0 group twaite shad have some similarities with juvenile herring (e.g. the scale pattern is similar) which can make identification difficult and the accuracy of identifying 0 group twaite shad has been queried by stakeholder. For the impingement sampling at SZB a very experienced team of fish taxonomists has been deployed including those who have handled juvenile twaite shad from Hinkley Point (where 0 group fish do occur). Based upon published taxonomic information and the experience with the Hinkley Point samples, the

monitoring team are confident that they can reliably distinguish twaite shad from other clupeids (based upon a combination of markings, body depth and eye size).

6.6.5 North Sea Herring and Blackwater Herring

The predicted effect of herring impingement in SZC is a negligible 0.01% SSB or 125 t per annum (Table 14).

The North Sea herring population has the following characteristics:

- North Sea Herring have multiple spawning grounds in the North Sea and eastern English Channel.
- The different spawning populations are not genetically distinct.
- Scientific hypothesis: There is one population which is mixed during summer feeding and which migrates down the North Sea with subsets separating off to breed on route using broadly defined spawning areas as autumn, winter and finally spring spawners. The migration proceeds from Shetland to the eastern Channel and then northwards again (including some movement along the east Anglian coast from the Thames to the Humber).
- Different breeding times produce different growth patterns but the morphometric differences between fish are generally unreliable as an indicator of spawning type (autumn, winter, spring). Patterns in otolith rings are considered the most reliable indicator of spawning type.
- Blackwater herring are a spring spawning stock from February to April that spawn on the Eagle Bank at the entrance to the Blackwater Estuary in Essex and uniquely for North Sea herring have their own catch quota.

The main North Sea and Blackwater herring stocks are very different:

- The Blackwater stock is small with a management target for an SSB of at least 410 t.
- Historically the Blackwater stock was of little commercial interest due to the small size of the fish until the North Sea herring stock collapse that started in 1955 and the consequent reduction in landings. This stimulated commercial interest in the Blackwater stock that grew from 1958 with peak catches in 1972/73 of 600 t shortly followed by stock collapse and a complete closure of the fishery in 1979/80. The fishery reopened in 1980/81 under a management control regime designed to prevent a reoccurrence of overfishing.
- Recruitment to the Blackwater population has been poor in recent years and catches are now limited to 10 t per year for monitoring purposes from an SSB of approximately 200 t. An SSB of approximately 200 t is regarded as the minimum SSB to avoid stock collapse.
- The North Sea herring population has recovered to an SSB of greater than 2M tonnes but markets and fish landings are much lower than they were historically despite the substantial stock size.

Stakeholders have asked why herring impingement at Sizewell is considered to be from the main North Sea stock rather than the Blackwater stock.

As described above, the different stocks cannot be distinguished by genetics and so a weight of evidence approach has been used to ascribe origin to the herring impinged at Sizewell that considered:

- The timing of herring impingement at Sizewell compared with spawning in the Thames Estuary.
- Trends in SZB impingement rates compared with trends in the Blackwater SSB.
- The size of the SZB herring impingement and the implications if these fish had originated from the Blackwater stock.

Sizewell impingement is dominated by adults with numbers peaking in February and March and declining in April (BEEMS Technical Report TR345). Blackwater herring typically spawn at Eagle bank from the end February to mid-April (Fox 2001) and then return to the southern North Sea feeding grounds. The coincident

timing of spawning in the Thames Estuary and impingement at Sizewell indicates the fish caught at Sizewell are unlikely to be from the Blackwater. If Sizewell B was intercepting post spawning fish moving northwards, the peak catches would be expected in April or May with minimal catches in February when Blackwater herring are congregated in the Thames Estuary close to Eagle Bank.

The trend in impingement numbers at SZB has been increasing rapidly (n = 8, tau = 0.93, p = 0.002, Theil-Sen slope = 157488 [2]; Figure 10) which is contrary to the flat, possibly declining trend in the Blackwater SSB (Figure 11).

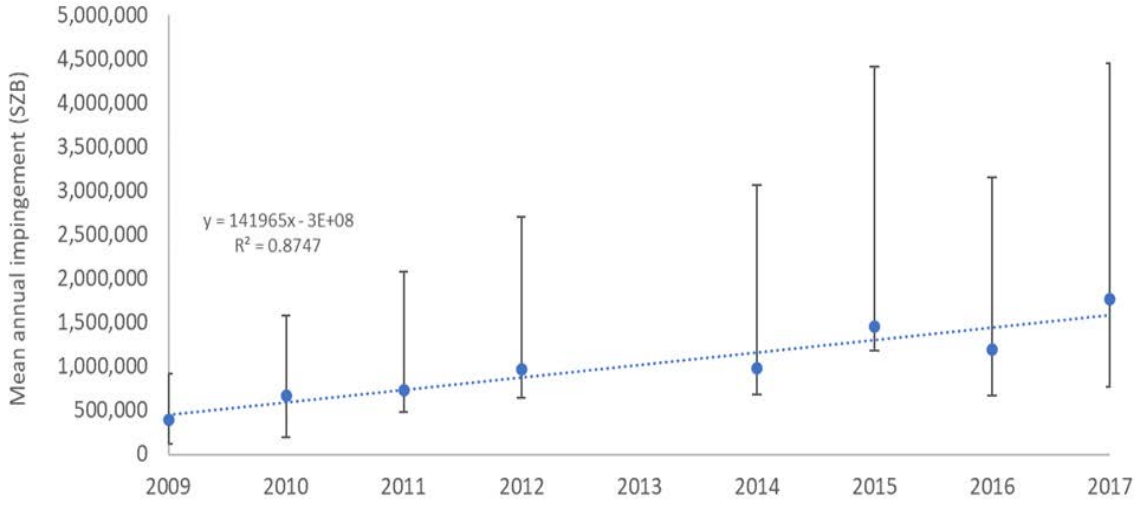


Figure 10 Predicted annual herring impingement numbers at SZB by year (mean ± 95th upper and lower percentiles).

[2] Herring statistics report the Kendall's tau statistic, which measures the rank correlation between impingement and year (with a range from -1 to +1, with 0 when there is no correlation), and the 2-sided p-value from the Mann-Kendall test. The Theil-Sen slope estimate is a median slope estimate for a linear change in impingement per year.

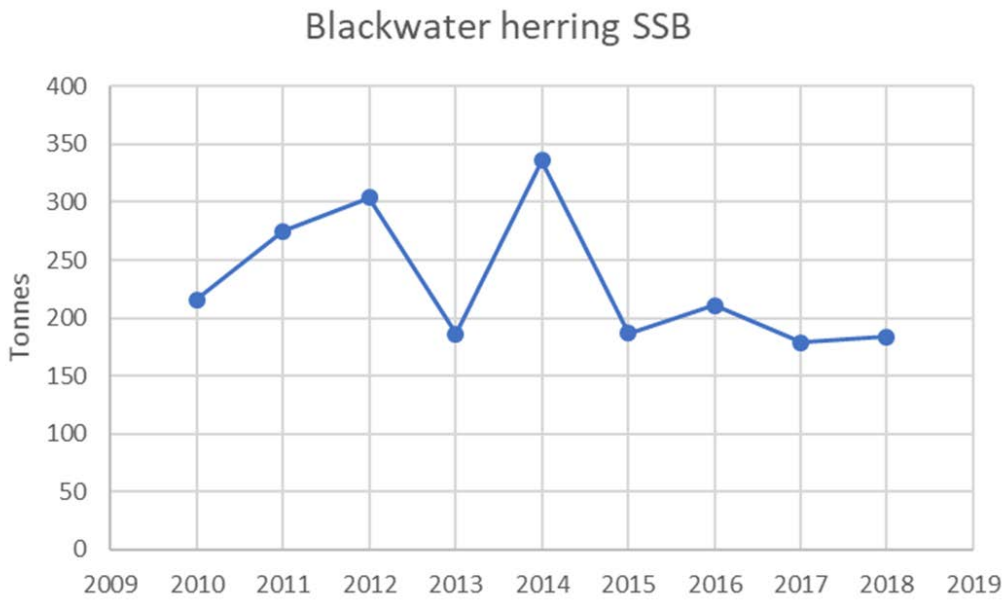


Figure 11 Change in the Blackwater herring SSB from 2010 to 2018 (Source Cefas).

Finally, the Blackwater SSB is less than or equal to 200 t at present (Figure 11). SZB annual mean herring impingement is equivalent to 137 t EAV (Appendix D2). If the SZB impingement was from the Blackwater herring stock that would represent an impingement impact of approximately 68.5% SSB, at the current SSB level would be completely unsustainable and lead to rapid total collapse of the stock to negligible levels and subsequently to minimal impingement at Sizewell. This has not happened, in fact SZB impingement numbers are increasing. This indicates that the SZB impingement is predominantly from the wider North Sea stock, not the Blackwater.

The weight of evidence therefore indicates that Sizewell impingement is from main North Sea stock (as assessed by ICES and used in this report) and not the Thames estuary Blackwater stock.

6.7 SZC predicted entrapment effects (impingement + entrainment)

In addition to the impingement of larger fish, invertebrates and other material, smaller organisms will also be abstracted by SZC but will not be retained on the drum screens. These smaller organisms will be entrained through the cooling water systems. The effects of entrainment on fish and other plankton is assessed in a separate report (BEEMS Technical Report TR318). Results indicated that apart from sand goby, the entrainment impacts for the key taxa present are significantly less than 1 %.

However, it is necessary to consider impingement and entrainment together, as a single entrapment impact for SZC. This is slightly complicated by the fact that the two data sets used for the respective assessments are different. For impingement the dataset spans 9 years, and the predictions are based on modelled mean monthly impingement values, using all year's data. Impingement losses are considered against a mean SSB or landings value for the years 2009 – 2017 (the years of sampling). For entrainment, the predictions are based on a single year's sampling (2010), and the losses were compared against the SSB and landings data for that year.

Entrapment has been estimated here by summing the % losses of impingement and entrainment. Although not a direct comparison of SSB or landings in a given year, given the extremely low entrainment losses of key taxa, when compared against SSB or landings, it would require annual changes in SSBs or landings of 2 or 3 orders of magnitude to significantly affect the combined total losses.

Combined impingement and entrainment estimated have been made for SZC:

a. without mitigation (Table 16);

b. with the proposed LVSE intake heads and FRR fitted (Table 17), and

c. With the impingement assessment with LVSE intakes and FRR systems fitted updated as detailed in section 6.5 for thin lipped grey mullet and bass (Table 18).

Results are similar to those obtained for impingement alone in that without mitigation only seabass, thin-lipped grey mullet, and European eel exceed 1 %. These species are joined by the sand goby for combined entrapment. With the proposed impingement mitigations, only sand goby exceeds 1 % of the stock comparator. However, sand goby is an unexploited short-lived species and in such circumstances the appropriate comparator for negligible effects is with 10% of the SSB. As such the predicted entrapment of sand goby is negligible.

Table 16 Annual mean SZC entrapment predictions (impingement + entrainment) **with no impingement mitigation**. For impingement, losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t) for the period 2009-2017. For entrainment, the worst-case losses have been converted to EAV numbers and weight and calculated as a % of the SSB and landings in 2010 only. Species where the entrapment weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout).

	Impingement						Entrainment						Entrapment			
Species	EAV number	EAV weight	Mean SSB	% of SSB	Mean Landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of landings
Sprat	5,199,288	54.61	220,757	0.02	151,322	0.04	199,715	2.00	225,041	0.00	143,500	0.00	5,399,003	56.61	0.03	0.04
Herring	1,864,587	351.78	2,198,449	0.02	400,244	0.09	23,992	4.18	2,023,720	0.00	187,600	0.00	1,888,579	355.96	0.02	0.09
Whiting	593,040	169.50	151,881	0.11	17,570	0.96	-	-	-	-	-	-	593,040	169.50	0.11	0.96
Bass	158,807	243.10	14,897	1.63	3,051	7.97	36	0.05	20,780	0.00	4,768	0.00	158,843	243.15	1.63	7.97
Sand goby	503,232	0.96	205,882,353	0.24	NA	NA	2,892,198	-	205,882,353	1.40			3,395,430		1.65	NA
Sole	49,802	10.66	43,770	0.02	12,800	0.08	631	0.14	31,358	0.00	12,603	0.00	50,433	10.80	0.02	0.08
Dab	63,810	2.60	NA	NA	5,309	0.05	21,810	0.87	NA	NA	8,279	0.01	85,620	3.48	NA	0.06
Anchovy	172,238	3.57	NA	NA	1,625	0.22	2,869	0.06	NA	NA	727	0.01	175,107	3.63	NA	0.23
Thin-lipped grey mullet	10,618	5.52	NA	NA	120	4.60	-	-	-	-	-	-	10,618	5.52	NA	4.60
Flounder	17,226	1.41	NA	NA	2,321	0.06	2	0.00	NA	NA	3,365	0.00	17,228	1.41	NA	0.06
Plaice	8,536	2.10	690,912	0.00	80,367	0.00	-	-	-	-	-	-	8,536	2.10	0.00	0.00
Smelt	15,834	0.26	50	0.52	8	3.11	-	-	-	-	-	-	15,834	0.26	0.52	3.11
Cod	5,899	15.35	103,025	0.01	34,701	0.04	-	-	-	-	-	-	5,899	15.35	0.01	0.04
Thornback ray	1,438	4.59	NA	NA	695	0.66	-	-	-	-	-	-	1,438	4.59	NA	0.66
River lamprey	2,929	0.23	62	0.37	1	20.77	-	-	-	-	-	-	2,929	0.23	0.37	20.77
Eel	3,031	1.00	79	1.27	14	7.18	-	-	-	-	-	-	3,031	1.00	1.27	7.18
Twaite shad	2,989	0.94	5,161,183	0.04	1	70.22	-	-	-	-	-	-	2,989	0.94	0.06	70.22
Horse mackerel	1,779	0.25	NA	NA	20,798	0.00	-	-	-	-	-	-	1,779	0.25	NA	0.00
Mackerel	473	0.15	3,888,854	0.00	1,026,828	0.00	-	-	-	-	-	-	473	0.15	0.00	0.00
Tope	70	0.48	NA	NA	498	0.10	-	-	-	-	-	-	70	0.48	NA	0.10
Sea trout	10	0.02	NA	NA	39,795	0.02	-	-	-	-	-	-	10	0.02	NA	0.02
Allis shad	3	0.00	27,397	0.01	0	1.31	-	-	-	-	-	-	3	0.00	0.01	1.31

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean Landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of landings
Sea lamprey	5	0.01	NA	NA	NA	NA	-	-	-	-	-	-	5	0.01	NA	NA
Salmon	0	0.00	NA	NA	38,456	0.00	-	-	-	-	-	-	0	0.00	NA	0.00

Table 17 Annual mean SZC entrapment predictions (impingement + entrainment) **considering the effect of the intake head design and with FRR systems fitted**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) are shaded red. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout).

	Impingement						Entrainment						Entrapment			
Species	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sprat	1,856,146	19.50	220,757	0.008831634	151,322	0.01	199,715	2	225,041	0.00	143,500	0.00	2,055,861	21.49	0.01	0.01
Herring	665,658	125.59	2,198,449	0.00571249	400,244	0.03	23,992	4	2,023,720	0.00	187,600	0.00	689,650	129.76	0.01	0.03
Whiting	116,541	33.31	151,881	0.021931594	17,570	0.19	-	-	-	-	-	-	116,541	33.31	0.02	0.19
Bass	31,214	47.78	14,897	0.320749526	3,051	1.57	36	0	20,780	0.00	4,768	0.00	31,250	47.83	0.32	1.57
Sand goby	37,009	0.07	205,882,353	0.02	NA	NA	2,892,198	-	205,882,353	1.40	0	0.00	2,929,206		1.42	0.00
Sole	3,663	0.78	43,770	0.001791388	12,800	0.01	631	0	31,358	0.00	12,603	0.00	4,294	0.93	0.00	0.01
Dab	12,267	0.50	NA	NA	5,309	0.01	21,810	1	NA	NA	8,279	0.01	34,077	1.37	NA	0.02
Anchovy	61,489	1.28	NA	NA	1,625	0.08	2,869	0	NA	NA	727	0.01	64,358	1.34	NA	0.09
Thin-lipped grey mullet	2,087	1.09	NA	NA	120	0.90	-	-	-	-	-	-	2,087	1.09	NA	0.90
Flounder	1,420	0.12	NA	NA	2,321	0.01	2	0	NA	NA	3,365	0.00	1,422	0.12	NA	0.01
Plaice	628	0.15	690,912	2.2347E-05	80,367	0.00	-	-	-	-	-	-	628	0.15	0.00	0.00
Smelt	5,653	0.09	50	0.19	8	1.11	-	-	-	-	-	-	5,653	0.09	0.19	1.11
Cod	1,268	3.30	103,025	0.003202508	34,701	0.01	-	-	-	-	-	-	1,268	3.30	0.00	0.01
Thornback ray	106	0.34	NA	NA	695	0.05	-	-	-	-	-	-	106	0.34	NA	0.05
River lamprey	215	0.02	62	0.03	1	1.53	-	-	-	-	-	-	215	0.02	0.03	1.53
Eel	223	0.07	79	0.093248721	14	0.53	-	-	-	-	-	-	223	0.07	0.09	0.53
Twaite shad	1,067	0.33	5,161,183	0.02	1	25.07	-	-	-	-	-	-	1,067	0.33	0.02	25.07
Horse mackerel	635	0.09	NA	NA	20,798	0.00	-	-	-	-	-	-	635	0.09	NA	0.00
Mackerel	169	0.05	3,888,854	1.38511E-06	1,026,828	0.00	-	-	-	-	-	-	169	0.05	0.00	0.00
Tope	5	0.04	NA	NA	498	0.01	-	-	-	-	-	-	5	0.04	NA	0.01
Sea trout	4	0.01	NA	NA	39,795	0.01	-	-	-	-	-	-	4	0.01	NA	0.01
Allis shad	1	0.00	27,397	0.004537717	0	0.47	-	-	-	-	-	-	1	0.00	0.00	0.47
Sea lamprey	0.36	0.00	NA	NA	NA	NA	-	-	-	-	-	-	0.36	0.00	NA	NA
Salmon	0	0.00	NA	NA	38,456	0.00	-	-	-	-	-	-	0	0.00	NA	0.00

Table 18 Annual mean SZC entrapment predictions (impingement + entrainment) **considering the effect of LVSE intake heads and FRR systems fitted and the corrections to thin lipped grey mullet and bass impingement assessment detailed in Section 6.5 (main changes shown in yellow)**. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight > 1 % of the relevant stock comparator (either SSB or landings – given in bold) would be shaded red (there are none in this Table) with the exception of sand goby where a 10% of SSB or landings comparator has been used. Numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon & sea trout).

	Impingement						Entrainment						Entrapment			
Species	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sprat	1,856,146	19.50	220,757	0.01	151,322	0.01	199,715	2	225,041	0.00	143,500	0.00	2,055,861	21.49	0.01	0.01
Herring	665,658	125.59	2,198,449	0.01	400,244	0.03	23,992	4	2,023,720	0.00	187,600	0.00	689,650	129.76	0.01	0.03
Whiting	116,541	33.31	151,881	0.02	17,570	0.19	-	-	-	-	-	-	116,541	33.31	0.02	0.19
Bass	3,121	4.78	14,897	0.3	3,051	1.16	36	0	20,780	0.00	4,768	0.00	3,157	4.83	0.03	1.16
Sand goby	37,009	0.07	205,882,353	0.02	NA	NA	2,892,198	-	205,882,353	1.40	0	0.00	2,929,206		1.42	0.00
Sole	3,663	0.78	43,770	0.00	12,800	0.01	631	0	31,358	0.00	12,603	0.00	4,294	0.93	0.00	0.01
Dab	12,267	0.50	NA	NA	5,309	0.01	21,810	1	NA	NA	8,279	0.01	34,077	1.37	NA	0.02
Anchovy	61,489	1.28	NA	NA	1,625	0.08	2,869	0	NA	NA	727	0.01	64,358	1.34	NA	0.09
Thin-lipped grey mullet	2,087	1.09	600	0.18	120	0.90	-	-	-	-	-	-	2,087	1.09	0.18	0.90
Flounder	1,420	0.12	NA	NA	2,321	0.01	2	0	NA	NA	3,365	0.00	1,422	0.12	NA	0.01
Plaice	628	0.15	690,912	0.00	80,367	0.00	-	-	-	-	-	-	628	0.15	0.00	0.00
Smelt	5,653	0.09	50	0.19	8	1.11	-	-	-	-	-	-	5,653	0.09	0.19	1.11
Cod	1,268	3.30	103,025	0.00	34,701	0.01	-	-	-	-	-	-	1,268	3.30	0.00	0.01
Thornback ray	106	0.34	NA	NA	695	0.05	-	-	-	-	-	-	106	0.34	NA	0.05
River lamprey	215	0.02	62	0.03	1	1.53	-	-	-	-	-	-	215	0.02	0.03	1.53
Eel	223	0.07	79	0.09	14	0.53	-	-	-	-	-	-	223	0.07	0.09	0.53
Twaite shad	1,067	0.33	5,161,183	0.02	1	25.07	-	-	-	-	-	-	1,067	0.33	0.02	25.07
Horse mackerel	635	0.09	NA	NA	20,798	0.00	-	-	-	-	-	-	635	0.09	NA	0.00
Mackerel	169	0.05	3,888,854	0.00	1,026,828	0.00	-	-	-	-	-	-	169	0.05	0.00	0.00
Tope	5	0.04	NA	NA	498	0.01	-	-	-	-	-	-	5	0.04	NA	0.01
Sea trout	4	0.01	NA	NA	39,795	0.01	-	-	-	-	-	-	4	0.01	NA	0.01
Allis shad	1	0.00	27,397	0.00	0	0.47	-	-	-	-	-	-	1	0.00	0.00	0.47

Species	Impingement						Entrainment						Entrapment			
	EAV number	EAV weight	Mean SSB	% of SSB	Mean landings	% of landings	EAV number	EAV weight	SSB 2010	% of SSB	Landings 2010	% of landings	EAV number	EAV weight	% of SSB	% of Landings
Sea lamprey	0.36	0.00	NA	NA	NA	NA	-	-	-	-	-	-	0.36	0.00	NA	NA
Salmon	0	0.00	NA	NA	38,456	0.00	-	-	-	-	-	-	0	0.00	NA	0.00

6.8 Contextualising Sizewell C entrapment losses against fisheries landings and discards

To place the predicted fish losses due to entrapment by SZC (Table 18) into context two analyses are presented:

i. Table 19 shows a comparison for those stocks where data are available between mean fishery landings as a percentage of SSB and SZC entrapment as a percentage of SSB for the period 2009-2017.

ii. Table 20 shows discarded fish weight as a percentage of landed weight by year for commercially exploited species compared with ICES records of fishery discards. The same table also shows predicted mean SZC entrapment weights as a percentage of the mean landings for each species.

Table 19 Comparison of mean fishery landings as a percentage of SSB with predicted SZC mean entrapment as a percentage of SSB for the period 2009-2017.

Species	Fisheries Landings as % SSB	SZC entrapment losses as % of SSB	Notes
Sprat	69%	0.01%	
Herring	18%	0.01%	Commercial market reduced after stock collapse and subsequent recovery
Whiting	20%	0.02%	
Bass	20%	0.03%	Including behavioural considerations.
Sole	29%	0.00%	
Plaice	12%	0.00%	
Cod	45%	0.00%	
Mackerel	26%	0.00%	
River lamprey	2%	0.03%	SSB is of Derwent population only. Landings and effort are restricted by licences.
Eel (silver eel)	18%	0.047%[3]	SSB is of Anglian RBD only.

Table 19 shows that the fishery impact on each stock is orders of magnitude greater than that of SZC entrapment.

[3] SZC effect overestimated due to inflated EAV see Section 7.6.4.

Table 20 Discards by year as a percentage of landed fish weight compared with predicted SZC entrapment as a percentage of landed fish weight.

Year	Cod	Sole	Plaice	Dab	Whiting	Flounder	Horse mackerel	Mackerel
2009	62.8	4.3	39.5	474.7	54.5	47.9	-	3.6
2010	33.6	8.3	35.5	513.2	73.5	97.5	-	0.4
2011	27.2	7.2	33.6	599.0	66.1	55.7	-	0.6
2012	26.9	9.0	44.2	746.9	73.6	55.0	-	0.5
2013	33.8	11.2	25.2	808.9	46.4	80.9	-	0.1
2014	30.4	6.5	39.6	1064.7	66.6	59.1	-	0.1
2015	33.5	6.9	35.1	932.2	109.8	69.7	20.0	0.0
2016	32.0	5.4	30.4	881.2	107.2	35.9	11.1	0.2
2017	23.1	5.8	31.5	875.9	91.0	47.3	9.1	0.1
SZC Entrapment (% of landed weight)	0.01	0.01	0.00	0.02	0.19	0.00	0.00	0.00

For commercially important species, SZC entrapment losses are lower than <1% of landings. Discards as a percentage of landings weights varied dramatically depending on the species between 2009-2017. However, entrapment losses are at least two orders of magnitude lower than the proportion of landed fish that was discarded annually between 2009-2017. Expressing the results as weights can provide an even more compelling illustration and for example, the mean weight of cod discarded between 2009 and 2017 was 12,980 t whereas the predicted mean SZC entrapment loss for the same period was 3.3 t.

7 Evaluation of uncertainty in impingement predictions

The confidence of impingement predictions is determined by the underlying assumptions used to parameterise the assessment and the appropriateness of the comparator. Section 5 details the appropriateness of applying ICES stock units with associated SSBs and fisheries landings to determine effect thresholds. Section 8 further considers the potential for local scale effects and concludes that whilst local-level effects may be estimated, ICES stock areas represent the most appropriate scale of assessment for contextualising population effects on mobile receptors in an open system.

In this section we consider the sensitivity of impingement predictions and effects to uncertainties in the defining parameters of the assessment including LVSE efficiency, FRR mortality, EAV values and changes in distribution of fish. It should be noted that Section 6 provides impingement predictions for all mitigation scenarios to demonstrate the relative effectiveness of mitigation , these include:

- unmitigated impingement;
- LVSE only mitigation;
- FRR only mitigation; and
- LVSE and FRR mitigation.

7.1 Sensitivity assessment

The first step of the sensitivity assessment process considers which species are potentially susceptible to uncertainties in the assessment of effects. At the fundamental level, unmitigated effects presented in Table 11 and Appendix C.1 demonstrate the mean and 95th percentile impingement predictions in the case of no FRR or LVSE intakes.

Except for bass, thin-lipped grey mullet and European eel, the mean and 95th percentile upper impingement estimates for key species remained below the appropriate 1% threshold for SSB or fisheries landings. This indicates that, in the case of most key species, the assessment is in-sensitive to uncertainties in impingement parameters and whilst impingement numbers may be subject to uncertainty there would be no significant population level effects from Sizewell C. For example, sprat, herring, and whiting are numerically the highest impinged species and contribute 60% of the unmitigated losses (EAV weight), mean losses for these species equate to 0.02%, 0.02% and 0.11% of SSB, respectively. Mitigation measures would reduce effects further (Table 21).

Table 21 Mitigation parameters and maximum impingement efficiency.

Species group	LVSE factor	FRR survival [1]	Mitigation efficiency*
Pelagic	0.357	0% [2]	2.8
Demersal	0.357	50%	5.0
Epi-benthic	0.357	80%	14

[1] Mitigation efficiency accounts for band screen mortality but does not account for species-specific mortality in the trash rack (Appendix C.3).
[2] In the case of pelagic species, such as the clupeids, the assessment assumes no benefit from the FRR. This is a reasonable, yet precautionary assumption, as species such as herring, sprat and smelt typically have low survival rates following passage through drum and band-screens (<10%, Environment Agency, 2005).

Bass (1.63%) and European eel (1.27%) exceed the 1% SSB threshold, whilst thin-lipped grey mullet (4.60%) exceeds 1% of landings. There is no commercial fishery for thin-lipped grey mullet and even by applying a highly precautionary estimate that the undirected fisheries harvest as much as 20% of SSB

(Section 6.5), unmitigated losses are below the 1% SSB threshold. Therefore, sensitivity checks focus on eel and bass.

It is noted that entrapment effects account for 1.42% of the sand goby SSB, however, entrainment (1.40%) is the primary source of mortality. As entrainment effects are not sensitive to LVSE or FRR parameterisation, and all sand gobies are given a precautionary EAV of 1 gobies are not considered further in this section. Furthermore, the effects on this small, short-lived species are considered negligible to population sustainability.

7.2 Sensitivity of impingement predictions on bass and eels

In the absence of any mitigation, and notwithstanding bass distribution the effects of Sizewell C are anticipated to represent 1.63% of bass SSB. In the case whereby yellow eels impinged at Sizewell are assumed to be silver eels and given an EAV of 1 impingement losses equate to 1.27% of the Anglian River Basin District silver eel biomass. This section considers uncertainties surrounding EAV estimates, FRR survival, bass distribution within the Greater Sizewell Bay (GSB) and LVSE benefits.

7.2.1 EAV sensitivity

The necessary biological and population data required to derive an EAV for eel is not available, as such a worst-case value of 1 has been assumed. The biologically sampling of the eye index of impinged individuals at SZB (n = 89) indicates all eels impinged were yellow eels and would have had an EAV of < 1 due to the natural mortality experienced by yellow eels in their many year growth period before maturation to silver eels. No silver eels were impinged at SZB indicating the EAV of 1 is a large overestimate.

Natural mortality of yellow eels is estimated to be 13% per annum (Dekker 2000). The yellow eels abstracted at SZB were in the age range 2 to >25 years and the majority would have remained at the yellow stage for many years before maturation. If the yellow eels impinged at Sizewell would have spent an average of 5 years before maturation, the predicted SZC impingement effect would be **reduced by 50%** of the RBD silver eel biomass (Section 6.6.1). Table 24 shows the effects of the EAV on eel SSB.

The EAV of seabass has been investigated in terms of the sensitivity to EAVs based on 50% maturity rather than 100 % applied here, and the appropriateness of applying the Environment Agencies proposed spawning production foregone approach. BEEMS Scientific Position Paper SPP102 concluded that there is no benefit from the use of SPF EAVs in impingement assessment and that their use is considered likely to increase, rather than decrease, uncertainty in assessments. In the case of the 50% maturity the EAV for bass would **increase by 3.6%** from 0.224 to 0.232 resulting in an increase in unmitigated impingement losses from 1.63% SSB to 1.69% SSB (Section 4.8.2). The change in EAV is considered inconsequential and is not considered further.

7.2.2 FRR mitigation

The Sizewell C FRR system mirrors the iterative Hinkley Point C design that was developed in consultation with the Environment Agency and based on BAT. Due to the reduced tidal range there is less raising required at Sizewell and two dedicated FRR outfalls, servicing each reactor unit, reduces the hydraulic complexity and prevent the requirement for an Archimedes screw. As such the Sizewell C FRR is anticipated to perform as well as the system designed for Hinkley Point C and possibly better.

The length frequency distribution of bass means most individuals would pass through the drum screen where 50% survival is assumed, whilst approximately 9% would be directed through the band screens (Appendix C.4). Survival through the band screens is precautionarily assumed to be 0% (Section 4.7.2). Varying survival rates through the different flow streams means approximately 45% of abstracted bass are predicted to survive impingement. The **45% survival** of bass is considered reasonably precautionary in comparison to the 50-80% survival reported for demersal species in Environment Agency (2005) and Turnpenny (2011). On this basis the FRR alone would be anticipated to reduce unmitigated losses to 0.86% of SSB.

European eel are anticipated to have **80% survival** through the FRR system resulting in effects falling to 0.26% of Anglian River Basin District silver eel biomass.

Table 23 and Table 24 demonstrates the effects of the FRR system along with other mitigation options for bass and eel, respectively.

7.2.3 LVSE mitigation

Following Environment Agency best practice guidance, the design of SZC would include LVSE intakes. The LVSE intakes are capped structures with the intake surfaces orthogonal to the direction of the tidal flows, thereby reducing the cross-sectional area available to intercept fish being transported in the tidal flows. Reducing the intercept area to fish impingement and minimising the velocity at face would have beneficial effects on fish impingement. However, there are currently no LVSE intakes in operation and the exact species-specific impingement factor can therefore not be measured (Environment Agency, 2020).

The approach for determining the LVSE factor is detailed in Scientific Position Paper SPP099 and is based on Environment Agency methodologies. The reduction in cross-sectional area combined with the low intake velocity is predicted to reduce the number of fish abstracted per cumec up to a factor of **0.357** when compared to SZB, which is fitted with conventional omnidirectional intake heads with intake velocities that exceed the ability of most fish to avoid being abstracted. Table 23 demonstrates effects with and without LVSE mitigation.

The LVSE intakes result in a tidally averaged worst case of approximately 0.7m either side of the intake faces experiencing velocities of greater than 0.3 m/s (BEEMS Scientific Position Paper SPP099). Swimming capabilities and behavioural responses to such side current velocities will determine the effectiveness of the LVSE design. Table 24 demonstrates effects with and without LVSE mitigation.

7.2.4 Juvenile bass attraction to thermal discharges

Juvenile bass are attracted to the warm water outfalls of power stations in winter (Jennings *et al.,* 1991). In the period of lowest seawater temperatures (December to March) bass impingement at Sizewell B is 96.1% of the annual total (Section 4.1.1) with 99.5% of bass impinged in Q1 and Q2 (BEEMS Technical Report TR339). Bass impingement at SZB consists mostly of juvenile fish with few mature adults (Appendix 4E).

The density of bass has been shown to be 20 times greater inshore of the Sizewell-Dunwich Bank in the vicinity of the SZB thermal plume than offshore of the Bank (BEEMS Technical Report TR380). To account for the distribution of bass, impingement predictions have assumed Sizewell C would abstract **10% of the bass in comparison to SZB** (Section 6.5). Stakeholders have questioned how the distribution of bass, and therefore impingement rates, at SZC may change following the decommissioning of SZB. The following extract is from BEEMS Technical Report TR318:

“Sea bass larvae move inshore over a period of 2−3 months and, at around 15 mm long, actively swim into coastal creeks, estuaries, backwaters and shallow bays from June on (Jennings and Pawson, 1992; Reynolds et al., 2003). Juvenile sea bass remain in these nursery areas through their first and second years, after which they migrate to overwintering areas in deeper water, and may return in summer to the larger estuaries for 3–5 years, depending on growth, until they are around 36 cm long (Pawson et al., 1987). In contrast to the regular migrations of adult sea bass, a substantial proportion of the juvenile population emigrates from its respective stock areas and disperses throughout large parts of the population’s distribution range (Pickett et al., 2004).

Climate and environment effects on the early life history of sea bass have significant implications for their distribution and abundance. There is a positive relationship between seawater temperature and the growth of the 0-group in summer (Pawson, 1992; Reynolds et al., 2003), and survival through the first winter is reduced at temperatures below 5–6 °C (Kelley, 2002; Lancaster, 1991). This may well explain the attraction for first-year sea bass of warm water effluents from coastal power station cooling systems, especially in autumn and winter (Kelley, 1986; Pawson and Eaton, 1999). This is likely to affect survival near Sizewell, where the warmed water might promote a longer growth season than elsewhere in the North Sea (without thermal warming), and also serve to retain juvenile sea bass in water that may be used for cooling.”

Table 22 shows the levels of warming at the SZB intakes and SZC intakes during December to March, when 96.1% of bass are impinged at SZB, for the current situation, when both stations are operational and once SZB comes offline. When SZB is operating alone the inshore waters at the intakes receive a median warming of 1.2°C and are 1.3°C warmer than the SZC intake locations. When both stations are operational the temperature differential drops to 0.9°C between the inshore and offshore intakes with the SZB intakes receiving a thermal uplift of 1.8°C compared to 1.0 - 1.1°C at SZC. Once SZB is decommissioned, the SZC thermal uplift at the offshore intakes is 0.9 - 1.0°C whilst a thermal uplift of 0.5°C is still felt at the SZB intakes within the Sizewell-Dunwich Bank. In this future scenario the temperature differential reverses with the SZC intakes 0.4°C warmer.

Table 22. Median bottom temperatures and thermal uplifts (°C) at the Sizewell B and C intakes under different operating scenarios during peak bass abundance, between 1st December – 31st March (data extracted from GETM).

Position	**Parameter (°C)** (1st December – 31st March)		**Operating Scenario**		
			SZB alone	SZC + SZB	SZC alone
SZB intakes	Temperature	Median	6.3	7.0	5.6*
	Uplift	Median	1.2	1.8	0.5
SZC I3b (Unit 1)	Temperature	Median	5.0	6.1	6.0*
	Uplift	Median	0.1	1.1	1.0
SZC I3b (Unit 2)	Temperature	Median	5.0	6.1	6.0*
	Uplift	Median	0.1	1.0	0.9

* Future absolute temperatures are not withstanding the of climate change. An in-combination climate change assessment with thermal discharges is provided in BEEMS Technical Report TR306.

During bass distribution surveys off Sizewell in February 2016, the largest catches occurred in water temperatures of 6.0, 6.2 and 6.8 ˚C. When survey catches were adjusted for date a significant trend was observed between catch and near-bed temperature, with water temperature explaining between 40 and 60% of the variance in the data. Low catch rates were observed at all offshore survey stations with 95% of bass caught in-shore of the Sizewell-Dunwich Bank suggesting the inshore distribution of juvenile bass was a wider phenomenon on the Suffolk coast and not just related to the immediate area of the plume (BEEMS Technical Report TR380). This corresponds to the established behaviour of juvenile bass utilising inshore coastal waters, where other factors such as food availability and predation threat are likely to drive distribution. Therefore, the small thermal benefit afforded in the deep waters at the offshore intakes once SZB is decommissioned and SZC is operating alone is unlikely to dramatically change bass impingement risks at SZC.

Furthermore, SZB may remain operational until the 2050s if the operational life is extended as has occurred at other stations. Attempting to predict the future distribution of juvenile bass is undermined by uncertainty. Over these time scales, the distribution and abundance of warm-water bass are subject to natural changes and climate change. Climate warming resulted in a range expansion of bass in UK waters in the 1990s and early 2000s (Wright *et al.,* 2020). Such effects can be seen when CIMP data from 2009-2017 is compared to the impingement data from a single year from Sizewell A (1981-1982), when bass impingement was 230-fold lower (Section 11.1). From a strictly local effects perspective, the end of operations of SZB would reduce the pressure of entrapment of bass in comparison to both stations operating.

7.3 Assessment sensitivity conclusions

The mean and 95th percentile upper impingement estimates for the key species remained below the appropriate 1% threshold for SSB or fisheries landings, for all species except bass and European eel. As such, in the absence of mitigation no significant effects on stock viability are predicted. The mitigation options proposed will reduce the impingement further, thereby minimising effects.

Whilst it is acknowledged that it is not possible to predict the precise mitigation efficiency on a species-by-species basis, the assessment for most species is in-sensitive to uncertainties in mitigation parameters. Whilst accurately predicting impingement numbers with different mitigation options may be subject to uncertainty there would be no significant population level effects from Sizewell C predicted for most species with an unmitigated station. To allow full transparency this report details effects for each mitigation measure individually.

Only bass and European eel exceed the relevant 1% effect threshold without mitigation. In the case of bass a combination of the LVSE intakes, FRR mitigation and applying knowledge of the distribution of bass within the GSB is anticipated to reduce effects to well below 1%. Depending on the efficiency of the mitigation measures effects of between 0.03 up to a maximum of 0.3% of SSB may be predicted (Table 23).

Table 23. Sensitivity of the assessment of effects on unmitigated bass SSB (1.63%) to impingement mitigation parameters.

Bass	Unmitigated	FRR (0.55)	LVSE (0.357)	Distribution (0.10)
Unmitigated	**1.63**	0.90	0.58	0.16
FRR	0.90	NA	0.32	0.09
LVSE	0.58	0.32	NA	0.06
Distribution	0.16	0.09	0.06	NA
LVSE + FRR	**0.32**	NA	NA	**0.032**

In the absence of mitigation, losses of eel due to impingement equates to 1 t per annum or 1.27% of the Anglian RDB SSB. Eel are a robust species and 80% survival is anticipated through well designed FRR systems. The efficiency of LVSE intakes for poor swimming species such as European eel has been questioned (Environment Agency, 2020) but the benefit from LVSE intakes are considered to be mostly due to the reduction in cross sectional area due being mounted orthogonal to the tidal current and the short range zone of influence on local hydrodynamics. Assuming no LVSE benefit at all, effects from the FRR alone equates to 0.13% of SSB if the more appropriate yellow eel EAV is applied (Table 24).

It can be concluded that the mitigation measures proposed will have a beneficial effect on reducing impingement rates. In all cases the proposed development would not have any significant effect on the sustainability of fish stocks.

Table 24. Sensitivity of the assessment of effects on unmitigated European eel Anglian RDB SSB (1.27%) to impingement mitigation parameters.

Eel	Unmitigated	EAV	FRR (0.206)	LVSE (0.357)
Unmitigated	**1.27**	0.64	0.26	0.45
EAV	0.64	NA	0.13	0.23
FRR	0.26	**0.13**	NA	0.09
LVSE	0.45	0.23	0.09	NA
LVSE + FRR	0.09	**0.047**	NA	NA

8 Consideration of local-level effects on the fish assemblage at Sizewell

In Section 6.7 the predicted Sizewell C entrapment effects on fish populations have been assessed against ICES derived spawning stock biomasses (SSB), or international landings as a precautionary proxy for SSB. Comparisons against SSBs is the internationally recognised best practice way that the much larger effects of fishing at either a fleet or individual boat level on fish stocks are assessed.

Sizewell C stakeholders have indicated that in principle they agree with this assessment methodology for effects on fish at the stock level but questioned the application of ICES stock units for assessing potential effects of a coastal power station on local populations. The concern pertains to the geographical scale of ICES stock units resulting in impacts from the station having small perceived effects on SSB and an inability to determine local effects should they occur.

Cefas' position remains that ICES stock units, as described in Section 4.10, are the most relevant scale of assessment for contextualising effects in relation to the sustainability of fish within safe biological limits. This position is consistent with that of the Marine Management Organisation (MMO). The MMO defines a stock as:

"*A fish stock refers to a fish population that is isolated from other stocks of the same species*"

(MMO, 2020).

In the Relevant Representation document of 30th September 2020, the MMO agreed in principle with the use of ICES stock areas to contextualise effects; *"The MMO conclude that the use of ICES stock areas for commercial fish species represents the current best scientific evidence available. There is currently no robust information that would support use of more local stock areas in the assessment".*

There are no isolated fish populations at Sizewell, fish live and move in an unconstrained coastal environment with most species undertaking wide spatial migrations throughout the year. As such, ICES stock units represent the best available evidence for assessing the impacts of the proposed development in relation to stock sustainability. However, assessments of effects on fish must consider aspects beyond stock sustainability and further information on the potential for localised effects of the station is provided in Version 7, herein.

This section of the assessment considers the potential for localised effects in relation to the proposed development in further detail. To address these concerns, the following points are considered:

- Can localised effects on fish be estimated?
- What local-level effects may be important at Sizewell?
- Is there evidence of localised impingement effects from other sites?
- How sensitive is the assessment to changes in stock area?
- How would future entrapment change in the face of changes in stock dynamics?

8.1 Can localised effects on fish be estimated?

Local level depletion of fish has been specifically considered in BEEMS Scientific Position Paper SPP103 by applying a simplified conceptual model of impingement relative to tidal replenishment. Whilst the model necessitates making assumptions relating to fish distribution and behaviour, it is possible to approximate the likely size of effects at a range of scales.

The assessment determines the predicted depletion of different fish species due to SZC acting alone and in-combination with SZB at the following scales

- the Greater Sizewell Bay (GSB) and the tidal excursion;
- ICES Statistical Rectangle 33F1 containing Sizewell, and;
- ICES Statistical Area 4c (IVc), containing 33F1.

Local depletion was computed for 365 days and was based on predicted mortality of different species with full mitigation in place. ICES Statistical Area 4c was considered to be a hard boundary for the purposes of this assessment i.e. there was no exchange beyond this areas as such all effects of depletion would be within this area. Table 8 demonstrated that in the case of most species the relevant stock area is appreciable larger than ICES Statistical Area 4c. The simplified model is not appropriate for determining sustainability of fish stocks, which follows ICES best practice and it detailed in Section 6. However, local depletion of representative pelagic, demersal and epibenthic species during their seasonal occurrence at Sizewell is estimated allowing further information to support HRA and WFD assessments.

The report focuses on the smallest unit incorporating the GSB and tidal excursion where local depletion would be concentrated. BEEMS Scientific Position Paper SPP103 found that:

1. With both stations operating together, local depletion reaches a plateau within 50 days, after which point abstraction losses and emigration are balanced by immigration into the local area. Depletion continues to rise at the scale of 33F1 and, to a lesser extent, within ICES 4c as the wider area dilutes impingement losses.

2. Local depletion is sensitive to the replenishment rate. However, even modest rates of replenishment of 5% per day result in modest local depletion at the scale of the GSB and tidal excursion.

3. Replenishment rates of 10% per day of fish within the GSB and tidal excursion result in minor reductions in pelagic species such as sprat and herring of 1.5% per day during periods of seasonal abundance with both Sizewell C and Sizewell B operating with full mitigation.

4. Demersal and epi-benthic species incur lower impingement mortality and species such as whiting and Dover sole are predicted to incur local depletion of 2.4% and 0.9%, respectively with both Sizewell C and Sizewell B operating with full mitigation.

5. Pelagic species such as smelt spend long periods foraging in the inshore waters off Sizewell. At just 5% daily exchange across the boundaries, effects within the GSB and tidal excursion result in 3.1% depletion of smelt compared to 1.5% local depletion with a 10% exchange.

6. Local depletion due to impingement is orders of magnitude below natural variability in abundance to which predator-prey relationships are adapted to. Therefore, impingement from Sizewell B and Sizewell C would not have any adverse food-web effects on designated features of HRA sites. This point is considered further in Section 8.2.3.

7. Local effects are not predicted to cause a deterioration in the WFD fish status of the Alde & Ore water body as local depletion is well below the sensitivity of the TFCI. This point is considered further in Section 8.2.4.

Figure 12 Show the local depletion of sprat with both stations operation in-combination with full mitigation, the output is based on 121 days reflecting the period from December to March when sprat are most abundant and assuming 10% daily exchange across the boundaries of the assessment cells.

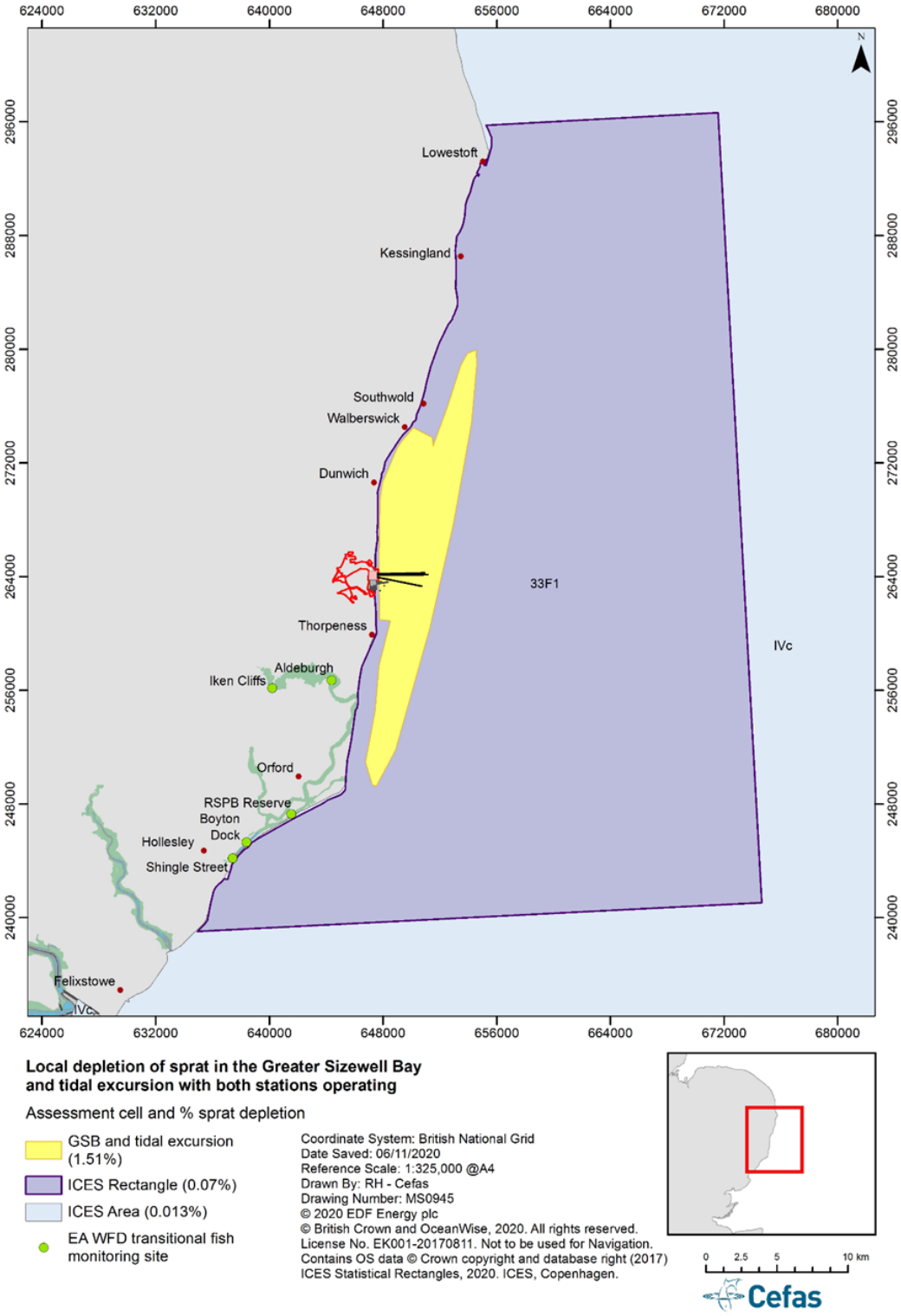


Figure 12 Areas of localised depletion for sprat assessed with both stations acting in-combination during the period December to March assuming 10% exchange across assessment cell boundaries. The GSB + tidal excursion (yellow), ICES statistical rectangle 33F1 (purple) and part of ICES Statistical Area 4c (blue) are shown.

8.2 What local effects of SZC entrapment might be important?

Having established the stock level effects of entrapment from Sizewell C (Section 6) and estimated minor local depletion from both stations operating in-combination (Section 8.1) we consider what local effects may be important. The potential for local effects is in part determined by the geography and hydrodynamics of the assessment area, and the ecology of the species in question. Site utilisation by different life-history stages, migratory routes, the degree of population isolation and genetic mixing and immigration versus emigration rates all determine the potential for effects on local populations. This section considers the potential for:

- Effects on migratory species with high spawning site fidelity.
- Effects on spawning or nursery habitats.
- Localised effects on fish availability in terms of their importance as prey species.
- Potential effects on the transitional fish status of the Alde-Ore.

8.2.1 Effects on spawning migrations.

Some species of fish impinged at Sizewell display diadromous behaviour, that is they migrate between freshwater and the marine environment to spawn. Catadromous species, such as European eel migrate from freshwater to the sea to spawn returning to estuaries as glass eels. Anadromous species such as smelt migrate to estuaries to spawn. Therefore, should the power station impede migratory routes the potential for localised impacts on fish populations exists. This would be particularly problematic should the power station be built in a semi-enclosed system where the risk of abstraction presented a non-trivial proportion of the migratory cross-sectional area.

At Sizewell, the intake headworks are located 3km offshore in an open system. Fish are only likely to be impinged on the migratory route past the station if they move into the zone of influence of the intake heads. For SZC, a tidally averaged worst case of approximately 0.7m either side of the intake faces would experience an intake velocity of greater than 0.3 m/s (BEEMS Scientific Position Paper SPP099). Compared with the spatial domain inshore, and offshore, of the intakes that migrating fish can move within; this abstraction risk zone is very small.

In the case of smelt, a spawning population may occur within the Alde-Ore Estuary. The mouth of the estuary is located 25 km south of the proposed development. Seasonal patterns of impingement have been observed in the Sizewell B CIMP. The vast majority of smelt are impinged between May and September and are dominated by smaller year group 1 and 2 fish for summer feeding. In February and March, when smelt migration to freshwater occurs, impingement rates are low but impinged individuals are larger (BEEMS Scientific Position Paper SPP101). It is likely these fish represent migratory adults. Whilst, the proposed development would introduce an additional abstraction risk to smelt migrating to the Alde-Ore Estuary the probability of impinging a significant proportion of the migratory population is very small given adults returning to the Estuary, 25 km south of the proposed development, may arrive from 180° of open coastline. Impingement rates of smelt at Sizewell B have shown no trend in the assessment period (BEEMS Scientific Position Paper SPP108) and natural mortality and fishing pressure would far exceed the effects of the proposed development.

8.2.2 Effects on spawning or nursery habitat in the vicinity of Sizewell.

If a reduction in local fish numbers adversely affected a spawning or nursery habitat in the vicinity of Sizewell localised effects could occur. Furthermore, it these areas were critical to the sustainability of a stock implications could be felt further afield.

The proposed development coincides with areas identified as spawning grounds of Dover sole, and ‘low intensity’ nursery grounds for plaice (Ellis *et al.,* 2012). Ichthyoplankton surveys completed as part of the fish characterisation identified Dover sole and Anchovy as contribution a total of 85% of measured egg numbers at Sizewell. However, the measured egg density indicates that the area at Sizewell is not of great importance to the species which are geographically widely distributed (BEEMS Technical Report TR318).

'High intensity' nursery grounds for herring and 'low intensity' nursery grounds for whiting, cod, and thornback ray occur off Sizewell on the east coast (Ellis *et al.,* 2012). However, nursery grounds for these species are widely distributed. The Sizewell region provides some nursery habitat predominantly for gobies and sprat with 86% of measured larval and juvenile fish numbers were associated with these species. Again the measured densities indicate that the area is of low importance for these two widely distributed species (BEEMS Technical Report TR318). For all of the species predicted to be entrained by SZC the predicted stock level effects are negligible. In the case of Dover sole, entrainment of eggs and larvae accounts for approximately 600 fish per annum, less than 0.0005% of SSB (BEEMS Technical Report TR318).

Unlike in some estuarine environments, there are no unique features or resources that fish populations are dependent upon in the vicinity. For example, estuaries often function as nursery areas for young fish but the Sizewell area does not have an extensive nursery role for most species. This point is illustrated by comparing the age classes of fish impinged at Sizewell B and Hinkley Point B in the Severn Estuary. At Sizewell older and larger fish are recorded compared with the fish taken from the Severn Estuary at Hinkley Point (Table 25). From Table 25 it can be seen that for the typical species listed, only Thornback Ray are larger at Hinkley Point than at Sizewell and for herring, whiting and cod, in particular, Sizewell has much lower numbers of 0 group (less than 1 year old) fish. The lower the EAV, in the range 0 to 1, the greater the proportion of juvenile fish. EAVs of around 0.1 or below consisting predominantly of 0 or possibly 1 group fish.

Table 25 Differences in age and maturity of fish caught at Sizewell and Hinkley Point as reflected in Equivalent Adult Values (EAV) where an EAV of 1 is a mature adult.

Species	SZ EAV	HP EAV
Sprat	0.751	0.556
Herring	0.715	**0.113**
Whiting	0.356	**0.098**
Bass	0.224	0.121
Dover sole	0.213	0.236
Cod	0.359	**0.018**
Thornback Ray	0.193	0.339
Plaice	0.345	0.192

Note: EAVs are from BEEMS Technical Reports TR383 (Sizewell) and TR426 (Hinkley Point) respectively.

For most species, the importance of the Sizewell Bay area for fish spawning and nursery habitat is of relatively low importance and there are much more important habitat areas over the southern North Sea and beyond (BEEMS Technical Report TR345). The fish assemblage at Sizewell reflects the seasonal migrations of fish into and out of the area.

8.2.3 Prey availability for predators

The local ecosystem is founded upon predator-prey relationships and localised depletion of a prey resource could adversely affect predator sustainability e.g. HRA protected marine birds predating on fish. Food webs are routinely subject to high levels of natural variability and predators have evolved adaptation strategies to cope including prey species switching and changes to foraging behaviour. Predators would be sensitive to sustained changes in prey abundance if their foraging range is small and their ability to change their foraging behaviour or dietary composition is limited.

Prey species of designated SPA, Ramsar and SAC species of relevance to the proposed development are detailed in BEEMS Technical Report TR431. The most seasonally abundant prey species at Sizewell that are considered important prey items for designated Annex II species include the pelagic species; sprat,

herring and anchovy, epi-benthic species such as plaice, Dover sole, flounder and gobies and demersal whiting and juvenile seabass.

Local scale depletion due to Sizewell C and Sizewell B acting in-combination is predicted to be less than 2.5% for demersal species, 1.5% for pelagic species and less than 1% of epibenthic species during periods of seasonal presence (Table 26) assuming 10% daily replenishment of fish from the wider area (BEEMS Scientific Position Paper SPP103). It should be noted that the level of depletion represents an average value across the GSB and tidal excursion and the effects would be tidally dynamic and concentrated around the intakes. Even at exchange rates as low as 5% per day local depletion of pelagic species is expected to be in the order of 3% with the abundance of epibenthic species reduce by less than 2% and demersal species by less than 5% with both stations operating. Acoustic survey results show no localised reduction of pelagic shoaling fish in the vicinity of the SZB intake (BEEMS Technical Report TR359) and it is apparent that localised losses at SZB are being replaced by the constantly moving shoals from the wider North Sea.

Local depletion in prey resources must be considered in the context of natural variability in resource availability. Table 26 illustrated the year-to-year variability impingement rates at Sizewell B, a proxy for natural abundance, in prey species. The large interannual variability clearly demonstrates that local depletion would be swamped in comparison. For example, local depletion of sprat is predicted to be approximately 1.5% within the GSB + tidal extent, whilst mean interannual variability is a factor of 2 with maximum between year variation a factor of 3.6. Local depletion of anchovy during its seasonal occurrence at Sizewell is predicted to be 1.4% for both stations operating (Table 26) whilst interannual variability is on average orders of magnitude greater at 5.02.

The foraging ranges of designated sandwich terns, lesser black-backed gulls, and common terns during the breeding season extends beyond the GSB and tidal excursions allowing them to access prey resources from a wider area. Marine mammals and overwintering red throated diver have greater foraging ranges still without the restriction of having to return to local breeding colonies near Sizewell. Little terns have the most area restricted foraging ranges and during the breeding season little tern forage close to their colonies out to a maximum distance of approximately 2.4 km offshore (BEEMS Technical Report TR431). Breeding colonies at Minsmere, Dingle marshes and Slaughden would forage to a large extent within the GSB and tidal extent, however, foraging would primarily be within the Sizewell-Dunwich Bank and more subject to the immediate effects of Sizewell B. Little tern breed during the period May to August and are designated in the Minsmere-Walberswick SPA ,however, colony locations for this species are known to be highly variable with time. At classification in 1991, the SPA's breeding population was 28 breeding pairs however, since classification, the numbers of little tern using Minsmere-Walberswick SPA has decreased by approximately 95% to 1.6 breeding pairs (5 year mean peak count 2014-2018) (Natural England, 2019).

The diet of breeding little tern at Sizewell is expected to consist of small schooling pelagic fish species that are found close to the sea surface and demersal fish in the shallows such as gobies. During the breeding season little tern forage close to their colonies out to a maximum distance of approximately 2.4 km offshore (BEEMS Technical Report TR431). Impingement data indicates that sprat and herring are the most common pelagic fish in Sizewell coastal waters (BEEMS Technical Report TR431). The predicted impingement mortality from SZB is 19.5 t of sprat and 125.6 t of herring per annum (Appendix D2). All abstracted fish are returned to sea from SZB but these pelagic species are not expected to survive impingement. Assuming sprat and/or herring accounted for the whole diet of little terns, the calculated biomass required to sustain the maximum recorded 28 breeding pairs observed in 1991 (with up to 3 chicks per pair) during the 4-month breeding season is less than 650 kg per annum. The quantities of fish required from an energetics point of view and the scale of local depletion of prey resources are well within the bounds of natural variability, which predator-prey relationships are adapted to (Table 26) reflecting the degree variability in natural abundance. The predicted effect of the SZB losses on the recognised stocks of sprat and herring are negligible (Appendix D2). If SZB was having a significant effect on the local sprat and herring abundance this would be apparent in the impingement numbers (which would collapse). However, the herring population at Sizewell

between 2009-2017 increased significantly (n = 8, tau = 0.93, p = 0.002, Theil-Sen slope = 157488 [4]; Figure 10) and the sprat population is the largest of any fish population at Sizewell and shows no discernible trend (n = 8, tau = 0.57, p = 0.063, Theil-Sen slope = 310686).

Furthermore, the FRR systems means that biomass is retained within the system resulting in bottom-up effects stimulating secondary production and, in some cases, affording opportunistic feeding opportunities for seabirds. Whilst the majority of FRR discards sink and would therefore not be accessible to surface feeding seabirds, floating discards would represent a potential foraging opportunity (BEEMS Technical Report TR501). Experiments have shown seabirds can take up to 71%, 8%, 12% and 4% of discarded roundfish, flatfish, elasmobranchs and invertebrates, respectively, in the southern North Sea (Garthe *et al.,* 1996).

Local depletion of prey availability, notwithstanding the retention of biomass from the FRR systems, is predicted to be minimal in comparison to natural variability. As such, no significant reduction in the prey availability of designated HRA species is anticipated.

Table 26 Seasonal abundance of selected prey species impinged at Sizewell B, predicted depletion during those periods and interannual variability in CIMP impingement rates. Values provided show local effects of Sizewell B and Sizewell C operating in-combination and are based on 10% exchange rates.

Species	Group	Period that the species is found in greatest numbers close to the Sizewell power stations (days)	Predicted depletion in the GSB and tidal excursion due to SZB + SZC	Mean interannual variability in impingement (2009-2017)	Maximum interannual impingement variability between any two adjacent years
Sprat	Pelagic	Dec. – March (121)	0.0151	2.02	3.64
Herring	Pelagic	Feb. – April (89)	0.0150	1.33	1.70
Anchovy	Pelagic	June (30)	0.0142	5.02	8.85
Whiting	Demersal	Oct. – April (221)	0.0239	1.56	2.04
Seabass	Demersal	Dec. – March (121)	0.0237	2.36	6.11
Sand goby	Epi-benthic	Aug. – Dec. (162)	0.0088	4.20	15.32
Dover sole	Epi-benthic	March – Oct. (245)	0.0088	1.53	2.26

8.2.4 Effect of SZC entrapment on the Water Framework Directive (WFD) status of the Alde & Ore water body.

Under the Water Framework Directive, the Environment Agency monitors the ecological status of the Alde-Ore Estuary as a single water body (Alde & Ore). To assess the status of fish communities in transitional waters a multi-metric index, termed the Transitional Fish Classification Index (TFCI) is applied. The TFCI is

[4] Herring statistics report the Kendall's tau statistic, which measures the rank correlation between impingement and year (with a range from -1 to +1, with 0 when there is no correlation), and the 2-sided p-value from the Mann-Kendall test. The Theil-Sen slope estimate is a median slope estimate for a linear change in impingement per year.

designed to detect impacts on the fish status due to a wide range of anthropogenic pressures. A decline in the status of the water body triggers further investigation and potentially additional monitoring.

Prior to the release of the 2019 status classification the Environment Agency expressed concerns relating to the potential for the proposed Sizewell C development to impact fish species of relevance to the Ade & Ore TFCI particularly smelt resulting in a deterioration in the 'good' status of the water body classified in 2016. The status of the Alde & Ore water body is complicated by an ongoing Environment Agency investigation into the use of different sampling gears within the assessment of the 2019 classification (further details are provided in Section 9). At the request of the Environment Agency, BEEMS Scientific Position Paper SPP108 assessed the sensitivity of the TFCI to manipulated reductions/removals of smelt, thin-lipped grey mullet, Dover sole, herring, seabass and Twaite shad from monitoring data.

To determine the sensitivity of the TFCI to smelt abundance, smelt numbers in samples were manipulated at a range of levels including complete smelt absence. A further test considered absence of smelt and Twaite shad and 50% reductions in herring and seabass. The sensitivity of the TFCI was also tested further through manipulated removals of thin-lipped mullet and Dover sole, as well as a scenario whereby smelt, thin-lipped mullet and Dover sole were all simultaneously reduced by 50%.

1. The calculated Ecological Quality Ratio (EQR) was insensitive to manipulated reductions in smelt abundance of 25% and 50%.

2. Total absence of smelt reduced the EQR by 11% but 'good' status remained (when fyke net data is not included in the TFCI calculation)

3. The extreme case requested by the Environment Agency including absence of shad (1 individual caught between 2013-2018) and smelt along with 50% reductions in herring and bass reduced the EQR by 10.3%, however, 'good' status remained (TFCI calculated without the inclusion of fyke net data).

4. Total absence of thin-lipped grey mullet and Dover sole reduced the EQR by less than 4% in each case and 'good' status remained. The status also remained 'good' following the combined 50% reduction of smelt, Dover sole and thin-lipped grey mullet.

Under all of the scenarios trested for fish manipulations, there was no deterioration of 'good' status when the 2019 TFCI was calculated without fyke net data[5]. The report concluded that it is highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore (BEEMS Scientific Position Paper SPP108).

8.3 Evidence of localised impingement effects from other sites

When Hinkley Point A (HPA) station ceased operations in 2000 a total seawater abstraction of 44 cumecs was removed from the Hinkley Point intake structure. For 24 years, the operation of HPA coincided with the operation of HPB that was commissioned in 1976. Therefore, if an impingement impact of the size of the HPA abstraction was having any effect on local fish populations then the closure should have been detectable in the HPB impingement records, which extend over 35 years.

In practice, no such effect could be detected, and it was concluded that the local fish assemblage was not sensitive to at least a 44 cumec reduction in unmitigated impingement pressure, BEEMS Technical Report TR456 (Note: HPA and HBP were not fitted with impingement mitigation measures).

[5] The TFCI calculated with fyke data causes the water body EQR to fall by 18.8% to 'moderate' status. BEEMS Scientific Position Paper SPP108 concluded that low sampling efficiency and statistical artefacts relating to the small fyke net sample size and low catches caused the apparent deterioration of the water body classification to 'moderate' (Section 12). Fish manipulations as described with the inclusion of fyke nets did not result in deterioration below 'moderate' status.

SZC would be fitted with the latest impingement mitigation technology e.g. LVSE intakes and an FRR system. To compare the impingement pressure exerted by the proposed station it is necessary to calculate an equivalent unmitigated abstraction (Table 27). For SZC this impact varies between 10.4 cumecs and 50.6 cumecs which for most species is less than the change in impingement of 44 cumecs that caused no detectable effect at Hinkley Point.

Table 27 SZC equivalent abstraction after impingement mitigation of LVSE intakes and FRR system in comparison with an unmitigated SZC abstracting 132 cumecs

Species group	Equivalent SZC abstraction in cumecs compared with an unmitigated station abstracting 132 cumecs
Pelagic	47.1
Demersal	25.9 – 28.3
Epibenthic	9.7 (11.2 flounder and 29.5 dab)

In the case of demersal and epibethic species, a fully mitigated SZC represents a considerably lower equivalent abstraction rate than the unmitigated system such as HPA. Whilst the assemblage at Sizewell differs from Hinkley Point it is reasonable to assume that the absence of discernible local effects observed at Hinkley would also be the case at Sizewell.

Pelagic species would experience a 3.1 cumec increase in impingement pressure than the 44 cumecs of HPA that caused no detectable effect at Hinkley Point. Pelagic species include sprat, herring, anchovy, smelt, mackerel, horse mackerel, twaite shad and allis shad. Of these species only sprat and herring and to a much lower extent anchovy play an important role in the local food web and as discussed above they are widely distributed species that migrate continuously over very large spatial areas. The other 5 species (smelt, mackerel, horse mackerel, twaite shad and allis shad) are not abundant enough at Sizewell to be important to the local food web with only smelt having a non-trivial presence (Table 18).

The additional abstraction from SZC is not considered large enough to have any significant effect on pelagic fish numbers because, as explained above, SZC losses will be continuously replaced by pelagic fish moving into the local area from the wider southern North Sea.

8.4 Local effects conclusions

Population structures of marine species fall along a continuum from panmictic (e.g. European eel, *Anguilla anguilla*) to numerous distinct sub-populations (e.g. North Sea herring, *Clupea harengus*) with the majority of species exhibiting complex structure. In the open sea, the sub-populations of many species mix to a considerable extent, especially during summer feeding and on nursery areas. Harvesting affects multiple components of the overall population simultaneously. ICES' definition of stock units integrates information on site fidelity to spawning, nursery and feeding areas together with knowledge of migration patterns and the degree of intermixing that takes place between sub populations.

There are no isolated fish populations at Sizewell, fish live and move in an unconstrained coastal environment with most species undertaking wide spatial migrations throughout the year. ICES stock units are the most relevant scale of assessment for contextualising effects in relation to the sustainability of fish within safe biological limits. Comparison against SSBs is the internationally recognised best practice way that the much larger effects of fishing at either a fleet or individual boat level are assessed.

However, whilst stock sustainability is central to managing fish populations within safe biological limits assessments of effects on fish must consider aspects beyond stock sustainability. This section considered

the potential for localised depletion to have consequences on prey availability for designated species Annex II species of relevance to the HRA and species of importance to the WFD TFCI.

An assessment of potential localised effects of SZC entrapment was undertaken and found no likely significant adverse effects, concluding that:

v. The proposed development would not have significant effects on local spawning or nursery areas in the vicinity of Sizewell.
vi. Sizewell C represents a low risk to migratory species.
vii. Local depletion due to impingement is orders of magnitude below natural variability in abundance to which predator-prey relationships are adapted to. Therefore, impingement from Sizewell B and Sizewell C in-combination would not have an adverse food-web effects on designated features of HRA sites.
viii. Local effects are not predicted to cause a deterioration in the WFD fish status of the Alde & Ore water body as local depletion is well below the sensitivity of the TFCI.
ix. Impingement monitoring from Hinkley Point B showed no discernible effect after Hinkley Point A ceased abstraction. The unmitigated Hinkley Point A was comparable or lower than the relative effects predicted Sizewell C indicating the system is unlikely to be sensitive to such changes.

8.4.1 What about adaptive management?

Stakeholder have also questioned whether a power station can be assessed in a similar manner to fishing. The effects of a power station, albethey small in comparison to fishing, are fixed spatially and for the duration of the operational phase whereas fisheries managers can implement adaptive fisheries management strategies in response to changes in stock dynamics. This may include, for example, reductions in total allowable catch (TAC), closed seasons, or redistribution of fisheries effort away from sensitive areas. Such strategies are not available to a power station once operational.

To maximise catch, fishing effort is targeted towards productive areas and fleets either operate seasonally, or fish different areas or target species at different times of the year. Whilst bycatch in some fisheries remains high, fishing effort is targeted and changes in effort, gear types, or location can be employed. Sizewell C would not be able to change abstraction rates or location. However, the station is a non-selective sampler i.e. rates of impingement are reflective of abundance. As a population increases, so to would impingement, conversely as a stock declines so impingement rates would decline reflecting local abundance. Consequently, impingement numbers are likely to change through time (e.g. due to climate change see Section 11) but the relative effect on the stock would only change if site utilisation changed dramatically.

9 Effect of SZC entrapment on the Water Framework Directive (WFD) status of local water bodies

The test for the Water Framework Directive (WFD) compliance assessment is dependent on whether SZC has the potential to cause deterioration in the status of the surface water bodies (both within and between status classes) by adversely affecting biological, hydromorphological and/or physico-chemical quality elements. In principle, SZC entrapment could affect the fish biological quality element of two nearest transitional water bodies to Sizewell:

a. Blyth (S) at approximately 12 km to the north of Sizewell

b. Alde & Ore at approximately 25 km to the south of Sizewell

The United Kingdom Technical Advisory Group for WFD (WFD-UKTAG) has produced an assessment method for fish in transitional water bodies - the Transitional Fish Classification Index (TFCI) (UKTAG 2014). The method is not applicable to coastal water bodies.

The TFCI is a multimetric index composed of 10 individual components known as metrics and listed in Table 28.

Table 28 WFD Transitional Fish Classification Index metrics

Number	Metric	Community characteristic
1	Species composition	Species diversity and composition
2	Presence of indicator species	
3	Species relative abundance	Species abundance
4	Number of taxa that make up 90% of the abundance	
5	Number of estuarine resident taxa (ER)	Nursery function
6	Number of estuarine-dependent marine taxa (MS & MJ)	
7	Functional guild composition	
8	Number of benthic invertebrate feeding taxa	Trophic integrity
9	Number of piscivorous taxa	
10	Feeding guild composition.	

Each metric is assessed by comparing the observed metric values with those expected metric values under reference conditions. A set of reference conditions have been developed for different water body types and sampling gears (the latter does not include power station impingement which provides a much greater sampling efficiency than the alternative net-based sampling methods).

The TFCI is calculated as the sum of all metric scores and converted into an Ecological Quality Ratio (EQR) operating over a range from zero (a severe impact) to one (reference/minimally disturbed). The four class boundaries are:

- High/Good = 0.81
- Good/Moderate = 0.58
- Moderate/Poor = 0.4
- Poor/Bad = 0.2.

With exception of metric 3 in Table 28, all the other metrics are counts of the number of species in functional, feeding or indicator species groups found in the population samples. As described in Section 5.1.2 the fish

abundance in the vicinity of Sizewell and in the two transitional water bodies will be subject to considerable in year and between year variability and also variability due to long term trends caused by climate change and changes in fishing pressure due to management action. Measurements of the TFCI will therefore be subject to variability and only developments that have a widescale, large impact on the community would be expected to make any significant changes to the index. In terms of WFD water body status the following conclusions are pertinent:

a. Marine fish in the transitional waters of the Blyth and Alde Ore are considered to be part of the same stock units as fish at Sizewell and the SZC entrapment effects have been assessed as negligible (Table 18) and much smaller than natural variability in the size of fish populations and would, therefore, be expected to have no effect on the calculated WFD fish biological quality element (Table 28). (SZC would have no effect on freshwater fish in the water bodies).
b. No losses of indicator species are expected due to SZC entrapment.
c. There are no predicted significant localised effects of SZC entrapment (Section 8.3) and none that would have any significant effect on the calculated WFD fish biological quality element.
d. There are no predicted changes due to SZC entrapment in the number of functional and feeding guilds in the transitional water bodies nor to the number of indicator species.

Given the above, no significant change in the EQR would be expected and certainly none that would result in a change in the WFD status of the Blyth (S) and Alde & Ore transitional water bodies due to SZC entrapment.

9.1 Alde & Ore TFCI sensitivity to changes in fish abundance

The Environment Agency has raised a number of concerns relating to the potential for the proposed Sizewell C development to impact fish species of relevance to the Ade & Ore TFCI. The primary concern is in relation to reductions in smelt. The status of the fish community in the Alde & Ore was classified as ‘Good’ in 2015 (EQR: 0.63) and ‘Good’ in 2016 (EQR: 0.61). During this period seine nets and beam trawls were the predominant sampling method but fyke nets were also deployed and used in the EQR calculation. No classification was produced in 2017 or 2018. The recently published 2019 results show the water body transitional fish status apparently fell to ‘Moderate’ in 2019 (Environment Agency Catchment data Explorer, 2020). The 2019 EQR provided by the Environment Agency is 0.56 and was calculated using a combination of seine nets, beam trawls and fyke nets using data from 2013-2018.

BEEMS Scientific Position Paper SPP108 calculated the TFCI baseline Ecological Quality Ratio (EQR) by applying six years (2013-2018) Environment Agency fish monitoring data for the Alde & Ore water body. Sampling data from the period includes 75 seine nets, 34 beam trawls and 8 fyke nets (fykes sampled in 2013 and 2014 only). Seine samples returned 26 species and the highest catches, whilst beam trawls caught 21 species in total. Fyke nets only caught a single seabass and one ballan wrasse.

The Environment Agency algorithms used to derive EQR values now incorporate bootstrapping approaches. The bootstrapping method permutes samples from each gear type 1,000 times to determine the mean result and confidence intervals that are averaged between gears to calculate an overall EQR. By putting equal emphasis on all sample gears, the bootstrapping approach differs from the previously applied weighted mean approach and produces different results particularly when sample numbers and catch efficiency is low. It is noteworthy that if the 2019 EQR is calculated with fyke nets using the weighted mean approach, as in 2015 (0.63) and 2016 (0.61), the water body status is ‘good’ (EQR 0.63). Conversely if the 2016 data is recalculated using the same bootstrapping method applied in 2019, the status falls to ‘moderate’ with an EQR of just 0.50. The Alde & Ore water body classification using data from 2013-2018 without the use of fyke net data, results in an EQR of 0.69 representing ‘good’ status and a confidence of class for good status or above of 0.92. BEEMS Scientific Position Paper SPP108 concluded that low sampling efficiency and statistical artefacts relating to the small fyke net sample size and poor catch rates caused the apparent deterioration of the water body classification to ‘moderate’ in 2019.

At the request of the Environment Agency, the sensitivity of the TFCI to smelt abundance, smelt numbers in samples were manipulated to four levels representing declines of 25%, 50%, a single smelt caught per annum (in seine samples), and total smelt absence. A further test considered absence of smelt and Twaite shad and 50% reductions in herring and seabass (multispecies manipulation). These tests were run with and without fyke nets. The sensitivity of the TFCI was also tested further through manipulated removals of thin-lipped mullet and Dover sole, as well as a scenario whereby smelt, thin-lipped mullet and sole were all simultaneously reduced by 50%. These manipulations were calculated with no fyke nets included.

1. The calculated Ecological Quality Ratio (EQR) was insensitive to manipulated reductions in smelt abundance of 25% and 50%.

2. Total absence of smelt reduced the EQR by 11% but 'good' status remained (when fyke net data is removed from the TFCI calculation)

3. The Environment Agency requested a scenario of total absence of both shad (1 individual caught between 2013-2018) and smelt along with 50% reductions in herring and bass. In this extreme scenario the EQR was reduced by 10.3%, however, 'good' status remains (TFCI calculated without the inclusion of fyke net data).

4. Total absence of thin-lipped grey mullet and Dover sole reduced the EQR by less than 4% in each case and 'good' status remained (TFCI calculated without the inclusion of fyke net data). The status also remained 'good' following the combined 50% reduction of smelt, Dover sole and thin-lipped grey mullet.

5. With the inclusion of fyke nets, the calculated TFCI did not change from 'moderate' status irrespective of the manipulated removal of fish described above.

Under all of the scenarios trested for fish manipulations, there was no deterioration of 'good' status when the TFCI was calculated without fyke net data. The report concluded that it is highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore (BEEMS Scientific Position Paper SPP108).

The potential for the proposed development to affect smelt, and other migratory fish (e.g. Dover sole and grey mullet), in the Alde & Ore by means of thermal occlusion and impingement was investigated and it was concluded that:

1. There is no evidence for thermal occlusion either at the mouth of the Alde & Ore where temperatures are ≤1°C as a 98th percentile, or across a hypothetical corridor at the frontage of Sizewell C, 25km to the north of the estuary mouth.

2. Impingement from Sizewell C is predicted to have a negligible effect on regional smelt stocks (0.19% of the conservatively estimated 'Anglian' SSB) and would not result in significant local reductions in smelt abundance entering the estuary (BEEMS Scientific Position Paper SPP103).

Based on the evidence presented in BEEMS Scientific Position Paper SPP108 the Alde & Ore TFCI was tested against effects far in exceedance of those anticipated from the proposed development and in all cases the EQR remained within the boundaries of 'good' status (in the absence of fyke net samples), it is therefore considered highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore.

10 Shellfish impingement predictions for Sizewell C

Four shellfish species (brown crab *Cancer pagurus*, European lobster *Homarus gammarus*, brown shrimp *Crangon crangon* and whelk *Buccinum undatum*) were defined as key benthic species on the basis of their socio-economic importance (all four species) and their ecological importance (brown shrimp only) (BEEMS Technical Report TR348). Of these, whelk were absent from the impingement dataset. The impact of SZC on whelks is therefore considered negligible and the species will not be considered further. Estimates of SZB and predictions of SZC impingement were made using the same methods described for the finfish species.

10.1 FRR system mortality

10.1.1 Trash rack mortality

As with finfish, the proportion of the shellfish species that would pass through the 75 mm trash rack was assessed (Table 29). For brown crab, carapace width (CW, mm) measurements were made on crabs sampled at the SZB site between 2014 and 2017 and used to calculate annual length distributions. The maximum width observed was > 75 mm, and the proportion that is likely to pass was calculated in a similar manner to finfish species. Only four lobsters were recorded in the CIMP dataset between 2009 and 2017, all of them prior to 2014. Only weights were recorded, and these were used to estimate the proportion that would pass through a 75 mm bar spacing.

Table 29 Proportion of shellfish, by species that will not pass through the 75 mm wide trash racks

Species	Calculation type	Size of largest SZB individual (mm)	Proportion not passing through trash rack	Comment
Brown shrimp	Group 1	Not measured	0.000	All shrimp will pass
Brown crab	Group 3	162	0.014	Calculate length at 75 mm width
Lobster		Not measured	0.500	Best estimate based on individual weights

10.1.2 FRR survival

According to Environment Agency (2005), crustaceans will have the same FRR survival rates as other epibenthic species. Therefore, for this report, the modified values for the epibenthic group (given in Section 4.7.2) were used here for shellfish (Table 30).

Table 30 Proportion mortality by species through the SZC drum and band screens

Species	Proportion lost		
	Drum	Band	Species group
Brown shrimp	0.206	0.206	epibenthic
Brown crab	0.206	0.206	epibenthic
Lobster	0.206	0.206	epibenthic

10.2 EAV conversions

An EAV value was calculated for brown crab. EAVs could not be calculated for brown shrimp and lobster due to a lack of size data and other biological data. For those species an EAV of 1 was used (Table 31), giving rise to impingement overestimates.

Table 31 EAV metrics and mean weight of individuals used to convert the numbers impinged to adult equivalent numbers and weights of shellfish at SZC. (See BEEMS Technical Report TR383 for brown crab EAV calculations)

Species	EAV	Mean adult weight per individual (kg)	Data source for mean weight
Brown shrimp	1.000	0.0013	Calculated from modelled mean number and mean weight of brown shrimp at SZB
Brown crab	0.219	0.5	Mean male and female weights at the minimum landings size, (87mm CW), and averaged assuming a 50:50 sex ratio
Lobster	1.000	0.379	Bannister et al. (1983)

10.3 Evaluating SZC impacts on shellfish

Assessments have been carried out by Cefas for lobsters and brown crabs in the southern North Sea, using slightly different assessment areas for the two species. Due to uncertainties in the data, SSB estimates are not available, and impingement losses were compared against the landings from the rectangles in the assessment regions (Table 32). None of the crustacean species is assigned to an ICES Working Group.

Table 32 Data sources used to provide information on relevant stock unit, landings and SSB

Species	Stock unit	Assessment type	Impingement effect comparator	Reference
Brown shrimp	Not defined	None	Landings, same ICES rectangles as lobster assessment	
Brown crab	Southern North Sea	Analytical assessment, but with many uncertainties	Landings from the Southern North Sea crab fishery unit (CFU) (as defined in Cefas 2017) ICES rectangles in the assessed area are shown in Table 33.	Cefas, 2017
Lobster	East Anglia	Analytical assessment, but with many uncertainties	Landings from East Anglia Lobster Fishery Unit (LFU). ICES rectangles in the assessed area are shown in Table 34.	Cefas, 2017b

Table 33 ICES Rectangles in Southern North Sea CFU

35F0	35F1	35F2	35F3	35F4	35F5	35F6	35F7 (no catch)	35F8
34F0	34F1	34F2	34F3	34F4				
	33F1	33F2	33F3	33F4				
32F0	32F1	32F2	32F3					

Table 34 ICES rectangles used in East Anglia LFU

35F0	35F1	35F2
34F0	34F1	34F2
	33F1	33F2
32F0	32F1	32F2
31F0	31F1	31F2

10.4 Predicted SZC impingement effects on shellfish without embedded mitigation measures

The predicted unmitigated SZC impingement effects for the three key crustacean species after adjusting to equivalent adults are given in Table 35. Predicted losses of brown shrimp and brown crab both exceeded the 1% threshold (2.89 % and 3.31 %, respectively). Unmitigated losses of lobsters were < 1 % and are therefore negligible.

Table 35 Annual mean SZC impingement predictions with no impingement mitigation for key shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of the mean international landings (t). Species where the impingement weight > 1 % of the landings – given in bold) are shaded red.

Species	Mean SZC	EAV number	EAV weight (t)	Mean landings (t)	% of landings
Brown shrimp	15,419,644	15,419,644	20.05	693	2.89
Brown crab	136,182	29,756	14.88	450	3.31
Lobster	37	37	0.01	114	0.01

10.5 Predicted SZC impingement effects with the LVSE intake heads fitted

Lobsters are not expected to derive any benefit from the proposed LVSE intake head design. Brown crab and brown shrimp would not be able to actively avoid the intake if they were in the water column, but any benefit from the proposed design has not yet been evaluated. Therefore, to be conservative, no benefit from the proposed intake head design has been assumed.

10.6 Predicted SZC impingement effects on shellfish with FRR systems fitted

With the inclusion of the FRR, the impingement losses of those species that exceeded 1 % in the absence of mitigation (brown shrimp and brown crab), were reduced to 0.6 % and 0.72 % of landings, respectively (Table 36). The fitting of the FRR systems alone therefore reduces the impingement losses of all three crustacean species to below 1 %. For brown shrimp, losses are regarded as an overestimate, due to the use of an EAV of 1.

10.7 Conclusions on the effects of SZC on shellfish

Four shellfish species (brown crab, European lobster, brown shrimp and whelk were defined as key benthic species on the basis of their socio-economic importance (all four species) and their ecological importance (brown shrimp only). Of these, whelk were absent from the impingement dataset and there is no predicted impingement effects on the species. The predicted losses of the other three shellfish species are less than 1

% of landings. The use of landings as an impingement comparator is highly conservative as SSBs will be larger. As such the predicted SZC effects on all four key shellfish species assessed are considered negligible.

Table 36 Annual mean SZC impingement predictions with FRR mitigation fitted for shellfish species. Losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a % of either the mean international landings (t). Any species where the impingement weight > 1 % of the landings – given in bold) are shaded red.

Species	Mean SZC	FRR mortality	EAV number	EAV weight (t)	Mean landings (t)	% of landings
Brown shrimp	15,419,644	3,176,447	3,176,447	4.13	693	0.60
Brown crab	136,182	29,523	6,451	3.23	450	0.72
Lobster	37	22	22	0.01	114	0.01

11 Consideration of climate change effects

Sea temperatures around the UK and Ireland have been warming at between 0.2 and 0.6 °C decade^{-1} over the past 30 years. Projected future changes in the temperature and chemistry of marine waters around the UK and Ireland are having, and will have, effects on the phenology (timing of lifecycle events), productivity and distribution of marine fish and shellfish (Heath *et al*., 2012). In a detailed study of terrestrial birds, butterflies and alpine herbs it was found that these species were undergoing northerly latitudinal change of 6.1 ± 2.4 km decade^{-1} and that there was an advancement of spring events of 2.3 d decade^{-1} (Parmesan and Yohe, 2003). Perry et al. (2005) described that distributions of both exploited and non-exploited North Sea fishes have responded to recent increases in sea temperature, with nearly two-thirds of species shifting in mean latitude or depth or both over 25 years. They found that species with shifting distributions have faster life cycles and smaller body sizes than non-shifting species and that the differential change between species could have consequences for predator-prey relationships. For species that shifted, the mean shift was 99 km northwards in 25y. Dulvy et al. (2008) found that North Sea winter bottom temperature had increased by 1·6 °C over 25 years and that during this period, the whole demersal fish assemblage deepened by ~3·6 m decade^{-1}. Simpson et al. (2011) found that most common northeast Atlantic fishes are responding significantly to warming with:

- Three times more species increasing in abundance with warming than declining.
- Local communities are being reorganized despite decadal stability in species composition.
- Species range shifts are the tip of iceberg compared to modification of local communities.

However, the effects of climate change on fish communities are hard to predict with accuracy because behaviour, genetic adaptation, habitat dependency and the impacts of fishing on species result in complex species responses (Heath et al., 2012). Petitgas et al., (2013) considered that the key issue for the significance of climate change impact on fishes is habitat availability and connectivity between lifecycle stages with climate driven changes in larval dispersion being a major unknown. Petitgas et al., (2013) considered that there was a significant risk for species with strict connectivity between spawning and nursery grounds.

The 2017 Marine Climate Change Impacts Partnership (MCCIP) review on fisheries describes the changes expected in fish and fisheries with climate change (Pinnegar *et al* 2017), and is summarised in this paragraph. *There has been a trend in recent decades for warm-affinity species to increase in abundance, and cold-affinity species to decrease in abundance, with many cold-water species moving northwards. For example, there has been a decline in abundance of Atlantic cod (linked with fishing pressure and climate), and a general northwards shift. Mackerel have shown complex changes in recent years, but with a general north and westward shift linked with sea temperature. Sea bass, a warm-affinity species, expanded distribution and increased in numbers in the early 2000s, but fishing mortality then reduced numbers again. Similarly, anchovy has expanded distribution in the North Sea in the past decade. There are exceptions to this general trend, such as sole which has shifted distribution southwards and are able to remain in shallow North Sea waters all year around. Changes in plankton phenology has resulted in changes in timing of fish spawning with a shift of approximately 1.5 weeks earlier per decade in the southern North Sea since 1970s* (Pinnegar *et al* 2017).

Modelling predicts that habitat suitability around the UK will increase this century for European squid, sea bass, pilchard, sprat, veined squid, John dory, anchovy, sole, plaice and whiting, and that it will decrease for saithe, hake, red mullet, haddock, halibut, mackerel and herring (Jones, 2013). Except for sole and whiting, the southerly distribution of all species is predicted to move northwards around the UK.

11.1 Changes in the Sizewell fish community

From data collected at the Sizewell A station in the 1980s, it is possible to observe changes in the Greater Sizewell Bay community in the 35+ year period up to the collection of the current SZB CIMP data. SZA impingement estimates in 1981-1982 were compared with those obtained for SZB in the current study. SZA

numbers were adjusted for the different pumping capacities (SZA = 30.4 cumecs vs SZB = 51.5 cumecs; ratio ≈1.7), but not for any differences in intake head design.

For several species, the estimated numbers impinged by SZB are significantly higher than those estimated for SZA, even adjusting for pumping capacity (Table 37). For example, seabass at SZB are 194 x more abundant than the SZA estimate, and the species has increased its contribution to the total number of fish impinged from 0.02 % to 5.22 %. Similarly, twaite shad are 7 x more abundant in impingement catches now than in the 1980's, and their contribution to the total has increased from 0.003 % to 0.02 %. River lamprey were not recorded in 40 sampling visits at SZA, but between 2009 and 2017, occurred in 33 % of all samples.

Conversely, the abundance of other species has declined. Greater pipefish *Syngnathus acus* were ranked as the 5th most abundant species by Turnpenny and Utting (1987) but in the current dataset, rank only 21st, and estimated numbers have decreased 10-fold (adjusted for pumping capacity). Similarly, Nilsson's pipefish *S. rostellatus* dropped from a rank of 6th to 14th and estimated numbers decreased by a fifth.

Smelt abundance showed little or no change over the 35+ year period, contributing similar numbers to both datasets. If the relative catching efficiency of the SZA and SZB intake heads for pelagic fish are considered, the smelt abundance may have increased since 1981/82 but the measured difference are within the bounds of natural variability (Table 9).

Table 37 Annual estimated numbers of fish impinged by SZA in 1981-1982 (Turnpenny and Utting, 1987) and by SZB in 2009-2017 (BEEMS Technical Report TR339), the number at SZA raised to the SZB pumping capacity, the percentage of the total number impinged and the species' rank

	Sizewell A 1981-1982				Sizewell B 2009-2017			
Species	**Number**	**Number (raised)**	**% of total numbers**	**Rank**	**Number**	**% of total numbers**	**Rank**	**Number SZB/SZA**
Seabass	685	1,160	0.02	28	276,943	5.22	4	238.7
European anchovy	240	407	0.01	37	69,058	1.30	7	169.7
River lamprey	0	0	0	-	1,144	0.02	33	-
Twaite shad	98	166	0.003	46	1,168	0.02	32	7.0
Smelt	6,764	11,459	0.18	14	8,122	0.15	18	0.7
Greater pipefish	66,074	111,935	1.77	5	4,672	0.09	21	0.04
Nilsson's pipefish	44,545	75,463	1.19	6	14,247	0.27	14	0.19
Flounder	22,855	38,718	0.61	8	14,569	0.27	13	0.38

11.2 Potential future changes

Some of the key observed trends in the Greater Sizewell Bay are likely to continue:

- Relative changes in species abundance with growing numbers for species that favour warmer water (in winter, in summer or both) and reducing abundance of species near to their southern latitudinal boundary.
- Effects on the phenology of some species (e.g. timing of the arrival of new recruits) and changes in migration patterns as some areas of the North Sea become more or less suitable habitat for each species and/or their prey.

- The presence of large numbers of juvenile species in the Greater Sizewell Bay is dependent upon the connectivity between spawning locations further offshore and their inshore nursery grounds. Some species have a lower tolerance to changes in winter temperatures than to summer temperatures (Dulvy et al., 2008; Perry et al., 2005) and it is possible that winter temperatures will reach a level such that some species may have to abandon fidelity to long established spawning locations which could produce a rapid reduction in the numbers of those species in the southern North Sea but not necessarily in the wider population biomass.

11.3 Effect on SZC impingement predictions

Differences in the two Sizewell impingement datasets show that the fish assemblage off Sizewell is changing due to a combination of climate change, changes in fishing pressure and other anthropogenic causes, both positive and negative (for example, improving water quality in continental rivers is attributed to increases in abundance of twaite shad in European rivers resulting in increases in abundance at Sizewell). SZC will efficiently sample the fish community at Sizewell. If a population increases in abundance then impingement numbers will increase, if a population declines in abundance then impingement numbers will reduce. In such circumstances climate change will have no effect on the predicted negligible effect of SZC impingement on the fish assemblage.

12 Conclusions

Estimates of the number of fish, invertebrates and other individuals impinged by the current SZB station and predictions for the proposed SZC station were based upon 205 samples collected between February 2009 and December 2017. This extensive dataset provides a substantial amount of information on the abundance, seasonality and size structure of individuals impacted by the SZB station.

Ninety-one finfish species were recorded at Sizewell over the 9-year study period. Of these 24 species have been selected as being representative of the assemblage and which include species of importance commercially, ecologically and of conservation value (BEEMS Technical Report TR345), with some species occurring in multiple groups.

Where possible species were compared with defined stock units for that species and compared against internationally coordinated stock assessments. If population data could not be obtained, losses were compared against international landings, which represent only a portion of the total stock biomass. Such assessments are therefore considered highly conservative.

The predicted SZC impingement losses were compared against the negligible effect thresholds derived in Section 5. For some stocks a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. In particular, a highly precautionary approach was adopted for European eel whereby the Anglian RBD SSB was used as the stock reference due the uncertainties surrounding both the current eel stock status and its stock dynamics (Sections 5.1.6, 6.6.1). This was equivalent to adopting a negligible effects threshold of approximately 0.005% SSB for the eel stock. This type of precautionary assessment of SZC effects against partial stock estimates has also been applied for twaite shad, allis shad, cucumber smelt and river lamprey

The mean and 95th percentile upper impingement estimate for the key species was predicted to be below the appropriate 1% threshold for SSB or fisheries landings, for all species except bass and European eel. For example, sprat, herring, and whiting are numerically the highest impinged species; mean losses for these species equate to 0.02%, 0.02% and 0.11% of SSB, respectively. Mitigation measures would reduce effects further by the factors described below:

Group	Example species	Impingement reduction
Pelagic fish	sprat, herring, anchovy, shad	64%
Demersal fish	bass, cod, whiting, grey mullet	78-80%
Epibenthic fish	eel, lampreys, sole, sand goby	93%
Shellfish	Crab, brown shrimp	93%

Of the 24 key species at Sizewell, impingement from Sizewell C is predicted to result is the loss of 3.93 million fish per annum with an equivalent adult weight of 191.5t. Sprat, herring and whiting are the three species with the highest rates of impingement contributing 93% of the weight of the impinged biomass.

In the case of bass, a combination of the LVSE intakes, FRR mitigation and applying knowledge of the distribution of bass within the GSB is anticipated to reduce effects to well below 1%. Depending on the efficiency of the mitigation measures effects of between 0.03 up to a maximum of 0.3% of SSB may be predicted (Section 7).

In the absence of mitigation, losses of eel due to impingement equates to 1 t per annum or 1.27% of the Anglian RDB SSB. Eel is a robust species, and 80% survival is anticipated through well designed FRR systems. The efficiency of LVSE intakes for poor swimming species such as European eel has been questioned but the benefit from LVSE intakes are considered to be mostly due to the reduction in cross sectional area due being mounted orthogonal to the tidal current and the short range zone of influence on local hydrodynamics. Assuming no LVSE benefit at all, effects from the FRR alone equates to 0.13% of SSB if the more appropriate yellow eel EAV is applied (Section 7).

The individual entrainment and impingement impacts are such that when combined into a single entrapment estimate, there is very little difference to the overall conclusions that are reached when each is viewed separately. With the proposed impingement mitigation fitted entrapment estimates (Table 18) show the only species that remains above the 1 % threshold is sand goby (entrainment = 1.40 % of abundance; impingement = 0.02 %, i.e. entrapment = 1.42 %). Sand gobies are a short-lived very abundant species that is ubiquitous in European coastal areas to at least a depth of 20 m. The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over the southern North Sea. Given that the species is not commercially-exploited, the appropriate negligible effects threshold is 10% of SSB (as discussed in Section 5.1). However, because of their short lifespan and early age of maturity, sand gobies have a sustainable harvesting rate of greater than 50% SSB (Section 5.1.1). Therefore, losses of 1.42 % of total abundance by SZC are regarded as negligible.

Conclusions on the predicted effects of SZC entrapment

a. Of the 24 key fish and 4 key shellfish species, no species exceeded the 1% impingement screening threshold for negligible effects with the proposed LVSE intake heads and FRR systems fitted when compared against stock estimates or, in the absence of these, international landings.

b. For European eel, twaite shad, allis shad, cucumber smelt and river lamprey assessments a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. For eel this was equivalent to adopting a highly precautionary screening threshold of approximately 0.005% SSB. The predicted impingement effects of SZC on all of the key taxa were below the effect threshold.

It is therefore concluded that the proposed SZC station with FRR systems and LVSE intake heads fitted will result in negligible entrapment losses of all 24 key finfish taxa and 4 shellfish taxa.

References

Almada, V.C., Pereira, J.L., Fonseca, J.P., Levy, A., Maia, C., Valenta, A., 2008. Mitochondrial DNA fails to reveal genetic structure in sea lampreys along European shores. Mol. Phylogenet. Evol. 46, 391–396.

Aprahamian, M.W., 1989. The diet of juvenile and adult twaite shad *Alosa fallax fallax* (Lacépède) from the rivers Severn and Wye (Britain). Hydrobiologia 179, 173–182.

Aprahamian, M.W., 1988. Age structure of eel (*Anguilla anguilla* (L.)) populations in the rivers Severn (England) and Dee (Wales). Aquac. Fish. Manag. 19, 365–376.

Aprahamian, M.W., 1982. Aspects of the biology of the twaite shad, *Alosa fallax fallax* (Lacépède), in the rivers Severn and Wye. PhD thesis, University of Liverpool.

Bannister, R.C.A., Addison, J.T., Lovewell, S.R.J., Howard, A.E., 1983. Results of a recent minimum size assessment for the fisheries for lobster, *Homarus gammarus*, in England and Wales. ICES CM 1983/K:4.

BEEMS Scientific Position Paper SPP071/s. Shad (*Alosa fallax* and *Alosa alosa*) impingement predictions for HPC, Edition 3. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP099. Predicted impingement performance of the SZC LVSE intake heads compared with the SZB intakes. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP100. Estimates of European populations of twaite shad and cucumber smelt of relevance to Sizewell. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP101. Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP102. Use of Spawning Production Foregone EAVs for impingement assessment. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP103. Consideration of potential effects on selected fish stocks at Sizewell. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP104. Worst case glass eel entrainment assessment for Sizewell C. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP108. Sensitivity of the Alde Ore Transitional Fish Classification Index (TFCI) to changes in smelt, *Osmerus eperlanus*, abundance.. Cefas, Lowestoft.

BEEMS Technical Report TR120. SZ Comprehensive Impingement Monitoring Programme 2009/10: Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR123. Review of commercial and recreational fisheries activity in the vicinity of Sizewell Power Station. Cefas, Lowestoft.

BEEMS Technical Report TR196. SZ Comprehensive Impingement Monitoring Programme Year II; Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR215. Comprehensive Impingement Monitoring Programme, Year III – Final Report. Cefas, Lowestoft.

BEEMS Technical Report TR270. Comprehensive Impingement Monitoring Programme 2012-2013 at Sizewell B: Annual report. Cefas, Lowestoft.

BEEMS Technical Report TR274. Dynamics of glass eels in the Bristol Channel 2012-2013. Cefas, Lowestoft.

BEEMS Technical report TR316. Evaluation of chlorination dosing options for Sizewell C (Balancing bio-fouling efficacy and non-target species effects). Cefas, Lowestoft.

BEEMS Technical Report TR318. Predictions of entrainment by Sizewell C in relation to adjacent fish and crustacean populations and their fisheries. Cefas, Lowestoft.

BEEMS Technical Report TR325. Jellyfish and ctenophores in relation to Sizewell. Cefas, Lowestoft.

BEEMS Technical Report TR333. Sizewell site: Modelling the optimal position for a fish recovery and return outfall for Sizewell C. Cefas, Lowestoft.

BEEMS Technical Report TR339. Cefas Comprehensive Impingement Monitoring Programme 2014-2017. Cefas, Lowestoft.

BEEMS Technical Report TR345. Sizewell characterisation report - fish. Cefas, Lowestoft.

BEEMS Technical Report TR348. Sizewell benthic ecology characterisation. Cefas, Lowestoft.

BEEMS Technical Report TR359 Acoustic surveying of pelagic fish off Sizewell (2015). Cefas, Lowestoft.

BEEMS Technical Report TR380. Sizewell juvenile fish survey 2016. Cefas, Lowestoft.

BEEMS Technical Report TR382. Smelt populations in the vicinity of Sizewell. Cefas, Lowestoft.

BEEMS Technical Report TR383. Sizewell Equivalent Adult Value (EAV) metrics. Cefas, Lowestoft.

BEEMS Technical Report TR423. The origins of smelt (*Osmerus eperlanus*) populations at Sizewell. Cefas, Lowestoft.

BEEMS Technical Report TR426. Hinkley Point. Equivalent Adult Value (EAV) metrics. Cefas, Lowestoft.

BEEMS Technical report TR431. Sizewell SPA/SAC features and associated marine prey species. Cefas, Lowestoft.

BEEMS Technical Report TR456. Revised predictions of impingement effects at Hinkley Point C - 2018. Cefas, Lowestoft.

BEEMS Technical report TR493. The effect of not fitting an AFD system at HPC on the operation of the HPC FRR systems. Cefas, Lowestoft.

BEEMS Technical report TR498. Sizewell C suspended sediment concentration at proposed cooling water intake locations. Cefas, Lowestoft.

BEEMS Technical Report TR511. Particle Tracking Study of Impinged Sprat from the Proposed Sizewell C Fish Recovery and Return. Cefas, Lowestoft.

BEEMS Technical Report TR520. Sizewell C influence of the fish recovery and return system on water quality and ecological receptors. Cefas, Lowestoft.

Berstedt, R.A., Seelye, J.G., 1995. Evidence for lack of homing by sea lampreys. Trans. Am. Fish. Soc. 124, 235–239.

Beullens, K., Eding, E.H., Ollevier, F., Komen, J., Richter, C.J.J., 1997. Sex differentiation, changes in length, weight and eye size before and after metamorphosis of European eel (*Anguilla anguilla* L.) maintained in captivity. Aquaculture 153, 151–162.

Bird, D.J., 2008. The biology and conservation of the fish assemblage of the Severn Estuary (cSAC). Report

Number CCW/SEW/08/1. 79 pp.

Bird, D. J., Rotchell, J. M., Hesp, S. A., Newton, L. C., Hall, N. G. and Potter, I. C. (2008) To what extent are hepatic concentrations of heavy metals in Anguilla anguilla at a site in a contaminated estuary related to body size and age and reflected in the metallothionein concentrations? Environmental Pollution, 151, 641-651.

Bracken, F.S.A., Hoelzel, A.R., Hume, J.B., Lucas, M.C., 2015. Contrasting population genetic structure among freshwater-resident and anadromous lampreys: the role of demographic history, differential dispersal, and anthropogenic barriers to movement. Mol. Ecol. 24, 1188–1204.

Caddy, J.F., Csirke, J., 1983. Approximations to sustainable yield for exploited and unexploited stocks. Ocean. Trop. 18, 3–15.

Cefas, 2017a. Edible crab (*Cancer pagurus*) Cefas Stock Status Report 2017. Report to Defra. Cefas, Lowestoft.

Cefas, 2017b. Lobster (*Homarus gammarus*) Cefas Stock Status Report 2017. Report to Defra. Cefas, Lowestoft.

Claridge, P.N., Gardner, D.C., 1978. Growth and movements of the twaite shad, *Alosa fallax* (Lacepede), in the Severn Estuary. J. Fish Biol. 12, 203–211.

Colclough, S., Coates, S., 2013. A review of the status of smelt *Osmerus eperlanus* (L.) in England and Wales – 2013. Report EA/001 to the Environment Agency, York, UK, by SC2, Chatham, Kent.

Colclough, S.R., Gray, G., Bark, A., Knights, B., 2002. Fish and Fisheries of the Tidal Thames: management of the modern resource, research aims and future pressures. J. Fish Biol. 61 (Supple, 64–73.

Defra, 2018. Report to the European Commission in line with Article 9 of the Eel Regulation 1100/2007: implementation of UK Eel Management Plans.

Defra, 2015. Report to the European Commission in line with Article 9 of the Eel Regulation 1100/2007: implementation of UK Eel Management Plans.

Defra, 2012. Report to the European Commission in line with Article 9 of the Eel Regulation 1100/2007: implementation of UK Eel Management Plans June 2012. http://webarchive.nationalarchives.gov.uk/20130402151656/http://archive.defra.gov.uk/foodfarm/fisheries/document.

Defra, 2010. Eel Management plans for the United Kingdom: Anglian River Basin District.

Dekker, W. 2000. A procrustean assessment of the European eel stock. Ices J. Mar. Sci. 57(4) pp 938-947.

Dekker, W. 2003. Did lack of spawners cause the collapse of the European Eel, Anguilla anguilla. Fish.Mar. Ecol. 10(6) pp 365-376.

Dekker, W. and Beaulaton, L. 2015. Climbing back up what slippery slope? Dynamics of the European eel stock and its management in historical perspective. ICES J Mar Sci. V73, issue 1, pp 5-13.

Dulvy, N.K., Rogers, S.I., Jennings, S., Stelzenmuller, V., Dye, S., Skjoldal, H.R., 2008. Climate change and deepening of the North Sea fish assemblage: a biotic indicator of warming seas. J. Appl. Ecol. 45, 1029–1039.

Environment Agency 2005. Screening for Intake and Outfalls: a best practice guide. Science Report SC030231. Environment Agency February 2005. Available from: http://publications.environmentagency.gov.uk.

Environment Agency 2010. Cooling Water Options for the New Generation of Nuclear Power Stations in the

UK. Evidence Report SC070015/SR3, Environment Agency. Available from: http://publications.environmentagency.gov.uk.

Environment Agency, 2013a. Salmonid & freshwater fisheries statistics for England & Wales 2012. 41pp.

Environment Agency, 2013b. Salmonid & freshwater fisheries statistics for England & Wales 2011 41pp.

Environment Agency, 2013c. Salmonid & freshwater fisheries statistics for England & Wales 2010. 41pp.

Environment Agency, 2014. Salmonid & freshwater fisheries statistics for England & Wales 2013. 41pp.

Environment Agency, 2014. Salmonid & freshwater fisheries statistics for England & Wales 2013. 41pp.

Environment Agency, 2015. Salmonid and freshwater fisheries statistics for England and Wales 2014. 40pp.

Environment Agency, 2017a. Salmonid and freshwater fisheries statistics for England and Wales 2016. 35pp.

Environment Agency, 2017b. Salmonid and freshwater fisheries statistics for England and Wales 2015. 38pp.

Environment Agency , 2018. Salmonid and freshwater fisheries statistics for England and Wales 2017. 37pp.

Environment Agency. 2020. Nuclear power station cooling waters: protecting biota. April 2020. SC180004/R1. 190pp.

Fox, C.J. 2001. Recent trends in stock-recruitment of Blackwater herring (Clupea harengus) in relation to larval production. ICES J.MAr. Sci. 58, pp 750-762.

Giannini, F., Hobsbawn, P.I., Begg, G.A., Chambers, M., 2010. Management strategy evaluation (MSE) of the harvest strategy for the Small Pelagic Fishery, Bureau of Rural Sciences. March 2010. ISBN 978-1-921192-50-0.

Harrison, A.J. ,Walker, A.M ., Pinder, A.C., Briand, C., Aprahamian. M. W. 2014. A review of glass eel migratory behaviour, sampling techniques and abundance estimates in estuaries implications for assessing recruitment, local production and exploitation. Rev. Fish Biol. Fisheries V24, Issue 1, pp 967-983

Hall, D.L., Hilborn, R., Stocker, M., Walters, C.J., 1988. Alternative harvest strategies for Pacific herring (*Clupea harengus pallasi*). Can. J. Fish. Aquat. Sci. 45, 888–897.

Hardisty, M.W., 1986. A general introduction to lampreys, in: Holcík (Ed.), The Freshwater Fishes of Europe. Vol. 1, Part 1. Petromyzontiformes. AULA-Verlag GmbH, Wiesbaden. Holcík, J., p. 313.

Heath, M.R., Neat, F.C., Pinnegar, J.K., Read, D.G., Sims, D.W., Wright, P.J., 2012. Review of climate change impacts on marine fish and shellfish around the UK and Ireland. Aquat. Conserv. 22, 337–367.

Heesen, H.J.L., Daan, N., Ellis,J.R. (2015). Fish atlas of the Celtic Sea, North Sea and Baltic Sea. Wageningen Academic Publishers.

Helcom, 2013. Species Information sheet: *Alosa fallax*. Downloaded from www.helcom.fi 1/June/2019.

ICES 2020.Official Nominal Catches 2006-2017. Accessed 16/04/2020 via http://ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx

ICES, 2019. Official Nominal Catches 2006 - 2017. Version 07-08-2018. Accessed 10-08-2018 via http://ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx. ICES, Copenhagen.

ICES WGCSE 2019. Report Section 29 -Seabass (Dicentrarchus labrax) in divisions 4.b–c, 7.a, and 7.d–h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea).

ICES. 2019b. Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK). ICES Scientific Reports. 1:7. 1271 pp. Available at: http://doi.org/10.17895/ices.pub.5402

ICES 2019c. Thornback ray (*Raja clavata*) in Subarea 4 and in divisions 3.a and 7.d (North Sea, Skagerrak, Kattegat, and eastern English Channel). ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion. Published 4 October 2019. Available at: http://ices.dk/sites/pub/Publication%20Reports/Advice/2019/2019/rjc.27.3a47d.pdf

ICES, 2018a. Report of the Herring Assessment Working Group for the Area South of 62°N (HAWG) 29-31 January 2018 and 12-20 March 2018 ICES HQ, Copenhagen, Denmark. ICESCM 2018/ACOM:07.

ICES, 2018b. Report of the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK) 24 April - 3 May 2018 Oostende, Belgium. ICES CM 2018/ACOM:22.

ICES, 2018c. Report of the Working Group on Celtic Seas Ecoregion (WGCSE) 9–18 May 2018 Copenhagen, Denmark (DRAFT). ICES CM 2018/ACOM:13.

ICES, 2018d. Report of the Working Group on Southern Horse Mackerel, Anchovy and Sardine (WGHANSA) 26–30 June 2018 Lisbon, Portugal. ICES CM 2018/ACOM:17.

ICES, 2018e. Report of the Working Group on Elasmobranch Fishes (WGEF) 19–28 June 2018 Lisbon, Portugal. ICES CM 2018/ACOM:16.

ICES, 2018f. Report of the Working Group on Widely Distributed Stocks (WGWIDE), 28 August- 3 September 2018, Torshavn, Faroe Islands. ICES CM 2018/ACOM: 23. 488 pp.

ICES. 2018e. Report of the Benchmark Workshop on Seabass (WKBASS), 20–24 February 2017 and 21–23 February 2018, Copenhagen, Denmark. ICES CM 2018/ACOM:44. 283 pp.

ICES, 2013. Report of the 2013 session of the Joint EIFAAC/ICES Working Group on Eels (WGEEL), 18-22 March 2013 in Sukarietta, Spain, and 4-10 September in Copenhagen, Denmark 850 pp.

Jacoby, D., Gollock, M., 2014. *Anguilla anguilla*. The IUCN Red List of Threatened Species 2014: e.T60344A45833138. http://dx.doi.org/10.2305/IUCN.UK.2014-1.RLTS.T60344A45833138.en. https://doi.org/http://dx.doi.org/10.2305/IUCN.UK.2014-1.RLTS.T60344A45833138.en

Jansen, H.M., Winter, H. V., Bult, T.P., 2007. Bijvangst van trekvissen in de Nederlandse fuikenvisserij. MARES report CO48/07.

Jennings, S., Lancaster, J.E., Ryland, J.S., Shackley, S.E. 1991. The age structure and growth dynamics of young-of-the-year bass, *Dicentrarchus labrax*, populations. JMBA 71, 799-810.

Jones, M. 2013. Investigating the Ecological and Economic Consequences of Marine Climate Change in UK Waters. PhD Thesis. University of East Anglia, UK, 2013

JNCC, 2013. Third Report by the United Kingdom under Article 17 on the implementation of the Directive from January 2007 to December 2012, Species: S1099 - River lamprey (*Lampetra fluviatilis*), 18 pp.

Jolly, M.T., Aprahamian, M.W., Hawkins, S.J., Henderson, P.A., Hillman, R., O Maoileidigh, N., Maitland, P.S., Piper, R., M.J., G., 2012. Population genetic structure of protected Allis shad (*Alosa alosa*) and twaite shad (*Alosa fallax*). Mar. Biol. 159, 675–687.

Knights, B., 2003. A review of the possible impacts of long-term oceanic and climate changes and fishing mortality on recruitment of anguillid eels of the Northern Hemisphere. Sci. Total Environ. 310, 237–244. https://doi.org/10.1016/S0048-9697(02)00644-7

Larsen, L.O., 1980. Physiology of adult lampreys, with special regard to natural starvation, reproduction and death after spawning. Can. J. Fish. Aquat. Sci. 37, 1762–1769.

Maes, G.E., Pujolar, J.M., Hellemans, B., Volckaert, F.A.M., 2006. Evidence for isolation by time in the European eel (*Anguilla anguilla* L.). Mol. Ecol. 15, 2095–2107.

Magath, V., Thiel, R., 2013. Stock recovery, spawning period and spawning area of the twaite shad *Alosa fallax* in the Elbe estuary, southern North Sea. Endanger. Species Res. 20.

Maitland, P.S., 2003a. The status of smelt *Osmerus eperlanus* in England, Conserving Natura 2000 Rivers Ecology Series No. 5.

Maitland, P.S., 2003b. Ecology of the River, Brook and Sea Lamprey, Conserving Natura 2000 Rivers Ecology Series No. 5.

Maitland, P.S., 1972. A key to the freshwater fishes of the British Isles with notes on their distribution and ecology. Freshwater Biological Association Scientific Publication, 27. 137 pp.

Masters, J.E.G., Jang, M., Ha, K., Bird, P.D., Frear, P.A., Lucas, M.C., 2006. The commercial exploitation of a protected anadromous species, the river lamprey *Lampetra fluviatilis* L., in the tidal River Ouse, north-west England. Aquat. Conserv. Mar. Freshw. Ecosyst. 16, 77–92.

McCleave,J.D. 1980. Swimming performance of European Eel (Anguilla anguilla) elvers. J.Fish.Biol. 16(4) pp 445-452.

McCleave, J.D., 1993. Physical and behavioural controls on the oceanic distribution and migration of leptocephali. J. Fish Biol. 40(Suppl., 243–273.

MMO 2020. Marine Management Organisation. UK sea fisheries statistics 2019. Available at: https://www.gov.uk/government/statistics/uk-sea-fisheries-annual-statistics-report-2019. Accessed on 16/10/2020.

Palm, S., Dannewitz, J., Prestegaard, T., Wickström, H., 2009. Panmixia in European eel revisited: no genetic difference between maturing adults from southern and northern Europe. Heredity (Edinb). 103, 82–89.

Parmesan, C., Yohe, G., 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421, 37–42.

Pawson, M.G., Pickett, G.D., Lebaalleur, J. Brown, M. and Fritsch, M. 2007. Migrations, fishery implications and management units of seabass (*Dicentrarchus labrax*) in Northwest Europe. ICES J.Mar. Sci. 64 pp 332-345.

Perry, A.L., Low, P.J., Ellis, J.R., Reynolds, J.D., 2005. Climate Change and Distribution Shifts in Marine Fishes. Science (80-.). 308, 1912–1915.

Petitgas, P., Rijnsdorp, A.D., Dickey-Collas, M., Engelhard, G.H., Peck, M.A., Pinnegar, J.K., Drinkwater, K., Huret, M., Nash, R.D.M., 2013. Impacts of climate change on the complex life cycles of fish. Fish. Oceanogr. 22, 121–139.

Pinnegar, J.K.., Garrett, A, Simpson, S.D., George H. Engelhard, G.H. and Van der Kooij, J. 2017. Fisheries. Marine Climate Change Impacts Partnership. MCCIP Science Review 2017: 73 - 89

Reiss, H., Kroncke, I., Ehrich, S., 2006. Estimating the catching efficiency of a 2-m beam trawl for sampling epifauna by removal experiments. ICES J. Mar. Sci. 63, 1453–1464.

Rogers, S.I., Millner, R.S., 1996. Factors affecting the annual abundance and regional distribution of English inshore demersal fish populations: 1973 to 1995. ICES J. Mar. Sci. 53, 1094–1112.

Rosenberg, A.A., Fogarty, M.J., Sissenwine, M.P., Bedington, J.R., Shepherd, J.G., 1993. Achieving sustainable use of renewable resources. Science (80-.). 262, 828–829.

Sabatino, S., Alexandrino, P., 2012. Genetic diversity and population structure of the Eurasian shads *Alosa alosa* and *Alosa fallax*. CIBIO report AARC project - Activity 4, December 2012.

Simpson, S.D., Jennings, S., Johnson, M.P., Blanchard, J.L., Schon, P.-J., Sims, D.W., Genner, M.J., 2011. Continental Shelf-Wide Response of a Fish Assemblage to Rapid Warming of the Sea. Curr. Biol. 21, 1565–1570.

Tesch, F.W., 2003. The Eel, 5th ed. John Wiley & Sons.

Thames Tideway Tunnel, 2013. Application for Development Consent, Application Reference Number: WWO10001. Needs Report. Doc Ref: 8.3. Appendix F - Tideway Fisheries Review, Thames Water Utilities Limited.

Turnpenny, A.W.H., Colclough, S., Holden, S.D.J., Bridges, M., Bird, H., O'Keeffe, N., Hinks, C., 2004. Thames Tideway Strategy: Experimental Studies on the Dissolved Oxygen Requirements of Fish. Consultancy Report no.FCR374/04 to Thames Water Utilities, Ltd. Fawley Aquatic Research, Fawley Southampton, April, 2004.

Turnpenny, A.W.H., 1989. The equivalent adult approach for assessing the value of juvenile fish kills, with reference to commercial species in British Water. CERL Report No. RD/L/3454/R89.

Turnpenny, A.W.H., Utting, N.J., 1987. The seasonal incidence of fish species at the cooling water intake of Sizewell Power Station. CEGB.

Turnpenny, A.W.H., Taylor, C.J.L., 2000. An assessment of the effect of the Sizewell power stations on fish populations. Hydroecol. Appl 12, 87–134.

Turnpenny, A. 2011. Trials and tribulations of fish recovery and return. International Fish Screening Techniques 2011. WIT Transactions on State of the Art in Science and Engineering, Vol 71; 111-123.

UKTAG (2014). Practitioners Guide to the Transitional Fish Classification Index (TFCI), Water Framework Directive: Transitional Waters. UKTAG Version 07 301112

Vøllested, L.A., 1992. Geographic variation in age and length at metamorphosis of maturing European eel: environmental effects and phenotypic plasticity. J. Anim. Ecol. 61, 41–48.

Wheeler, A., 1969. The fishes of the British Isles and North West Europe. Macmillan and Co Ltd, London.

Appendix A Cooling water system design

The cooling water system for SZC will essentially be the same as for HPC. (There will be detailed differences for example in the layout of the FRR system, the need or not for an Archimedes screw but none of these will affect the calculations in this report). The key design features are:

a. the total cooling water abstraction at SZC will be approximately 131.86 cumecs with a maximum of 9% of the total cooling water flow supplying the essential and auxiliary cooling water systems via band screens and the remaining 91% (120 cumecs) supplying the main cooling water systems (CRF) via the station drum screens.

b. the SZC band screens will be fitted with their own FRR systems

c. the trash rack bar spacing for SZC will be 75 mm. The SZC trash rack will have a rake which returns impinged materials (including fish) to the FRR system.

d. the SZC FRR system will not be chlorinated unless there is a major change in the future water quality conditions that would facilitate the rapid growth of biofouling organisms but this is considered unlikely.

A.1 Main cooling water systems in each pumping station

SZC will consist of two EPR units. Each unit has its own forebay, pumping station, debris recovery building (HCB) and discharge pond. Each pumping station is divided into four distinct sectors: two central sectors (four channels (or 'trains') each) with high flow volume drum screens (ds2 and ds3) and two lateral sectors (one channel (or 'train') each) with lower flow volume band screens (bs1 and bs4).

Each pumping station supplies seawater to a number of systems; the main ones of which are:

CRF: Cooling Water System used to extract waste heat from the turbine steam condensers.

SEC: Essential Cooling Water system (Nuclear Island)

SEN: Auxiliary Cooling Water system (Conventional Island)

SRU: Ultimate cooling water system (Emergency use only)

CFI: Circulating Water Filtration system: supplies wash water for the drum and band screens.

The schematic layout of each pump station is shown in Figure 13.

At Mean Sea Level (MSL) the system flow rates per unit are as follows:

CRF 2*30 cumecs per unit (supplied from the 2 drum screens in each pump station)

SEC 2*1.2 cumecs per unit (can be supplied from the drum screens or band screens in any combination)

SEN 2*1.61 cumecs per unit (normally supplied from the 2 band screens in each pump station)

SRU Negligible flow (only used when testing the system or in emergency)

CFI additional to SEC flow consisting of 2*0.117 cumecs for the 2 drum screens and a worst case of 2*0.039 cumecs for the 2 band screens.

As the SEC/CFI seawater sources can be from the drum screens or band screens there is a range of different water flows through the different filtration systems at SZC.

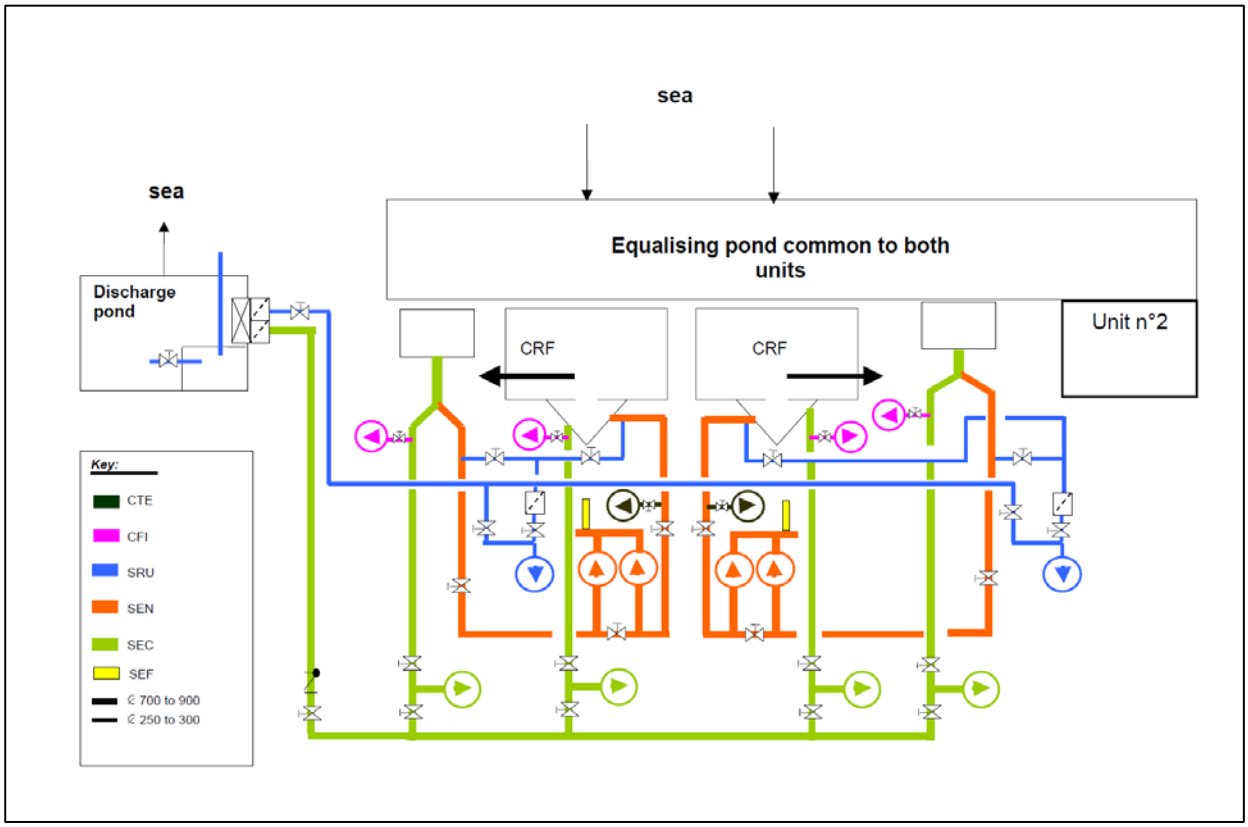


Figure 13 Illustrative schematic of EPR cooling water circuits for each unit (Source EDF CNEPE E.T.DOMA/09 0119 A1 Approved). The equalising pond shown in the figure is the station forebay and SZC has 1 forebay for each unit

Table 38 details the minimum flow at MSL (mean sea level) through the drum screens and Table 39 shows the maximum flow through the drum screens at MSL. Dependent upon the system configuration the seawater flow through the band screens can, therefore, vary between 4.9% and 9% of the total seawater abstraction of 131.86 cumecs.

Table 38 Cooling water flow volumes when SEC/CFI systems are supplied from the band screens

	Channel	flow (cumecs)	Flow through	cumecs		cumecs
	bs1	2.966	drum screens	60		
	ds2	30	band screens	5.932		
	ds3	30	Total CW flow	65.932	of which CRF	60
	bs4	2.966				
Total flow/EPR		65.932				
	2 EPRs	131.86	Flow through drum screens			120
			Total CW flow			131.86
			Band screen flow as % of total flow			**9.0%**

Table 39 Cooling water flow volumes when SEC/CFI systems are supplied from the drum screens

	Channel	flow (cumecs)	Flow through	cumecs		cumecs
	bs1	1.61	drum screens	62.712		
	ds2	31.356	band screens	3.22		
	ds3	31.356	total CW flow	65.932	of which CRF	60
	bs4	1.61				
Total flow/EPR		65.932				
	2 EPRs	131.86	Flow through drum screens			125.42
			Total CW flow			**131.86**
			Band screen flow as % of total flow			**4.9%**

Appendix B Calculated annual impingement by number at SZB and SZC without mitigation – all species

B.1 Calculated annual impingement by number at SZB and SZC without mitigation – all species

Annually raised and unmitigated number of individuals that are estimated to be impinged by SZB and predicted to be impinged by SZC, based on data from 2009-2017. Blue shading indicates the 24 key finfish species in the Greater Sizewell Bay (BEEMS Technical Report TR345) and the type of model that was used to estimate impingement numbers for each species (BEEMS Technical Report TR345). The 31 invertebrate species marked with an asterisk (*) were only enumerated from 2014 onwards, all other species were enumerated from 2009 onwards.

Key finfish and shellfish species

		SZB - estimate			SZC - prediction		
Common name	**Scientific name**	**Mean**	**Lower**	**Upper**	**Mean**	**Lower**	**Upper**
Finfish							
Sprat	*Sprattus sprattus*	2,703,033	1,427,342	4,549,040	6,920,815	3,654,550	11,647,309
Herring	*Clupea harengus*	1,018,211	598,405	1,550,795	2,607,016	1,532,149	3,970,636
Whiting	*Merlangius merlangus*	650,479	452,300	888,210	1,665,479	1,158,064	2,274,163
European seabass	*Dicentrarchus labrax*	276,943	124,683	500,639	709,082	319,238	1,281,830
Sand gobies	*Pomatoschistus spp*	196,545	81,220	383,493	503,232	207,954	981,891
Dover sole	*Solea solea*	91,370	63,510	125,792	233,942	162,609	322,077
European anchovy	*Engraulis encrasicolus*	69,058	18,123	166,131	176,816	46,401	425,360
Dab	*Limanda limanda*	56,054	33,869	93,069	143,519	86,719	238,293
Thin-lipped grey mullet	*Liza ramada*	49,751	13,394	97,783	127,382	34,295	250,361
Transparent goby	*Aphia minuta*	36,461	12,857	79,828	93,354	32,920	204,390
Bib	*Trisopterus luscus*	24,975	12,650	40,450	63,944	32,389	103,568
Lesser weever fish	*Echiichthys (trachinus) vipera*	20,222	12,839	30,149	51,777	32,872	77,192
Flounder	*Platichthys flesus*	14,569	10,943	19,251	37,303	28,018	49,291
Nilsson's pipefish	*Syngnathus rostellatus*	14,247	2,315	29,139	36,478	5,927	74,608

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Pogge (hooknose)	*Agonus cataphractus*	10,410	7,185	14,587	26,654	18,397	37,348
European plaice	*Pleuronectes platessa*	9,653	6,328	14,355	24,716	16,202	36,755
Five-bearded rockling	*Ciliata mustela*	8,941	5,773	13,071	22,892	14,782	33,466
Cucumber smelt	*Osmerus eperlanus*	8,122	5,405	11,295	20,795	13,840	28,921
Atlantic cod	*Gadus morhua*	6,415	2,440	11,433	16,426	6,248	29,272
Lesser spotted dogfish	*Scyliorhinus canicula*	5,359	2,543	9,415	13,722	6,512	24,106
Great pipefish	*Syngnathus acus*	4,672	2,132	7,734	11,962	5,459	19,802
Grey mullets	*Mugilidae*	4,081	0	17,752	10,449	0	45,451
Thornback ray	*Raja clavata*	2,913	1,808	4,292	7,460	4,629	10,989
Common sea snail	*Liparis liparis*	2,902	1,514	4,584	7,431	3,876	11,736
Tub gurnard	*Trigla (chelidonichthys) lucerna*	2,259	1,379	3,497	5,783	3,531	8,955
Pilchard	*Sardina pilchardus*	1,769	374	5,736	4,530	958	14,687
Unidentified herrings	*Clupeidae*	1,695	0	7,410	4,340	0	18,974
Common dragonet	*Callionymus lyra*	1,614	690	2,993	4,133	1,766	7,662
Poor cod	*Trisopterus minutus*	1,444	298	3,114	3,697	762	7,973
Starry smooth hound	*Mustelus asterias*	1,379	683	2,202	3,531	1,749	5,638
European eel	*Anguilla anguilla*	1,184	762	1,719	3,031	1,951	4,402
Twaite shad	*Alosa fallax*	1,168	574	2,036	2,989	1,470	5,213
River lamprey	*Lampetra fluviatilis*	1,144	630	1,932	2,929	1,614	4,946
Three-spined stickleback	*Gasterosteus aculeatus*	1,121	314	3,012	2,869	804	7,711
Common sandeel	*Ammodytes tobianus*	1,021	377	2,179	2,615	965	5,578
Bullrout	*Myoxocephalus scorpius*	972	447	1,694	2,488	1,145	4,336
Scald fish	*Arnoglossus laterna*	803	416	1,323	2,057	1,066	3,388
Horse-mackerel	*Trachurus trachurus*	695	220	1,715	1,779	564	4,391
Witch	*Glyptocephalus cynoglossus*	664	4	2,043	1,701	11	5,231
Black goby	*Gobius niger*	576	72	1,699	1,475	183	4,350

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Great sandeel	*Hyperoplus lanceolatus*	565	201	1,001	1,448	514	2,563
Brill	*Scophthalmus rhombus*	445	148	950	1,138	380	2,433
Rock goby	*Gobius paganellus*	405	12	1,511	1,036	30	3,869
Snake pipefish	*Entelurus aequoreus*	362	3	1,638	927	9	4,193
Lemon sole	*Microstomus kitt*	333	125	652	853	319	1,669
Tompot blenny	*Parablennius gattorugine*	247	26	802	632	66	2,053
Montague's seasnail	*Liparis montagui*	231	9	752	592	22	1,926
Solenette	*Buglossidium luteum*	221	68	446	567	175	1,141
Sand Smelt	*Atherina boyeri*	188	8	462	481	20	1,183
Mackerel	*Scomber scombrus*	185	17	502	473	43	1,287
Red mullet	*Mullus surmuletus*	145	41	309	372	104	790
Butter fish	*Pholis gunnellus*	138	47	257	353	120	657
Sea scorpion	*Taurulus bubalis*	129	14	324	331	35	831
Jeffrey's goby	*Buenia jeffreysii*	110	0	606	282	0	1,551
Sandeels	*Ammodytidae*	95	0	296	243	0	757
Baillons wrasse	*Symphodus (crenilabrus) balloni*	83	0	259	213	0	662
Grey gurnard	*Eutrigla (chelidonicthys) gurnardus*	82	15	197	210	38	505
Garfish	*Belone belone*	71	13	165	183	34	423
Sand smelt	*Atherina presbyter*	70	0	292	179	0	747
Eelpout (Viviparous blenny)	*Zoarces viviparus*	68	0	211	174	0	541
Turbot	*Scophthalmus maximus (psetta maxima)*	64	6	155	163	17	396
Lesser forkbeard (tadpolefish)	*Raniceps raninus*	60	2	174	153	5	445
Frie's goby	*Lesueurigobius friesii*	55	0	168	142	0	431
Corkwing wrasse	*Crenilabrus melops*	40	0	118	101	0	303
Northern rockling	*Ciliata septentrionalis*	40	0	128	101	0	327
John dory	*Zeus faber*	35	0	96	89	0	246

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Sandeel	*Ammodytes marinus*	27	0	110	70	0	283
Tope	*Galeorhinus galeus*	27	0	106	70	0	271
Lumpsucker	*Cyclopterus lumpus*	24	0	68	62	0	175
Four-bearded rockling	*Enchelyopus cimbrius*	23	0	61	59	0	156
Ballan wrasse	*Labrus bergylta*	22	0	86	58	0	219
Spotted ray	*Raja montagui*	21	0	67	53	0	171
Saithe	*Pollachius virens*	20	0	117	51	0	300
Crystal goby	*Crystallogobius linearis*	16	0	91	42	0	233
Thick-lipped grey mullet	*Crenimugil labrosus*	12	0	68	30	0	175
Norway bullhead	*Micrenophrys lilljeborgii*	9	0	45	24	0	115
Black seabream	*Spondyliosoma cantharus*	9	0	42	23	0	108
Cuckoo wrasse	*Labrus mixtus*	9	0	50	22	0	129
Snake blenny	*Lumpenus lampretaeformis*	8	0	46	20	0	118
Goldsinny	*Ctenolabrus rupestris*	8	0	46	19	0	118
Bigeye rockling	*Antonogadus macropthalmus*	6	0	20	16	0	51
Deep-snouted pipefish	*Syngnathus typhle*	5	0	31	13	0	78
Sea Trout	*Salmo trutta*	4	0	23	10	0	58
Norway pout	*Trisopterus esmarkii*	3	0	19	8	0	50
Shore rockling	*Gaidropsarus mediterraneus*	3	0	15	8	0	39
Red gurnard	*Aspitrigla (chelidonichthys) cuculus*	3	0	17	7	0	42
Sea lamprey	*Petromyzon marinus*	2	0	11	5	0	29
Spotted dragonet	*Callionymus maculatus*	2	0	10	4	0	26
Sand sole	*Pegusa lascaris*	2	0	10	4	0	26
Pollack	*Pollachius pollachius*	2	0	10	4	0	25
Allis shad	*Alosa alosa*	1	0	8	3	0	21

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Invertebrates							
Ctenophores*		63,722,358	27,658,442	113,953,359	163,153,983	70,816,352	291,764,853
Brown shrimp	*Crangon crangon*	6,022,385	4,293,876	8,357,183	15,419,644	10,993,990	21,397,635
Pink shrimp	*Pandalus montagui*	1,173,611	798,148	1,660,660	3,004,899	2,043,569	4,251,936
Common prawn	*Palaemon serratus*	1,130,928	603,033	1,846,382	2,895,616	1,543,998	4,727,454
Common swimming crab	*Polybius (liocarcinus) holsatus*	591,467	316,403	1,018,651	1,514,384	810,114	2,608,143
	Crangon allmanni	391,498	180,585	668,201	1,002,386	462,367	1,710,854
Jellyfish*		232,726	1,856	931,667	595,869	4,751	2,385,429
Plumose anemone*	*Metridium senile*	225,690	150,164	325,554	577,855	384,479	833,546
Isopod*	Idoteidae	108,544	18,886	282,587	277,915	48,356	723,534
Dahlia anemone*	*Urticina (tealia) felina*	82,123	56,554	111,967	210,266	144,799	286,679
Edible crab	*Cancer pagurus*	53,188	28,452	82,122	136,182	72,847	210,263
Little cuttlefish	*Sepiola atlantica*	28,028	19,254	40,383	71,762	49,298	103,397
Velvet swimming crab	*Necora puber*	26,707	8,220	55,431	68,379	21,046	141,924
Edible mussel*	*Mytilus edulis*	20,796	0	55,958	53,245	0	143,273
Green shore crab	*Carcinus maenas*	16,703	7,837	29,840	42,767	20,067	76,401
Unidentified spider crab*	Macropodia spp	14,609	920	42,144	37,405	2,355	107,905
Common starfish	*Asterias rubens*	12,098	2,694	24,294	30,975	6,896	62,201
European common squid	*Loligo (alloteuthis) subulata*	9,511	5,872	14,634	24,352	15,034	37,469
Opossum shrimps*	Mysidacea	9,417	0	33,445	24,112	0	85,632
Ragworms*	*Nereis* spp.	9,286	4,269	15,291	23,777	10,929	39,151
Long-leg spider crab	*Macropodia rostrata*	7,100	91	27,939	18,179	232	71,533
Scaleworms*		5,260	2,120	9,283	13,467	5,428	23,769
Anemone unidentified*	Anemone unidentified	4,748	84	12,486	12,157	216	31,968

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Crangonid (brown) shrimps*	Crangonidae	3,564	0	14,727	9,125	0	37,706
Sea slugs*	Nudibranchia	3,249	232	8,957	8,318	594	22,933
Hairy crab	*Pilumnus hirtellus*	2,980	386	7,589	7,629	989	19,430
Ghost shrimp	*Pasiphaea sivado*	1,848	0	10,189	4,733	0	26,089
Rissos crab*	*Xantho pilipes*	1,755	0	6,782	4,494	0	17,364
*	*Bolocera tuediae*	1,486	0	3,808	3,804	0	9,750
Long clawed porcelain crab*	*Pisidia longgicornis*	1,330	0	6,869	3,406	0	17,588
Lug-worm*	*Arenicola marina*	1,325	285	2,997	3,393	730	7,672
Swimming crab	*Liocarcinus depurator*	1,041	0	6,256	2,665	0	16,018
Slender spider crab*	*Macropodia tenuirostris*	792	8	2,276	2,028	20	5,827
Beadlet anemone*	*Actinia equina*	680	0	2,462	1,740	0	6,305
Contracted crab	*Hyas coarctatus*	518	13	1,635	1,327	33	4,186
Hermit crab	*Eupagurus bernhardus*	466	0	2,581	1,193	0	6,608
*	*Psammechinus miliaris*	438	15	1,464	1,121	39	3,748
	Processa canaliculata	427	31	1,184	1,094	79	3,032
Scorpion spider crab*	*Inachus dorsettensis*	412	0	2,275	1,055	0	5,825
Spider crabs*	Majidae	405	0	2,103	1,036	0	5,384
Hairy crab*	*Pilumnus spinifer*	402	0	2,445	1,030	0	6,259
Unidentified sea urchin		374	0	1,493	957	0	3,822
Unidentified brittlestar		321	0	1,599	821	0	4,094
Great spider crab	*Hyas araneus*	169	0	614	432	0	1,571
Dwarf-swimming crab	*Liocarcinus pusillus*	130	0	789	334	0	2,019
Marbled swimming crab	*Liocarcinus marmoreus*	93	0	489	239	0	1,253
Common cuttlefish	*Sepia officinalis*	68	0	361	175	0	924
European squid	*Loligo vulgaris*	59	0	349	152	0	893

Common name	Scientific name	SZB - estimate			SZC - prediction		
		Mean	Lower	Upper	Mean	Lower	Upper
Northern squid	*Loligo forbesi*	43	0	206	111	0	528
Xanthid crab*	Xanthidae	43	0	140	110	0	358
Necklace shell*	*Euspira (polinices) catena*	43	0	245	109	0	628
	Processidae	29	0	137	73	0	349
Bristle worms*	Polychaeta	25	0	106	65	0	271
Purple heart urchin*	*Spatangus purpureus*	16	0	92	40	0	236
Lobster	*Homarus gammarus*	15	0	51	37	0	132
*	*Upogebia deltaura*	13	0	78	34	0	200
*	Gammaridae	13	0	64	34	0	165
Brittlestar*	*Ophiura ophiura*	13	0	79	33	0	202
Cuttle-fish*	*Sepia elegans*	6	0	27	15	0	69
Sea cucumbers*	Holothuroidea	4	0	27	11	0	69
Unidentified swimming crab	*Liocarcinus* spp.	3	0	15	7	0	39
	Axius stirhynchus	2	0	11	5	0	29

* Species enumeration began in 2014.

Appendix C Mean, lower and upper numbers of fish estimated (SZB) and predicted SZC) to be impinged annually – full calculation tables

C.1 Predicted impingement without embedded mitigation measures

	Annually raised SZB estimate			Annually raised SZC prediction			SZC EAV equivalent numbers		
Species	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,703,033	1,427,342	4,549,040	6,920,815	3,654,550	11,647,309	5,199,288	2,745,494	8,750,084
Herring	1,018,211	598,405	1,550,795	2,607,016	1,532,149	3,970,636	1,864,587	1,095,822	2,839,874
Whiting	650,479	452,300	888,210	1,665,479	1,158,064	2,274,163	593,040	412,361	809,779
Bass	276,943	124,683	500,639	709,082	319,238	1,281,830	158,807	71,497	287,081
Sand goby	196,545	81,220	383,493	503,232	207,954	981,891	503,232	207,954	981,891
Sole	91,370	63,510	125,792	233,942	162,609	322,077	49,802	34,616	68,564
Anchovy	69,058	18,123	166,131	176,816	46,401	425,360	172,238	45,200	414,346
Dab	56,054	33,869	93,069	143,519	86,719	238,293	63,810	38,556	105,947
Thin-lipped grey mullet	49,751	13,394	97,783	127,382	34,295	250,361	10,618	2,859	20,869
Flounder	14,569	10,943	19,251	37,303	28,018	49,291	17,226	12,938	22,761
Plaice	9,653	6,328	14,355	24,716	16,202	36,755	8,536	5,596	12,694
Smelt	8,122	5,405	11,295	20,795	13,840	28,921	15,834	10,538	22,021
Cod	6,415	2,440	11,433	16,426	6,248	29,272	5,899	2,244	10,512
Thornback ray	2,913	1,808	4,292	7,460	4,629	10,989	1,438	892	2,118
Eel	1,184	762	1,719	3,031	1,951	4,402	3,031	1,951	4,402
Twaite shad	1,168	574	2,036	2,989	1,470	5,213	2,989	1,470	5,213
River lamprey	1,144	630	1,932	2,929	1,614	4,946	2,929	1,614	4,946
Horse mackerel	695	220	1,715	1,779	564	4,391	1,779	564	4,391
Mackerel	185	17	502	473	43	1,287	473	43	1,287
Tope	27	0	106	70	0	271	70	0	271
Sea trout	4	0	23	10	0	58	10	0	58
Sea lamprey	2	0	11	5	0	29	5	0	29
Allis shad	1	0	8	3	0	21	3	0	21
Salmon	0	0	0	0	0	0	0	0	0

Species	EAV equivalent weight (t)			Mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
	Mean	Lower	Upper		Mean	Lower	Upper		Mean	Lower	Upper
Sprat	54.6	28.8	91.9	220,757	0.02	0.01	0.04	151,322	0.04	0.02	0.06
Herring	351.8	206.7	535.8	2,198,449	0.02	0.01	0.02	400,244	0.09	0.05	0.13
Whiting	169.5	117.9	231.5	151,881	0.11	0.08	0.15	17,570	0.96	0.67	1.32
Bass	243.1	109.4	439.5	14,897	1.63	0.73	2.95	3,051	7.97	3.59	14.40
Sand goby	1.0	0.4	1.9	205,882,353	0.24	0.10	0.48	NA	NA	NA	NA
Sole	10.7	7.4	14.7	43,770	0.02	0.02	0.03	12,800	0.08	0.06	0.11
Anchovy	3.6	0.9	8.6	NA	NA	NA	NA	1,625	0.22	0.06	0.53
Dab	2.6	1.6	4.3	NA	NA	NA	NA	5,309	0.05	0.03	0.08
Thin-lipped grey mullet	5.5	1.5	10.9	NA	NA	NA	NA	120	4.60	1.24	9.05
Flounder	1.4	1.1	1.9	NA	NA	NA	NA	2,321	0.06	0.05	0.08
Plaice	2.1	1.4	3.1	690,912	0.00	0.00	0.00	80,367	0.00	0.00	0.00
Smelt	0.26	0.17	0.36	50	0.52	0.35	0.73	8	3.11	2.07	4.32
Cod	15.3	5.8	27.4	103,025	0.01	0.01	0.03	34,701	0.04	0.02	0.08
Thornback ray	4.6	2.8	6.8	NA	NA	NA	NA	695	0.66	0.41	0.97
Eel	1.0	0.6	1.4	79	1.27	0.82	1.84	14	7.18	4.62	10.43
Twaite shad	0.9	0.5	1.6	5,161,183	0.06	0.03	0.10	1	70.22	34.53	122.46
River lamprey	0.2	0.1	0.4	62	0.37	0.21	0.63	1	20.77	11.44	35.08
Horse mackerel	0.2	0.1	0.6	NA	NA	NA	NA	20,798	0.00	0.00	0.00
Mackerel	0.2	0.0	0.4	3,888,854	0.00	0.00	0.00	1,026,828	0.00	0.00	0.00
Tope	0.5	0.0	1.9	NA	NA	NA	NA	498	0.10	0.00	0.38
Sea trout	0.0	0.0	0.1	NA	NA	NA	NA	39,795	0.02	0.00	0.15
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Allis shad	0.0	0.0	0.0	27,397	0.01	0.00	0.08	0.2	1.31	0.00	7.72
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	0.00	0.00	0.00

C.2 Predicted SZC impingement with the effect of LVSE intake heads

	Annually raised SZB estimate			Annually raised SZC estimate			Intake head mortality			SZC EAV equivalent numbers		
Species	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,703,033	1,427,342	4,549,040	6,920,815	3,654,550	11,647,309	2,470,731	1,304,674	4,158,089	1,856,146	980,141	3,123,780
Herring	1,018,211	598,405	1,550,795	2,607,016	1,532,149	3,970,636	930,705	546,977	1,417,517	665,658	391,208	1,013,835
Whiting	650,479	452,300	888,210	1,665,479	1,158,064	2,274,163	594,576	413,429	811,876	211,715	147,213	289,091
Bass	276,943	124,683	500,639	709,082	319,238	1,281,830	253,142	113,968	457,613	56,694	25,525	102,488
Sand goby	196,545	81,220	383,493	503,232	207,954	981,891	179,654	74,239	350,535	179,654	74,239	350,535
Sole	91,370	63,510	125,792	233,942	162,609	322,077	83,517	58,052	114,981	17,779	12,358	24,477
Anchovy	69,058	18,123	166,131	176,816	46,401	425,360	63,123	16,565	151,853	61,489	16,136	147,922
Dab	56,054	33,869	93,069	143,519	86,719	238,293	51,236	30,959	85,070	22,780	13,764	37,823
Thin-lipped grey mullet	49,751	13,394	97,783	127,382	34,295	250,361	45,475	12,243	89,379	3,791	1,021	7,450
Flounder	14,569	10,943	19,251	37,303	28,018	49,291	13,317	10,002	17,597	6,150	4,619	8,126
Plaice	9,653	6,328	14,355	24,716	16,202	36,755	8,824	5,784	13,121	3,047	1,998	4,532
Smelt	8,122	5,405	11,295	20,795	13,840	28,921	7,424	4,941	10,325	5,653	3,762	7,861
Cod	6,415	2,440	11,433	16,426	6,248	29,272	5,864	2,230	10,450	2,106	801	3,753
Thornback ray	2,913	1,808	4,292	7,460	4,629	10,989	2,663	1,652	3,923	513	319	756
Eel	1,184	762	1,719	3,031	1,951	4,402	1,082	696	1,572	1,082	696	1,572
Twaite shad	1,168	574	2,036	2,989	1,470	5,213	1,067	525	1,861	1,067	525	1,861
River lamprey	1,144	630	1,932	2,929	1,614	4,946	1,046	576	1,766	1,046	576	1,766
Horse mackerel	695	220	1,715	1,779	564	4,391	635	201	1,568	635	201	1,568
Mackerel	185	17	502	473	43	1,287	169	15	459	169	15	459
Tope	27	0	106	70	0	271	25	0	97	25	0	97
Sea trout	4	0	23	10	0	58	4	0	21	4	0	21
Sea lamprey	2	0	11	5	0	29	2	0	10	2	0	10
Allis shad	1	0	8	3	0	21	1	0	7	1	0	7
Salmon	0	0	0	0	0	0	0	0	0	0	0	0

Species	EAV equivalent weight (t)			Mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
	Mean	Lower	Upper		Mean	Lower	Upper		Mean	Lower	Upper
Sprat	19.5	10.3	32.8	220,757	0.01	0.00	0.01	151,322	0.01	0.01	0.02
Herring	125.6	73.8	191.3	2,198,449	0.01	0.00	0.01	400,244	0.03	0.02	0.05
Whiting	60.5	42.1	82.6	151,881	0.04	0.03	0.05	17,570	0.34	0.24	0.47
Bass	86.8	39.1	156.9	14,897	0.58	0.26	1.05	3,051	2.84	1.28	5.14
Sand goby	0.3	0.1	0.7	205,882,353	0.09	0.04	0.17	NA	NA	NA	NA
Sole	3.8	2.6	5.2	43,770	0.01	0.01	0.01	12,800	0.03	0.02	0.04
Anchovy	1.3	0.3	3.1	NA	NA	NA	NA	1,625	0.08	0.02	0.19
Dab	0.9	0.6	1.5	NA	NA	NA	NA	5,309	0.02	0.01	0.03
Thin-lipped grey mullet	2.0	0.5	3.9	NA	NA	NA	NA	120	1.64	0.44	3.23
Flounder	0.5	0.4	0.7	NA	NA	NA	NA	2,321	0.02	0.02	0.03
Plaice	0.7	0.5	1.1	690,912	0.00	0.00	0.00	80,367	0.00	0.00	0.00
Smelt	0.09	0.06	0.13	50	0.19	0.12	0.26	8	1.11	0.74	1.54
Cod	5.5	2.1	9.8	103,025	0.01	0.00	0.01	34,701	0.02	0.01	0.03
Thornback ray	1.6	1.0	2.4	NA	NA	NA	NA	695	0.24	0.15	0.35
Eel	0.4	0.2	0.5	79	0.45	0.29	0.66	14	2.56	1.65	3.72
Twaite shad	0.3	0.2	0.6	5,161,183	0.02	0.01	0.04	1	25.07	12.33	43.72
River lamprey	0.1	0.0	0.1	62	0.13	0.07	0.23	1	7.42	4.09	12.52
Horse mackerel	0.1	0.0	0.2	NA	NA	NA	NA	20,798	0.00	0.00	0.00
Mackerel	0.1	0.0	0.1	3,888,854	0.00	0.00	0.00	1,026,828	0.00	0.00	0.00
Tope	0.2	0.0	0.7	NA	NA	NA	NA	498	0.03	0.00	0.13
Sea trout	0.0	0.0	0.0	NA	NA	NA	NA	39,795	0.01	0.00	0.05
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Allis shad	0.0	0.0	0.0	27,397	0.00	0.00	0.03	0.2	0.47	0.00	2.76
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	0.00	0.00	0.00

C.3 Predicted SZC impingement with FRR systems fitted

	Annually raised SZB estimate			Annually raised SZC estimate			FRR mortality			SZC EAV equivalent numbers		
Species	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,703,033	1,427,342	4,549,040	6,920,815	3,654,550	11,647,309	6,920,815	3,654,550	11,647,309	5,199,288	2,745,494	8,750,084
Herring	1,018,211	598,405	1,550,795	2,607,016	1,532,149	3,970,636	2,607,016	1,532,149	3,970,636	1,864,587	1,095,822	2,839,874
Whiting	650,479	452,300	888,210	1,665,479	1,158,064	2,274,163	916,779	637,468	1,251,836	326,445	226,988	445,751
Bass	276,943	124,683	500,639	709,082	319,238	1,281,830	390,396	175,762	705,732	87,434	39,364	158,057
Sand goby	196,545	81,220	383,493	503,232	207,954	981,891	103,666	42,838	202,270	103,666	42,838	202,270
Sole	91,370	63,510	125,792	233,942	162,609	322,077	48,192	33,498	66,348	10,259	7,131	14,124
Anchovy	69,058	18,123	166,131	176,816	46,401	425,360	176,816	46,401	425,360	172,238	45,200	414,346
Dab	56,054	33,869	93,069	143,519	86,719	238,293	77,287	46,699	128,324	34,362	20,763	57,054
Thin-lipped grey mullet	49,751	13,394	97,783	127,382	34,295	250,361	70,135	18,882	137,846	5,846	1,574	11,490
Flounder	14,569	10,943	19,251	37,303	28,018	49,291	8,613	6,470	11,382	3,978	2,988	5,256
Plaice	9,653	6,328	14,355	24,716	16,202	36,755	5,091	3,338	7,571	1,758	1,153	2,615
Smelt	8,122	5,405	11,295	20,795	13,840	28,921	20,795	13,840	28,921	15,834	10,538	22,021
Cod	6,415	2,440	11,433	16,426	6,248	29,272	9,890	3,762	17,625	3,552	1,351	6,329
Thornback ray	2,913	1,808	4,292	7,460	4,629	10,989	1,537	953	2,264	296	184	436
Eel	1,184	762	1,719	3,031	1,951	4,402	624	402	907	624	402	907
Twaite shad	1,168	574	2,036	2,989	1,470	5,213	2,989	1,470	5,213	2,989	1,470	5,213
River lamprey	1,144	630	1,932	2,929	1,614	4,946	603	332	1,019	603	332	1,019
Horse mackerel	695	220	1,715	1,779	564	4,391	1,779	564	4,391	1,779	564	4,391
Mackerel	185	17	502	473	43	1,287	473	43	1,287	473	43	1,287
Tope	27	0	106	70	0	271	14	0	56	14	0	56
Sea trout	4	0	23	10	0	58	10	0	58	10	0	58
Sea lamprey	2	0	11	5	0	29	1	0	6	1	0	6
Allis shad	1	0	8	3	0	21	3	0	21	3	0	21
Salmon	0	0	0	0	0	0	0	0	0	0	0	0

	EAV equivalent weight (t)			mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
Species	Mean	Lower	Upper		Mean	Lower	Upper		Mean	Lower	Upper
Sprat	54.6	28.8	91.9	220,757	0.02	0.01	0.04	151,322	0.04	0.02	0.06
Herring	351.8	206.7	535.8	2,198,449	0.02	0.01	0.02	400,244	0.09	0.05	0.13
Whiting	93.3	64.9	127.4	151,881	0.06	0.04	0.08	17,570	0.53	0.37	0.73
Bass	133.8	60.3	242.0	14,897	0.90	0.40	1.62	3,051	4.39	1.97	7.93
Sand goby	0.2	0.1	0.4	205,882,353	0.05	0.02	0.10	NA	NA	NA	NA
Sole	2.2	1.5	3.0	43,770	0.01	0.00	0.01	12,800	0.02	0.01	0.02
Anchovy	3.6	0.9	8.6	NA	NA	NA	NA	1,625	0.22	0.06	0.53
Dab	1.4	0.8	2.3	NA	NA	NA	NA	5,309	0.03	0.02	0.04
Thin-lipped grey mullet	3.0	0.8	6.0	NA	NA	NA	NA	120	2.53	0.68	4.98
Flounder	0.3	0.2	0.4	NA	NA	NA	NA	2,321	0.01	0.01	0.02
Plaice	0.4	0.3	0.6	690,912	0.00	0.00	0.00	80,367	0.00	0.00	0.00
Smelt	0.26	0.17	0.36	50	0.52	0.35	0.73	8	3.11	2.07	4.32
Cod	9.2	3.5	16.5	103,025	0.01	0.00	0.02	34,701	0.03	0.01	0.05
Thornback ray	0.9	0.6	1.4	NA	NA	NA	NA	695	0.14	0.08	0.20
Eel	0.2	0.1	0.3	79	0.26	0.17	0.38	14	1.48	0.95	2.15
Twaite shad	0.9	0.5	1.6	5,161,183	0.06	0.03	0.10	1	70.22	34.53	122.46
River lamprey	0.0	0.0	0.1	62	0.08	0.04	0.13	1	4.28	2.36	7.23
Horse mackerel	0.2	0.1	0.6	NA	NA	NA	NA	20,798	0.00	0.00	0.00
Mackerel	0.2	0.0	0.4	3,888,854	0.00	0.00	0.00	1,026,828	0.00	0.00	0.00
Tope	0.1	0.0	0.4	NA	NA	NA	NA	498	0.02	0.00	0.08
Sea trout	0.0	0.0	0.1	NA	NA	NA	NA	39,795	0.02	0.00	0.15
Sea lamprey	0.0	0.0	0.0	NA	NA	NA	NA	NA	NA	NA	NA
Allis shad	0.0	0.0	0.0	27,397	0.01	0.00	0.08	0.2	1.31	0.00	7.72
Salmon	0.0	0.0	0.0	NA	NA	NA	NA	38,456	0.00	0.00	0.00

C.4 Predicted SZC impingement with the effect of LVSE intake heads and FRR systems fitted

	Annually raised SZB estimate			Annually raised SZC estimate			SZC Intake head mortality		
Species	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,703,033	1,427,342	4,549,040	6,920,815	3,654,550	11,647,309	2,470,731	1,304,674	4,158,089
Herring	1,018,211	598,405	1,550,795	2,607,016	1,532,149	3,970,636	930,705	546,977	1,417,517
Whiting	650,479	452,300	888,210	1,665,479	1,158,064	2,274,163	594,576	413,429	811,876
Bass	276,943	124,683	500,639	709,082	319,238	1,281,830	253,142	113,968	457,613
Sand goby	196,545	81,220	383,493	503,232	207,954	981,891	179,654	74,239	350,535
Sole	91,370	63,510	125,792	233,942	162,609	322,077	83,517	58,052	114,981
Anchovy	69,058	18,123	166,131	176,816	46,401	425,360	63,123	16,565	151,853
Dab	56,054	33,869	93,069	143,519	86,719	238,293	51,236	30,959	85,070
Thin-lipped grey mullet	49,751	13,394	97,783	127,382	34,295	250,361	45,475	12,243	89,379
Flounder	14,569	10,943	19,251	37,303	28,018	49,291	13,317	10,002	17,597
Plaice	9,653	6,328	14,355	24,716	16,202	36,755	8,824	5,784	13,121
Smelt	8,122	5,405	11,295	20,795	13,840	28,921	7,424	4,941	10,325
Cod	6,415	2,440	11,433	16,426	6,248	29,272	5,864	2,230	10,450
Thornback ray	2,913	1,808	4,292	7,460	4,629	10,989	2,663	1,652	3,923
Eel	1,184	762	1,719	3,031	1,951	4,402	1,082	696	1,572
Twaite shad	1,168	574	2,036	2,989	1,470	5,213	1,067	525	1,861
River lamprey	1,144	630	1,932	2,929	1,614	4,946	1,046	576	1,766
Horse mackerel	695	220	1,715	1,779	564	4,391	635	201	1,568
Mackerel	185	17	502	473	43	1,287	169	15	459
Tope	27	0	106	70	0	271	25	0	97
Sea trout	4	0	23	10	0	58	4	0	21
Sea lamprey	2	0	11	5	0	29	2	0	10
Allis shad	1	0	8	3	0	21	1	0	7
Salmon	0	0	0	0	0	0	0	0	0

Species	Trash rack mortality			Drum screen mortality			Band screen mortality		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	0	0	0	2,248,365	1,187,254	3,783,861	222,366	117,421	374,228
Herring	665,637	391,196	1,013,803	241,212	141,761	367,380	23,856	14,020	36,334
Whiting	0	0	0	273,778	190,367	373,836	53,512	37,209	73,069
Bass	60	27	108	116,534	52,465	210,663	22,777	10,255	41,175
Sand goby	0	0	0	33,678	13,917	65,711	3,331	1,376	6,499
Sole	0	0	0	15,656	10,882	21,554	1,548	1,076	2,132
Anchovy	0	0	0	57,442	15,074	138,187	5,681	1,491	13,667
Dab	21,457	12,965	35,626	5,582	3,373	9,269	552	334	917
Thin-lipped grey mullet	13	3	25	20,934	5,636	41,144	4,092	1,102	8,042
Flounder	418	314	552	2,418	1,816	3,195	239	180	316
Plaice	0	0	0	1,654	1,084	2,460	164	107	243
Smelt	0	0	0	6,756	4,496	9,395	668	445	929
Cod	37	14	65	2,970	1,130	5,292	524	199	935
Thornback ray	0	0	0	499	310	735	49	31	73
Eel	0	0	0	203	131	295	20	13	29
Twaite shad	942	463	1,643	114	56	198	11	6	20
River lamprey	0	0	0	196	108	331	19	11	33
Horse mackerel	0	0	0	578	183	1,426	57	18	141
Mackerel	0	0	0	154	14	418	15	1	41
Tope	0	0	0	5	0	18	0	0	2
Sea trout	4	0	21	0	0	0	0	0	0
Sea lamprey	0	0	0	0	0	2	0	0	0
Allis shad	1	0	7	0	0	0	0	0	0
Salmon	0	0	0	0	0	0	0	0	0

Species	Total FRR mortality			EAV equivalent numbers			EAV equivalent weight (t)		
	Mean	Lower	Upper	Mean	Lower	Upper	Mean	Lower	Upper
Sprat	2,470,731	1,304,674	4,158,089	1,856,146	980,141	3,123,780	19.50	10.30	32.81
Herring	930,705	546,977	1,417,517	665,658	391,208	1,013,835	125.59	73.81	191.28
Whiting	327,290	227,576	446,905	116,541	81,035	159,133	33.31	23.16	45.48
Bass	139,372	62,747	251,946	31,214	14,053	56,426	47.78	21.51	86.38
Sand goby	37,009	15,293	72,210	37,009	15,293	72,210	0.07	0.03	0.14
Sole	17,205	11,959	23,686	3,663	2,546	5,042	0.78	0.55	1.08
Anchovy	63,123	16,565	151,853	61,489	16,136	147,922	1.28	0.33	3.07
Dab	27,591	16,672	45,812	12,267	7,412	20,368	0.50	0.30	0.83
Thin-lipped grey mullet	25,038	6,741	49,211	2,087	562	4,102	1.09	0.29	2.13
Flounder	3,075	2,310	4,063	1,420	1,067	1,876	0.12	0.09	0.15
Plaice	1,818	1,192	2,703	628	412	934	0.15	0.10	0.23
Smelt	7,424	4,941	10,325	5,653	3,762	7,861	0.09	0.06	0.13
Cod	3,531	1,343	6,292	1,268	482	2,260	3.30	1.25	5.88
Thornback ray	549	340	808	106	66	156	0.34	0.21	0.50
Eel	223	143	324	223	143	324	0.07	0.05	0.11
Twaite shad	1,067	525	1,861	1,067	525	1,861	0.33	0.16	0.58
River lamprey	215	119	364	215	119	364	0.02	0.01	0.03
Horse mackerel	635	201	1,568	635	201	1,568	0.09	0.03	0.22
Mackerel	169	15	459	169	15	459	0.05	0.00	0.15
Tope	5	0	20	5	0	20	0.04	0.00	0.14
Sea trout	4	0	21	4	0	21	0.01	0.00	0.04
Sea lamprey	0.36	0	2	0.36	0	2	0.00	0.00	0.00
Allis shad	1	0	7	1	0	7	0.00	0.00	0.00
Salmon	0	0	0	0	0	0	0.00	0.00	0.00

Note: Total FRR mortality is the sum of the trash rack, drum screen and band screen mortalities, applied to the numbers impinged after the effect of the intake head has been considered.

Species	mean SSB (t)	EAV weight as % of SSB			Mean landings (t)	EAV weight as % of landings		
		Mean	Lower	Upper		Mean	Lower	Upper
Sprat	220,757	0.01	0.00	0.01	151,322	0.01	0.01	0.02
Herring	2,198,449	0.01	0.00	0.01	400,244	0.03	0.02	0.05
Whiting	151,881	0.02	0.02	0.03	17,570	0.19	0.13	0.26
Bass	14,897	0.32	0.14	0.58	3,051	1.57	0.71	2.83
Sand goby	205,882,353	0.02	0.01	0.04	NA	NA	NA	NA
Sole	43,770	0.00	0.00	0.00	12,800	0.01	0.00	0.01
Anchovy	NA	NA	NA	NA	1,625	0.08	0.02	0.19
Dab	NA	NA	NA	NA	5,309	0.01	0.01	0.02
Thin-lipped grey mullet	NA	NA	NA	NA	120	0.90	0.24	1.78
Flounder	NA	NA	NA	NA	2,321	0.01	0.00	0.01
Plaice	690,912	0.00	0.00	0.00	80,367	0.00	0.00	0.00
Smelt	50	0.19	0.12	0.26	8	1.11	0.74	1.54
Cod	103,025	0.00	0.00	0.01	34,701	0.01	0.00	0.02
Thornback ray	NA	NA	NA	NA	695	0.05	0.03	0.07
Eel	79	0.09	0.06	0.14	14	0.53	0.34	0.77
Twaite shad	5,161,183	0.02	0.01	0.04	1	25.07	12.33	43.72
River lamprey	62	0.03	0.02	0.05	1	1.53	0.84	2.58
Horse mackerel	NA	NA	NA	NA	20,798	0.00	0.00	0.00
Mackerel	3,888,854	0.00	0.00	0.00	1,026,828	0.00	0.00	0.00
Tope	NA	NA	NA	NA	498	0.01	0.00	0.03
Sea trout	NA	NA	NA	NA	39,795	0.01	0.00	0.05
Sea lamprey	NA	NA	NA	NA	NA	NA	NA	NA
Allis shad	27,397	0.00	0.00	0.03	0	0.47	0.00	2.76
Salmon	NA	NA	NA	NA	38,456	0.00	0.00	0.00

Appendix D Mean numbers of fish estimated impinged annually at SZB

Annually raised mean SZB estimates of impingement for 24 key species, with no mitigation and with the embedded FRR mitigation. Total losses have been converted to adult equivalent (EAV) numbers and weights (t) and calculated as a percentage of either the mean stock SSB (t) or mean international landings (t). Species where the impingement weight exceed 1 % of the relevant stock comparator are shaded in red. Note: numbers in red font are either estimates of the population numbers (e.g. sand goby) or reported catch numbers (salmon and sea trout).

D.1 Unmitigated impingement effects

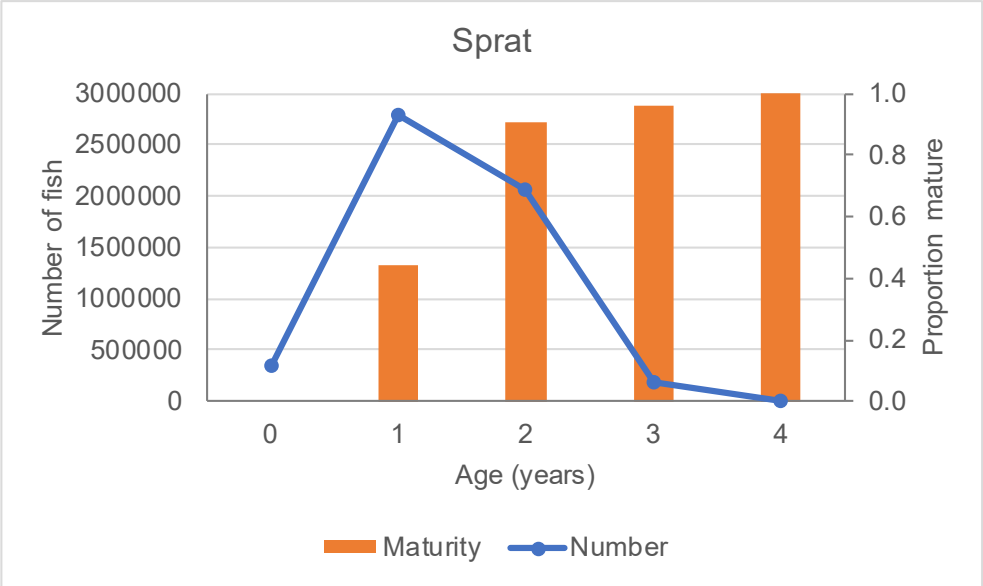
Species	Mean SZB estimate	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	2,703,033	2,030,664	21.33	220,757	0.01	151,322	0.01
Herring	1,018,211	728,244	137.39	2,198,449	0.01	400,244	0.03
Whiting	650,479	231,621	66.20	151,881	0.04	17,570	0.38
Bass	276,943	62,025	94.95	14,897	0.64	3,051	3.11
Sand goby	196,545	196,545	0.37	205,882,353	0.10	NA	NA
Sole	91,370	19,451	4.16	43,770	0.01	12,800	0.03
Anchovy	69,058	67,270	1.40	NA	NA	1,625	0.09
Dab	56,054	24,922	1.02	NA	NA	5,309	0.02
Thin-lipped grey mullet	49,751	4,147	2.16	NA	NA	120	1.80
Flounder	14,569	6,728	0.55	NA	NA	2,321	0.02
Plaice	9,653	3,334	0.82	690,912	0.00	80,367	0.00
Smelt	8,122	6,184	0.10	50	0.20	8	1.21
Cod	6,415	2,304	5.99	103,025	0.01	34,701	0.02
Thornback ray	2,913	562	1.79	NA	NA	695	0.26
Eel	1,184	1,184	0.39	79	0.50	14	2.80
Twaite shad	1,168	1,168	0.37	5,161,183	0.02	1	27.42
River lamprey	1,144	1,144	0.09	62	0.15	1	8.11
Horse mackerel	695	695	0.10	NA	NA	20,798	0.00
Mackerel	185	185	0.06	3,888,854	0.00	1,026,828	0.00
Tope	27	27	0.19	NA	NA	498	0.04
Sea trout	4	4	0.01	NA	NA	39,795	0.01
Sea lamprey	2	2	0.00	NA	NA	NA	NA
Allis shad	1	1	0.00	27,397	0.00	0	0.51
Salmon	0	0	0.00	NA	NA	38,456	0.00

D.2 Impingement effects with embedded FRR mitigation

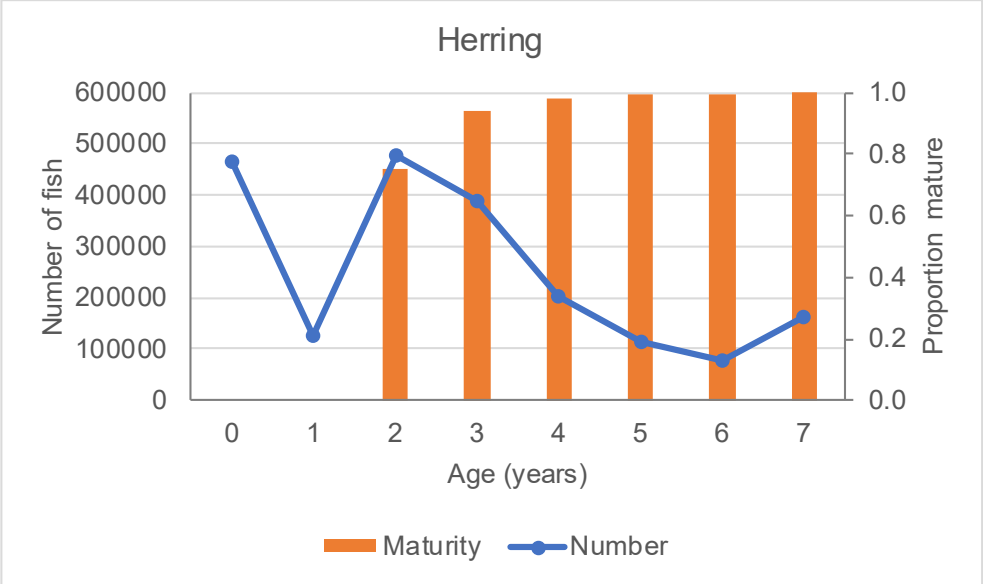
Species	Mean SZB estimate	FRR mortality	EAV number	EAV weight (t)	Mean SSB	% of SSB	Mean landings (t)	% of landings
Sprat	2,703,033	2,703,033	2,030,664	21.33	220,757	0.01	151,322	0.01
Herring	1,018,211	1,018,211	728,244	137.39	2,198,449	0.01	400,244	0.03
Whiting	650,479	325,239	115,811	33.10	151,881	0.02	17,570	0.19
Bass	276,943	138,472	31,012	47.47	14,897	0.32	3,051	1.56
Sand goby	196,545	39,309	39,309	0.07	205,882,353	0.02	NA	NA
Sole	91,370	18,274	3,890	0.83	43,770	0.00	12,800	0.01
Anchovy	69,058	69,058	67,270	1.40	NA	NA	1,625	0.09
Dab	56,054	11,211	4,984	0.20	NA	NA	5,309	0.00
Thin-lipped grey mullet	49,751	24,876	2,074	1.08	NA	NA	120	0.90
Flounder	14,569	2,914	1,346	0.11	NA	NA	2,321	0.00
Plaice	9,653	1,931	667	0.16	690,912	0.00	80,367	0.00
Smelt	8,122	8,122	6,184	0.10	50	0.20	8	1.21
Cod	6,415	3,208	1,152	3.00	103,025	0.00	34,701	0.01
Thornback ray	2,913	583	112	0.36	NA	NA	695	0.05
Eel	1,184	237	237	0.08	79	0.10	14	0.56
Twaite shad	1,168	1,168	1,168	0.37	5,161,183	0.02	1	27.42
River lamprey	1,144	229	229	0.02	62	0.03	1	1.62
Horse mackerel	695	695	695	0.10	NA	NA	20,798	0.00
Mackerel	185	185	185	0.06	3,888,854	0.00	1,026,828	0.00
Tope	27	5	5	0.04	NA	NA	498	0.01
Sea trout	4	2	2	0.00	NA	NA	39,795	0.00
Sea lamprey	2	0.38	0.38	0.00	NA	NA	NA	NA
Allis shad	1	1	1	0.00	27,397	0.00	0	0.51
Salmon	0	0	0	0.00	NA	NA	38,456	0.00

Appendix E Number at age and proportion maturity of commonly impinged fish species

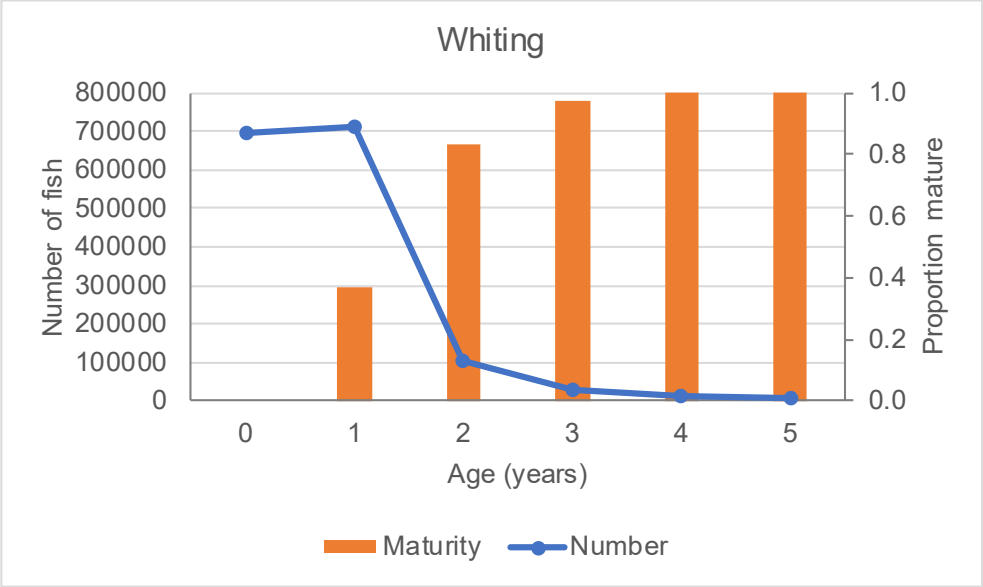
E.1 Sprat



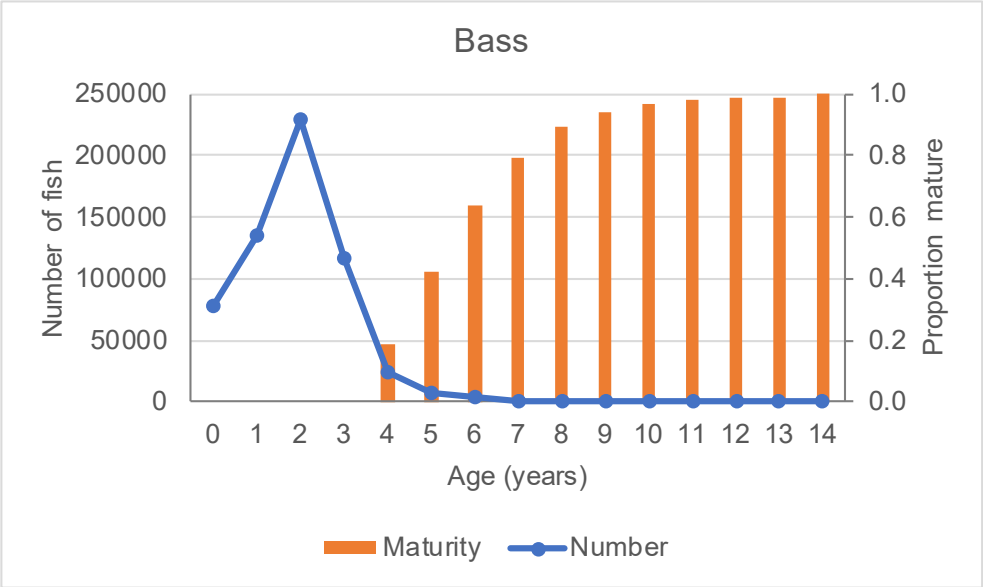
E.2 Herring



E.3 Whiting



E.4 Bass



E.5 Sole

Sole
140000
120000
100000
80000
60000
40000
20000
0
Number of fish
1.0
0.8
0.6
0.4
0.2
0.0
Proportion mature
0
1
2
3
Age (years)
Maturity
Number

Appendix F Full version history

F.1 BEEMS Technical Report TR606: Full version history

A detailed version History have been provided in Appendix F.

V1 dated 03/06/2019

Version 1 of this report was released (03/06/2019). The report superseded the preliminary impingement assessment reports produced to allow consultation and iterative development of the methodology.

This report is the primary report for impingement assessments.

V2 report dated 9/12/2019

Since Version 1 of this report was released (03/06/2019), new details and clarifications on the proposed station design have emerged. Additional work has been undertaken to address issues raised by stakeholders. The major changes/impacts on the assessments that are included in Version 2 are:

2. After a detailed modelling programme EDF Energy have decided to fit low velocity side entry (LVSE) intake heads at SZC. These intakes are designed to substantially reduce impingement impacts. Calculations have been undertaken to assess the effects of the proposed head design on SZC impingement and in conjunction with the proposed FRR mitigation.

3. The fine filtration mesh size is proposed to be 10 mm and the associated trash rack bar spacing proposed 75mm. No adjustments are therefore required for the mesh size as this is consistent with that of the current SZB station. A proposed wider trash rack spacing required an update to the proportion of each species that will either pass through or be retained on the trash racks. Trash rack mortality calculations have been updated accordingly.

4. Improved estimates of the populations of the conservation species; twaite shad, cucumber smelt and river lamprey have been included. This has substantially improved the confidence in impact assessment for these species. All tables have been updated accordingly.

5. Following stakeholder comments, estimates of the sand goby population have been updated. A calculation error was noted in the adjustment for beam trawl sampling efficiency. The population estimate has been re-scaled to the expected abundance if the trawl was fishing at 100 % efficiency.

6. Impingement estimates for key crustacea (brown crab, brown shrimp, lobster and whelks) has been included.

7. All Appendices have been updated to the latest values. Following stakeholder requests, the final Appendix table for SZC (including all mitigation), now includes all calculation steps, including trash rack mortality.

8. An overall entrapment assessment has been included that presents the combined effects of impingement plus entrainment at SZC (Section 6.7).

V3 report dated 17/01/2020

1. Following the release of Version 2 of this report, an error was identified with some of the stock data in that catch data (landings plus discards) were used, rather than landings only. This error affected four species only (whiting, cod, horse mackerel and dab). Assessments for the first 3 species were unaffected by the error as the primary stock comparator for these species is SSB and not landings. For dab, the result of replacing catches with landings was negligible – in the absence of mitigation, losses of dab changed from 0.01 % of catches to 0.04 % of landings. All calculations and tables for the four species have been updated.

2. Sections 4.7 on factors that could influence FRR mortality and 4.8 on EAV calculations have been expanded to provide clarifications in response to stakeholder comments. In particular, this report now provides:

    - An assessment of the potential for clogging of the FRR system by dead fish and ctenophores
    - A critique of potential alternative methods of calculating EAVs

These were editorial changes only and they resulted in unchanged assessment results.

3. Following discussions with stakeholders at MTF meetings, further clarifications and explanations have been provided on the assessment of the predicted effects of SZC on:

    i. the endangered European eel stock in Sections 5.1.6 and 6.6.1

    ii. the North Sea herring and the Blackwater herring in Section 6.6.5.

    iii. the local fish assemblage in Section 8.

4. A new Section 6.8 has been added to contextualise predicted SZC entrapment losses. The data included are an expanded version of those provided to stakeholders at the Sizewell MTF meeting on 18 December 2019.

V4 report dated 28/01/2020

1. Clarification of the development site red line boundary information added after the Table of Figures

V5 report dated 19/02/2020

In response to stakeholder comments clarifications have been added about:

1. The measured differences in bass abundance at the SZB and SZC intake locations; in particular how the surveys were intended to quantify previously well-established scientific facts about bass thermal preference in winter. Section 4.1.1.1.

2. Details of impinged eel length frequency distributions from SZB impingement data added in Section 6.6.2.3.

3. In response to Environment Agency comments dated 12 February 2012 on the v2 report. In particular, the following sections have been added:

    - Section 2 to summarise the extensive range of studies and the decision-making process that took place on the selection of impingement mitigation options to be fitted at SZC.

    - Section 4.10 which provides a justification of the use of SSBs derived from ICES's stock units to assess the effects of SZC on fish populations. A specific SZC question on the validity of the bass stock unit size is addressed in Section 4.10.2

    - Section 9 - Effect of SZC entrapment on the Water Framework Directive (WFD) status of local transitional water bodies. This assessment included a specific consideration of the likelihood of any significant effects of SZC entrapment on smelt in the Alde Ore (and potentially the Blyth) at the request of the Environment Agency.

V6 report dated 27/02/2020 – Version submitted for DCO

1. Minor editorial correction to Section 5.1.1.1

    Table 9 updated with Sizewell C data to replace previous reference to Hinkley Point data.

V7 report dated 06/11/2020

Version 6 of this report of 27/02/2020 was part of the DCO submission. Version 7 of this report has been modified following stakeholder consultation, Marine Technical Forum (MTF) meetings and initial responses to Relevant Representations provided on the 30th September 2020.

The Environment Agency produced a differences document 'EA Sizewell C - Marine Ecology Gap Analysis' based on version 6 of the report in February 2020 which preceded a Level 4 Marine Ecology meeting on 19th March 2020. On 23rd September 2020 a MTF was hosted including the Marine Management Organisation, Environment Agency, Natural England and Eastern IFCA in relation to key issues pertaining to fish impingement assessments provided in the DCO.

Version 7 of this report captures the main discussion and action points from the meetings and written comments from statutory stakeholders. Furthermore is provides additional evidence based on a series of supplementary reports to support the evidence base for impingement assessments.

The most totable changes to this report include:

5. **An updated LVSE factor.** Following Environment Agency consultation on BEEMS Scientific Position Paper SPP099, modelling was refined to capture the full range of tidal directions. Modelling indicates that the SZC station will abstract 0.357 per cumec of the fish abstracted by SZB (rather than 0.383 reported in version 6), because of the intake head design. The methodology used to derive this impingement reduction factor is described in BEEMS Scientific Position Paper SPP099 v4 and is based upon the methodology proposed by the Environment Agency for impingement studies at Hinkley Point C. Further information is provided in Section 4.6.

6. **Updated Sizewell B impingement predictions:** Following statistical review, a bootstrapping approach, akin to the methods employed at Hinkley Point, with an additional step to account for periods when sampling was not possible due to station outages, has been applied. The bootstrapping method randomly reallocates samples from adjacent years to account for sampling gaps. The bootstrap outputs for each species provides the mean numbers impinged by quarter for Sizewell B, which is converted to percentage impingement by quarter to show the seasonal patterns. The bootstrapping approach has been applied to calculate annual impingement numbers and 95th percentile confidence intervals for Sizewell B these numbers form the basis for scaled Sizewell C predictions. Further information is provided in Section 4.4.

   The development of the model and bootstrap approaches, the advantages, and limitations along with differences in impingement predictions between the approaches for Sizewell B is provided in BEEMS Technical Report TR339 v3.

7. **Evaluating uncertainty in impingement parameters:** The confidence of impingement predictions is determined by the underlying assumptions used to parameterise the assessment and the appropriateness of the comparator. **:** Section 7 provides an assessment of the sensitivity of impingement predictions and effects to uncertainties in the defining parameters including LVSE efficiency, FRR mortality, EAV values and changes in distribution of fish.

8. **Local effects assessment:** To address specific concerns relating to the potential for local effects Section 9 provides an additional local-level assessment.

Version 7 of this report has been updated further following ongoing evidence gathering, technical review and specific stakeholder concerns. A series of new Scientific Position Papers provides the evidence base that underpins edits to this report, these include:

- SPP101 - Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell.

- SPP102 - Use of Spawning Production Foregone EAVs for impingement assessment. Section 4.8.1.

- SPP103 - Consideration of potential effects on selected fish stocks at Sizewell. Section 4.10.2 and Section 8.

- SPP104 - Worst case glass eel entrainment assessment for Sizewell C. Section 6.6.2.3.

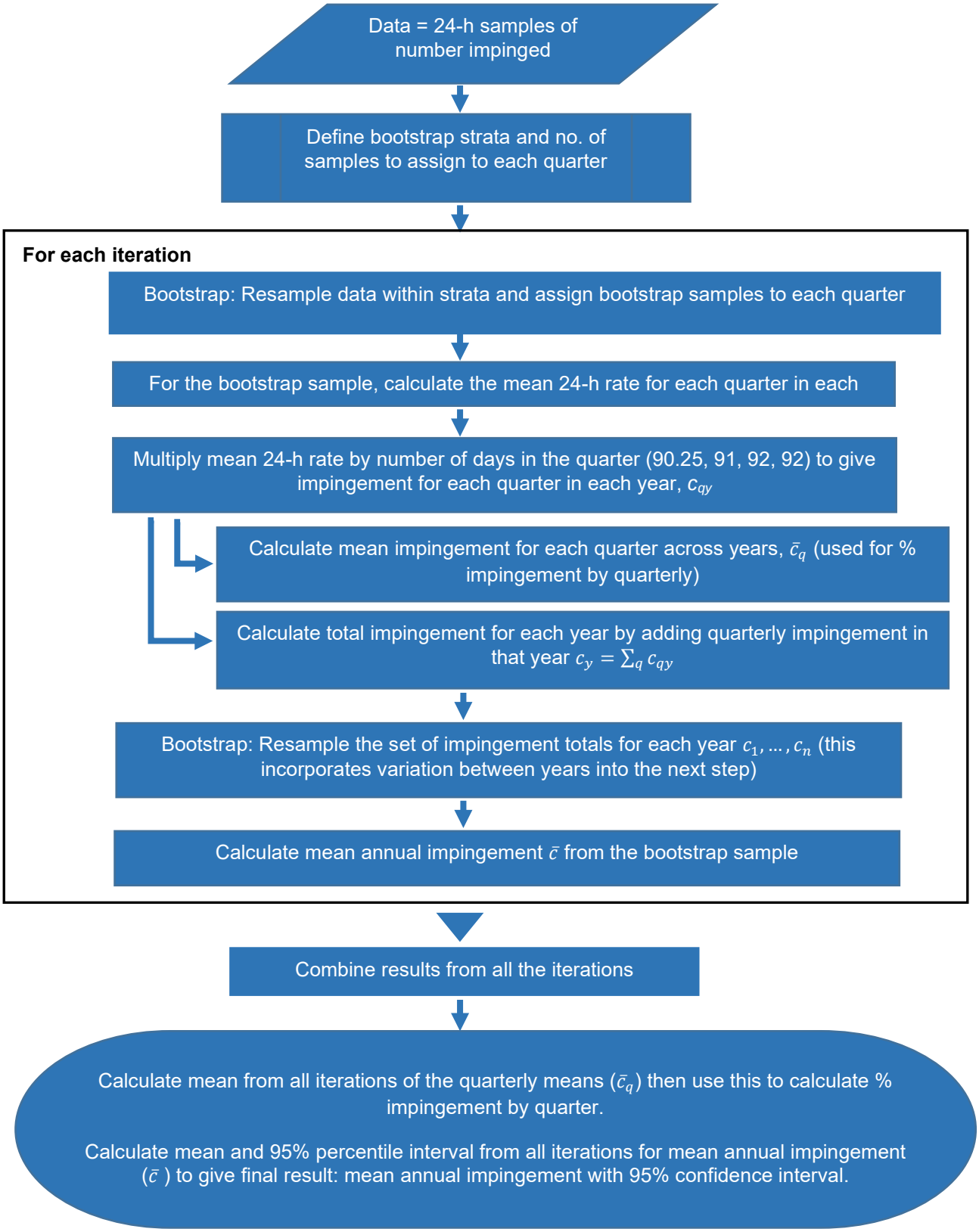
- SPP108 - Sensitivity of the Alde Ore Transitional Fish Classification Index (TFCI) to changes in smelt, *Osmerus eperlanus*, abundance. Section 9.

Whilst the edits to the report have in some cases resulted in changes to absolute impingement predictions, in terms of numbers and biomass, the scale of effects have not altered, and the conclusions remain unchanged.

Appendix G Sizewell B impingement prediction methodology

G.1 Sizewell B impingement prediction methodology (BEEMS TR339.v3)

Figure 14 The bootstrap process applied to the Sizewell B Comprehensive Impingement Monitoring Programme data to calculate mean and 95th percentile confidence intervals.


Data = 24-h samples of number impinged
Define bootstrap strata and no. of samples to assign to each quarter
For each iteration
Bootstrap: Resample data within strata and assign bootstrap samples to each quarter
For the bootstrap sample, calculate the mean 24-h rate for each quarter in each
Multiply mean 24-h rate by number of days in the quarter (90.25, 91, 92, 92) to give impingement for each quarter in each year, c_{qy}
Calculate mean impingement for each quarter across years, $\bar{c}_q$ (used for % impingement by quarterly)
Calculate total impingement for each year by adding quarterly impingement in that year $c_y = \sum_q c_{qy}$
Bootstrap: Resample the set of impingement totals for each year $c_1, \ldots, c_n$ (this incorporates variation between years into the next step)
Calculate mean annual impingement $\bar{c}$ from the bootstrap sample
Combine results from all the iterations
Calculate mean from all iterations of the quarterly means ($\bar{c}_q$) then use this to calculate % impingement by quarter.
Calculate mean and 95% percentile interval from all iterations for mean annual impingement ($\bar{c}$) to give final result: mean annual impingement with 95% confidence interval.

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. TR520

NOT PROTECTIVELY MARKED

TR520: Sizewell C Influence of the fish recovery and return system on water quality and ecological receptors

UNCONTROLLED WHEN PRINTED
NOT PROTECTIVELY MARKED

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Please note that the red line boundary used in the figures within this document was amended after this document was finalised, and therefore does not reflect the boundaries in respect of which development consent has been sought in this application. However, the amendment to the red line boundary does not have any impact on the findings set out in this document and all other information remains correct.

Executive summary

Introduction

As a result of the direct cooling of the Sizewell C (SZC) power station with seawater, fish will be abstracted into the cooling water (CW) infrastructure. To avoid fish, invertebrates and other debris passing into the station condensers, cooling water from the intakes passes through rotating drum or vertical band screens. Impinged fish and invertebrates are washed from the screens and are returned to sea via the Fish Recovery and Return (FRR) outfall. Survival rates for delicate pelagic species such as herring, sprat and shad are usually <10% (Turnpenny and O'Keeffe, 2005). However, for demersal species such as the robust gadoids (e.g. cod) survival rates may be 50 - 60% (BEEMS TR406) and for more robust species such as eel *Anguilla anguilla*, plaice *Pleuronectes platessa* and flounder *Platichthys flesus* survival rates may be 80 - 100%.

BEEMS Technical Report TR406 has considered the effects of fish and crustacea loading on the survivability of fish within the SZC FRR system and BEEMS Technical Report TR511 has modelled the locations where dead fish discharged from the FRR system are likely to wash up on local beaches and how long they will remain before being consumed by gulls. This report (BEEMS Technical Report TR520) considers the potential effects on local water quality from decomposing dead fish discharged from the SZC FRR system. Sizewell B (SZB) also discharges impinged fish back to sea but the influence of these inputs contributes to baseline monitoring data that forms part of the assessment.

The biomass of moribund organisms discharged from the SZC FRR system was calculated from the results of the SZC impingement assessment presented in BEEMS Technical Report TR406 v7. That assessment is based upon the estimated performance of the two impingement mitigation systems that are planned for SZC:

a. Low Velocity Side Entry (LVSE) intakes
b. The FRR System

In this report the estimated reductions in impingement mortality from these two systems are as detailed in BEEMS Technical Report TR406. The report assumes that an Acoustic Fish Deterrent (AFD) is not fitted to SZC.

The purpose of this assessment is to determine whether there would be any deterioration in the water body status under the Water Framework Directive (WFD) from the FRR discharges. The assessment considers effects on the WFD water bodies and associated Marine Strategy Framework Directive (MSFD) sea area within the local area of the SZC FRR outfall:

a. Suffolk (coastal water body, C21): the SZC FRRs would discharge into this water body. The SZB intake and outfall also abstract from and discharge to this water body.

b. Adjacent WFD water bodies

c. MSFD sea area: The Greater North Sea. This assessment considers the potential effects of the SZC FRR discharge on nutrient concentrations, biochemical oxygen demand, un-ionised ammonia, phytoplankton production and organic enrichment of the seabed sediments due to smothering and resulting potential habitat loss.

The assessment methodology considers whether there would be any deterioration in status of the Suffolk coastal water body; if none were identified then no deterioration could be concluded for adjoining water bodies both upstream and downstream of the discharges. If a potential deterioration were identified, the resulting effect on other WFD water bodies outside of those initially selected would be undertaken within the WFD 'Further Assessment' stage.

Assessment Results

Water Quality

Dissolved inorganic nitrogen and Dissolved inorganic phosphorus

The FRR discharge is predicted to release dissolved inorganic nitrogen (DIN) into the Suffolk coastal water body. Under the WFD standards, the Suffolk coastal water body has 'Good' chemical status but is moderate for DIN. Discharges from an operational SZC with an FRR fitted (Appendix A,4) would result in a very small elevation in DIN in the receiving water body representing <0.3% of the mass present in daily tidal exchange for Sizewell Bay (with ca. 26% of this from the FRR) for daily operational inputs in combination with those from the FRR. For dissolved inorganic phosphorus (DIP) combined inputs from operation and the FRR would result in a very small elevation in DIP in the receiving water body representing 0.11% of the mass present in the daily tidal exchange for Sizewell Bay (with around 73% of this from the FRR). This level of input for both DIN and DIP is negligible and would not change the current nutrient status of the water body or have a significant effect for wider sea areas.

Biochemical oxygen demand (BOD)

The decaying biomass of moribund fish from the FRR would have negligible impact on the oxygen levels for the Suffolk coastal water body. The predicted oxygen demand of the decaying biomass (based on a 95th percentile) was estimated for Quarter 1, the period of maximum predicted impinged biomass, with LVSE in place and takes account of the influence of a temperature elevation of 2°C in the vicinity of the FRR. Under this scenario the oxygen demand would be equivalent to an area of 0.08 km^2 (7.83 ha) that would move from good to moderate status for oxygen concentration. This is a worst-case assessment as it assumes no predation and no remobilisation of partially decayed fish. In practice both effects will take place reducing the predicted area of influence on dissolved oxygen levels.

Un-ionised ammonia

The un-ionised ammonia contribution from FRR biomass was assessed considering conditions that would maximise the derived concentration (i.e. 95th percentile biomass during Quarter 1 when the largest numbers of fish are impinged). Taking account of thermal influence and the most extreme background pH and salinity an area of 0.007 ha could be exposed to an un-ionised ammonia concentration at or above the EQS of 21 μg l^{-1}. This input would be expected to dissipate more widely and have minimal influence on the water column or sediments.

Organic enrichment of benthic habitat due to smothering

For organic carbon deposition a benchmark value is defined as 100g organic carbon/m^2/year. An equivalent daily value would be 0.3g organic carbon/m^2/day. During the winter the highest numbers of fish are discharged from the FRR and adopting the peak value of associated carbon for the equivalent moribund biomass of fish this would contribute to an organic carbon loading at the benchmark standard level over an area of 1.5 km^2 in the worst-case period for largest numbers of fish not surviving impingement (based on 95th percentile biomass in Quarter 1). Based on annual average biomass inputs the area at benchmark value would be equivalent to 0.44 km^2.

Potential effects of elevated nutrients on phytoplankton status

SZC will discharge small volumes of nutrients in the form of phosphate and nitrates, which will tend to increase local phytoplankton productivity. For much of the year light availability limits phytoplankton growth and the addition of relatively small quantities of nutrients has no effect. In the summer, nitrate is a limiting nutrient (when light is not limiting) and is consumed rapidly. However, the nutrient exchange with the wider marine environment is much greater than the maximum proposed discharges, during either operation or construction, such that the predicted increase in

phytoplankton growth due to the SZC discharges would be negligible (<0.3% increase in annual gross production of carbon, BEEMS TR385).

The potential for nutrient additions to enhance opportunistic macroalgae in the Blyth or the more distant Alde-Ore Estuary has been raised. The point source is located a considerable distance from the estuaries, over 10km from the mouth of the Blyth and 23km from the mouth of the Alde-Orr Estuary at Shingle Street. Furthermore, the inshore estuaries and tidal areas of Eastern England, monitored for opportunistic macroalgal as part of the WFD, do not appear to exhibit excessive growth (Painting et al., 2017). As such, no discernible effects on opportunistic algae because of nutrient discharges from the proposed development are anticipated.

The FRR discharge would therefore cause no deterioration in the water body status under the WFD for phytoplankton nor is it likely to influence macroalgal growth in adjacent transitional water bodies.

Test for inclusion of habitats in the WFD assessment

The tests for inclusion of habitats in a WFD assessment are if the footprint of the FRR discharge is any of the following:

Test		Predicted activity footprint	Result
i.	0.5 km² or larger	Organic enrichment peak period Q1 95th percentile with LVSE 1.5 km²	Exceeds test value
ii.	1% or more of the water body's area	Organic enrichment peak period Q1 95th percentile with LVSE 1.5 km² and 1.03% of the Suffolk Coastal water body	Exceeds test value
iii.	within 500 m of any higher sensitivity habitat	Organic enrichment peak period Q1 95th percentile with LVSE, the effect footprint would be at 583 metres from the nearest point of the coralline crag and associated polychaete (*Sabellaria spinulosa*) reef habitat	Is below test value
iv.	1% or more of any lower sensitivity habitat	Organic enrichment peak period Q1 95th percentile with LVSE, the effect footprint overlaps 10.98% of lower sensitivity habitat	Exceeds test value

Tests i., ii. and iv. are met.

Potential effects on WFD habitats

Higher sensitivity habitats

The footprint of the predicted organic carbon discharge of the FRR with LVSE in place is not within 500 m of higher sensitivity habitat polychaete reef and so there in no potential overlap of the *Sabellaria* sublittoral reef habitat area for Sizewell Bay. The predicted plume areas for reduced dissolved oxygen (to a level that would represent a change in status from good to moderate) and for un-ionised ammonia (to reach a concentration equal to the EQS of 21 $\mu g\ l^{-1}$) are very small and there is also no overlap with polychaete reef habitat. The area potentially affected by the discharge footprints is based on a most conservative assessment and considers the peak biomass (which occurs in Quarter 1 and applies 95th percentile value) discharge and assumes no predation of dead fish. *Sabellaria* reef habitat is also not considered a sensitive receptor for the additional organic carbon input as it is typically associated with dispersive conditions that would reduce settlement of

organic material and evidence suggests (Walker and Rees, 1980) an ability of *Sabellaria* to adapt to organically enriched conditions.

Lower sensitivity habitats

The FRR discharge would overlap with lower sensitivity infralittoral fine sand habitat. The potential impact footprint for the maximum winter discharge during Quarter 1 for elevated un-ionised ammonia (to reach a level equal to the EQS) is negligible at 0.004 ha increasing to 0.007 ha under thermal influence. For assessment of the influence of the maximum biomass discharge decay upon dissolved oxygen (DO), together with the influence of a temperature uplift from the thermal effluent, the area equivalent to a reduction from good to moderate DO status would extend over 7.83 ha of low sensitivity habitat. This would represent less than 0.6% of the low sensitivity Infralittoral fine sand habitat within the Suffolk coastal water body.

The organic carbon deposition with LVSE in place has a larger footprint of potential area affected and this could overlap with an area of ca., 11% Infralittoral fine sand habitat within the Suffolk coastal water body. The footprint of the area predicted to be affected by increases in organic matter concentration above benchmark levels would not overlap with polychaete reef habitat and is over 500 metres away at the nearest point.

Overall, discharges from the FRR are likely to have neutral to localised minor effects on core species of Infralittoral fine sand habitat in Sizewell Bay. No significant changes to the community structure or function of these habitats are expected.

Summary

The decay of fish that do not survive passage through the FRR and that are part of the FRR discharge has been assessed. The assessment is conservative accounting for the highest potential biomass discharge from the FRR and takes account of physicochemical extremes and influence of thermal elevation upon potential water quality effects that could be contributed by the moribund fish biomass discharged from the FRR. The zone of potential impact at the point of discharge of the FRR is predicted to affect a relatively small area.

The influence of the FRR discharge is not within 500 m of polychaete reef habitat, a WFD higher sensitivity habitat. There is no predicted overlap of this *Sabellaria* habitat with the predicted effect area for dissolved oxygen reduction and elevated un-ionised ammonia or for organic carbon enrichment above benchmark level.

However, the FRR discharge will overlap with low sensitivity infralittoral fine sand habitat. The overlap of the un-ionised ammonia discharge would be negligible. For reduced dissolved oxygen the influence of the discharge footprint may extend over 7.83 ha of low sensitivity habitat and this represents less than 0.6% of this habitat in the Suffolk coastal water body.

The effect of elevated organic carbon loading discharge on lower sensitivity habitats from the FRR whilst affecting ca., 11% of this habitat in the Suffolk water body is likely to have neutral to localised minor effects on core species of infralittoral fine sand habitat. No significant changes to the community structure or function of these habitats are expected.

This assessment is conservative and the predicted changes in water quality, to inorganic enrichment of benthic habitat and to phytoplankton production in Sizewell Bay and the Suffolk coastal water body due to the FRR discharges are predicted to be negligible and to cause no deterioration to status. No deterioration can therefore be concluded for adjoining water bodies both upstream and downstream of the discharge. The small areas of elevated BOD and ammonia and relevant loadings would not change the status of the Greater North Sea sub-region.

There is a potential for some fish to beach over a total extent of 18.0 km stretch of coast, with particles beaching 5.5 km south of the SZC FRR and 12.5 km north and that these would rapidly be consumed by seabirds within 1-2 hours of daybreak. TR511 further showed that of the 12% of the dead fish that did not sink immediately in the vicinity of the FRR outfall, a total of ca., 3.5% would be consumed by seabirds, with approximately 7.9% sinking over a wide area over a period of 24 hours.

To be precautionary, the assessments in this report assume that 100% of dead fish discharged from the FRR outfall will sink immediately and no reduction is made for the 12% that will be advected over a larger area. The effects due to decay of this larger discharge have been assessed in this report as negligible and as having no effect on the status of the Suffolk coastal water body or the associated Marine strategy area. There is, therefore, no need to consider the water quality and biological effects of the more diffuse spread of dead fish as these will also be negligible.

1 Background

As a result of the direct cooling of the Sizewell C (SZC) power station with seawater, fish will be abstracted into the cooling water (CW) infrastructure. To avoid fish, invertebrates and other debris passing into the station condensers, cooling water from the intakes passes through rotating drum or band screens. Impinged fish and invertebrates are washed from the screens and are returned to sea via the dedicated subtidal Fish Recovery and Return (FRR) outfall.

FRR systems have been reported to achieve 80−100% survival rates for robust epibenthic species such as plaice *Pleuronectes platessa* and flounder *Platichthys flesus*, and moderate rates (~50−60%) for demersal species such as the robust gadoids (e.g. cod). However, survival rates for delicate pelagic species such as herring, sprat, and shad (twaite shad *Allosa fallax* and Allis shad *A. alosa*) are usually low (<10%, Turnpenny and O'Keeffe, 2005). This report assesses the effects on water quality that result from the degradation of dead fish that do not survive passage via the SZC FRR system.

The environmental impact assessment for the SZC power station requires consideration of the effects of cooling water abstraction and discharge on the coastal ecosystem. Under the Water Framework Directive (WFD), it is necessary to assess the effects of SZC upon chemical and biological indicators of water quality with one of the biological indicators being the phytoplankton index.

SZC will abstract cooling water from the marine environment at a rate of approximately 132 $m^3 s^{-1}$. The cooling water will be heated by approximately 11.6°C and treated with a chlorine-based biocide to prevent biofouling. The cooling water will be returned to the marine environment at approximately the same location from which it was extracted.

1.1 Water Framework Water Bodies

The purpose of this assessment was to determine whether there would be any deterioration in the water body status under the WFD from the FRR discharges. The assessment began with the selection of the relevant WFD water body within the local area of the SZC FRR outfall.

The FRR discharges will be approximately 400 metres offshore into the Suffolk Coastal (coastal water body, C21).

The assessment methodology considered whether there was any deterioration in status in this water body; if none were identified then no deterioration could be concluded for adjoining water bodies both upstream and downstream of the discharges. If a potential deterioration were identified, the resulting effect on other WFD water bodies outside of the Suffolk coastal water body would be undertaken within the WFD 'Further Assessment' stage. The WFD water body Suffolk coastal is shown in Figure 3.

Data for the Suffolk coastal water body has been obtained from the second River Basin Management Plan (RBMP) status objectives published by the Environment Agency, as presented in the online Catchment Data Explorer and the 'Cycle 2 Extended Water Body Summary Report' produced for each water body and presented in Table 1.

Table 1 Summary of WFD water body information

Parameter	Information
WFD	Suffolk
Water body ID	GB650503520002
River basin district name	Anglian
Water body type (estuarine or coastal)	Coastal
Water body total area (km^2)	146.53
Overall water body status (2016)	Moderate
Ecological status	Moderate
Fish classification status	Not assessed as coastal water body
Chemical status	Good
Target water body status and deadline	GEP by 2027
Hydromorphology status of water body	Not assessed
Heavily modified water body and for what use	Yes – Coastal and Flood Protection
Higher sensitivity habitats present	Polychaete reef and Saltmarsh
Lower sensitivity habitats present	Cobbles, Gravel and shingle, Intertidal soft sediment, Rocky shore, Subtidal rocky reef, Subtidal soft sediments
Phytoplankton status	Good
History of harmful algae	Not monitored
WFD protected areas within 2 km	Yes (European designated sites only)

1.2 The Sizewell power stations in context

There is an existing operational B station at Sizewell. The Sizewell B outfall is positioned 150m offshore (from mean water level on the beach face) and the SZB intake is located 600 m offshore (Figure 2). Sizewell B therefore extracts and discharges within Sizewell Bay and within the Suffolk coastal water body. The Sizewell C cooling water intakes and outfalls are located 3km offshore, beyond the Sizewell-Dunwich Bank, and therefore operational discharges would be within a different and larger tidal excursion with Sizewell Bay and the open sea.

Two Fish Recovery and Return (FRR) systems would also be part of the SZC cooling water system. The exact positions of the FRR outfalls are still subject to final engineering design, however, assessment suggests that the optimal easting is between BNG Easting 647977 and 648127. The northing positions of each FRR are aligned with the two EPR forebays, allowing a minimum tunnel length of ca. 400m offshore from the hard-coastal defence feature. As a result of the direct cooling of SZC with seawater, fish will be impinged through the cooling water infrastructure. To avoid fish, invertebrates and other debris passing into the station condensers, cooling water from the intakes passes through rotating drum and band screens. These organisms (fish and crustaceans >25 mm in length) are removed through impingement on rotating fine-mesh (10 mm at Sizewell B and this value is also proposed for Sizewell C) drum screens which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages.

Impinged organisms will be returned to the sea via a Fish Recovery and Return system (FRR). Not all will survive this process and EDF Energy is required to evaluate the impact of the loss of these organisms on marine communities. The FRR outfalls will be into the subtidal waters of Sizewell Bay, discharging at a depth of ca. 4 m.

The area enclosed by the Sizewell Bay is not a closed system and there is greater daily exchange of water between that area and the Southern North Sea and daily extraction of water within this area due to the SZB power station. The volume of the Sizewell Bay at Mean Sea Level (MSL) is 3.64×10^8 m^3. The Suffolk coastal water body intersects Sizewell Bay and is defined by the area enclosed by a boundary line extending from the coast with a northern boundary extending from North Beach Gunton (North of Lowestoft) and southern boundary at the North of old Felixstowe (Figure 3).

A typical value for the exchange rate coefficient in partially mixed estuaries is 5% volume exchange on each tide (Dyer, 1979), thus 0.1 per day is the value used here for various assessments. In the open sea greater variability could be expected, however, it is during calm periods, when only tidal processes operate that algal blooms are most likely, and this value is therefore used as a starting test value.

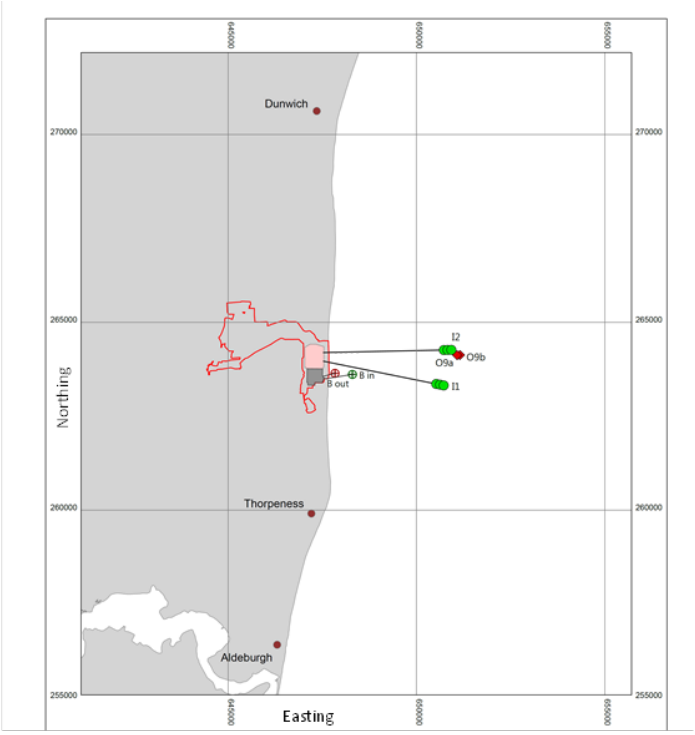

Figure 1: Overview of the SZC and SZB intakes and outfalls (top).

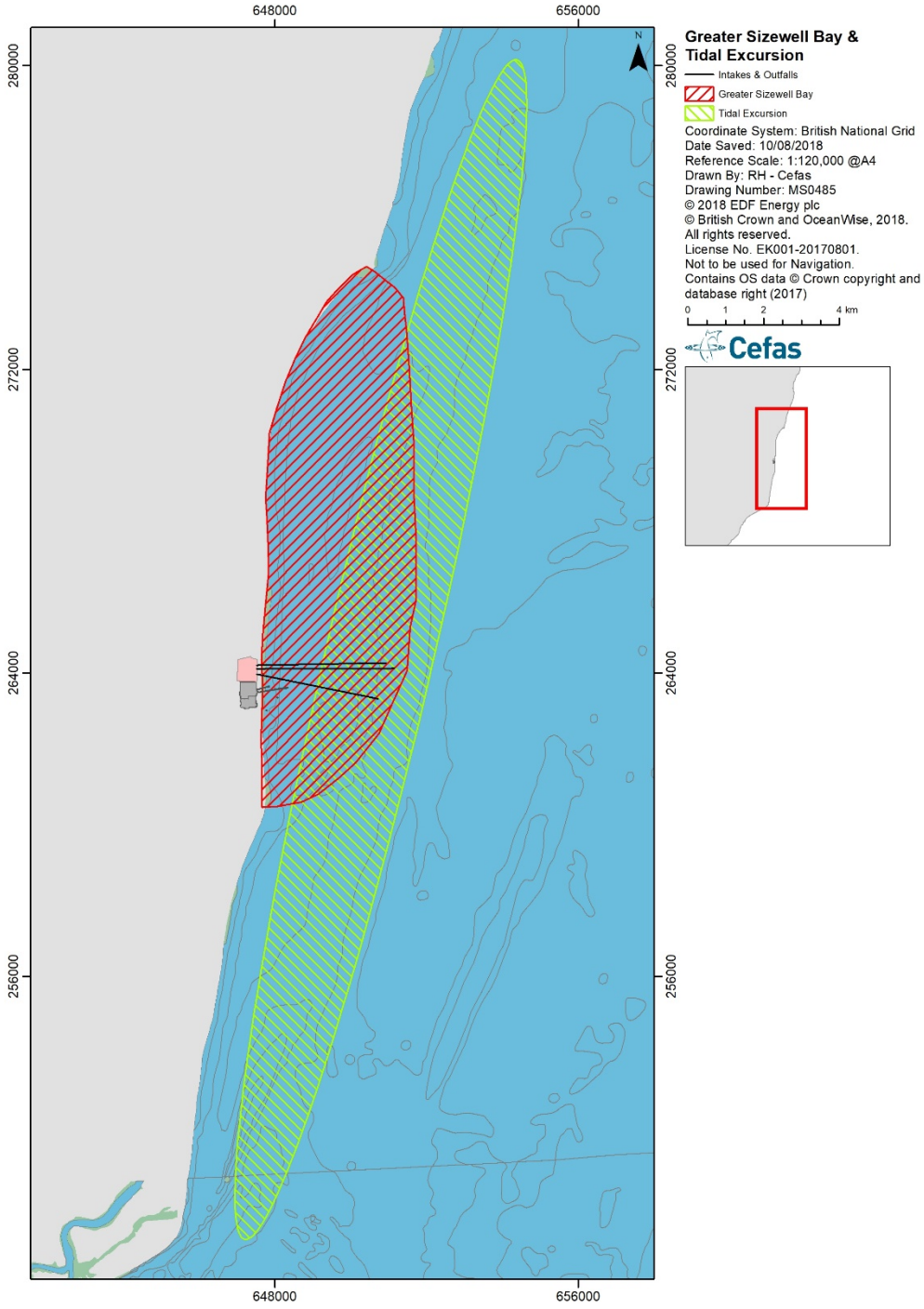

Figure 2: Areas from which the Sizewell B and Sizewell C intakes extract cooling water (used to calculate volumes for use in the simple phytoplankton box model, BEEMS TR385). Red inner hash area encapsulates the Sizewell Bay region used in association with the Sizewell B intake and outfall and FRRs for Sizewell C and the green outer hash area marks the area used in association with the Sizewell C intakes and outfall.

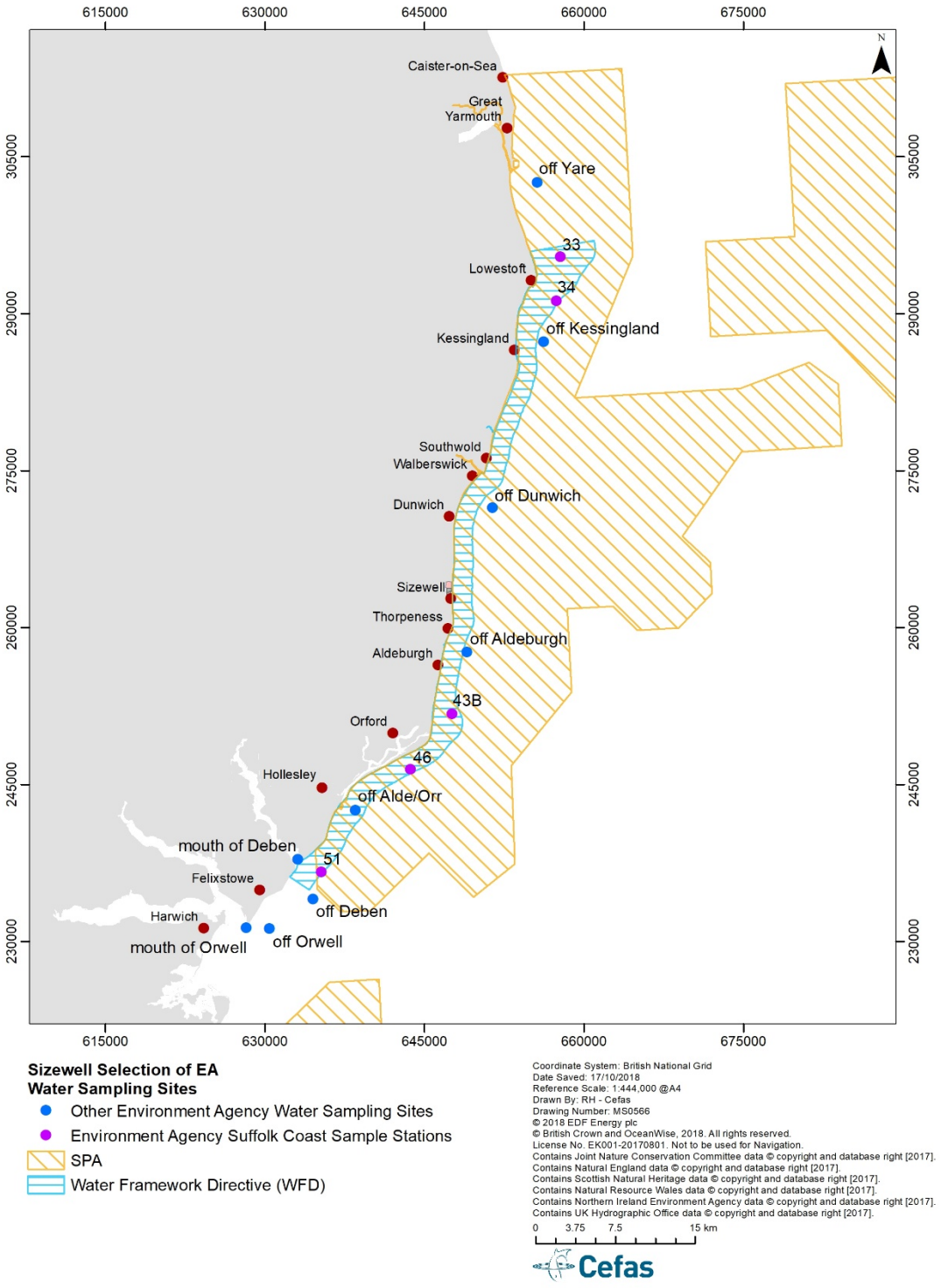

Figure 3: Map showing Suffolk Coastal boundary and that for the Outer Thames Special Protected Area. The red dots are geographical locations and the other points shown are Environment Agency (EA) water quality sampling stations shown in relation to Sizewell Power Station.

1.3 Background to assessments

During the operation of SZC, discharges of cooling water conditioning chemicals and treated sewage will increase dissolved inorganic nitrogen (DIN) and phosphorus (DIP) inputs to the marine environment (Appendix A). The SZC Fish Recovery and Return (FRR) is planned to provide a safe return of the more robust organisms from the drum and band screens directly into the marine environment. However, some of these organisms will not survive passage through the FRR system and after discharge, their subsequent decay will also introduce additional DIN and DIP into the local environment around the FRR outfall.

The Sizewell C cooling water system is chlorinated after the drum and band screens but routing of the water sources that supply the FRR would mean that it would not be chlorinated.

The SZC FRR will discharge into the Suffolk Coastal water body. Estimates of moribund biomass discharged from the FRR have been calculated and values for nutrient input and influence on other parameters have been derived (Appendix B) and are discussed in the following sections.

1.3.1 Assessment thresholds

The predicted amount of change for a given impact is assessed in relation to regulatory thresholds or standardised pressure benchmarks, for example, Environmental Quality Standards (EQS). In the absence of established standards, applied thresholds based on a 'weight of evidence approach' and pressure benchmarks proposed in Marine Evidence-based Sensitivity Assessments (MarESA) (Tyler-Walters *et al* 2018) are used to inform impact magnitude. Pressure benchmarks provide a basis for assessing the sensitivity of a given receptor to the site-specific impacts relative to recognised standards. Exceedance of benchmark thresholds would trigger further ecological investigation and does not necessarily infer sensitivity of all receptor groups.

1.3.2 Calculation of FRR discharges

The biomass of moribund organisms discharged from the SZC FRR system is calculated from the results of the SZC impingement assessment presented in BEEMS Technical Report TR406 v7.

The assessment here also considers the impingement reductions expected from the SZC low velocity side entry intake designs and provides an update to the assessment previously made (BEEMS TR193 Edition 5). The assessment here assumes that an AFD is not fitted.

2 Fish Recovery and Return System - assessment of potential effects on physicochemical parameters

The FRR system is designed to minimise impacts on impinged fish and invertebrate populations. However, some species such as clupeids are highly sensitive to mechanical damage caused by impingement on the screens and incur high mortality rates. The return of dead and moribund biota retains biomass within the local food web but represents a source of organic carbon with the potential to enhance secondary production of carnivorous zooplankton and through the detrital pathways. In addition to organic loading, the potential for increases in nutrients, un-ionised ammonia concentration and reductions in dissolved oxygen due to the decaying biomass are considered.

The total biomass of moribund biota that may potentially be discharged from the FRR is estimated based on the level of abstraction (pump rates) for the planned SZC intakes and the information on

seasonal distribution of species and length weight distribution of the species impinged for the existing Sizewell C (BEEMS TR406 and Appendix B, Tables 5 - 9). The assessment considers the total discharge loading of biomass from the moribund biota from both FRRs together at a single location FRR1 (the discharged material from the two FRRs separately would cover a slightly different footprint but assessing total discharge from FRR1 the most southerly produces a worst-case assessment as it is slightly closer to polychaete reef habitat). The main area of habitat overlap is also the same for the footprint derived assuming all biomass is discharged via FRR1 so does not affect the overall assessment.

The SZC data indicate that the highest biomass discharged occurs in Quarter 1 (January to March), when a mean of 965.76 kg per day (this would be 3239.6 kg without LVSE) is predicted to be discharged from the FRRs (**Appendix B**, **Table 9**). To provide a more conservative assessment 95th percentile loading values (1319.7 kg per day with LVSE and 4369.8 kg without) were used for all assessments other than for nutrient release which are modelled for a whole year and so use annual mean daily biomass.

Observations reported in BEEMS TR511 indicate that 88% of moribund fish discharged from the FRR would sink immediately and be deposited in an area around the FRR discharge. The remaining ca.,12% would be passively transported from the FRR outfall by the strong tidal currents. This group of dead fish would either:

- sink to the seabed before reaching land and either decompose or more likely be consumed by benthic organisms, or
- be consumed by foraging piscivorous birds (whilst the fish are floating but also once any fish beach.

There is a potential for some fish to be deposited on surrounding beaches and become a nuisance. However, modelling reported in TR511 demonstrated that for the peak impingement period of January to March only ca.,12% of the discharged dead sprat (the most abundant species in that period that would not survive impingement) would beach over a total extent of 18.0 km stretch of coast, with particles beaching 5.5 km south of the SZC FRR and 12.5 km north and that these would rapidly be consumed by seabirds within 1-2 hours of daybreak. TR511 further showed that of the 12% of the dead fish that did not sink immediately in the vicinity of the FRR outfall, a total of ca., 3.5% would be consumed by seabirds, with approximately 7.9% sinking over a wide area over a period of 24 hours. The chemical and biological effects of this latter group would be very diffuse and much less (e.g. in terms of chemical concentrations, effects on dissolved oxygen levels and in terms of organic enrichment of the seabed) than those due to the larger group that sank immediately after discharge. If the larger group caused no effect on water body status, then it was considered that the effects of the more diffuse group would be less and not merit further consideration.

For the following assessment, the water quality monitoring data for Sizewell (BEEMS TR314) provides the background parameters against which the inputs estimated from SZC are considered. The SZB inputs are, therefore, included in this measured baseline.

2.1 Calculated effects of decaying moribund biomass from the FRR on DIN/DIP and phytoplankton growth

For the operation run abstraction mortality was accounted for. The Fish Return and Recover system (FRR) aims to discharge fish live to the receiving waters. However, clupeids such as sprat and herring are highly sensitive to mechanical damage during passage through the cooling water system and incur high mortality rates (assumed to be 100%). The return of dead and moribund

biota retains biomass within the system, but decay of organic material would release nutrients into the system. There is a strong seasonal bias to impingement numbers, and the return of dead biomass. The largest quantity of dead biomass is returned in January, February, and March, however during this period light is the limiting factor for phytoplankton growth. Dead biomass returned during the summer months, coinciding with periods of nutrient limitation, is much lower than in winter. However, as a highly precautionary measure the total biomass discharged per annum (with LVSE) was modelled as a daily average equating to approximately 382.91 kg of fish (wet weight, **Appendix B1 Table 10**). These values are based on rates of impingement at Sizewell B and extrapolated to account for abstraction volumes and they also account for the impingement reductions expected from the SZC low velocity side entry intake designs. A conservative assumption was applied whereby all of this mass of fish was assumed to be available as nitrogen and phosphorus sources (i.e. no predation was assumed) leading to an additional 13.40 kg per day of nitrogen (based on Walker et. al., 2011) and 1.91 kg of phosphorus (based on Gende et. al., 2004) per day (**Appendix B2, Table 11**), in addition to inputs from the discharge due to sewage and conditioning chemicals.

The discharge of nutrients during the SZC construction phase is primarily into the Sizewell Bay area inside of the Bank, as it is mostly via the combined discharge outfall (CDO). However, it is possible that there will be discharge of some nutrients via the main cooling water outfalls during functional testing. As the suspended sediment is less in offshore surface waters, i.e. where there is a better light regime, a worst-case scenario has been assumed that all these nutrients are discharged offshore. As it is likely there will be periods of SZB outage during the construction of SZC the increased mortality due to SZB has been ignored, so that these results will slightly over predict the role of the increased nutrients (i.e. a conservative assumption).

The FRR discharge is predicted to release dissolved inorganic nitrogen (DIN) into Sizewell Bay. Under the WFD standards, the Suffolk coastal water body has ‘Moderate’ status for DIN. Discharges from an operational SZC (**Appendix A**, **Table 4** and **Appendix B**2 Table 11) would result in a very small elevation in DIN in the receiving water body representing 0.3% of the mass making up the daily volume exchange for Sizewell Bay (with around 26% of this from the FRR). The daily phosphate input from the FRR and operational discharge would represent ca. 0.11% of the mass of phosphate making up the daily exchange for Sizewell Bay (with most ca. 73% of this from the FRR).

This mean level of input for both DIN and DIP from combined operational and FRR inputs has been assessed using a Combined Phytoplankton and Macroalgae (CPM) model to predict the effect of Sizewell C on phytoplankton community biomass (BEEMS TR385). This model simulates the dynamics of phytoplankton biomass using data on known environmental drivers such as nutrients and light.

During construction, the exact mass of the discharges has considerable variation, depending on which activity is occurring and varies considerably with the workforce on site. The details of this variation are described in BEEMS Technical Report TR193, where worst cases have been used and reproduced in **Appendix A Table 4**.

During operation, the maximum number of people on site occurs when there are refuelling outages, during this time nitrate and phosphate loads are increased above background concentrations and these contributions are represented by the peak 24 hour loading during operation. The refuelling outages typically last four to six weeks but can occur at any time of year. During the winter period light is limiting and there is no effect resulting from the additional supply of nutrients. It is only in summer that the discharge needs to be considered.

Typically, in offshore waters of the UK nutrient concentrations are reduced to very low levels due to phytoplankton uptake, but in the near shore coastal waters (3-5 km from the coast) off Sizewell due to the turbid nature of the coastal environment and continual freshwater input from the south of

the area (River Orwell and River Deben) there are background summertime inputs of nutrients. Observations (BEEMS Technical Report TR314) show these to be around 10 µMol l^{-3} for nitrate and 0.65 µMol l^{-3} for phosphate. As the daily exchange of water is around 10%, the total additional mass of nitrate per day in summer is the volume x 10% x concentration, this gives a daily exchange of 16.9 tons of nitrate and 2.4 tons of phosphate.

During construction (**Appendix A Table 4**) all daily inputs are ≤1.1% of the normal daily exchange of nutrients at the boundary and therefore would be indistinguishable from the modelled situation without the SZC discharges. During the operational phase, peak daily loading for nitrogen does reach 1.9% of the daily exchange but it is expected that this peak value is for short periods only. The more realistic average daily value of N (including FRR discharges) is low at 0.3% of the daily exchange (45.4/16900). For phosphate, the maximum daily value can be 5% of the daily exchange, however phosphate is not a limiting nutrient and therefore the addition of more phosphate does not produce more growth. In addition, over a whole year the phosphate contribution (including FRR) is very small at less than 0.11%.

To provide greater reassurance of this assessment, model runs were conducted to assess the influence on phytoplankton growth of representative inputs of nutrients (i.e., total annual loadings) from the power station during operation and during construction including commissioning inputs. Model runs reported in BEEMS TR385 did not include mitigation by the LVSE and were set at operational levels including the FRR of 69.3 kg/day for nitrogen and 6.04 kg of phosphorus. These values are therefore conservative and would exceed the 95th percentile FRR with LVSE and combined operational loadings for nitrogen (53.03 kg) and phosphorus (3.33 kg).

Using the CPM operational inputs have a slightly higher influence on phytoplankton production but for both construction and operation there is less than a negligible 0.3% increase in annual gross production of carbon.

The potential for nutrient additions to enhance opportunistic macroalgae in the Blyth or the more distant Alde Orr Estuary has been raised. Direct links between nutrient loading and macroalgal growth have been established but evidence supporting direct causal linkages between effluent discharges and the macroalgae community structure is limited (Wells et al., 2010). Prolific macroalgal growth during periods of high nutrients is dependent on other factors such as the position on the littoral zone, tidal action, light availability, and seasonal temperature (Wells et al., 2010). During the construction and operational phases of the proposed development, additional nutrients would be added to the system. However, nutrient additions represent a very small fraction of the daily exchange with wider sea areas (**Appendix A Table 4**) and modelling (without LVSE mitigation) has demonstrated no negligible enhancement of phytoplankton growth above present background. The point source is located a considerable distance from the estuaries, over 10km from the mouth of the Blyth and 23km from the mouth of the Alde-Orr estuary at Shingle Street. Furthermore, the inshore estuaries and tidal areas of Eastern England, monitored for opportunistic macroalgal as part of the WFD, do not appear to exhibit excessive growth (Painting et al., 2017). As such, no discernible effects on opportunistic algae because of nutrient discharges from the proposed development are anticipated.

Considering the influence of thermal uplift due to Sizewell C upon phytoplankton growth in combination to nutrient inputs a value of 0.5 °C has been used as representative of the whole GSB, along with a 1°C value for comparative purposes (BEEMS TR385).

Whilst phytoplankton growth is mostly unchanged by thermal uplifts, increased zooplankton activity and grazing on phytoplankton occurs. The overall result is a small decrease in simulated chlorophyll concentration and total phytoplankton production.

The assessments described for nutrient inputs for the combined operational and FRR are covered in more detail in BEEMS TR385 however this assessment does not account for biomass reductions with the LVSE included and is precautionary.

In terms of the most recent Marine Strategy eutrophication assessment, the elevated BOD and ammonia have very localised influence and would not change the current Marine strategy status as "non problem" for the Greater North Sea. The most recent eutrophication assessment published in 2019 (https://moat.cefas.co.uk/pressures-from-human-activities/eutrophication/) by Defra. The UK National Report 2016 (Painting et al., 2017), showed that only a small number of eutrophication problems remain in coastal and estuarine waters, representing 0.03% of the total UK Exclusive Economic Zone, and 0.41% of estuarine and coastal waters. The closest "problem area" to SZC according to this assessment is Chichester Harbour. Currently, there are no major outstanding issues for eutrophication in the UK as a whole and the inputs indicated for this assessment would make a negligible contribution to the overall loading for the Southern North Sea area.

2.2 Calculated effects of decaying moribund biomass from the FRR on dissolved oxygen levels

The decaying fish biomass has the potential to contribute to the biological oxygen demand (BOD). An estimated 3.5 g of oxygen is required for complete oxidation of one gram of dry mass and this value is derived based on a study of particulate organic matter from fish cages (Stigbrandt, 2001) (see Appendix B1 and B2). This source-term for oxygen demand is used to derive an estimate of the BOD contribution from the daily biomass. The source BOD value is used to derive an estimate of the BOD contribution from the period of highest biomass inputs, Quarter 1 (Based on 95th percentile biomass, 1319.73 kg with LVSE in place and 4369.83 kg without LVSE, **Appendix B2**, BOD section). The estimate is 1662.86 kg BOD/day with LVSE and 5505.99 kg BOD/day without LVSE (after allowing for a wet weight/dry weight adjustment see **Appendix B2,** BOD section).

Any area that exceeds 1.5 mg l^{-1} deviation in BOD from background is expected to generate less than 0.5 mg l^{-1} impact/reduction on dissolved oxygen (OSPAR, 1997). Therefore, dividing the BOD loading by 1.5 and multiplying by 0.5 produces an estimate of the total oxygen reduction potential due to the BOD input which is 554.29 kg/day with LVSE (**Appendix B2** BOD section). Without LVSE the oxygen reduction potential would be 1835.33 kg/day.

Based on a background concentration of 6.96 mg/l dissolved oxygen the calculated O_2 demand requirement (554.29 kg with LVSE) is equivalent to oxygen available in 79,639 m^3. This volume represents 0.22% of the daily exchange for GSB (with no LVSE this value increases to 0.73% of the daily exchange).

Reaeration at the sea surface would also replenish oxygen levels. Typical values of oxygen flux are 100mmol m^{-2}d^{-1} (Hull, 2016) or 3.2 gm^{-2} d^{-1} therefore daily reaeration across 0.2 km^2 (17.32 ha with LVSE in place, **Appendix B2**, BOD section) would be expected to compensate for the estimated daily oxygen consumption by decaying fish biomass.

In practice both reaeration and tidal exchange in combination would replenish dissolved oxygen.

As increasing temperature would reduce the capacity for seawater to carry oxygen the heated discharge from Sizewell C would reduce the potential oxygen levels in Sizewell Bay so this extreme condition is considered for Quarter 1 when maximum biomass discharge occurs. Considering this scenario of temperature elevation during the maximum biomass discharge in Q1 (as a 95th percentile) the calculated oxygen reduction due to biomass degradation is considered relative to a 1.77 mgl^{-1} reduction to the background oxygen level under thermal influence (from 5.77 mg l^{-1} to 4 mg/l^{-1}) and this would result in the affected area moving from Good to Moderate status. The theoretical area that would be affected based on the maximum biomass discharge and under a thermal uplift of 2°C in Q1 with LVSE fitted is calculated as 7.83 ha and would be defined by a tidal ellipse of 766.89 x 129.98 metres (**Appendix B2**, BOD section).

This assessment makes worst case assumptions of instantaneous breakdown of all available +biomass and no losses through predation. Reduction of oxygen concentration will only occur if the rate of oxygen use due to BOD is greater than daily exchange of Sizewell Bay and the oxygen transfer across the water surface. The waters off Sizewell are well mixed vertically facilitating reaeration at the surface. The additional BOD loading from the FRR discharge is very small compared to the natural background level. This level of change would not influence the current dissolved oxygen status of the Suffolk coastal water body and represents 0.05% of the WFD water body area.
This is a worst-case assessment as it assumes no predation and no remobilisation of partially decayed fish. In practice both effects will take place reducing the predicted area of influence on dissolved oxygen levels.

2.3 Calculated effect of decaying moribund biomass from the FRR on un-ionised ammonia

Decaying biomass will result in local increases in un-ionised ammonia. Studies on cod tissue show ammonia contribution of 125 mg kg^{-1} NH_4-N (Timm and Jorgensen, 2002) (see **Appendix B2**, un-ionised ammonia section). This value is used as a proxy in the un-ionised ammonia calculator (Clegg and Whitfield, 1995), along with relevant site background conditions for pH, temperature and salinity, to indicate the potential un-ionised ammonia contribution from decaying biomass in Sizewell Bay. Based on the daily 95th percentile biomass of fish discharged during Quarter 1 with LVSE in place (and mean pH, and salinity and mean temperature for this period) the estimated NH_3 loading could be at or above the EQS (NH_3-N, 21 $\mu g\ l^{-1}$) in a volume of 159,000 litres of seawater or an area equivalent to 0.004 ha (0.013 ha without LVSE). This is a conservative assessment accounting for the highest potential biomass discharge from the FRR.

Temperature elevation of 2°C that might occur around the FRR due to the added thermal influence of Sizewell C will elevate the proportion of un-ionised ammonia relative to the influence of natural background temperatures alone (which take account of Sizewell B inputs). Taking account of conditions that would maximise the un-ionised ammonia concentration a 2°C elevation above Q1 natural background temperature at Sizewell, a lower salinity 31.7 (5th percentile) and high pH 8.23 (95th percentile) the volume of seawater affected by elevated un-ionised ammonia would increase to 281,000 litres (an area equivalent to 0.007 ha). Without an LVSE in place this latter value would increase to 0.023 ha. This most extreme scenario may be representative for thermal uplift and for reduced salinity due to potential higher freshwater inputs during winter however pH values during winter are likely to be much lower due to limited primary production at this time of year so it is a very conservative assessment.

This is a worst-case assessment as it assumes no predation and no remobilisation of partially decayed fish. In practice both effects will take place further reducing the predicted area above the EQS.

2.4 Calculated effect of decaying moribund biomass from the FRR on organic carbon deposition

The peak biomass of moribund fish discharged from the FRR, 1319.73 kg/day (95th percentile value with LVSE in place), for Quarter 1 would contribute an estimated input of ca. 411.75 kg organic carbon (see **Appendix B2**, organic enrichment section). Settlement of moribund biomass could result in smothering of the seabed and organic enrichment contributing to deoxygenation in the sediments and disturbance on sediment geochemistry. These changes can subsequently lead to decreasing numbers of sediment dwelling species, individuals, and biomass.

There are no regulatory standards for assessing organic smothering of benthic organisms. In the absence of established standards, applied thresholds based on a ‘weight of evidence approach’

and pressure benchmarks proposed in Marine Evidence-based Sensitivity Assessments (MarESA) (Tyler-Walters *et al* 2018) have been used in the assessment of potential effects. Pressure benchmarks provide a basis for assessing the sensitivity of a given receptor to the site-specific impacts relative to recognised standards. For organic carbon deposition the appropriate benchmark is defined as 100g organic carbon/m^2/year. Exceedance of this benchmark threshold would trigger further ecological investigation and does not necessarily infer sensitivity of all receptor groups.

An equivalent daily benchmark would be 0.3 g organic carbon/m^2/day. During the winter (Quarter 1) the highest numbers of fish are discharged from the FRR and adopting the peak value of associated carbon as a 95th percentile this would contribute to an organic carbon loading at the benchmark standard level over an area of 1.50 km^2 (4.98 km^2 with no LVSE). Based on annual average biomass inputs the area at the benchmark value would be equivalent to 0.44 km^2 (1.47 km^2 with no LVSE). The annual average biomass input equivalent area of elevation of background organic carbon loading with no LVSE in place represents 1.01% of the Suffolk coastal water body and is not expected to alter overall water body status. This latter value is ca., 0.3% of the Suffolk coastal water body area when taking account of the LVSE reductions.

Using the plume shape factors described in BEEMS TR306, with LVSE in place the total length of the worst case organic enrichment footprint in Quarter 1 above the pressure benchmark would be approximately 5.9 times the width i.e. the equivalent tidal ellipse with LVSE in place would have a total length of approximately 3360 m (radius 1680) and a total width of 569 m (radius 285). For Q1 the average loading of biomass organic carbon, would be equivalent to an area encompassed by a tidal ellipse of approximate total length 2874 m (radius 1437) and a total width of 487 m (radius 244). With no LVSE the average biomass loading for Q1 would be increased with equivalent ellipse of total length 5264 m (radius 2632) and width 892 m (radius 446). However, from April through to December in quarters 2 to 4 the mean daily biomass is considerably reduced and leads to a lower mean annual loading of organic carbon for which the equivalent tidal ellipse would be of approximate total length 1810 m (radius 905) and of total width 307 m (radius 153).

This is a worst-case assessment as it assumes no predation and no remobilisation of partially decayed fish. In practice both effects will take place reducing the predicted area above the benchmark level.

3 Fish Recovery and Return System assessment of potential overlap of FRR discharge on habitats

Figure 4 and Table 2 show the EUNIS habitats in the vicinity of the Sizewell C FRR discharge outfall. The decay of dead fish discharged from the FRR may affect surrounding habitat due to the un-ionised ammonia released, the resulting deoxygenation or the enrichment and smothering effect of the organic matter. Based on the likely plume shape dimensions an ellipse representing the area over which a given parameter has an affect is derived. For un-ionised ammonia, this plume area is evaluated relative the EQS of 21μgl^{-1}. For dissolved oxygen, the oxygen demand of the organic matter is calculated and the equivalent area over which this demand would reduce oxygen status from the current Good background to a Moderate status level was calculated. The area effected by discharged organic matter was calculated relative to the benchmark loading value (section 2.4) and this latter area is shown on Figure 4. The largest areas are likely to be affected by organic enrichment, so this is the parameter considered in the following tests however all parameters are considered in the discussion for higher and lower sensitivity habitats.

The tests for inclusion of habitats in a WFD assessment are if the footprint of FRR discharge is any of the following:

i. 0.5 km² or larger
ii. 1% or more of the water body's area
iii. within 500 m of any higher sensitivity habitat
iv. 1% or more of any lower sensitivity habitat

3.1 Applying the habitats tests

i. The effect footprint (organic enrichment of seabed sediments) for the Q1 as a 95th percentile is 1.50 km^2 with LVSE included in the assessment (Appendix B2), i.e. >0.5 km². The effect footprint increases if LVSE is not included in the assessment (Appendix B2).

ii. 1% of the water body area (Suffolk Coastal water body, 14653.27 ha, derived directly from the WFD GIS data Defra, 2017) is 1.46 km^2. The largest effect footprint for the Q1 95th percentile organic matter is 1.50 km^2 with LVSE included in the assessment and this represents 1.03% of the Suffolk Coastal WFD water body (details **Appendix B2,** organic enrichment section). The effect footprint for organic matter would increase if LVSE is not included in the assessment (**Appendix B2**).

Table 2 EUNIS Level 4 habitat total area information for Greater Sizewell Bay of relevance to FRR discharge

EUNIS* code	Description	Area (ha)	Area in WFD (ha)
A4.22	*Sabellaria* reefs on circalittoral rock	402.88	340.17
A5.13	Infralittoral coarse sediment	677.55	629.99
A5.23/A5.26 (transitional habitat)	Infralittoral fine sand/ Circalittoral muddy sand	1490.57	392.48
A5.23	Infralittoral fine sand	2,208.9	1362.05
A5.26	Circalittoral muddy sand	249.02	0.00
A5.33/A5.26 (transitional habitat)	Circalittoral muddy sand/ Infralittoral sandy mud	3428.17	1627.40
A5.43	Infralittoral mixed sediments	73.41	59.88
Engineering Feature**	Engineering Feature**	1.11	1.11
	TOTAL	*8531.61*	*4413.07*

*The EUNIS classification covers the local and sub regional area around Sizewell. The extent of the area is only limited by the extent covered by three acoustic surveys, see BEEMS Technical Report TR087.

** Sandbank feature has been factored out, 851.459 ha in full data and 346.6812 ha within WFD. The sandbank feature (8m contour Sizewell Dunwich bank) overlaps the EUNIS areas and therefore if left in the area (ha) would be double counted.

iii. Distance from higher sensitivity habitats

With LVSE included in the assessment, the effect footprint (organic enrichment) for Q1 as a 95th percentile would not overlap coralline crag and associated polychaete (*Sabellaria spinulosa*) reef habitat and would be at a minimum distance of 583 metres from the nearest point of this feature (A4.22 in **Figure 4**). The effect footprint would marginally overlap (1.54%) this feature in the unlikely scenario that LVSE is not used.

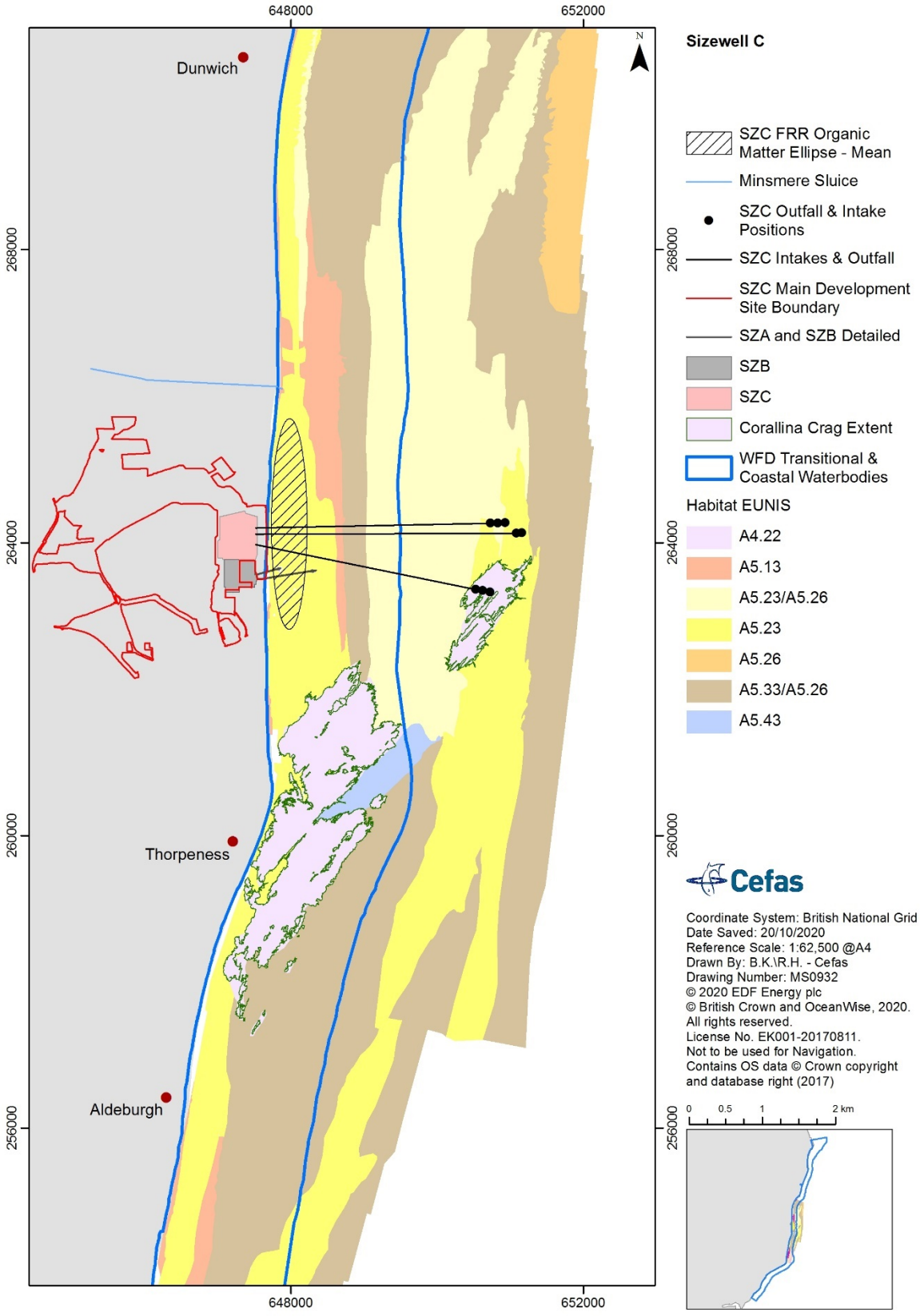

Figure 4. The hatched ellipse shows the organic enrichment footprint of the FRR discharge for the Q1 mean organic carbon with LVSE included. The colour shaded areas correspond to different EUNIS Level 4 habitats with EUNIS codes shown in the legend and in more detail in **Table 2**.

iv. Lower sensitivity habitats

Of the EUNIS habitats within Sizewell Bay (**Table 2**), the 95th percentile organic matter deposition from FRR discharges during Quarter 1 would overlap 'Infralittoral fine sand' (A5.23) with LVSE in place. The area of overlap of the effect footprint with these habitats within the WFD water body is > 1% (10.98%, **Table 3**). In the unlikely scenario that LVSE is not used, two additional lower sensitivity habitats would be overlapped by the organic enrichment footprint: 'Infralittoral coarse sediment' (A5.13), and a transitional habitat between A5.23 and A5.26 Infralittoral fine sand/Circalittoral muddy sand (**Table 2**). However, the effect footprint would extend over <1% (0.7%) of the latter transitional A5.23/A5.26 habitat within the Sizewell Bay.

Table 3 Areas of intersection of FRR effect footprint for the March maximum with LVSE included in terms of organic matter enrichment and lower sensitivity EUNIS habitats in Sizewell Bay.

EUNIS Habitat codes	Area in Sizewell Bay (ha)	Area in WFD (ha)	FRR effect footprint area of intersection (ha)	Percentage Total	Percentage WFD
A5.23	2208.90	1362.05	149.37	6.76	10.98

3.2 Habitat assessment

A habitat assessment is required based on the results of tests i, ii and iv (and test iii if no LVSE is used). The potential for benthic biota within Sizewell Bay to be affected by organic matter enrichment (food supply, smothering and deoxygenation of sediments) from FRR discharges is therefore assessed below.

3.2.1 Higher sensitivity habitats

Higher sensitivity habitats would not be affected by organic enrichment from the FRR in the expected scenario that LVSE is used, as polychaete (*S. spinulosa*) reef is not within 500 m of the effect footprint.

Regardless of whether LVSE is used, polychaete reef habitat is not within 500 m of the area likely to be affected by elevated un-ionised ammonia concentration due to the decay of discharged biomass, which at a concentration of un-ionised ammonia EQS of 21μg l^{-1} would be equivalent to a small area of 0.004 ha for Quarter 1 for a 95th percentile biomass loading with LVSE when highest biomass discharge is predicted (0.007 ha without LVSE). This area predicted for Quarter 1 would increase marginally to 0.013 ha if thermal elevation of 2°C is accounted for (0.023 ha without LVSE).

Based on background dissolved oxygen level of 6.96 mg l^{-1} the calculated oxygen reduction due to biomass degradation with LVSE in place is 0.22% of the tidal exchange for Sizewell Bay (0.73% without LVSE in place). Oxygenation would also be affected by reaeration at the sea surface (see section 2.2).

Under thermal influence the dissolved oxygen concentration is predicted to decrease from a background of 6.96 mg l^{-1} to 5.77 mg l^{-1}. A further reduction of this background oxygen concentration under thermal influence from 5.77 mg l^{-1} to 4 mg l^{-1} (a reduction of 1.77 mg l^{-1}) would move it from Good to Moderate status. The theoretical area that could be affected by a reduction of 1.77 mg l^{-1} based on the predicted oxygen demand of the 95th percentile biomass discharge in Quarter 1 would be equivalent to an area of 7.83 ha. If this area is derived as a discharge ellipse around the FRR it would have a total length 766.89 m and total width 129.98 m. This ellipse footprint for dissolved oxygen would be >1600 metres from the higher sensitivity habitat polychaete reef.

These assessments are conservative based upon two implausible assumptions:

1. no predation of dead fish either by other fish or benthic organisms; and
2. no remobilisation of partially decayed fish.

In practice, there is likely to be substantial predation, and the worst-case discharge month of March is in the period of high storm intensity and wave energy which will produce high rates of remobilisation and

subsequent tidal dispersal of decaying matter. Moreover, *S. spinulosa* is not considered vulnerable to organic enrichment as reefs generally develop in areas of high-water movement where material would tend to disperse rather than settle out (Gibb et al., 2014). The presence of *S. spinulosa* adjacent to a sewage sludge dumping site also suggests that this feature is tolerant to high levels of organic enrichment (Walker and Rees, 1980).

As the effect footprint would not overlap polychaete (*S. spinulosa*) reef habitat with LVSE in place, and *S. spinulosa* appears to be resistant to organic enrichment, discharges of organic matter from the FRR are not expected to have a significant effect on WFD higher sensitivity habitats.

3.2.2 Lower sensitivity habitats

The subtidal surveys suggest there is one overall infaunal and epifaunal community spanning most of Sizewell Bay (BEEMS Technical Report TR348), meaning that benthic invertebrate species have a largely ubiquitous distribution across habitats. The response of the benthos to organic enrichment is, therefore, unlikely to differ depending on which lower sensitivity habitats are affected (i.e. the size of the footprint whether LVSE is included, **Table 3** and **Appendix B.3**).

There is negligible overlap of lower sensitivity habitat with a potential un-ionised ammonia plume (with LVSE in place) at a concentration of un-ionised ammonia equal to the EQS of 21μg l^{-1}. Based on natural background physicochemical conditions a 95th percentile biomass discharge for Q1 with LVSE is calculated to produce un-ionised ammonia levels that would be equivalent to a small area of 0.004 ha (0.013 ha without LVSE). This area predicted for Q1 would increase marginally to 0.007 ha if thermal elevation of 2°C is accounted for (0.023 ha without LVSE).

As described for higher sensitivity habitats, the theoretical area that would be affected by a dissolved oxygen reduction equivalent to a reduction from Good to Moderate status is calculated as 7.83 ha (with LVSE) and would represent 0.6% of this habitat in the Suffolk water body, but the habitats concerned are considered not sensitive to this change.

The effects of biomass discharges contributing to organic matter deposition on lower sensitivity habitats, which are overlapped by the effect footprint by up to 11% of their area within the Suffolk water body, could also occur through increases in food supply. Scavengers, predators, and surface deposit feeders are most likely to respond to this change to the environment. Few benthic invertebrates within Sizewell Bay obtain their nutrition from scavenging, with <5% of infaunal and epifaunal individuals exhibiting this feeding mode (BEEMS Technical Report TR348). However, most epifaunal individuals are predators and ca., 40% of infaunal individuals are surface deposit feeders (BEEMS Technical Report TR348). Therefore, a substantial component of both infaunal and epifaunal communities could benefit from organic matter input.

Population densities of scavengers, predators and surface deposit feeders may increase within the effect footprint (1.5 km^2 for the likely LVSE scenario) due to the increased food supply. This is supported by experimental evidence from UK waters relating to whelks, crabs, amphipods, shrimps, and echinoderms (Ramsay et al. 1997; Groenewold and Fonds 2000). While this can be considered a positive response, the amount of change would be small at the scale of Sizewell Bay.

As negative effects due to ammonia and deoxygenation are limited, while any positive effects of increased food supply are likely to be localised, discharges of organic matter from the FRR are not expected to have a significant effect on WFD lower sensitivity habitats.

4 Discussion

The results of the assessment of the influence of the HPC FRR inputs in combination with cooling water discharges from HPB and HPC are as follows:

Water quality effects

1. Discharges from Sizewell C with an FRR fitted (**Appendix A**, **Table 4**) would result in a very small elevation in DIN in the receiving water body representing ca., 0.3% of the mass present in the daily

tidal exchange for Sizewell Bay (with ca. 26% of this from the FRR). For dissolved inorganic phosphorus (DIP) combined inputs from operation and the FRR would result in a very small elevation in DIP in the receiving water body representing ca., 0.11% of the mass present in the daily tidal exchange for Sizewell Bay (with most, 73% of this from the FRR). This level of input for both DIN and DIP is negligible and would not change the current nutrient status of the Suffolk coastal water body or have significant influence of the Marine Strategy area Greater North Sea.

2. The decaying biomass of moribund fish from the FRR would have negligible impact on the oxygen levels for the Suffolk coastal water body with the predicted demand being equivalent to an area of 0.08 km^2 (with LVSE in place, **Appendix B2,** BOD section) that would be subject to a change from Good to Moderate status for dissolved oxygen including the influence of a 2°C thermal uplift.

3. Based on the daily average biomass of fish discharged during Q1 with LVSE (and relevant pH, salinity and temperature) the estimated NH_3 loading is equivalent to a relatively small volume of water at the EQS concentration (NH_3-N, 21 µg/l) of ca. 159,000 litres around the FRR (**Appendix B2**, un-ionised ammonia section). As the ammonia is produced from decaying organic matter it is likely to be rapidly mixed and dispersed by tidal currents so that negligible areas are affected by un-ionised ammonia levels at or above the EQS. Hypothetically if mixed evenly through the full depth of the water column (4m) the volume of seawater at the EQS would affect a very small area of ca. 39.8 m^2 (0.004 ha). Accounting for a temperature elevation of 2°C that might occur around the FRR (with LVSE) due to the influence of cooling water discharge, the volume affected by elevated un-ionised ammonia would increase to 281,00 litres. Hypothetically if mixed evenly through the full depth of the water column (4 m) the volume of seawater at the EQS would affect a slightly larger area of ca. 70.3 m^2 (0.007 ha).

4. For organic carbon deposition a benchmark is defined as 100 g organic carbon/m^2/year (**Appendix B2**, organic enrichment section). An equivalent daily value would be 0.3 g organic carbon/m^2/day. During the winter (Q1) the highest numbers of fish are discharged from the FRR and adopting the 95th percentile (with LVSE) of associated carbon this would contribute to an organic carbon loading at the benchmark standard level over an area of 1.5 km^2.

These assessments are considered conservative as they assume instantaneous breakdown of biomass discharged from the FRR, no predation from benthic organisms and no redistribution of decaying material by tidal or wave action. In practice all these effects will occur and the predicted areas of effect on water quality status will be further reduced.

Ecological effects

1. Phytoplankton production levels within the Sizewell Bay and the small overlapping area of the Suffolk coastal water body would not increase due to the added nutrients from the FRR.

2. The discharge of the FRR with LVSE is not within 500 m of higher sensitivity habitat polychaete reef and there is no organic carbon deposition overlap with the *Sabellaria* sublittoral reef habitat area for Sizewell. The much smaller potential effect footprints for dissolved oxygen reduction or elevated un-ionised ammonia are also very unlikely to have any effect on this habitat. The area potentially affected by the discharge footprints is based on a most conservative assessment and considers the 95th percentile biomass discharge and assumes no predation of dead fish.

3. *Sabellaria* reef habitat is in any case not considered a sensitive receptor for the additional organic carbon input as they are typically associated with dispersive conditions that would reduce settlement of material and evidence suggests ability to adapt to organically enriched conditions.

4. The FRR discharge would overlap with lower sensitivity infralittoral fine sand habitat (A5.23). The potential impact footprint for elevated un-ionised ammonia is negligible. For reduced dissolved oxygen the influence of the plume may extend over 7.83 ha of low sensitivity habitat and this represents a maximum of 0.6% of the area of habitat concerned within the Suffolk Coastal water body. The low sensitivity habitats are not assessed as sensitive to the low level of reduced dissolved oxygen.

5. The organic carbon deposition has a larger footprint of potential area affected and this could overlap with an area of 10.97% infralittoral fine sand.

6. Benthic invertebrate species have a largely ubiquitous distribution across habitats in Sizewell Bay. An assessment of the core species indicates that many are tolerant of smothering and deoxygenation and a substantial component of both infaunal and epifaunal communities could benefit from organic matter input.

7. Overall, discharges from the FRR are likely to have neutral to localised minor effects on core species of Sizewell Bay. Population densities of scavengers, predators and surface deposit feeders may increase within the effect footprint (1.5 km^2 for the likely LVSE scenario) due to the increased food supply. While this can be considered a positive response, the amount of change would be small at the scale of Sizewell Bay.

4.1 Conclusion

The predicted changes in water quality, to inorganic enrichment of benthic habitat and to phytoplankton production in Sizewell Bay and the adjacent Suffolk coastal water body due the FRR discharges are negligible and would cause no deterioration to the status. The overlap of predicted areas of potential effect for dissolved oxygen and un-ionised ammonia are negligible. There is no overlap of the organic carbon footprint with sensitive *Sabellaria* reef habitat when the planned LVSE is fitted. However, the sensitivity of *Sabellaria* to the influence of elevated organic carbon loading is considered to be low. Insignificant areas of low sensitivity habitat would be affected by reduced dissolved oxygen or elevated un-ionised ammonia although small areas ca. 11% of one habitat within the Suffolk water body is also affected by elevated organic carbon loading. However, the core species either are tolerant of smothering and or have short life histories and are able to rapidly recolonise new areas of habitat and as the assessments are conservative (not accounting for predation or initial dispersal of discharged biomass or the turbulent mixing within the water column the effects are minor and localised. No deterioration is therefore concluded for adjoining water bodies both upstream and downstream of the discharge.

In terms of the most recent Marine Strategy eutrophication assessment, the elevated BOD and ammonia have very localised influence and would not change the current MSFD status as “non problem” for the Southern North Sea coastal area. The most recent eutrophication assessment published in 2019 (https://moat.cefas.co.uk/pressures-from-human-activities/eutrophication/) by Defra. The UK National Report 2016 (Painting et al., 2017), showed that only a small number of eutrophication problems remain in coastal and estuarine waters, representing 0.03% of the total UK Exclusive Economic Zone, and 0.41% of estuarine and coastal waters. The closest “problem area” to SZC according to this assessment is Chichester Harbour. Currently, there are no major outstanding issues for eutrophication in the UK as a whole and the inputs indicated for this assessment would make a negligible contribution to the overall loading for the Southern North Sea area.

5 References

BEEMS Technical Report TR193. Sizewell C H1 Assessment. Cefas, Lowestoft.
BEEMS Technical Report TR306 Sizewell - Marine Water and Sediment Quality Synthesis MSR2/4. Cefas, Lowestoft.
BEEMS Technical Report TR314. Supplementary Water Quality Report. Cefas, Lowestoft.
BEEMS Technical Report TR339 Sizewell Comprehensive Impingement Monitoring Programme 2009 – 2017. Cefas, Lowestoft.
BEEMS Technical Report TR348. Sizewell Benthic Ecology Characterisation Report. Cefas, Lowestoft.
BEEMS Technical Report TR381. Sizewell- Acoustic surveying of pelagic fish off Sizewell. Cefas, Lowestoft.
BEEMS Technical Report TR385 Modelling the effect of Sizewell C entrainment on the phytoplankton of Sizewell Bay. Cefas, Lowestoft.
BEEMS Technical Report TR406 Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft.
BEEMS Technical Report TR511. Particle Tracking Study of Impinged Sprat from the Proposed Sizewell C Fish Recovery and Return. Cefas, Lowestoft.
Dyer, K.R. (1979) Estuaries: A Physical Introduction. Wiley & Sons pp140.
Environment Agency. Catchment Data Explorer. Available from: Https://Environment.Data.Gov.Uk/Catchment-Planning/. Available from: https://environment.data.gov.uk/catchment-planning/. Accessed 2019.
Gende, S.M., Quinn, T.P., Willson, M.F., Heintz, R. and Scott, T.M. 2004. Magnitude and Fate of Salmon-Derived Nutrients and Energy in a Coastal Stream Ecosystem, Journal of Freshwater Ecology, 19:1, 149-160, DOI: 10.1080/02705060.2004.9664522
Gibb, N., Tillin, H., Pearce, B. and Tyler-Walters, H. Assessing the sensitivity of *Sabellaria spinulosa* reef biotopes to pressures associated with marine activities. 2014. JNCC Report No. 504. ISSN 0963 8901
Groenewold, S. and Fonds, M. 2000. Effects on benthic scavengers of discards and damaged benthos produced by the beam-trawl fishery in the southern North Sea. ICES Journal of Marine Science, 57 (5), 1395–1406.
Hull, T., Greenwood, N., Kaiser, J., and M. Johnson 2016 Uncertainty and sensitivity in optode-based shelf-sea net community production estimates Biogeosciences, 13, 943–959.
OSPAR, 1997 Comprehensive studies for the purpose of article 6 and 8.5 of DIR 91/271 EEC, The Urban Wastewater Treatment Directive. Second Edition. 10236 Research Report Collection.
Painting, S., Garcia, L., and Collingridge, K. 2017. Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area. UK National Report 2017. 205 pp.
Ramsay, K., Kaiser, M.J., Moore, P.G. and Hughes, R.N. 1997. Consumption of fisheries discards by benthic scavengers: utilization of energy subsidies in different marine habitats. Journal of Animal Ecology, 66, 884–896
Silva J. F., Ellis J. R. and Ayers R. A. 2013. Length-weight relationships of marine fish collected from around the British Isles. Sci. Ser. Tech. Rep., Cefas Lowestoft, 150: 109 pp..
Stigebrandt, A., 2001. FjordEnv – a water quality model for fjords and other inshore waters. Gothenburg University Report, Sweden, 41pp.
Storebakken, T. Shearer, K.D., and Roem, A.J. 2006. Growth, uptake and retention of nitrogen and phosphorus, and absorption of other minerals in Atlantic salmon Salmo salar fed diets with fish meal and soy-protein concentrate as the main sources of protein. Aquaculture Nutrition 6;103-108
Timm, M., Jorgensen, B.M. 2002. Simultaneous determination of ammonia, dimethylamine, trimethylamine and trimethylamine-n-oxide in fish extracts by capillary electrophoresis with indirect UV-detection. Food Chemistry 76 (2002) 509–518.
Turnpenny, A.W.H., O'Keeffe, N., 2005. Screening for intake and outfalls: a best practice guide. Environment Agency Science Report, SC030231. Environment Agency, Bristol.
Tyler-Walters, H., Tillin, H.M., d'Avack, E.A.S., Perry, F., Stamp, T., 2018. Marine Evidence-based Sensitivity Assessment (MarESA) – A Guide. Marine Life Information Network (MarLIN). Marine Biological Association of the UK, Plymouth, pp. 91. Available from https://www.marlin.ac.uk/publications
Walker, A.J.M. and Rees, E.I.S. 1980. Benthic ecology of Dublin Bay in relation to sludge dumping: fauna. Irish Fisheries Investigations Series, 22, 1-59.
Walker, M.J., Ellison, S., Burns, D.T., Gray, K. 2011. Nitrogen Factors for Atlantic Salmon, Salmo salar, farmed in Scotland and in Norway and for the derived ingredient, "Salmon Frame Mince", in Fish Products.

Journal of the Association of Public Analysts (Online) 2011 39 44-78 P Colwell et al.

Wells, E., Best, M., Scanlan, C., and Foden, J. 2010. Water Framework Directive development of classification tools for ecological assessment: Opportunistic Macroalgae Blooming. United Kingdom Technical Advisory Group (WFD-UKTAG). 66 pp.

Appendix A Nutrient chemicals potentially present in discharges during Hinkley Point C operation

A.1 Dissolved inorganic nitrogen and phosphorus inputs during operation

Source nutrient data for the operational discharge from the Sizewell C development including the FRR is based on information provided in EDF, 2014 and is summarised in **Table 4**.

Table 4 Summary of phosphate and nitrogen discharge loading during operational phase and including the FRR inputs based on annual average loadings without LVSE and expressed as a percentage relative to the daily exchange with the wider environment.

Substance	Daily loading during operation kg (annual loading kg) plus including FRR discharge	Peak 24 hr load during operation kg d^{-1}	Peak loading in construction (2 EPR units + sewage) per day kg.	Daily exchange with wider environment, kg	% of daily loading in operation	% max possible daily loading in operation	Fraction of maximum possible daily loading in construction
Nitrogen (as N) Including FRR	32 (11725) 45.4 (16571)[3]	332[1]	17.3[2]	16900 (as N)	0.2% 0.27%	1.9%	0.1%
Phosphates as P Including FRR	0.71 (257) 2.62 (956.3)[4]	115[1]	28.2[2]	2440	0.03% 0.11%	5%	1.1%

1 Extracted from BEEMS TR193 (operational loadings as N and PO_4 loading converted to P). 2 The Phosphate as P load values are derived from groundwater, sewage and EPR commissioning inputs as described in BEEMS TR193 section on 'Assessment of construction discharge'. 3 The FRR 95th percentile biomass loading would contribute 21.03 kg nitrogen and a total with operational input of 53.03 kg. 4 The FRR 95th percentile biomass loading would contribute 3 kg Phosphate as P and a total with operational input of 3.33 kg.

Appendix B Calculation of moribund biomass from the FRR and potential contribution to nutrients,

B.1 Calculation of moribund biomass of fish discharged from the Fish Recovery and Return system

This section provides the supporting data and calculation used to derive the values for the FRR assessment. The values are the outputs of the modelling approach used to estimate impingement at SZB and are the same values that form the basis of the final impingement estimates. In TR339 V2, only the final annual estimates for SZB are given. Since completion of this report an error in subsampling has been corrected and this has led to minor increases in annual impingement predictions at Sizewell B (TR339 V3). Mean impingement weight increases from 201.3 tonnes to 203.2 tonnes per annum and this represents a 1% increase and is considered non material to the assessments based on these values. Results here are also based on 95th percentiles impingement weights and so provide a more precautionary assessment.

Mean daily estimated numbers impinged were modelled using data collected during onshore impingement sampling between 2009 and 2017 at SZB (**Table 5**). Full details of the modelling approach are given in BEEMS Technical Report TR339. The mean estimated daily values for SZB were raised to predicted daily values for SZC (**Table 6**) on the ratio of the pumping capacities of the two stations (i.e. the SZB daily mean values were multiplied by (131.86/51.5 cumecs). Next, survival through the FRR was considered, as some of the more robust species will not all die during their passage through the cooling water systems (**Table 7**). In this case, the same values of FRR survival that were used to predict impingement at SZC were used (BEEMS Technical Report TR406). The resulting numbers lost to impingement were converted to weight using the mean length of each species in the impingement samples and published length-weight relationships (**Table 8, no LVSE in place**). The size distributions used are shown in BEEMS Technical Report TR339. The length-weight relationships used were taken from Silva et al (2013). Comparison of values in **Table 8** and **Table 9** show the reduction in the weight of fish impinged for each species without and with the LVSE fitted (LVSE factor 0.357 multiplier for values in Table 8 to derive reductions in **Table 9**). **Table 10** shows the mean daily weight of fish impinged for Sizewell C based on annual estimates.

Table 5 Sizewell B mean quarterly estimated number of fish impinged

Species	Q1 Mean	Q1 L95	Q1 U95	Q2 Mean	Q2 L95	Q2 U95	Q3 Mean	Q3 L95	Q3 U95	Q4 Mean	Q4 L95	Q4 U95
Sprat	2322303	1475929	3251790	77749	61334	97268	60966	46961	77292	238783	145548	365627
Herring	585301	377679	814024	132915	66740	211419	81209	48329	118910	217438	103043	372106
Whiting	322920	246442	417006	79694	50599	110754	65819	46235	87255	181231	141701	226871
Bass	210347	162709	257939	1058	580	1629	318	171	491	54204	29233	82361
Sand gobies	15302	11591	19316	3153	2069	4507	74130	31287	131161	102540	67596	166084
Sole	13835	7913	20928	44714	34827	55071	21843	15556	28521	11253	8477	14121
Dab	18505	14854	23267	3048	2015	4189	17785	6951	30218	16666	13354	20338
Anchovy	8	2	15	55900	19624	91906	12589	6884	19683	87	35	148
Thin-lipped grey mullet	35704	25438	46679	131	0	317	0	0	0	13176	3186	26167

Table 6 Sizewell C mean quarterly estimated number of fish impinged

Species	Q1 Mean	Q1 L95	Q1 U95	Q2 Mean	Q2 L95	Q2 U95	Q3 Mean	Q3 L95	Q3 U95	Q4 Mean	Q4 L95	Q4 U95
Sprat	5945998	3778951	8325844	199068	157040	249043	156098	120239	197898	611377	372660	5945998
Herring	1498598	967005	2084217	340315	170882	541316	207927	123741	304455	556725	263830	1498598
Whiting	826802	630988	1067698	204049	129554	283572	168521	118380	223406	464021	362809	826802
Bass	538570	416599	660424	2709	1484	4170	815	438	1256	138784	74848	538570
Sand gobies	39179	29677	49457	8073	5297	11539	189801	80107	335822	262543	173073	39179
Sole	35424	20261	53585	114485	89170	141002	55926	39829	73025	28812	21705	35424
Dab	47380	38033	59572	7804	5159	10726	45538	17797	77370	42672	34190	47380
Anchovy	20	4	39	143126	50246	235315	32232	17625	50396	224	91	20
Thin-lipped grey mullet	91415	65130	119516	336	0	812	0	0	0	33735	8157	91415

Table 7 Sizewell C mean quarterly estimated number of fish impinged taking account of species mortalities with FRR fitted[1]

Species	Q1 Mean	Q1 L95	Q1 U95	Q2 Mean	Q2 L95	Q2 U95	Q3 Mean	Q3 L95	Q3 U95	Q4 Mean	Q4 L95	Q4 U95
Sprat	5945998	3778951	8325844	199068	157040	249043	156098	120239	197898	611377	372660	936146
Herring	1498598	967005	2084217	340315	170882	541316	207927	123741	304455	556725	263830	952737
Whiting	418362	319280	540255	103249	65554	143488	85272	59900	113043	234795	183582	293924
Bass	272516	210799	334175	1371	751	2110	412	222	636	70225	37873	106703
Sand gobies	8071	6114	10188	1663	1091	2377	39099	16502	69179	54084	35653	87599
Sole	7297	4174	11039	23584	18369	29046	11521	8205	15043	5935	4471	7448
Dab	9760	7835	12272	1608	1063	2209	9381	3666	15938	8790	7043	10727
Anchovy	20	4	39	143126	50246	235315	32232	17625	50396	224	91	379
Thin-lipped grey mullet	46256	32956	60475	170	0	411	0	0	0	17070	4127	33901

1: Based on BEEMS TR406 Species specific mortalities used to derive fish numbers impinged with FRR fitted: Sprat 100%; Herring 100%; Whiting 50.6%; Bass 50.6%; Sand gobies 20.6%; Sole 20.6%; Dab 20.6%; Anchovy 100%; Thin-lipped grey mullet 50.6%

Table 8 Sizewell C mean daily estimated weight (kg) of fish impinged with an FRR fitted

Species	mean weight at length (kg)	Q1 Mean	Q1 L95	Q1 U95	Q2 Mean	Q2 L95	Q2 U95	Q3 Mean	Q3 L95	Q3 U95	Q4 Mean	Q4 L95	Q4 U95
Sprat	0.009	614	390	860	21	16	26	16	12	20	63	39	97
Herring	0.095	1553	1002	2160	353	177	561	215	128	316	577	273	987
Whiting	0.050	230	175	297	57	36	79	47	33	62	129	101	161
Bass	0.095	284	220	348	1	1	2	0	0	1	73	39	111
Sand gobies	0.001	0	0	0	0	0	0	1	0	1	1	1	1
Sole	0.015	1	1	2	4	3	5	2	1	2	1	1	1
Dab	0.018	2	2	2	0	0	0	2	1	3	2	1	2
Anchovy	0.008	0	0	0	13	5	21	3	2	5	0	0	0
Thin-lipped grey mullet	0.041	21	15	27	0	0	0	0	0	0	8	2	15
Total daily biomass kg		614	390	860	21	16	26	16	12	20	63	39	97

Table 9 Sizewell C mean daily estimated weight of fish impinged (kg) per quarter– including Low Velocity Side Entry head and with an FRR fitted (LVSE factor 0.357)

Species	Q1 Mean	Q1 L95	Q1 U95	Q2 Mean	Q2 L95	Q2 U95	Q3 Mean	Q3 L95	Q3 U95	Q4 Mean	Q4 L95	Q4 U95
Sprat	219.313	139.383	307.091	7.342	5.792	9.186	5.758	4.435	7.299	22.550	13.745	34.529
Herring	554.452	357.773	771.120	125.910	63.223	200.276	76.929	45.782	112.642	205.977	97.612	352.494
Whiting	82.055	62.622	105.962	20.251	12.857	28.143	16.725	11.748	22.172	46.051	36.006	57.648
Bass	101.420	78.451	124.367	0.510	0.279	0.785	0.153	0.082	0.237	26.135	14.095	39.711
Sand gobies	0.046	0.035	0.058	0.010	0.006	0.014	0.224	0.095	0.397	0.310	0.205	0.503
Sole	0.418	0.239	0.633	1.351	1.053	1.664	0.660	0.470	0.862	0.340	0.256	0.427
Dab	0.699	0.561	0.879	0.115	0.076	0.158	0.672	0.262	1.141	0.629	0.504	0.768
Anchovy	0.001	0.000	0.001	4.596	1.613	7.556	1.035	0.566	1.618	0.007	0.003	0.012
Thin-lipped grey mullet	7.354	5.240	9.615	0.027	0.000	0.065	0.000	0.000	0.000	2.714	0.656	5.390
Total daily biomass kg	965.758	644.304	**1319.7**	160.112	84.900	247.848	102.156	63.441	146.368	304.714	163.083	491.482

[1]Q1 95th percentile daily mean biomass value and this is used for calculation (1319.7 kg a day).

Table 10 Sizewell C mean daily weight of fish impinged (kg) based on annual estimates – with and without various mitigations (NM- no mitigation, FRR- mitigation, FRR+LVSE- mitigation.

Species	NM Mean	NM L95	NM U95	FRR Mean	FRR L95	FRR U95	FRR+LVSE Mean	FRR+LVSE L95	FRR+LVSE U95
Sprat	177.96	92.99	299.90	177.96	92.99	299.90	63.53	33.20	107.07
Herring	674.27	394.33	1028.75	674.27	394.33	1028.75	240.71	140.78	367.26
Whiting	228.02	158.03	311.76	115.38	79.96	157.75	41.19	28.55	56.32
Bass	177.90	81.16	329.60	90.02	41.07	166.78	32.14	14.66	59.54
Sand gobies	2.02	0.82	3.95	0.42	0.17	0.81	0.15	0.06	0.29
Sole	9.38	6.52	12.92	1.93	1.34	2.66	0.69	0.48	0.95
Dab	7.19	4.35	11.95	1.48	0.90	2.46	0.53	0.32	0.88
Anchovy	3.98	1.04	9.57	3.98	1.04	9.57	1.42	0.37	3.41
Thin-lipped grey mullet	14.09	3.71	27.83	7.13	1.88	14.08	2.55	0.67	5.03
Total daily biomass kg	1294.82	742.96	2036.22	1072.56	613.68	1682.77	**382.91**	219.08	600.75

B.2 Calculation of dead biomass discharged from the Sizewell C FRR system potential contribution to nutrient input, influence on dissolved oxygen, un-ionised ammonia, and organic enrichment

The total biomass of moribund biota that potentially may be discharged from the FRR has been estimated based on the level of abstraction (pump rates) for the planned Sizewell C intakes and the information on seasonal distribution of species and length weight distribution of the species impinged at the existing Sizewell B (BEEMS TR339 and TR381). The derived Sizewell C data indicate that the highest biomass discharged occurs during the winter months of Quarter 1 and the 95th percentile value for this period is used to provide a conservative assessment for the contribution of un-ionised ammonia and oxygen concentration reduction. For the assessment of additional nutrients from decaying biomass the mean annual daily biomass 382.91 kg per day (with LVSE in place, **Table 10**) as well as the more conservative 95th percentile daily biomass 600.75 kg per day (with LVSE, **Table 10**) is used to derive the estimates of tissue concentration for nitrogen and phosphorus from several studies and are shown in **Table 11**.

Table 11 Phosphorus and nitrogen concentration data for fish tissue derived from several studies and which is used in calculation of potential nutrient loadings released during decay of dead fish released from the FRR

Nutrient	g/kg	Percentage (wet weight)	Average Daily biomass (382.91 kg) nutrient content (January -December)	95th Percentile Daily biomass (600.75 kg) nutrient content (January - December)	Literature Source
P content	3.74-4.7 (ww)	0.47	(382.91*0.47)/1000 =1.8 kg	(600.75*0.47)/1000 =2.82 kg	Storrebakken et al., 2000
P content		0.45-0.5	1.91 kg	3.00 kg	Gende et. al., 2004
N content		3.2-3.5	13.40 kg	21.03 kg	Walker et. al., 2011
N content		3.4	13.02 kg	20.43 kg	Gende et. al., 2004

BOD

For BOD calculation the Quarter 1 95th percentile daily biomass value is used to provide the most conservative assessment = 1319.7 kg biomass (this is the value with LVSE, **Table 9** Q1 95th percentile)
BOD generated from this biomass is estimated based on an equivalent value of 3.5g BOD/kg dry weight of fish tissue (Stigebrandt *et al*., 2001).
The estimate of BOD load per day is:

(i) 1319.7 x (3.5 x dry weight/wet weight conversion 0.36)= 1662.86 kg BOD

Total oxygen reduction potential based on OSPAR information BOD of 1.5mgl^{-1} = 0.5mgl^{-1} oxygen reduction:

(i) 1662.86/1.5=554.29 kg/ day O_2 reduction
(ii) Based on a background dissolved oxygen level 6.96mg/l this is equivalent to oxygen present in 79639 m^3.
(iii) The daily volume exchange of 10% of GSB = 36,380,000 m^3
(iv) 79639/36,380,000=0.22% of daily exchange
(v) Also, daily reaeration contributes 3.2gm^{-2}d^{-1}

Therefore, reaeration over 554.29/0.0032= 173,214 m^{-2} (17.32 ha) would meet this daily oxygen demand.

The lowest background dissolved oxygen for Sizewell Bay under thermal influence as reported in BEEMS TR306 was 5.77 mg l^{-1} which would still be equivalent to Good status. To reduce status of a water body under thermal influence (5.77 mg l^{-1}) from Good to Moderate (4 mg l^{-1}) would require a reduction of 1.77 mg l^{-1} oxygen. The oxygen demand calculated for Q1 95th percentile biomass loading with LVSE (1319.7 kg) is 554.29 kg O_2 and this demand could reduce oxygen levels by 1.77 mg l^{-1} in a volume of (554.29/0.00177) = 313155.5 litres or an area of (square route (313155.5/ 4 metres depth)= (280 x 280 metres) or 7.83 ha.

Un-ionised ammonia

The highest biomass discharged is predicted for Q1 95th percentile (1319.72 kg with LVSE). Based on a value of 125 mg of ammonia nitrogen per kg of cod tissue (Timm and Jorgensen, 2002) this loading is equivalent to NH_4-N of 1319.72 x 125= 164965.85 mg NH_4-N. The total NH_4 loading generated when used as a source value in the un-ionised ammonia calculator with an average temperature for Q1 of 11.73°C and 5th percentile salinity and 50th percentile pH= 5,198,601µg NH_3-N. Taking into account an un-ionised ammonia background of 1.6 µg l^{-1} this loading of un-ionised ammonia is equivalent to a seawater volume of 159,000 litres at the EQS (21 µg l^{-1} NH_3-N). A volume of 159,000 litres at a depth of 4 metres (FRR discharge) would be equivalent to an area of 6.31 m x 6.31 m=39.82 m^2 (0.004 ha at the EQS).

Temperature elevation of 2°C that might occur around the FRR due to the added thermal influence of Sizewell C will elevate the proportion of un-ionised ammonia relative to the influence of natural background temperatures alone (which take account of Sizewell B inputs). Taking account of conditions that would maximise the un-ionised ammonia concentration a 2°C elevation above March natural background temperature at Sizewell (11.73 +2°C), a lower salinity 31.7 (5th percentile) and high pH 8.23 (95th percentile) the volume of seawater affected by elevated un-ionised ammonia would increase to 281,000 litres with LVSE in place (an area equivalent to 0.12 ha) and 281,000 litres with no LVSE in place (an area equivalent to 0.023 ha)

Organic enrichment

The area potentially effected by organic enrichment resulting from the decomposition of moribund fish biomass was determined relative to a benchmark of 100g organic carbon/m^2/year (Tyler-Walters *et. al.*, 2018). The carbon content of fish biomass was derived from results reported by Alves *et. al.*, 2019 that the carbon content of fish processing waste was 64.7% of the dry weight and the wet weight to dry weight conversion factor was 0.48.

Area calculations were as follows:

(i) Peak biomass of fish discharged from FRR **(LVSE) was in** Q1 95th percentile at 1319.73 kg per day
(ii) 1319.73 kg converted to dry weight and then to weight of carbon (1319.73 x 0.48) x 0.65 = 411.75 kg
(iii) Daily carbon load is divided by daily benchmark carbon as referenced above and converted to a daily value 411.75/(0.1kg/365 days)= 1,502,905 m^2 affected
(iv) Convert metres to hectares and square kilometre = **150.29 ha or 1.50 km^2**.
(v) With no **LVSE fitted** Q1 95th percentile value is 1363.39 kg per day (organic carbon)=**497.64 ha or 4.98 km^2**.
(vi) For **annual average** biomass fish FRR **(no LVSE** 95th percentile**)** 2036.22 kg per day converted to 635.30 kg organic carbon=**231.88 ha or 2.32 km^2**.
(vii) For **annual average** biomass fish FRR with **LVSE** 95th percentile 600.75 kg per day converted to 187.43 kg organic carbon=**68.41 ha or 0.68 km^2**.

B.3 Habitat Effect footprint for organic enrichment due to discharges from the FRR when LVSE is not included in the assessment.

Table 12 Areas of intersection of the FRR effect footprint for Q1 95th percentile without LVSE included in terms of organic matter enrichment and lower sensitivity EUNIS habitats in Sizewell Bay.

EUNIS Habitat codes	Area in Sizewell Bay (ha)	Area in WFD (ha)	FRR effect footprint area of intersection (ha)	Percentage EUNIS habitat in WFD
A5.13	677.55	629.99	60.55	9.61
A5.23	2208.90	1362.05	321.28	23.59
A5.23/A5.26	1490.57	392.48	2.73	0.07
Engineering Feature	1.11	1.11	1.11	100
A 5.22	402.88	340.17	9.72	1.54

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP099

NOT PROTECTIVELY MARKED

SPP099 - Predicted performance of the SZC LVSE intake heads compared with the SZB intakes

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Executive summary

The proposed Sizewell C (SZC) cooling water system would be fitted with four low velocity side entry (LVSE) intake heads mounted on the seabed approximately 3 km offshore. The proposed intake heads would be capped structures with the intake surfaces orthogonal to the direction of the tidal flows. These state-of-the-art intakes are specifically designed to reduce the cross-sectional area available to intercept any fish being transported in the tidal flows. The reduction in cross-sectional area combined with their low intake velocity is predicted to substantially reduce the number of fish abstracted per cumec (cubic metre per second) of seawater compared with Sizewell B (SZB) which is fitted with two capped omnidirectional intake heads with intake velocities that exceed the ability of most fish to avoid being abstracted.

The results of preliminary modelling of the SZC LVSE intakes are described in BEEMS Technical Report TR406 v6. These results indicated that the predicted ratio of the SZC to SZB intake intercept areas was 0.383 which implied that SZC would abstract 38.3% of the fish and crustacea per cumec that SZB currently abstracts. The methodology used to calculate the intercept areas of the LVSE intakes in TR406 was based upon that proposed by the Environment Agency (Environment Agency 2019) but was not fully compliant as the method was still being evaluated by Cefas at the time.

This modelling work has since been further refined and the methodology used in this report (BEEMS Scientific Position Paper SPP099) is now fully compliant with Environment Agency (2019). The existing SZB intakes are omnidirectional and therefore the directional analysis has had to be removed from of the Environment Agency methodology, but the resulting algorithm is still compliant with the remaining principles of the method; in particular it incorporates the same treatment of slack water effects.

Consultee comments on SPP099 v2 requested that the methodology should be extended to capture the full range of tidal directions likely to occur in the real world. The Environment Agency methodology uses hydrodynamic data (current speeds and directions) from a validated 3D hydrodynamic model of Sizewell Bay. This model is forced by European Centre for Medium-Range Weather Forecasts meteorological data but the temporal and spatial resolution of those data means that the model is not able to fully reproduce the high frequency variability caused by local meteorology. This lack of high frequency components would reduce the spread of the estimated tidal directions compared to the real world. To evaluate this effect, the GETM model outputs used to derive the intake intercept areas were replaced by Acoustic Doppler Current Profiler (ADCP) measurement data from the locations of the proposed SZC northern and southern intakes measured at a height of approximately 2 m above the sea bed i.e. slightly below the centre of the proposed SZC intake faces at 2.5m above the seabed. The tidal velocity is slightly lower and the tidal directional spread is slightly wider at 2m than 2.5m and the use of these ADCP data therefore produced over estimated intercept areas. No other changes were made to the methodology.

As expected the spread in measured tidal direction was larger with ADCP measurements than from the model. Part of this will have been due to measurement noise, but the majority is considered to be due to the instruments ability to capture high frequency events.

This paper describes how the Environment Agency methodology has been applied and the parameterisation of the modelling. The Sizewell project will use the results of this paper (SPP099 v3) to update the results in TR406. A spreadsheet is available separately that details the calculations (Appendix A).

The results of modelling the SZC LVSE intakes in the ANSYS computational fluid dynamics (CFD) software are also presented.

Conclusions

1.LVSE intake velocities

CFD modelling of the proposed LVSE intakes at Sizewell shows that:

i. The hydrodynamic influence of the LVSE intakes is only felt over a short distance from the structure with intake velocities falling to 0.1 m s^{-1} within approximately 1.5 to 2.5 m from the intake surfaces and to 0.05 m s^{-1} at a range of approximately 3m from the intake surfaces over the full tidal cycle;

ii. The range from the intake surface for the inward velocity to fall to the Environment Agency's required figure of 0.3 m s^{-1} is 0.0 to 0.68 m dependent upon the tidal current speed (faster tidal velocities have the effect of increasing the intake velocity). Over the tidal cycle the median distance from the intake where the inward velocity criterion of 0.3 m s^{-1} is met is 0.38 m (based upon the distribution of tidal currents at the SZC intakes).

2. Ratio of SZC to SZB intake intercept areas

i. Using the methodology proposed by Environment Agency (2019) to calculate the intercept area of the SZC LVSE intakes but with ADCP data rather than model data, the ratio of the intercept cross-sectional areas of the proposed four SZC LVSE intakes compared with the two omnidirectional SZB intakes is 0.357 as a best estimate with values of 0.371 and 0.363 at the 5 percentile and 95 percentile head misalignment values respectively at a slack tide threshold of 0.2 m s^{-1}. This means that SZC fitted with LVSE intakes is expected to abstract 35.7% of the fish and crustacea per cumec that SZB does.

ii. This ratio is smaller than that provisionally estimated in BEEMS Technical Report TR406 but confidence in the result is much higher because it:

   a. has now been calculated fully in line with the methodology in Environment Agency (2019);
   b. it has been subject to an extended sensitivity analysis that incorporates measured as opposed to only model data;
   c. benefits from much more realistic modelling of the SZB intake heads; in particular how their intercept area varies during the tidal cycle.

3. Alignment of the LVSE intakes at Sizewell.

The SZC LVSE inatkes will be aligned in the field by the use of ADCP survey data. Such surveys can determine the tidal axis to a directional accuracy in the range ±1 to ± 2 degrees dependent upon whether the instrument is individually calibrated or the manufacturer's worst case calibration is used. This measurement accuracy, together with a realistic estimate of the accuracy that the heads can be placed on the seabed (± 1 degree) has been built into the sensitivity analyses presented in this report.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture science (Cefas) supported by a network of subcontractors has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

SZC would need to abstract approximately 132 cumecs ($m^3 s^{-1}$) compared with approximately 51.5 cumecs for the existing SZB. SZB was commissioned in 1995 and is currently the most modern operational nuclear power station in the UK. As a first in the UK, SZB was specifically designed to reduce the losses of impinged fish and crustacea and incorporated:

a. omnidirectional intakes mounted in deeper water than at SZA in order to reduce abstraction of juvenile flatfish that use shallow water near to the Sizewell coast as a nursery area;
b. capped intake heads which reduce vertical velocities that fish are ill equipped to avoid. Previous studies in the USA had demonstrated that capped intakes could reduce pelagic fish abstraction by up to 85%;
c. intake surfaces mounted off the seabed to minimise abstraction of benthic fish and organisms; and
d. an early, rudimentary example of a Fish Recovery and Return (FRR) system designed to return as many abstracted fish as possible safely back to sea. The SZB system was focussed on reducing the mortality of juvenile sole, an important commercial species in the southern North Sea.

Trials showed that these measures successfully reduced losses of fish and crustacea per cumec abstracted compared with SZA (Turnpenny and Taylor (2000)). The SZB intake designs were state-of-the-art in 1995 and whilst representing a substantial improvement over those of SZA, their design can be further improved to reduce fish abstraction. In particular, the omnidirectional design of the SZB intakes means that whilst at slack water the intake velocity is approximately 0.23 m s^{-1}, at peak tidal velocity the intake velocity rises to over 1 m s^{-1} which is beyond the ability of nearly all fish to avoid. Environment Agency guidance (Environment Agency 2005; 2010) states that “for most power plant intake purposes a design fish-escape velocity (i.e. intake velocity) of 0.3 m s^{-1} will be suitable and meet best requirements”.

The SZC LVSE intake heads have been designed to substantially reduce the abstraction of fish and crustacea per cumec compared with SZB and similar LVSE intakes are currently being manufactured for installation at Hinkley Point C (HPC) which is under construction. The HPC and SZC LVSE designs are the first examples of this technology to be constructed worldwide as far as EDF Energy is aware. These very large and highly engineered intake structures are designed to minimise impingement by:

a. limiting the exposure of the intake surfaces to the tidal stream and in so-doing reduce the risk of impingement for fish swimming with the tidal stream. i.e. they reduce the cross-sectional area of the intake presented to the prevailing tidal directions by mounting the head orthogonally to the tidal flow;
b. reducing intake velocities into the head to a target velocity of 0.3 m s^{-1} over as much of the length of the intake surface as practical during all tidal states in order to maximise the possibility of most fish avoiding abstraction; and
c. in common with the SZB design, reducing vertical velocities which fish are ill equipped to resist by means of velocity caps on the intakes.

In addition, SZC would feature other key measures to reduce the risk to fish and crustacea:

- the intakes will be situated in much deeper water than SZB that would mean that fish swimming near to the surface would be largely invulnerable to abstraction. For energetic reasons many fish use tidal stream transport to migrate in surface waters and these fish would be at much lower risk of abstraction than at SZB;
- an advanced FRR, designed from the outset to reduce fish mortality for a wider range of species than SZB, particularly for conservation species such as eel and lamprey; and
- an anti-biofouling chlorination policy whereby chlorination will only occur downstream of the filtration systems and hence not within the FRR system. This would reduce mortality for demersal species such as bass, whiting and cod compared with SZB where the intakes and intake tunnels are chlorinated all year round.

All of these measures are expected to reduce losses of fish and crustacea but this paper focusses on the predicted performance of the LVSE intakes compared to the omnidirectional intake heads at SZB in reducing the abstraction of biota per cumec abstracted.

LVSE intakes have the key advantages over other potential mitigation measures that they are expected to minimise the abstraction of biota in the first place, they are predicted to work for all fish species and they are entirely passive and therefore not prone to failure due to electrical or mechanical breakdowns.

2 Description of the SZB and proposed SZC intake heads.

2.1 SZB Intakes

The intake structure consists of two intake heads located on a line at approximately 90 degrees to the tidal flow. Each head is octagonal, approximately 11.5 m across and is omnidirectional. The structure sits ~1.5 m above the seabed and the intake aperture is 3 m high. The tidal flows at Sizewell are highly rectilinear and peak at 1 m s^{-1} on spring tides. During peak tides, a tidal stream line of approximately the width of the intake enters the intake, whereas at slack water the water is drawn from all round around the intake.

Each of the 8 intake faces on each head has a face width of 4.76 m and a height of 3m i.e. the area of each face is 14.29 m^2. The total intake surface area per head is therefore 114.32 m^2 and for the two heads the intake surface area is 228.65 m^2.

2.2 SZC intakes

The proposed SZC intakes will be low velocity side entry (LVSE) structures. A total of four LVSE heads will be installed with two heads fitted on each of the two intake tunnels (Table 1). The intake heads will be located approximately 3 km offshore, on the eastern side of the Sizewell-Dunwich Bank in approximately 15m (ODN) depth (Figure 1). The detailed design of the intake heads is not yet complete but the performance of the design with the worst hydrodynamic performance has been used for this assessment in order to envelope worst case modelled performance. This design has intake surfaces of the same size as those being constructed for Hinkley Point C (HPC) with the bottom of the intake surfaces being at approximately 1.5 m off the seabed (as at HPC) in order to reduce the abstraction of benthic organisms.

The following LVSE head dimensions have been modelled:

- Each intake surface is 35.5 m long and 2m high with the middle of the intake face 2.5 m off the sea bed.
- Area per intake face = 71 m^2, 2 faces/head=142 m^2 per head. With 4 heads the total surface area = 568 m^2.
- When the tide is flowing only 4 intake faces are exposed to the tide i.e. intake exposed surface area = 284 m^2.

Table 1 Coordinates of the SZB and proposed SZC cooling water intake and outfall heads

	Latitude WGS84 (degrees N)	Longitude WGS84 (degrees E)	Easting BNG (m)	Northing BNG (m)	Depth ODN (m)
Sizewell B					
IB	52.21472	1.63332	648297	263612	9.0
OB	52.21525	1.62658	647834	263647	5.1
Sizewell C					
I3a	52.21948	1. 66931	650726	264262	12.9
I3b	52.21945	1. 67077	650826	264264	13.6
I4a	52.21148	1. 66572	650526	263360	11.5
I4b	52.21126	1. 66714	650624	263341	13.5
O9a	52.21807	1.67435	651080	264125	16.9
O9b	52.21803	1.67544	651155	264125	16.8
WGS84: World Geodetic system 1984, BNG: British National Grid, ODN: Ordnance Datum Newlyn					

Notes:

1. IB is at the centre of the two SZB intakes, I3b and I4b are the SZC intake locations modelled in this report. OB is the SZB outfall and O9a and O9b are the two proposed SZC outfall locations.
2. I3b and I4b are shown in Figure 1 as OS22 and OS12

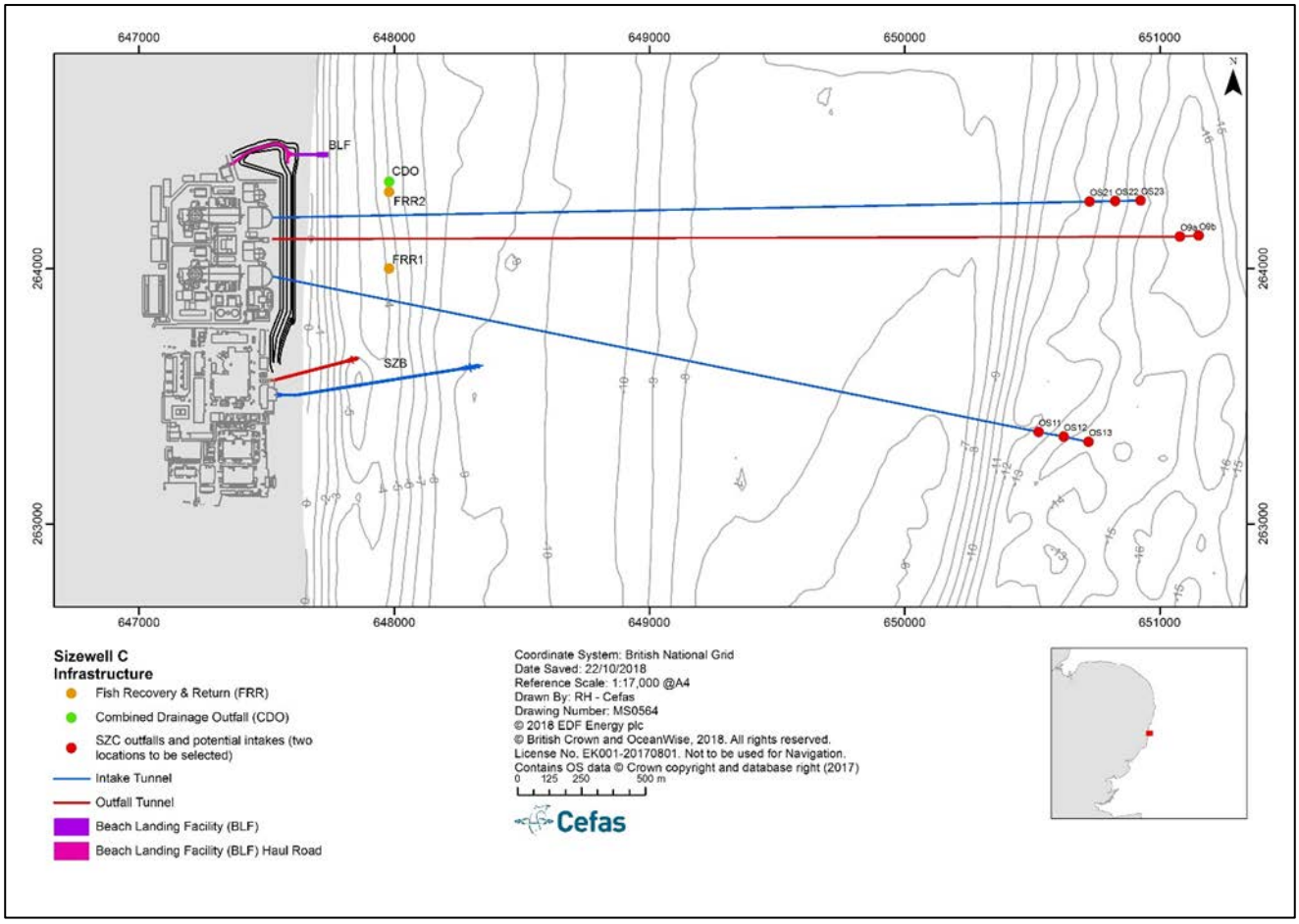

Figure 1 Location of the SZB and proposed SZC cooling water intake and outfall heads

2.3 Hydrodynamic considerations

2.3.1 Intake head performance

At slack water, both the SZB and SZC intakes will abstract from all intake faces equally. Fish will not be transported to the heads by the tidal current and therefore only fish local to the heads or which swim into the abstraction zone (a zone parallel and close to the intake surfaces) will be abstracted. At slack water the intake velocities of both the SZC and SZB intakes is designed to be approximately 0.23 m s^{-1}.

In theory, with the SZC intakes orthogonal to the tidal flow, the current will flow past the SZC intake surfaces and only fish that are in an abstraction risk zone will be likely to be abstracted. In practice, the size of this abstraction zone (i.e. the distance from the intake surface) will depend on the effects of turbulence when the tidal current interacts with the intake head. To this end, the SZC intakes are designed to linearise flow velocities along the head and to reduce the formation of initial eddies by the use of shaped wedges at each end of the intake. The intake velocity of the SZC intakes will not increase with the tidal flow if the intakes are perfectly aligned to the tidal flow. In practice there is a spread around the modal directions of the tidal flow and there is also some tidal asymmetry, i.e. the tidal flow directions are not exactly 180 degrees apart, and so one side of the SZC intakes will be exposed to the tide for part of each tidal cycle. This will increase the abstraction cross-sectional area of the intakes and serve to increase intake velocities.

For HPC studies, an initial rule of thumb assumption was made that the abstraction risk zone extended approximately 2 m from the head based upon physical modelling studies on the HPC LVSE heads (HRW,

2013). This would give an abstraction cross-sectional area of 2 m × 2 m height per intake face, 8 m^2 per head or a total of 32 m^2 for the four intakes designed for HPC. It has been suggested that this assumption is overly simplistic and, in particular, ignores effects at slack water when the intakes would have a larger cross-sectional area, and an alternative methodology to calculate the cross-sectional area of the LVSE intakes has been produced (Environment Agency, 2019). This methodology takes account of slack water effects and tidal spread and asymmetry on the LVSE head performance. We have adopted the Environment Agency (2019) methodology in this paper.

The SZB intakes are omnidirectional and at slack water the intake cross sectional area is 228.65 m^2. As the tidal velocity increases, water abstraction will increasingly take place on those intake surfaces facing into the tidal flow and the intake velocity on the surface facing most directly into the flow will increase to the approximate tidal current speed. In theory, at peak tidal flow the intake cross-sectional area of the SZB intakes would be expected to reduce to the physical dimensions of the head facing into the tidal flow i.e. 11.5 m wide × 3 m depth per head = 34.5 m^2 per head or a total of 69 m^2 for both heads.

2.3.2 What is the definition of slack water

As velocity increases from 0 m s^{-1}, the behaviour of the intakes will change from omni directional to preferentially abstracting from intake surfaces facing into the tidal current. In marine hydrodynamics, slack water is often characterised as velocities less than or equal to 0.1 m s^{-1}. Environment Agency (2019) considered the effects on the LVSE performance using a slack water definition of 0.1 to 0.3m s^{-1}. This point is examined in more detail in section 4.1.

3 SZC LVSE Intakes – predicted intake velocities

The proposed SZC LVSE intakes were modelled using ANSYS computational fluid dynamics (CFD) software to determine the distribution of intake velocities across the intake surfaces in tidal currents ranging from 0 m s^{-1} to 1.0 m s^{-1} (which are representative of those experienced at the proposed locations of the SZC intake heads).

Figure 2 shows that the distance from the intake surface where the inward velocity (the velocity into the intake surfaces) falls to 0.3 m s^{-1} is 0.0 m to 0.68 m. Over the tidal cycle the median range to 0.3 m s^{-1} is 0.38 m based upon the modelled distribution of tidal currents at the SZC intakes.

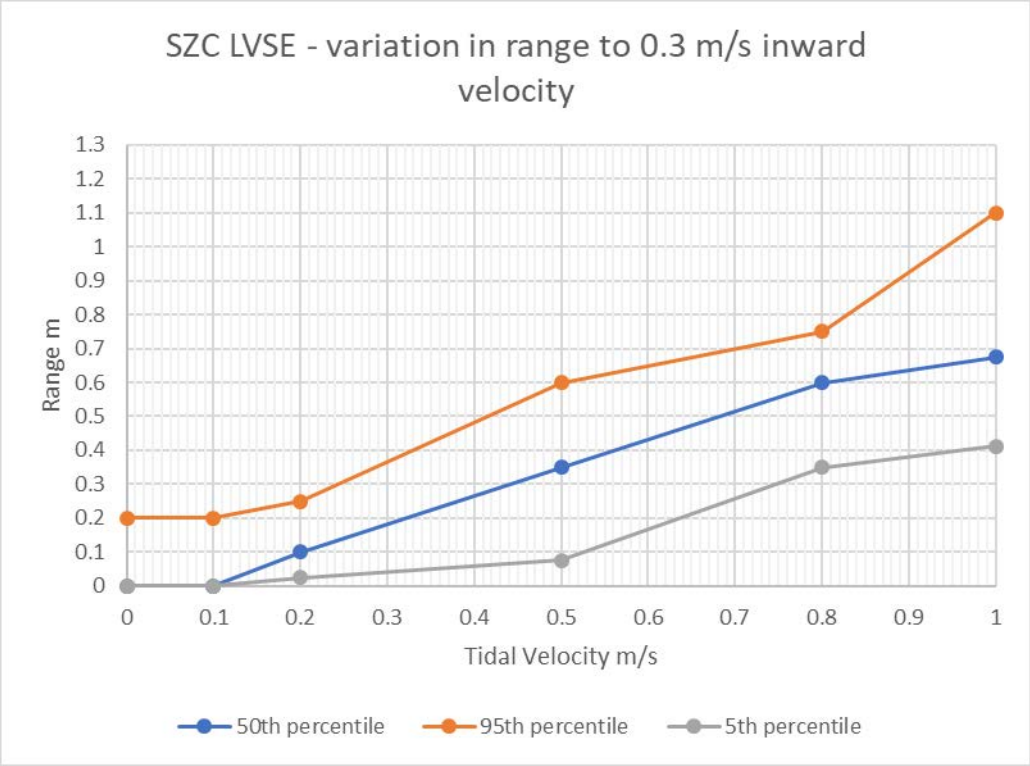

Figure 2 Modelled range to 0.3 m/s inward velocity at different tidal current speeds

The influence of the LVSE intakes is only felt over a short distance with inward velocities falling to 0.1 m s^{-1} over a range of approximately 1.5 m to 2.5 m from the intake surfaces and further to 0.05 m s^{-1} at a distance of approximately 3 m from the intake surfaces over the full tidal cycle (Figure 3).

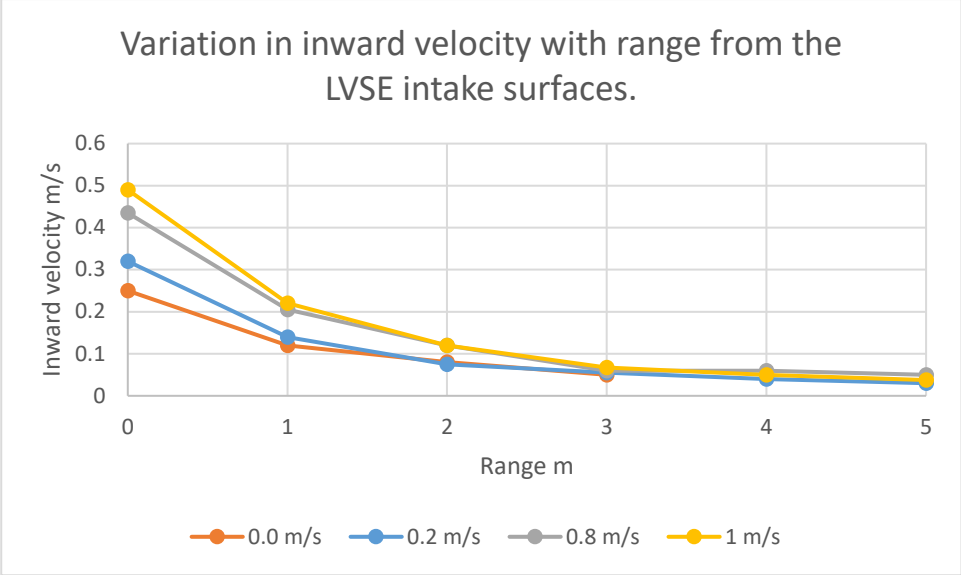

Figure 3 Variation in inward velocity with distance from the LVSE intake surfaces with tidal current speed

4 Methodology used to predict the intake head intercept cross-sectional areas.

4.1 Sizewell C LVSE intakes

As described in section 2.3.1, we have used the methodology described in Environment Agency (2019) to calculate the intercept cross-sectional area of the four SZC LVSE intakes. The data source for the required tidal current speed and directions in this method was the validated GETM 3D model of Sizewell Bay. In subsequent comments the Environment Agency requested that that the methodology should be extended to capture the full range of tidal directions likely to occur in the real world. As a consequence we have changed the data source from model output to field measurements as described below.

Environment Agency (2019) describes the results of 3 variants to the preferred approach with different slack water thresholds of 0.1, 0.2 and 0.3 m s^{-1}. We have adopted the same methodology as Environment Agency (2019) and with the parameterisation described below:

i. The tidal current data are from BEEMS 2018 ADCP monitoring data at the location of the proposed SZC northern and southern intakes at a height of approximately 2m off the seabed (2.04m and 2.09m respectively). These data replaced results used in v2 of this report from the validated Sizewell GETM 3D hydrodynamic model (BEEMS Technical Report TR302) at the proposed location of two of the LVSE intakes at a height 2.5 m above the sea bed (i.e. at the midpoint of the LVSE intake surfaces). There are three possible intake locations on each of the two SZC intake tunnels for two LVSE intake heads and so we have selected the mid positions (I3b and I4b on each tunnel alignment). The ADCP measurements were made with two landers placed within 50m of the I3b and I4b locations. The separation of the intakes on each tunnel alignment is sufficiently small that the local hydrodynamics do not vary significantly but we considered the two tunnel runs were sufficiently different due variations in water depth and underwater topology to merit modelling heads on each tunnel.

ii. The ADCP data were for 30 days (720h) in November to December or two full spring-neap tidal cycles at a 10-minute interval.

iii. The ADCP output was transformed into a 2-dimensional histogram with rows of direction and columns of current speed with the directions and velocities grouped into bins. From the modelling data it had previously been determined that tidal direction is very tightly distributed around the modal direction at Sizewell (Figure 4) and it is, therefore, very important to use a sufficiently small directional bin size to avoid biasing the analysis results. It was found that a 1 degree bin size was satisfactory. For current speed the same bin size was used as in Environment Agency (2019) (0.1m/s).

iv. Head alignment. EDF's construction engineers have confirmed that they can readily place the heads at any desired alignment to an accuracy of at least 1 degree. Due to the slight tidal asymmetry at each location, we adjusted the head alignment at each location to achieve minimum intercept cross-sectional areas at each location.

v. The intercept projected area of the LVSE intakes was calculated as described in Environment Agency (2019).

vi. Slack water conditions were treated differently. At slack water, the area scale factor of the LVSE intakes compared with the SZB intakes was set at 1 i.e. differences in abstraction were determined by the differences in flow rates. The flow velocity at slack was set to 0.3 m s^{-1} (i.e. greater than the design specification of 0.23 m s^{-1}. That parameter choice only produced a small overestimate in calculated intercept area.). Both parameters are as described in Environment Agency (2019).

vii. The threshold speed for considering the tide to be slack was subjected to uncertainty analysis as the calculation is sensitive to this parameter. Based upon inspection of model and measured ADCP data at SZC and modelling data at SZB, slack water can conservatively be represented by speeds less than or equal to 0.2 m s^{-1}.

viii. The calculation of the intercept area for the omnidirectional SZB intakes is described in section 4.2.

ix. Spreadsheets showing the calculations are provided as described in Table A1.

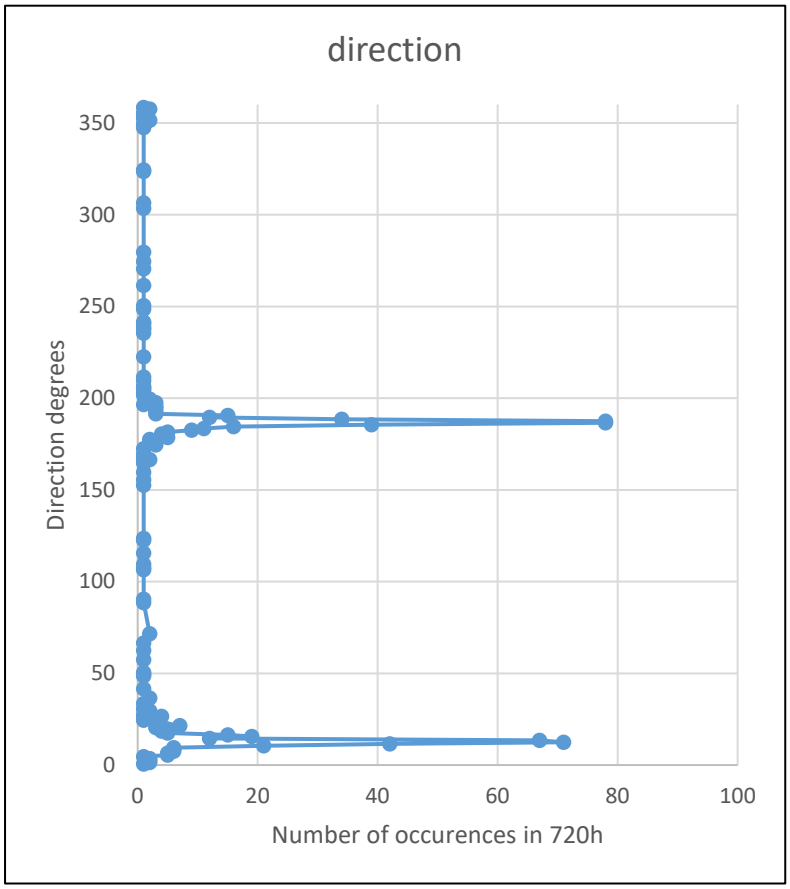

Figure 4 Modelled SZC current direction histogram at 2.5m above the seabed at the location of LVSE intake

4.2 SZB intakes

As described in section 2.1, the combined area of the two SZB intake surfaces is 228.65 m^2 and in accordance with Environment Agency (2019) this is used as the SZB intercept area at slack water. At peak tidal flow, the intercept area is expected to approach the physical cross section of the structures of 69 m^2 (11.5 m width × 3 m height × 2 heads). Simplistically, the intercept area could be regarded as 228.65 m^2 at slack water (for example, at tidal velocities up to 0.1 m s^{-1} and then falling as a step function to 69 m^2 for higher tidal velocities; though this is clearly an unrealistic over simplification).

To gain a greater understanding of the SZB intake head performance, a hydrodynamic model of one of the two heads was constructed based upon scaled drawings provided by EDF Energy. A 2D hydrodynamic model of the SZB intake head was made using Telemac2D. The Telemac system was designed to provide realistic simulations of hydrodynamic flows around and through structures for engineering purposes and the system produces results that compare well with physical models. For the purposes of this study a model consisting of a simple channel domain 300 m long and 100 m wide, with the SZB intake centred in the middle of the domain was created, as shown in Figure 5. Figure 6 shows a zoomed portion of the model domain, focussing on the SZB intake, based on designs provided by EDF. The resolution of the model domain varies between 2.5 m along the boundary down to 5 cm at its finest around the intake structure. As the model is 2D, the depth of the domain is 3 m representing the vertical slice of the water column that runs through the open aperture of the intake head. This is a reasonable assumption given the velocity caps on the intake head reducing the vertical drawdown of currents above and below the intake head. Given the scale of the node density, a k-ϵ turbulence model has been used in the model.

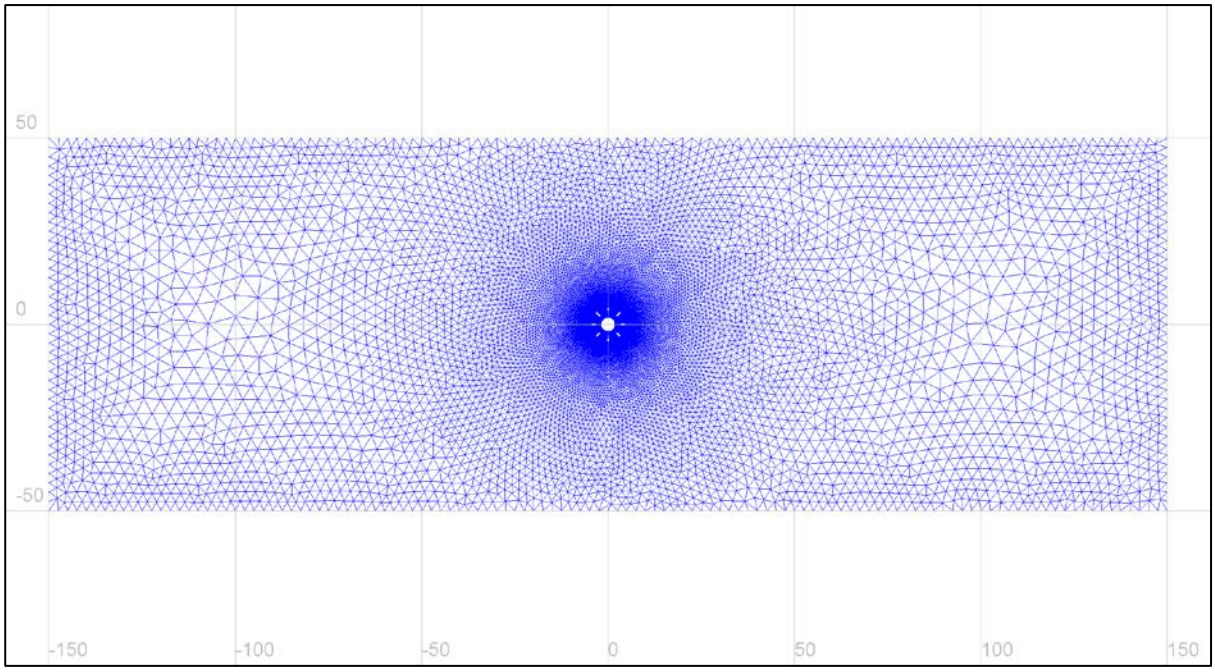

Figure 5: SZB intake – Telemac total model domain.

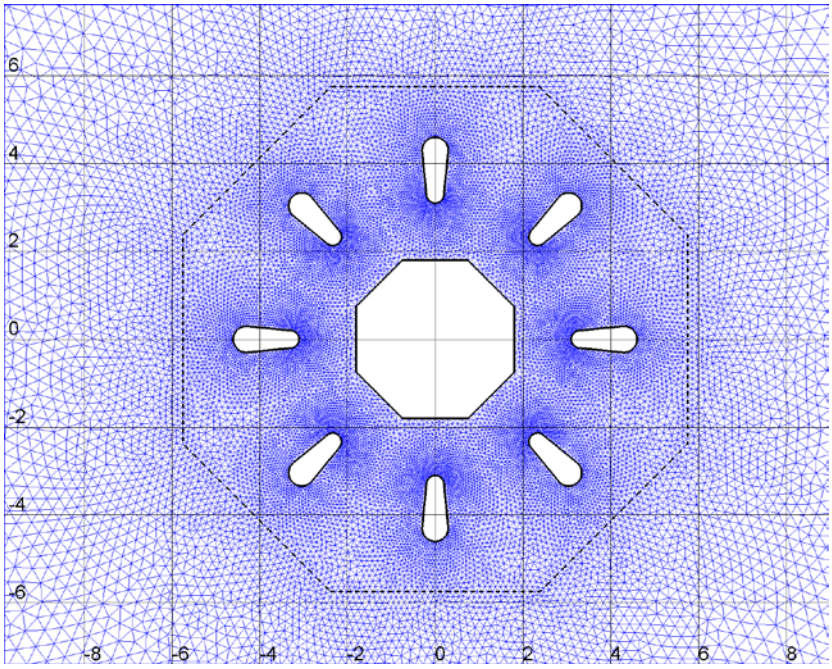

Figure 6: Zoomed portion of model mesh showing the SZB intake. The dashed black line represents the extent of the intake caps, but are not included in the model domain.

The head is omnidirectional and does not incorporate any special hydrodynamic features to shape the flow field and, therefore, the head was modelled at different tidal speeds in Telemac 2D. The flow fields were determined at tidal current speeds from 0m s^{-1} to 1m s^{-1} (at intervals of 0.1 m s^{-1}) to determine at what velocity the head performance deviated from omnidirectional and to find out how the intercept area varied with current speed. As the model is a simple channel model of just the intake head and is running at steady state intervals, there is no way to calibrate or validate the model. This is akin to CFD modelling in that

calibrated/validated models are not used and reliance is placed upon the use of industry standard algorithms (indeed validation is not normally possible except by physical models).

To further explore that issue, the head model was also subject to passive particle tracing. Passive particles were released 50 m upstream of the intake along the width of the channel, at a lateral spacing of 25 cm for the first 25 m centred on the origin, then increasing to every 50 cm for a further 15 m and then increasing to 1 m thereafter. An indicative plot of the particle release is shown in Figure 7 and is not representative of a particular flow speed.

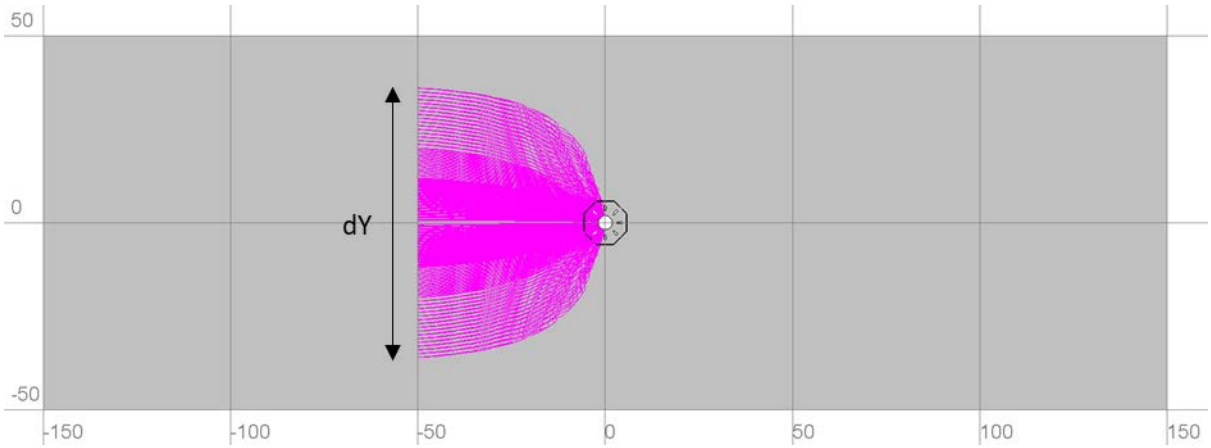

Figure 7: Model domain and extent of particles released 50 m from SZB intake. Tidal flows enter model from left hand side and exit right hand side, with SZB intake centred at middle of domain, abstracting at a flow of 25.75 m^3/s.

The head was found to be fully omnidirectional at tidal currents of up to 0.1 m s^{-1}, i.e. the intake velocity at each of the 8 intake faces was virtually identical (Figure 8). Figure 8 and Figure 9 show vector plots of the current, with the arrow's direction showing the current direction. The vector length varies and is proportional to the current speed, with larger arrows representing faster flow. The magnitude of current speed is shown by the colour of the arrow. The variation in arrow density is due to the model's unstructured mesh, with a higher node density around the intake structure.

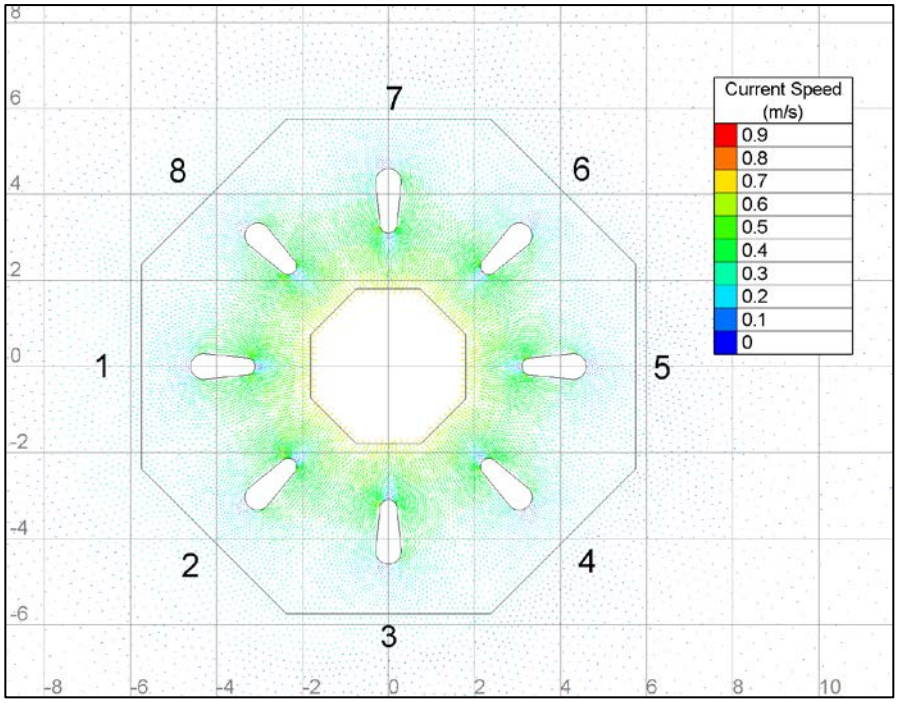

Figure 8 Flow pattern around each SZB intake at a tidal current of 0.1 m/s. Horizontal and vertical scales are metres.

As the tidal current increases after slack water, abstraction increases progressively on those entrances on the side of the head facing more directly into the incoming tidal flow. Figure 9 shows the modelled flow at a tidal current of 0.76m/s and it can seen that water is abstracted predominantly by faces 1, 2, 3, 7 and 8 with much reduced flow into faces 4 and 6 and negligible flow into face 5. Due to turbulence created around the head caused by the head shape, even at tidal currents of 0.76 m s^{-1} faces 3 and 7 abstract water. This was explored further by tracking passive particles released around the modelled head to predict the intercept zone of the head. At a tidal speed of 1m s^{-1} (see Figure 10) the intercept width was only slightly greater than the physical width of the head (12 m versus the physical width of 11.5 m) which is as expected. At 0.5 m s^{-1} the modelled intercept width increased to 16.5 m (see Figure 11) and to 40 m at 0.2 m s^{-1} (see Figure 12).

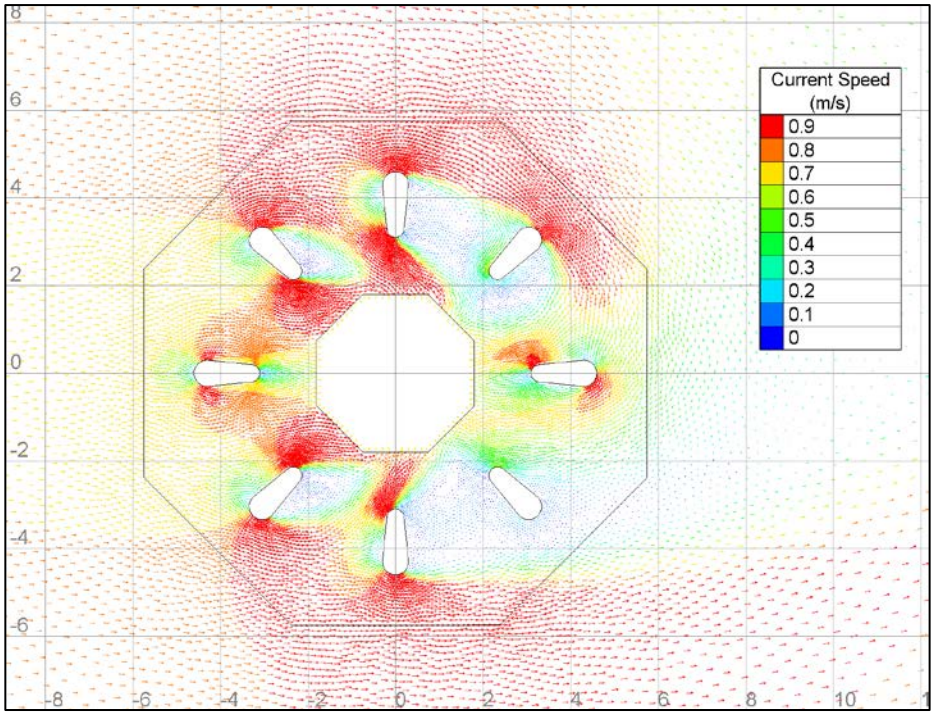

Figure 9 Flow pattern around each SZB intake at near peak tidal flow (0.76 m/s). Tidal direction is at 90° to face 1. Horizontal and vertical scales are metres.

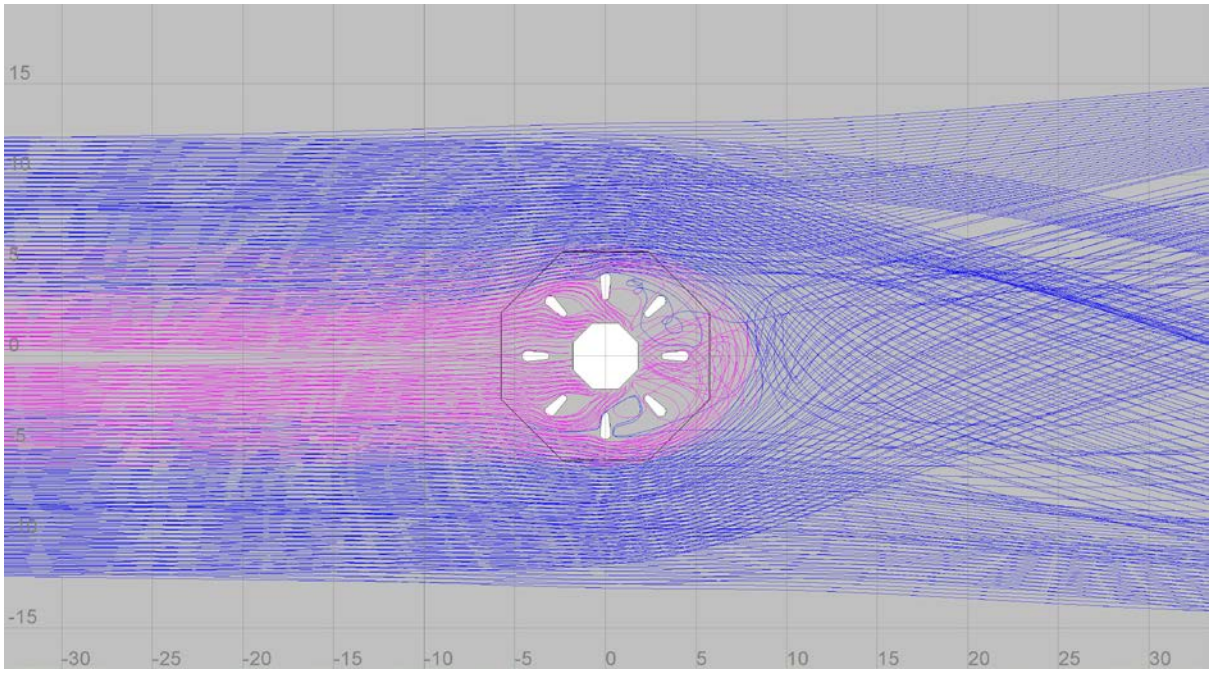

Figure 10: Particle tracks around SZB intake head, extracting at 25.75 m^3/s, with 1.0 m/s input flow conditions. Pink lines represent the particles abstracted by the intake.

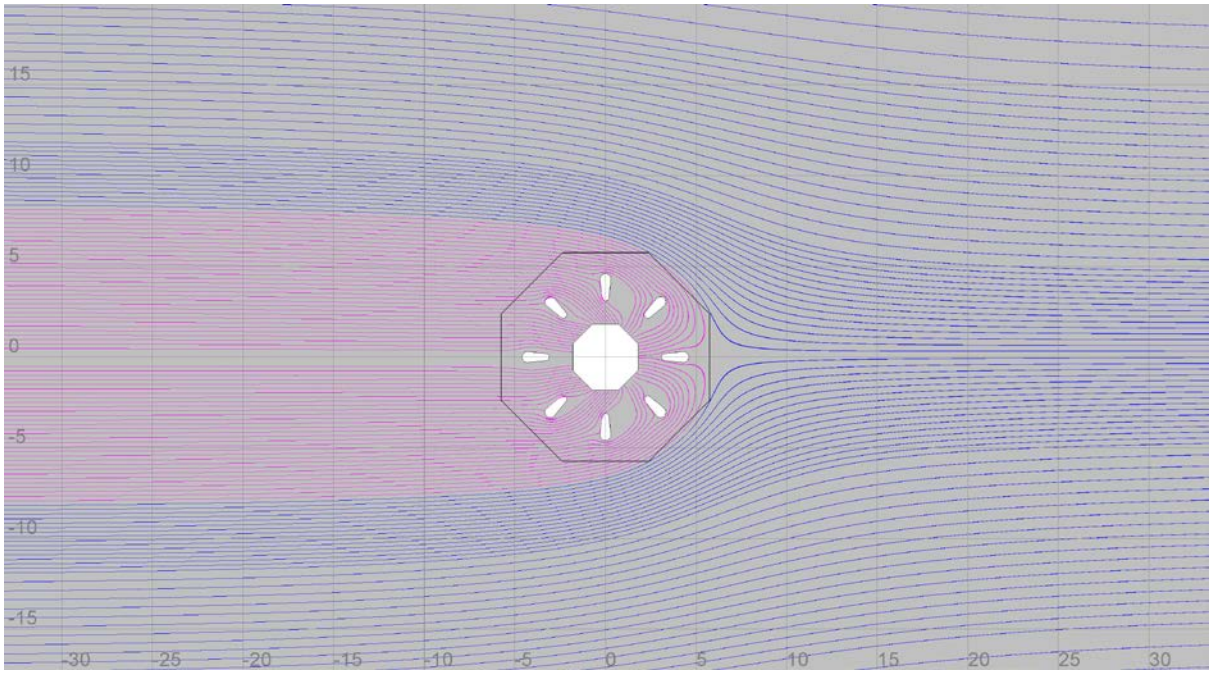

Figure 11: Particle tracks around SZB intake head, extracting at 25.75 m^3/s, with 0.5 m/s input flow conditions. Pink lines represent the particles abstracted by the intake.

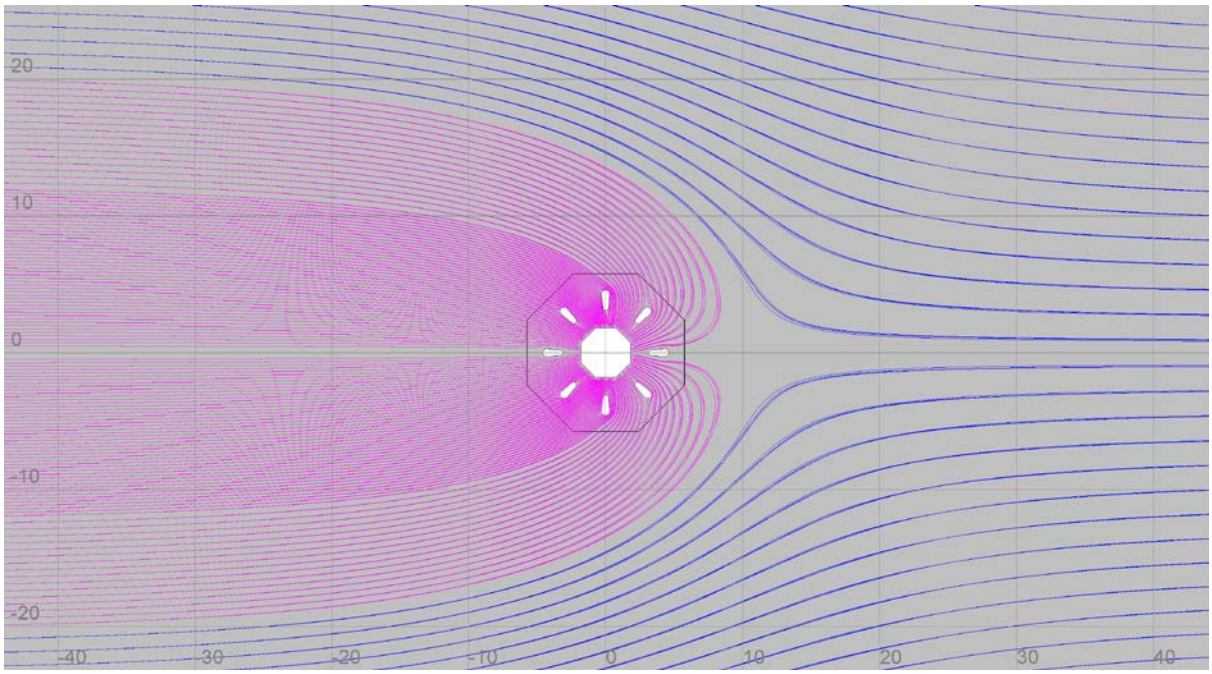

Figure 12: Particle tracks around SZB intake head, extracting at 25.75 m^3/s, with 0.2 m/s input flow conditions. Pink lines represent the particles abstracted by the intake.

Due to the use of a 2D model, these are approximations to reality in that they do not take into account changes in abstraction from outside of the height of the intake surface, however, due to the use of the velocity cap, the predicted intercept zones are expected to be reasonable at mid to high current speeds but overestimated at tidal speeds near to slack water. The SZB intercept area for this report was, therefore,

calculated from particle tracking (abstract width × height of the intake surface) for tidal velocities greater than 0.3 m s^{-1} but set at the physical area of the intake surface for tidal velocities less than or equal to 0.3 m s^{-1} (Figure 13). The abstraction width of the particle tracking is the width Δy shown in Figure 7.

It should be noted in Figure 13 that the area of interception, as calculated by particle tracking, decreases with increasing tidal flow but with a small increase above 0.7 m s^{-1} before decreasing again. This is caused by eddy shedding off the internal support structures with flows above 0.7 m s^{-1}, initially increasing the width of influence over particles travelling past the structure. Thereafter, with increasing strength of the tidal currents, the impact of the eddy shedding is reduced.

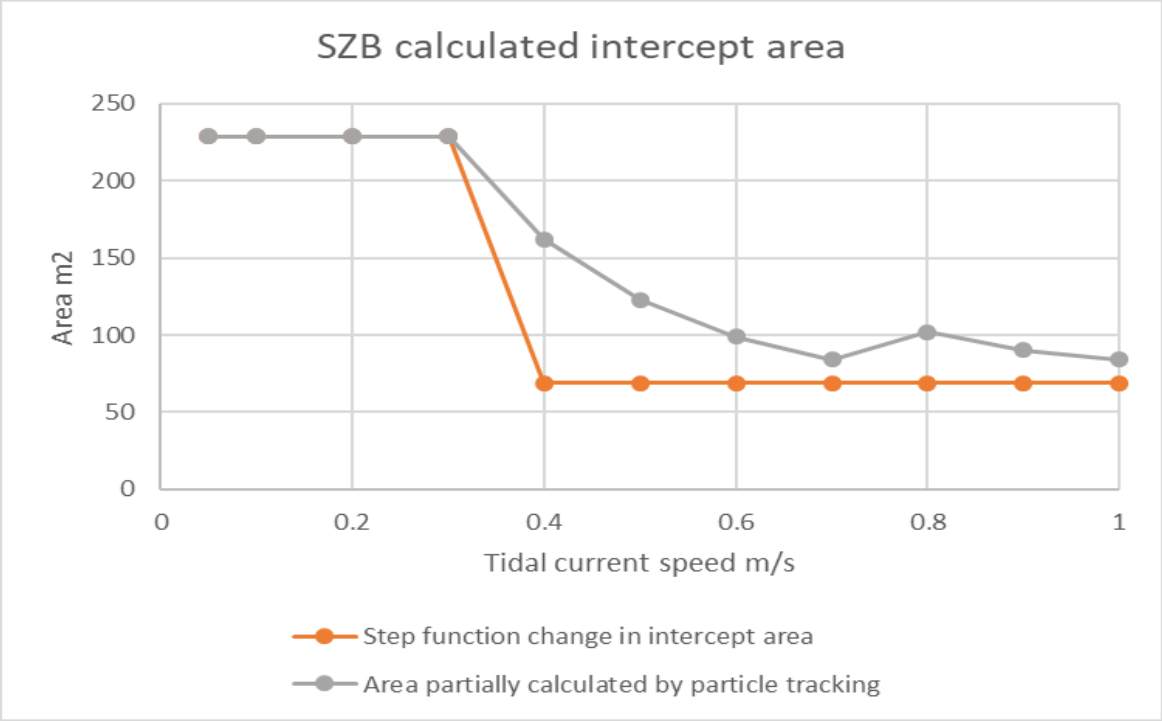

Figure 13 Options for the calculation of the intercept area of the SZB intake heads; a step change is physically unrealistic

Figure 14 shows the modelled histogram of tidal current speed at the Sizewell B intakes from the validated Sizewell GETM model at 2.5 m above the sea bed.

The Sizewell C intake area calculations use ADCP data whereas the SZB intake area has been calculated with GETM model data. This then provokes the question whether the continued use of model data leads to an underestimate in the SZB calculated intercept area. Comparison of the SZC model and ADCP data at the same location does indeed show that the velocity histograms are different with the ADCP data showing a higher percentage of velocities in the range 0.1 to 0.5 m/s. However, if the same pattern is assumed at SZB, sensitivity tests show that this change only results in a 1% underestimate in the SZB intercept area because there is no directional component in the SZB analysis (due to the omnidirectional heads at SZB). This small error has been ignored, thereby slightly overestimating the calculated ratio of the SZC/SZB intercept area which is conservative.

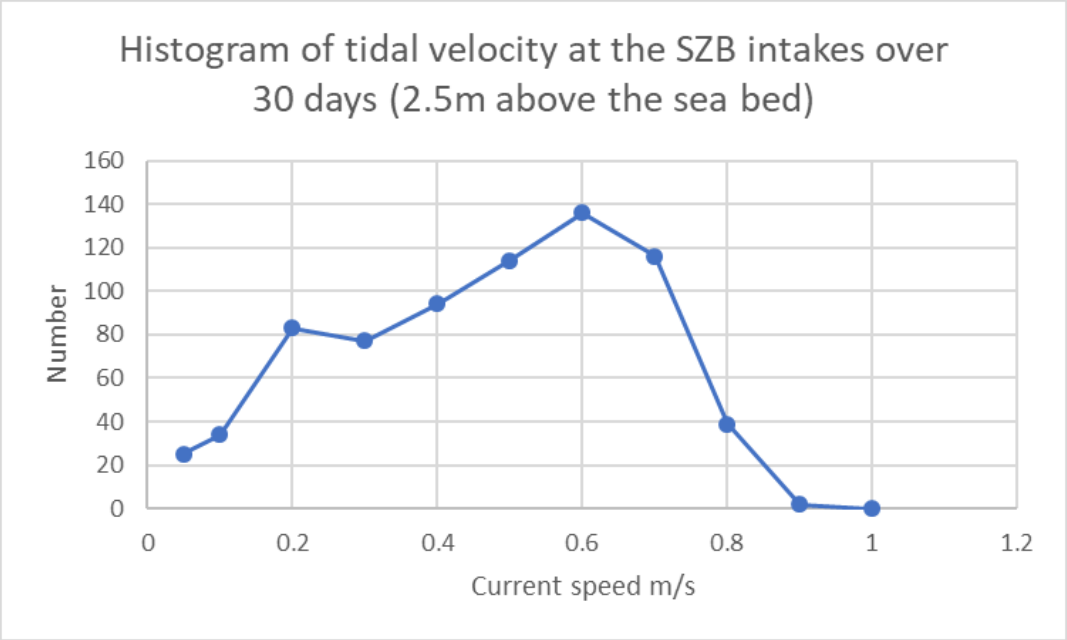

Figure 14 Tidal velocity histogram at the SZB intakes.

5 Results

The following results are from the provided supplementary spreadsheets (Appendix A).

The calculated intercept areas of the SZC intakes are shown in Table 2

Table 2 Calculated intercept area of the 4 SZC LVSE intakes (assuming that the intakes were all at locations I3b or I4b)

Assumed slack water threshold m s^{-1}	Intake I3b (northern) intercept area m^2	Intake I4b (southern) intercept area m^2
0.1	33.19	33.61
0.2	41.61	43.04
0.3	50.23	53.3

Table 3 shows the calculated intercept area of the SZB intakes.

Table 3 Calculated intercept area of the SZB intakes over a 30 day tidal cycle

Assumed slack water threshold m s^{-1}	SZB intakes intercept area m^2
0.1	118.4
0.2	118.4
0.3	123.5

Notes

1. the calculated area does not change with an assumed slack water threshold of 0.1 m s^{-1} as the physical area of the intake surfaces is used at that current speed.
2. The SZB areas are slightly larger than reported in SPP099 v2 because the bin velocities in the analysis of the tidal currents at SZB were set at maximum rather than the correct mid bin velocities.

Model and measurement data show that flow into the SZB head deviates significantly from omnidirectional after the tidal velocity increases past 0.1 m s^{-1}. The use of a slack threshold of 0.2 m s^{-1} is therefore precautionary. By extending the definition of slack water to 0.3 m s^{-1} the duration of 'slack' increases beyond one hour and since the calculations of SZB intercept area do not take account of this extended duration, the intercept area calculations at 0.3 m s^{-1} shown in Table 3 are underestimates, possibly large underestimates.

Table 4 shows the ratio of the SZC to SZB intercept area at each of the slack water thresholds. The ratio differs slightly at the SZC northern and southern intake locations (I3b and I4b respectively) and so the mean result is also shown as there will be 2 intake heads on each tunnel alignment.

Table 4 Calculated SZC/SZB intercept area ratio

Slack water threshold m s^{-1}	Northern	Southern	Mean ratio
0.1	0.280	0.284	0.282
0.2	0.351	0.364	0.357
0.3	0.407	0.432	0.419

5.1 Sensitivity analyses

The LVSE intakes will be aligned in the field by the use of ADCP survey data. Such surveys can determine the tidal axis to a directional accuracy in the range ±1 to ± 2 degrees dependent upon whether the instrument is individually calibrated or the manufacturer's worst case calibration is used.

EDF Energy's construction constractors have confirmed that the intake heads can be placed on the seabed to an accuracy of ± 1 degree.

For sensitivity tests it has been assumed that the ADCP will not be individually calibrated and that the instrument will produce normally distributed errors with one standard deviation of 1 degree. Head placement errors will also be normally distributed with one standard deviation of 0.5 degrees. Simulating the combined error from the two independent error sources over 10,000 trials gives 5 and 95 percentile errors of ± 1.85 degrees.

Table 5 shows the calculated intercept area for the 4 SZC intakes for up to ±3 degrees misalignment around the optimum head alignment directions. The cells in green provide the mean (0°), and 5 percentile and 95 percentile head misalignment values.

Table 5 Calculated intercept areas (m^2) assuming that all of the SZ LVSE intakes were either at the northern or southern intake locations with a realistic slack velocity threshold of 0.2 m s^{-1}. Green cells are 5 and 95 percentile values

SZC intake	-3°	-2°	-1.85°	0°	+1.85°	+2°	+3°
Northern	45.27	43.39	43.20	41.61	42.25	42.35	43.66
Southern	46.77	44.89	44.70	43.04	43.80	43.86	45.28

Table 6 shows the ratio of the SZC to SZB intercept areas calculated from Table 5 and Table 3. The cells in green provide the mean (0°), and 5 percentile and 95 percentile head misalignment values.

Table 6 Calculated sensitivity of the SZC/SZB intercept area ratio for a slack threshold =0.2 m s^{-1}

SZC intake	-3°	-2°	-1.85°	0°	+1.85°	+2°	+3°
Northern	0.382	0.366	0.365	0.351	0.357	0.358	0.369
Southern	0.395	0.379	0.378	0.364	0.370	0.376	0.382
Mean ratio	0.389	0.373	0.371	0.357	0.363	0.364	0.376

The mean ratio of 0.357 is 50.8% larger than that calculated using the SZC GETM model data at 0.237.

The calculations have also been repeated for completeness at an assumed slack water threshold of 0.3 m s^{-1}. As stated in Section 4.1, this threshold is considered unrealistically high based upon model and measurement data at Sizewell. The cells in orange provide the calculated mean (0°), and 5 percentile and 95 percentile head misalignment values without compensation for the length of slack tide at this threshold. This will mean that the estimated SZC/SZB intercept ratio is likely to be overestimated.

Table 7 Calculated intercept areas (m^2) assuming that all of the SZ LVSE intakes were either at the northern or southern intake locations with an unrealistic slack velocity threshold of 0.3 m s^{-1}

SZC intake	-3°	-2°	-1.85°	0°	+1.85°	+2°	+3°
Northern	53.85	52.00	51.81	50.23	50.84	50.94	52.21
Southern	57.00	55.15	54.97	53.30	53.95	54.04	55.40

Table 8 Calculated sensitivity of the SZC/SZB intercept area ratio assuming a slack threshold of 0.3 m s^{-1}

SZC intake	-3°	-2°	-1.85°	0°	+1.85°	+2°	+3°
Northern	0.436	0.421	0.420	0.407	0.412	0.412	0.423
Southern	0.462	0.447	0.445	0.432	0.437	0.438	0.449
Mean ratio	0.449	0.434	0.432	0.419	0.424	0.425	0.436

The mean ratio of 0.419 is 42.1% larger than that calculated from the GETM model data at 0.295.

6 Conclusions

6.1 LVSE intake velocities

Computational Fluid Dynamic (CFD) modelling of the proposed Sizewell C LVSE intake heads shows that:

i. The hydrodynamic influence of the LVSE intakes is only felt over a short distance with inward velocities falling to 0.1 m s^{-1} over a distance of approximately 1.5 to 2.5 m from the intake surfaces and to 0.05 m s^{-1} at a distance of approximately 3m from the intake surfaces over virtually the full tidal cycle;

ii. The range from the intake surface for the inward velocity to fall to the Environment Agency (2005, 2010) criterion of 0.3 m s^{-1} is 0.0 to 0.68 m dependent upon the tidal current speed. Over the tidal cycle the median distance to an inward velocity of 0.3 m s^{-1} is 0.38 m based upon the distribution of tidal currents at the SZC intakes.

6.2 Ratio of SZC to SZB intake intercept areas

Using the methodology proposed by Environment Agency (2019) to calculate the intercept area of the SZC LVSE intakes but with ADCP data rather than model data, the ratio of the intercept cross-sectional areas of the proposed four SZC LVSE intakes compared with the two omnidirectional SZB intakes is 0.357 as a best estimate with values of 0.371 and 0.363 at the 5 percentile and 95 percentile head misalignment values respectively at a slack tide threshold of 0.2 m s^{-1}. This means that SZC fitted with LVSE intakes is expected to abstract 35.7% of the fish and crustacea per cumec that SZB does. This ratio is appropriate to impingement assessments rather than entrainment assessments; the notable exception being for glass eels that are entrained because of their long, thin morphology.

This ratio is slightly smaller than that provisionally estimated in BEEMS Technical Report TR406 but confidence in the result is much greater because it:

a. has now been calculated fully in line with the methodology in Environment Agency (2019);
b. it has been subject to an extended sensitivity analysis that incorporates measured as opposed to only model data;
c. benefits from much more realistic modelling of the SZB intake heads; in particular how their intercept area varies during the tidal cycle

6.3 Alignment of the LVSE intakes at Sizewell.

The SZC LVSE intakes will be aligned in the field by the use of ADCP survey data. Such surveys can determine the tidal axis to a directional accuracy in the range ±1 to ± 2 degrees dependent upon whether the instrument is individually calibrated or the manufacturer's worst case calibration is used. This measurement accuracy, together with a realistic estimate of the accuracy that the heads can be placed on the seabed (± 1 degree) has been built into the sensitivity analyses presented in this report.

References

BEEMS Technical Report TR302 Ed. 3. Sizewell Thermal Plume Modelling: GETM Stage 3 results with the preferred SZC cooling water configuration. Cefas, Lowestoft

BEEMS Technical Report TR406 Ed 2 v6. Sizewell C – Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft

Environment Agency 2005 Screening for Intake and Outfalls: a best practice guide. Environment Agency Science Report SC030231/SR3. Environment Agency.

Environment Agency 2010. Cooling Water Options for the New Generation of Nuclear Power Stations in the UK SC070015/SR3. Environment Agency, Bristol UK.

Environment Agency 2019. Low Velocity Side Entry Intake Design; effect of intake intercept area. Technical Brief: TB006 draft 3.

HRW (2013) Numerical & Physical Modelling of the Hinkley Point C Intake & Outfall Structures. Task 1 – Physical Modelling of Flows at Intake Heads (TN-10). HR Wallingford Ltd. May 2013.

Turnpenny, A.W.H., Taylor, C.J.L., 2000. An assessment of the effect of the Sizewell power stations on fish populations. Hydroecol. Appl 12, 87–134.

Appendix A

Table A1 Spreadsheets showing detailed calculations provided in support of this Scientific Position Paper

Filename	Contents
SZC intake velocities awac 6 July 2020.xlsx	Intercept area calculations SZC (using ADCP data)
SZB intake velocities 6 July 2020.xlsx	Intercept area calculations SZB (using GETM model)
LVSE Supplemental calculations 5May2020.xlsx	LVSE intake velocities (CFD results)

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP100

SPP100 - Estimates of European populations of twaite shad and cucumber smelt of relevance to Sizewell.

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Executive summary

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture Science (Cefas) has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

BEEMS Technical Report TR406 v7 provides an assessment of the predicted losses for fish and crustacea due to entrapment (impingement plus entrainment) in the SZC cooling water system. For assessment purposes 24 key fish taxa in the Greater Sizewell Bay were selected as representative of the assemblage at Sizewell based on either their commercial value, their ecological importance, or their conservation status. Two of the conservation species (Twaite shad (*Alosa fallax*) and Cucumber smelt (*Osmerus Eperlanus*)) have no known spawning stock biomass estimates. The purpose of this report is to identify the possible spawning populations and to estimate their sizes based upon existing monitoring data.

Conclusions

The following conclusions have been reached in this report:

1. The mean annual adult population estimate for the period 2009-2017 for:

    a. Twaite shad from the Elbe and Scheldt is 5.16M fish.

    b. Cucumber smelt from the Elbe only is 23.9M fish. The timing of the European surveys were not optimal for peak smelt migration which occurred earlier in the year. The smelt population estimates are therefore likely to have been underestimated.

2. There is no known spawning population of twaite shad on the UK east coast. Twaite shad caught at Sizewell are genetically distinct from the known breeding population in the Severn. The most likely origin of the twaite shad caught at Sizewell is fish originating from the Elbe and to a lesser extent the Scheldt. Twaite shad on the European southern North Sea coast have already been found to be from one population. It is recognised that it remains to be proven that the Sizewell twaite shad are part of this single southern North Sea population but based upon the weight of evidence available to date this is considered the most logical conclusion.

3. Cucumber smelt are known to spawn in East Anglian rivers. These fish may also be part of a single southern North Sea population, again centred on at least the Elbe and Scheldt but also including other estuaries. This also remains to be proven.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture Science (Cefas) has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC Development Consent Order (DCO) application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

SZC would need to abstract approximately 132 cumecs (m^3 s^{-1}) compared with approximately 51.5 cumecs for the existing SZB station.

BEEMS Technical Report TR406 v7 provides an assessment of the predicted losses for fish and crustacea due to entrapment (impingement plus entrainment) in the SZC cooling water system. For assessment purposes 24 key fish taxa in the Greater Sizewell Bay were selected as representative of the assemblage at Sizewell based on either their commercial value, their ecological importance, or their conservation status. Two of the conservation species (Twaite shad (*Alosa fallax*) and Cucumber smelt (*Osmerus Eperlanus*)) have no known spawning stock biomass estimates. The purpose of this report is to identify the possible spawning populations and to estimate their sizes based upon existing monitoring data.

1.1 Twaite shad population in the North Sea

Twaite shad (*Alosa fallax*) belongs to the herring family. The species is anadromous with adults spending most of their lives in the marine environment but migrating through estuaries to spawn in freshwater.

Historically, the twaite shad *Alosa fallax* was a widely distributed species in Europe, occurring in the Baltic and North Sea (Thiel *et al.,* 2008), the Northeast Atlantic (Taverny & Elie 2001) and the Mediterranean and Black Seas (Turan & Basusta 2001, López *et al.,* 2007). However, populations suffered from anthropogenic impacts, in particular water pollution and hydraulic engineering work, including the building of weirs (Aprahamian & Aprahamian 1990, Aprahamian *et al.,* 2003).

Twaite shad declined throughout Europe and even became extinct in some rivers. For this reason, the twaite shad has been included in a number of European national Red Lists, e.g. listed as extinct in the Netherlands and Luxembourg (Aprahamian *et al.,*2003), endangered in Belgium, Denmark and Germany (Fricke *et al.,* 1994, Berg 1998, Aprahamian *et al.,* 2003), and vulnerable in Spain, Portugal, Poland, Ireland and the UK (Aprahamian *et al.,* 2003).

In the recent past, improvements in water quality have led to a recovery of some twaite shad spawning populations in the Baltic Sea (Thiel *et al.,* 2008) and in the Mediterranean (López *et al.,* 2007). Increased numbers of twaite shad have been found off the European coast and in offshore areas of the southern North Sea from surveys completed during the period 1996 to 2004 (Stelzenmüller *et al.,* 2004, Neudecker & Damm 2005, Thiel & Backhausen 2006), although no significant increase was observed in the Wadden Sea (Jager *et al.,* 2009). Furthermore, there is evidence of a stock increase in the Weser estuary (Scholle & Schuchardt 2012) and the reappearance of twaite shad in the Scheldt (Maes *et al.,* 2005, Maes *et al.,* 2008).

One of the most important spawning stocks of twaite shad in the North Sea region occurs in the Elbe estuary, one of the largest estuaries in Europe. The twaite shad spawning stock of the Elbe estuary increased significantly in numbers between 1992/1993 and 2010/2011 with median abundance increases of between 13 and 30 times and with expansion of the spawning areas including to locations in both freshwater and areas of low salinities (0.5 ± 0.1), which was considered an exceptional finding (Magrath and Thiel 2013). The Elbe population was still increasing in recent years (Tulp *et al.,* 2017).

The Belgian population of the Scheldt estuary is also exhibiting signs of “genuine improvement” (Wilson and Venerata, 2019) although recovery is later than in the Elbe after action to improve water quality and more

recently to improve access to spawning grounds. Very large numbers of migrating twaite shad have been reported in the Scheldt in 2020 with “vast areas of spawning in the river” (pers. comm: Pieterjan Verheist, University of Ghent, 8 May 2020). Numbers of a small population of twaite shad in the Weser Estuary are stable without clear trends in the recent years (Tulp *et al.,* 2017). This process of quick stock recovery occurred during operation of Sizewell B.

There are no known UK east coast spawning sites for twaite shad. Genetic analyses of twaite shad from Sizewell demonstrate that they do not originate from the Severn catchment (Jolly *et al.,* 2012). Sabatino and Alexandrino (2012) identified a North Sea twaite shad population with low genetic diversity between fish sampled off Belgium (Scheldt) and Denmark and also the Solway Firth. These analyses identified separation between the Baltic and North Sea populations. The twaite shad caught at Sizewell range from >1 yr old juveniles to sexually mature adults that are probably a part of the North Sea mixed population widely dispersed across feeding grounds. In contrast to allis shad, twaite shad are iteroparous so adults may return to the sea to forage several times during their lifetime after each seasonal spawning. Given the distance of the proposed development from the spawning rivers in mainland Europe and the likelihood of population mixing during feeding in the marine environment it is not logical to associate all the fish impinged at Sizewell to a single river system. Given the geography, it is much more likely that the origin of fish caught at Sizewell would be approximately in proportion to the size of the spawning populations in the European Rivers. For example, the history of shad recovery in European rivers is firstly the Weser and then the Scheldt, in all likelihood both reseeded from the Elbe. No adult shad were detected in the Scheldt until 2012 whereas Sizewell B has been abstracting adults since at least 2009 (when impingement monitoring began) so those fish could not have originated from the Scheldt. Sizewell C is expected to impinge fish from different European rivers on a pro-rata basis according to their abundance and it is therefore considered highly unlikely that there would be a significant effect on the population in any given river.

The Wadden Sea status report (Tulp *et al.,* 2017) provides an overview of North Sea twaite shad populations in the nearby German rivers and describes a situation that varies greatly between estuaries:

- The estuaries of the Ems and Eider are of no significance for twaite shad nowadays. The decline in juvenile smelt in the Ems is possibly due to extreme quantities of suspended matter (fluid mud) and oxygen deficiency in the inner parts of the estuary. These adverse environmental conditions also prevent successful reproduction of twaite shad in this potential spawning area.
- Twaite shad regularly occur in the Weser and reproduce there successfully, without clear trends in the recent past.
- The Elbe estuary contributes most to twaite shad numbers and reproduction in German estuaries. In contrast to the other Wadden Sea estuaries, numbers in the Elbe appeared to be increasing in recent years. Against this background the Red List status of the twaite shad for the North Sea was recently changed from critically endangered to “early warning” (early warning list, Thiel *et al.,* 2014).

Maes *et al.,* 2008 described the situation in the Scheldt whereby poor water quality conditions just upstream of the freshwater–saltwater boundary acted as an effective migration barrier for upstream movement. As a consequence, spawning grounds were inaccessible in 2008 and the population was dominated by seasonal adults occurring in the lower estuarine part of the watershed. Based upon trends in dissolved oxygen, Maes *et al.,* 2008 predicted that the entire river gradient between the ocean and historical freshwater spawning sites would become a suitable habitat for twaite shad by the year 2010.

No adult fish were detected in anchor net surveys in the Scheldt at Antwerp or further up estuary at Steendorp in spring of 2011 (Goudswaard and Breine 2011). The only twaite shad that were caught were in Dutch waters in the Westerschelde (the part of the Scheldt between the Belgian border and the North Sea). These fish were all immature juvenile fish (8-18cm length) that had hatched in another estuary, probably the Elbe. However, in spring of 2012, Breine *et al.,* 2012 reported that adult twaite shad had been caught in anchor net surveys at Antwerp and further up-estuary, and juveniles were detected in summer indicating spawning had taken place.

1.2 Twaite shad reproduction

Figure 1 shows twaite shad monitoring data from the Weser. The pattern for shad presence in the estuary is as follows (Scholle 2011):

- Adults migrate into the estuary from the North Sea starting from the beginning of April. The peak migration period is approximately for 6 weeks from late April to the end of May with inward migration ending by late June.
- Adults return to sea after the spawning period is complete.
- Eggs and larvae appear in spawning areas from the end of April to mid-June and rapidly disperse in the estuary.
- Juveniles appear from mid-June to early September and then either migrate out of the estuary to the Wadden Sea nursery areas or remain in the outer reaches of the Weser Estuary.
- Sub adults disperse to feed in the southern North Sea and associated coastal zones but are also found in estuary mouths.

Genetic studies show no significant differences between any of the four German estuaries (Scholle 2011). Sabatino and Alexandrino (2012) identified a North Sea twaite shad population with low genetic diversity between fish sampled off Belgium (Scheldt) and Denmark and also the Solway Firth. The lack of genetic diversity on the European North Sea coast is not surprising since it would appear likely that the Elbe has seeded the recently re-established populations in the Weser and probably the Scheldt.

1.3 Cucumber Smelt populations in the southern North Sea

Cucumber smelt are found all along the Anglian coast, in the southern North Sea and on the European coast from the English Channel to Denmark. Particularly large populations exist in the Scheldt and the Elbe.

Smelt (*Osmerus eperlanus*) are found in coastal waters and estuaries around the western coast of Europe, from southern Norway to north-west Spain (Maitland, 2003). Although there are several non-migratory populations in large freshwater lakes in Scandinavia, it is usually found in coastal waters and migrates into large clean rivers to spawn. Adults live in the marine environment, but migrate to estuarine or slightly brackish rivers in early spring (February to April) to spawn, after which the adults return to sea (Maitland, 2003).

Genetic analysis of 215 smelt collected from the SZB CIMP programme and from the Thames, Waveney, Great Ouse and Tamar estuaries showed that East Anglian smelt are genetically homogeneous with no genetic structuring seen within the region. Smelt from the Tamar was clearly distinct from the East Anglian collections (BEEMS Technical Report TR406). Given the genetic information on the smelt at Sizewell, it is probable that the smelt impinged are from multiple locations on the east coast of the UK and, based on the comparable distances, from European estuaries of at least the Scheldt (Belgium) and the Elbe in Germany (this hypothesis is considered reasonable, but it is recognised that it has not yet been proven).

Fish monitoring programmes in German and Belgian estuaries are undertaken to determine trends in fish populations. The dates of these surveys (end April – to early May) have been selected to be optimal for twaite shad and as such are too late to measure peak smelt migration which occurs earlier (February to early April) than for twaite shad.

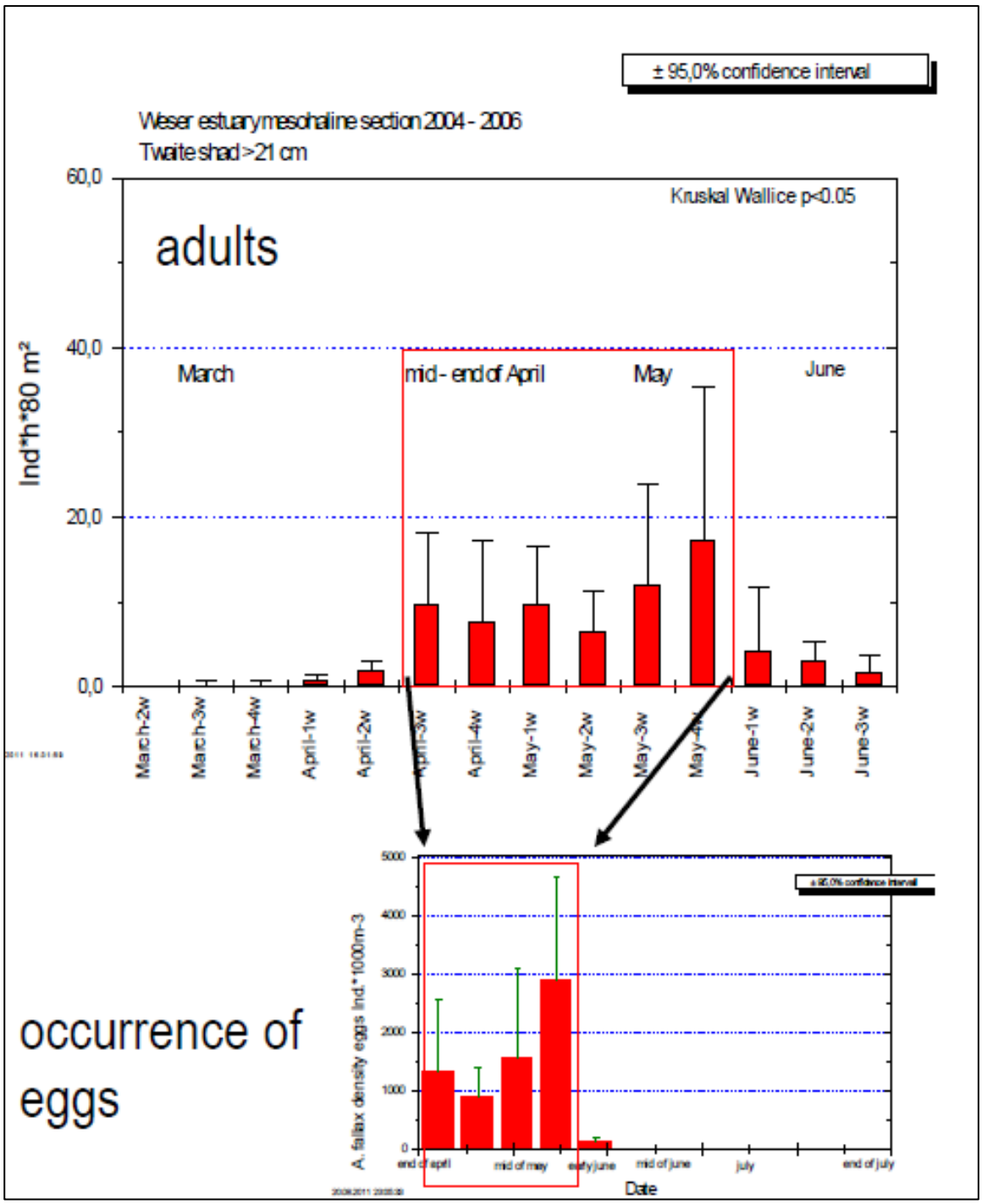


Figure 1 Typical shad migration and recruitment timing in the Weser (from Scholle 2011)

2 Twaite Shad monitoring in the Elbe and Scheldt

Both the German and Belgian authorities undertake monitoring for the state of fish populations in the Elbe and Scheldt respectively. These studies are focussed on trend monitoring for WFD and HRA purposes. In Germany, the quantitative trend data for the Elbe are compared with an estimated reference condition inferred from historical landings information. Monitoring in both estuaries is done by anchor netting which is a type of stow net deployed from an anchored vessel which samples from the water surface to the bed (‘hamen’ net in German and ‘ankerkuil’ in Dutch).

2.1 Scheldt Estuary

Fish in the Belgian Scheldt are sampled with anchor nets installed on a flat-bottomed ship, ‘De Harder'; and owned by the fishing company Bout-Van Dijke (

Figure 2). The anchor net consists of two 8 metre wide steel beams, the bottom of which is lowered to the bottom and the upper net on or above the waterline (Figure 3). Between these beams the grid is stretched to cover the total water column over a width of 8 meters. The vertices of the beams are connected to the ship's anchor to which the vessel is also moored. The water current opens the net and that filters all objects in the end of the net which has a 20mm mesh size. When weather permits, two anchor nets are deployed (one on each side of the vessel) per tide to avoid lost data, with one net deployed and then the other. The catch statistics integrate all of the catch data per tide.

Figure 2 Fishing vessel De Harder (source: INBO).

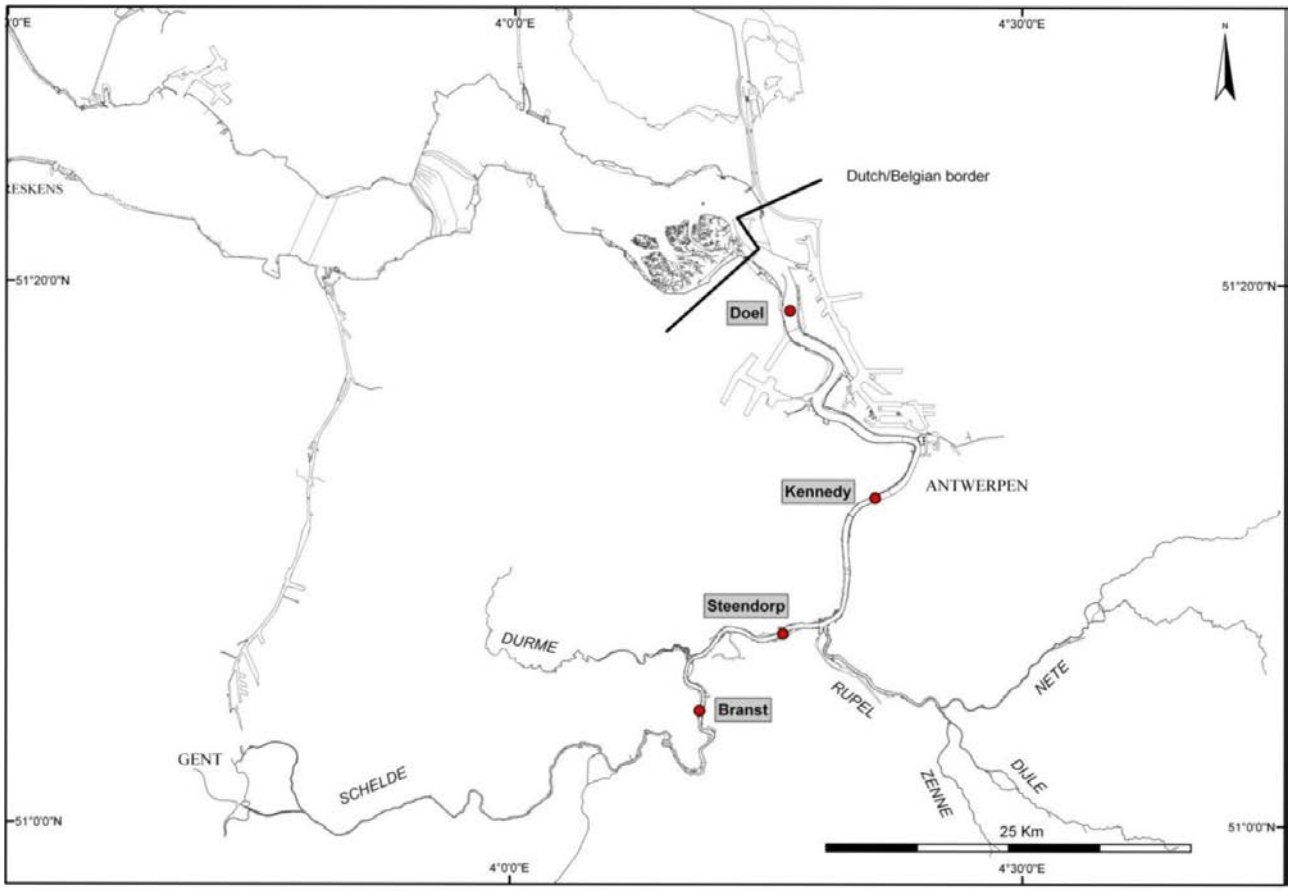
Figure 3. One anchor net prior to deployment, showing the top and bottom beams together above the surface of the estuary with the boat facing into the current (source INBO).

Monitoring is conducted by Instituut voor Natuur- en Bosonderzoek (INBO - The Institute for Nature and Forest Research). The INBO is the Flemish research and knowledge centre for nature and its sustainable management and use.

The INBO surveys using anchor nets started in 2011 in a joint survey with IMARES (a Dutch institute) of the different salinity zones in the Scheldt Estuary. Prior to 2011 surveys were conducted using fyke nets which are not very efficient at catching pelagic species such as twaite shad and cucumber smelt. Thereafter an expanded Belgian time series commenced from 2012 using anchor netting and is ongoing.

2.1.1 Scheldt Data Source

For the purpose of this report data on twaite shad and cucumber smelt catches have been obtained from INBO annual publications (2012-2017). The datasets are published slightly differently from year to year and, as such, in some years information on fish length-frequencies is not clear. The shad and smelt estimates are therefore best estimates in some years due to a lack of access to original raw catch data at this time. This uncertainty largely affects the smelt data and so those data have not been used in BEEMS Technical Report TR406.



Figuur 1. Met ankerkuil bemonsterde locaties in het Zeeschelde-estuarium in 2012

Figure 4 The four Belgian monitoring stations in the Scheldt.

2.2 Elbe Estuary

Monitoring in the Elbe is conducted in a similar way to that in the Scheldt. Due to the availability of suitable local vessels in the Elbe, only one anchor net is deployed at each station. Cefas has been provided data for the four most western monitoring stations shown in Figure 5 which (dependent upon weather) are monitored in spring, summer and autumn on 2 tides in a day. As described in Section 1.2 above, the pattern of shad migration is that adults arrive in the estuary to spawn from April. Eggs and larvae are found over a large area of the upper reaches of the estuary in summer and large numbers of juveniles are fund everywhere in autumn. The outer estuary is important for subadults.

Typically, the net is set for 2-5 hours per tide. Some stations are on the left bank, some on the right bank and the station at Medem is in the centre of the channel at the Estuary mouth. The Elbe is a major commercial waterway for the port of Hamburg and has been extensively reengineered over the past hundred years to deepen and widen the channel such that much of the shallow water fringes no longer exist

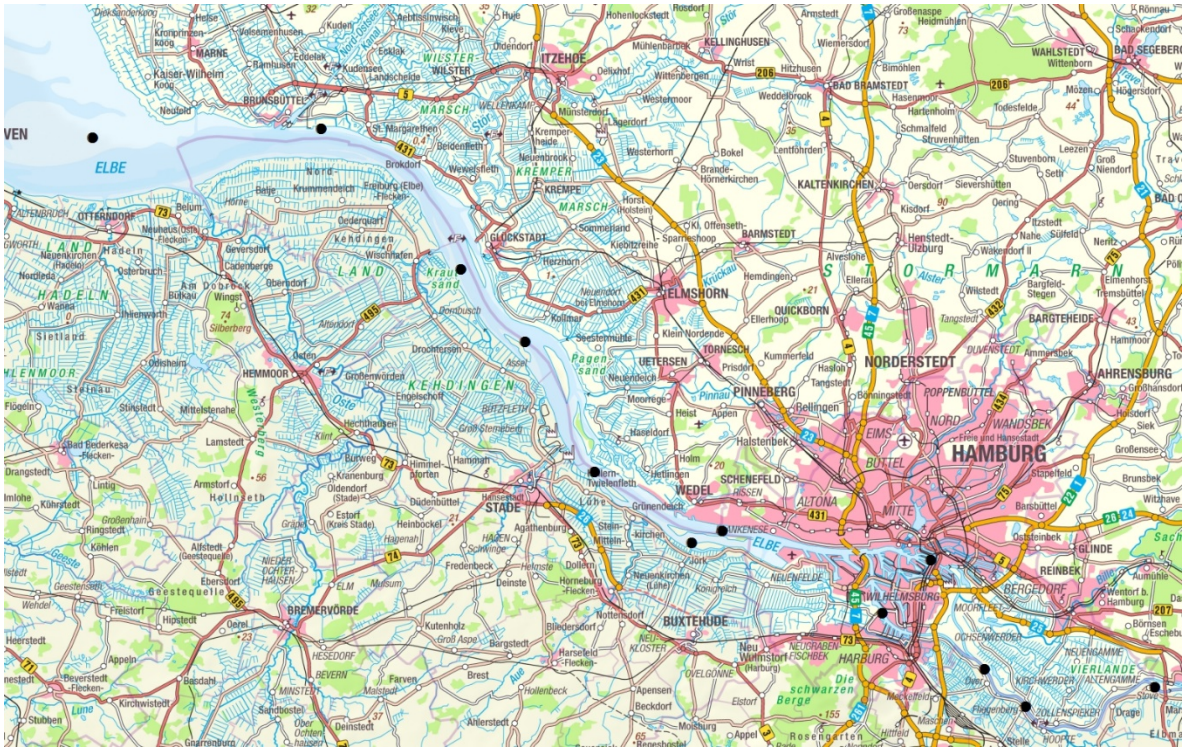


Figure 5 Elbe monitoring stations (from the west: Medem, Brunsbuttel, Krausand/Gluckstadt, Kollmar).

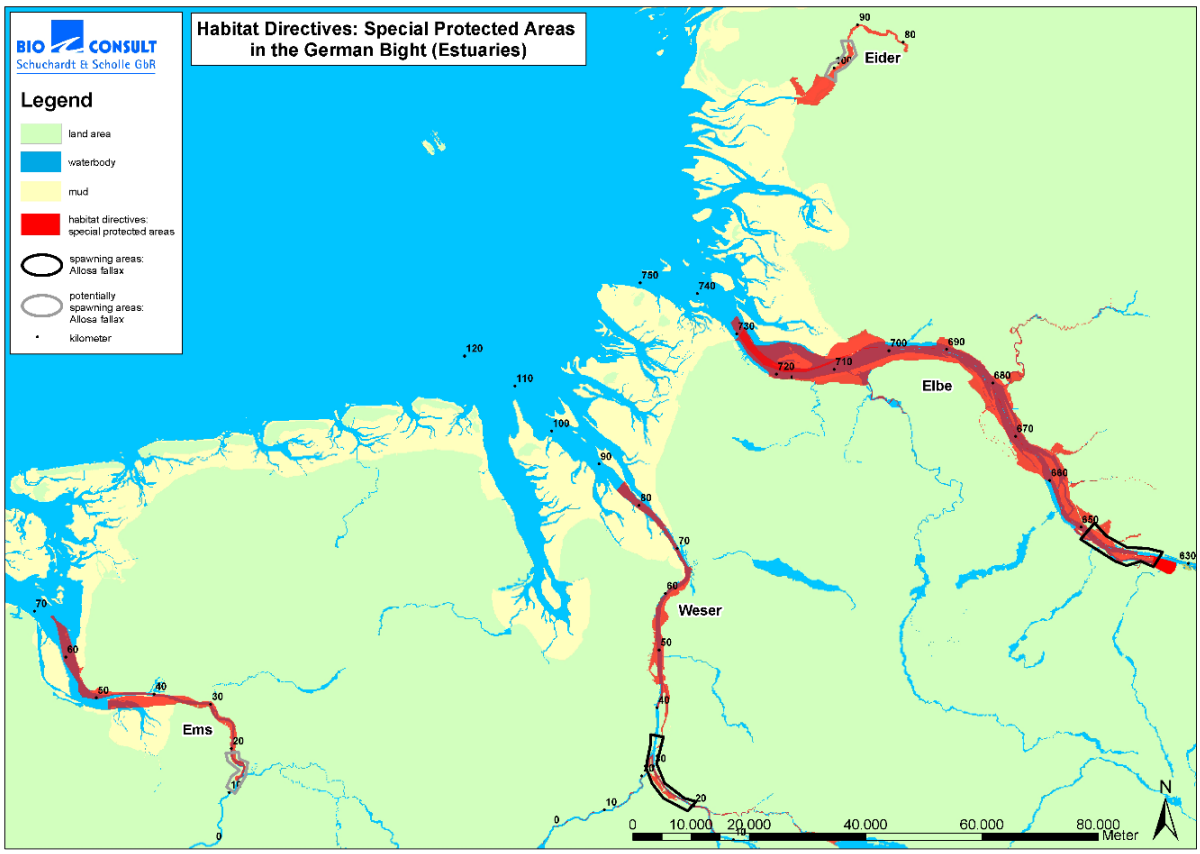


Figure 6 German North Sea estuaries in which twaite shad is an HRA feature.

Monitoring in German estuaries is the responsibility of two different federal authorities:

1. LAVES (Department Inland Fisheries in Lower Saxony) - Ems and Weser estuaries.
2. NLWKN Stade (Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency) - Elbe and Eider estuaries

Both bodies use BioConsult Schuchardt & Scholle GbR to undertake monitoring programmes.

2.2.1 Shad in the Elbe data sources

In this report we have used data from the Elbe as this is by far the most important of the four German North Sea estuaries. The Weser does show some shad recruitment in most years, but the population is smaller than in the Elbe. The populations in the Ems and Eider are negligible. By not including the Weser population in the total population estimate used to assess Sizewell C effects we have been precautionary.

Data were kindly provided by:

Behörde für Umwelt und Energie – Hamburg“ (Daten 2000-2012) und „Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küstenschutz und Naturschutz – Betriebsstelle Stade“ (Daten 2013-2018)

(Authority for Environment and Energy - Hamburg" (data 2000-2012); and Lower Saxony State Water Management, Coastal Protection and Nature Conservation - Stade Office" (data 2013-2018).)

3 Adult Population Estimates

3.1 Population Estimation

For both Twaite shad and cucumber smelt the spring survey data from selected monitoring stations were used, where possible for the same assessment period (2009-2017) used in BEEM Technical Report TR406.

For the Elbe the results from 2 lower estuary stations were averaged (Gluckstadt and Kollmar) as at these locations fish would much more definitely be migrating to spawn. At the upper estuary locations it was considered that there may have been behavioural artefacts which would bias the results. It was also much more difficult to estimate the channel width due to the more complex bathymetry at these locations.

In the Scheldt uncertainties over fish length frequencies from the published reports meant that only the data from Antwerp were used.

The respective shad and smelt catches were scaled up to 24 hours and then to 30 days (the assumed migration period when the overwhelming majority of fish were migrating up estuary). This is known to be an underestimate for shad as biological sampling showed that the fish caught were in pre-spawning condition for the first 4 weeks of the migration season with approximately 10-20% spent in the next 2 weeks (Magrath and Thiel 2013). Finally, the numbers were scaled to the full estuary width by multiplying by the estuary width at the sampling location divided by the anchor net width (8m). This calculation assumes that fish migrate in the full width of the estuary. The density gradient across the estuary is unknown but that caveat also applies to the monitoring data which was collected at various positions across the width of the estuaries. The use of the anchor net means that the entire water column is sampled but undoubtedly there will have been some avoidance at the edges of the net so the sampled area may in fact be smaller than assumed. The monitoring was predominantly done in the Elbe in the period 0600-2100h at the end of April which would have been in daylight. Some Elbe surveys were done as late as 2 am but there are insufficient data to determine if there is any density variation between day and night. The Scheldt surveys were all done in daylight.

3.2 Shad Population

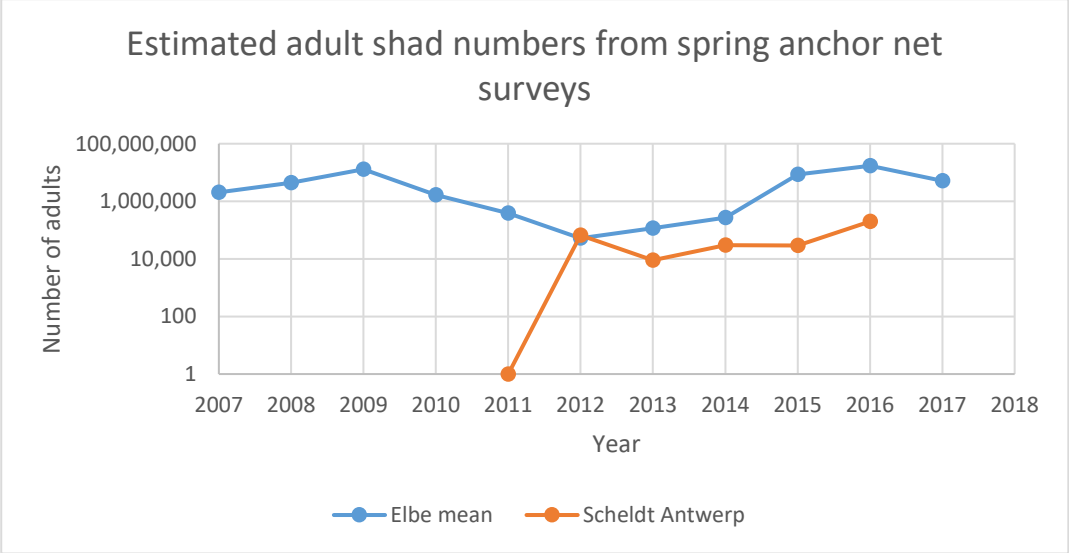


Figure 7 Estimated annual adult shad numbers for the Elbe and Scheldt.

Note the Scheldt catch in 2011 was zero but has been shown as 1 in the Figure 7 so that it could be displayed on the log axis.

Table 1 Annual twaite shad population estimates for the Elbe and Scheldt (Number of adult fish).

Year	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	mean 2009-17
Elbe Kolmar	3,480,000	8,655,853	1,194,067	2,568,000		71,339	210,162	330,353	14,985,000	16,032,990	4,561,920	4,994,229
Elbe Gluckstadt	567,717	92,571	24,698,520	751,546	385,132	32,764	25,138	210,162	1,965,130	18,225,000	5,601,789	5,766,131
Elbe mean	2,023,858	4,374,212	12,946,294	1,659,773	385,132	52,052	117,650	270,258	8,475,065	17,128,995	5,081,855	**5,124,119**
Scheldt Antwerp					1	66,385	8,904	30,300	29,281	198,705		**66,715**
Total shad population	**2,023,858**	**4,374,212**	**12,946,294**	**1,659,773**	**385,133**	**118,437**	**126,554**	**300,558**	**8,504,346**	**17,327,700**	**5,081,855**	**5,161,183**

Note:

a. Grey cell = No survey conducted.
b. The Elbe data were clearly categorised into larvae, sub adults and adults but the Scheldt data relied on length frequency data for apportionment and this was uncertain in some years and abundances are therefore approximate.
c. The Scheldt catch in 2011 was zero but has been shown as 1 allowing it to be plotted on Figure 7 log-axis.

3.3 Smelt Population

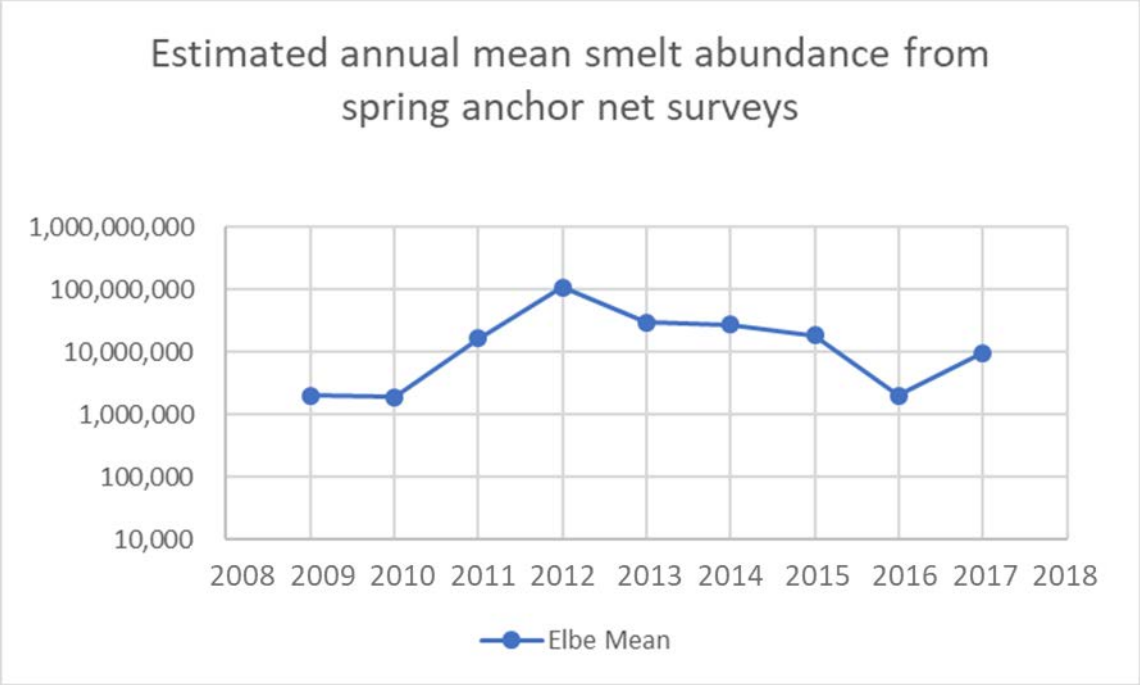


Figure 8 Estimated annual adult smelt numbers for the Elbe

Table 2 Annual cucumber smelt population estimates for the Elbe and Scheldt (Number of adult fish).

Station	2,009	2010	2,011	2012	2,013	2014	2,015	2016	2,017	Mean 2009-2017
Elbe Kolmar	1,397,933	1,152,000		63,016,514	34,011,243	24,217,412	12,096,000	1,282,639	7,153,920	18,040,958
Elbe Gluckstadt	2,624,400	2,705,567	16,450,642	152,156,225	25,012,241	29,685,405	25,198,043	2,733,750	12,162,789	29,858,785
Elbe Mean	**2,011,166**	**1,928,784**	**16,450,642**	**107,586,369**	**29,511,742**	**26,951,409**	**18,647,022**	**2,008,195**	**9,658,355**	**23,861,520**
Scheldt Antwerp							4,264,684	9,647,214		
Total smelt population	**2,011,166**	**1,928,784**	**16,450,642**	**107,586,369**	**29,511,742**	**26,951,409**	**22,911,706**	**11,655,409**	**9,658,355**	**25,407,287**

a. Grey cell = No survey conducted.
b. The Elbe data were clearly categorised into larvae, sub adults and adults but the Scheldt data relied on length frequency data for apportionment and this was uncertain in some years and abundances are therefore approximate. The Scheldt smelt data have not been used in BEEMS Technical Report TR406.
c. Pink cells length frequency data inadequate.

3.4 Shad and smelt population genetics in the southern North Sea

The intention was to undertake comparative genetic studies between fish caught in the Elbe and the Scheldt monitoring programmes and fish caught in East Anglian estuaries and at Sizewell. Fish samples have been obtained and have been frozen but problems created by the COVID-19 virus have prevented those samples from being analysed as at November 2020. It is not yet known when those studies will be able to take place.

4 Conclusions

The following conclusions have been reached in this report:

1. The mean annual adult population estimate for the period 2009-2017 for:

    a. Twaite shad from the Elbe and Scheldt is 5.16M fish.

    b. Cucumber smelt from the Elbe only is 23.9M fish. The timing of the European surveys were not optimal for peak smelt migration which occurred earlier in the year. The smelt population estimates are therefore likely to have been underestimated.

2. There is no known spawning population of twaite shad on the UK east coast. Twaite shad caught at Sizewell are genetically distinct from the known breeding population in the Severn. The most likely origin of the shad caught at Sizewell is fish originating from the Elbe and to a lesser extent the Scheldt. Twaite shad on the European southern North Sea coast have already been found to be from one population. It is recognised that it remains to be proven that the Sizewell twaite shad are part of this single southern North Sea population but based upon the weight of evidence available to date this is considered the most logical conclusion.

3. Cucumber smelt are known to spawn in East Anglian rivers. These fish may also be part of a single southern North Sea population, again centred on at least the Elbe and Scheldt but also including other estuaries. This also remains to be proven.

References

Aprahamian MW, Aprahamian CD (1990) Status of the genus Alosa in the British Isles; past and present. J Fish Biol 37: 257−258.

Aprahamian MW, Baglinière J, Sabatié MR, Alexandrino P, Thiel R, Aprahamian CD (2003) Biology, status, and conservation of the anadromous Atlantic twaite shad Alosa fallax fallax. Am Fish Soc Symp 35: 103−124.

BEEMS Technical Report TR406 v7. Sizewell C – Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft

Berg S (1998) 7.15 Ferskvandsfisk. In: Stoltze M, Pihl S (eds) Rødliste (1997) over planter og dyr i Danmark. Skov- og Naturstyrelsen, Copenhagen, p 165−169.

Breine, J., Van Thuyne G., De Bruyn L. (2012), Opvolging van het visbestand van de Zeeschelde met ankerkuilvisserij: resultaten voor 2012. INBO.R. 2012.38. Rapporten van het Instituut voor Natuur- en Bosonderzoek 2012 (INBO.R. 2012.38], Instituut voor Natuur- en Bosonderzoek, Brussel.

Fricke R, Berghahn R, Rechlin O, Neudecker T, Winkler H, Bast H, Hahlbeck E (1994) Rote Liste und Artenverzeichnis der Rundmäuler und Fische (Cyclostomata & Pisces) im Bereich der deutschen Nord- und Ostsee. In: Nowak E, Blab J, Bless R (eds) Rote Liste der gefährdeten Wirbeltiere in Deutschland, Vol. 42. Kilda Verlag Greven, p 157−176.

Goudswaard, P.C. & J. Breine (2011). Kuilen en schieten in het Schelde-estuarium. Vergelijkend vissen op de Zeeschelde in België en Westerschelde in Nederland. Rapport C139/11, IMARES & INBO. 35 pp.

Jager Z, Bolle L, Dänhardt A, Diederichs B, Neudecker T,Scholle J, Vorberg R (2009) Fish. Thematic Report No. 14. In: Marencic H, Vlas J de (eds) Quality status report 2009, Vol. 25. Trilateral Monitoring and Assessment Group, Common Wadden Sea Secretariat, Wilhelmshaven, p 1−41.

Jolly, M.T., Aprahamian, M.W., Hawkins, S.J., Henderson, P.A., Hillman, R., O Maoileidigh, N., Maitland, P.S., Piper, R., M.J., G. (2012). Population genetic structure of protected Allis shad (Alosa alosa) and twaite shad (Alosa fallax). Mar. Biol. 159, 675–687.

López, MA, Gázquez, N, Olmo-Vidal, JM, Aprahamian, MW, Gisbert, E. (2007) The presence of anadromous twaite shad (Alosa fallax) in the Ebro River (western Mediterranean, Spain): an indicator of the population's recovery?J. Appl. Ichthyol. 23: 163−166.

Maes J, Stevens M, Ollevier F. (2005) The composition and community structure of the ichthyofauna of the upper Scheldt estuary: synthesis of a 10-year data collection (1991–2001). J. Appl. Ichthyol. 21: 86−93

Maes J, Stevens M, Breine J. (2008) Poor water quality constrains the distribution and movements of twaite shad *Alosa fallax fallax* (Lacépède, 1803) in the watershed of river Scheldt. Hydrobiologia 602: 129−143.

Magrath, V. and Thiel, R. (2013). Stock recovery, spawning period and spawning area expansion of the twaite shad *Alosa fallax* in the Elbe estuary, southern North Sea. Endang. Species Res. Vol 20 109-119.

Maitland, P.S. (2003). The status of smelt *Osmerus eperlanus* in England, Conserving Natura 2000 Rivers Ecology Series No. 5.

Neudecker T, Damm U. (2005) Maifische an der deutschen Nordseeküste—zum Auftreten von Finte (Alosa fallax) und Alse (Alosa alosa). Inf Fischforsch 52: 43−50.

Sabatino, S., Alexandrino, P. (2012). Genetic diversity and population structure of the Eurasian shads *Alosa alosa* and *Alosa fallax*. CIBIO report AARC project - Activity 4, December 2012.

Scholle, J. (2011). Status of the Anadromous Twaite Shad (*Alosa fallax*) in the Wadden Sea Estuaries – assessment according to EU directives. Wadden Sea Day presentation 2011.

Scholle J, Schuchardt B (2012) A fish-based index of biotic integrity − FAT-TW an assessment tool for transitional waters of the northern German tidal estuaries. Coastline Rep 18: 1−73.

Stelzenmüller V, Maynou F, Ehrich S, Zauke G (2004) Spatial analysis of twaite shad Alosa fallax (Lacepède, 1803), in the southern North Sea: application of non-linear geostatistics as a tool to search for Special Areas of Conservation.Int. Rev. Hydrobiol. 89: 337−351.

Taverny C, Elie P. (2001) Répartition spatio-temporelle de la grande alose Alosa alosa (Linné, 1766) et de l'alose feinte Alosa fallax (Lacépède, 1803) dans le Golfe de Gascogne.. Bull Fr Peche Piscic 362/363: 803−821.

Thiel R, Backhausen I. (2006). Survey of Natura 2000 Fish Species in the German North and Baltic Seas. In: von Nordheim H, Boedeker D, Krause JC (eds) Progress in marine conservation in Europe. Springer, Berlin, p 157−178.

Thiel, R, Riel, P, Neumann, R, Winkler, HM, Böttcher, U, Gröhsler, T. (2008). Return of twaite shad Alosa fallax (Lacépède, 1803) to the southern Baltic Sea and the transitional area between the Baltic and North Seas. Hydrobiologia 602: 161−177.

Thiel R., Winkler H., Böttcher U., Dänhardt A., Fricke M.R., George R., Kloppmann M., *et al.,* (2014). Rote Liste und Gesamtartenliste der etablierten Neunaugen und Fische (Petromyzontida, Elasmobranchii & Actinopterygii) der marinen Gewässer Deutschlands. - Naturschutz und Biologische Vielfalt.

Tulp, I., Bolle, L. J., Dänhardt, A., de Vries, P., Haslob, H., Jepsen, N., van der Veer, H. W. (2017). Fish. Wadden Sea Quality Status Report 2017. Wilhelmshaven: Common Wadden Sea Secretariat.

Turan C, Basusta N. (2001). Comparison of morphometric characters of twaite shad (Alosa fallax nilotica, Geoffroy Saint-Hilaire, 1808) among three areas in Turkish seas. Bull Fr Peche Piscic 362/363: 1027−1035.

Wilson, K., and Veneranta, L. (Eds). (2019). Data-limited diadromous species – review of European status. ICES Cooperative Research Report No. 348. 273 pp. Available at: http://doi.org/10.17895/ices.pub.5253

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP101

NOT PROTECTIVELY MARKED

SPP101 - Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell

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Executive summary

Consultation with statutory stakeholders on the Sizewell C (SZC) fish entrapment predictions presented in BEEMS Technical Report TR406 v6 resulted in requests for additional information on potential effects on smelt in the Alde-Ore transitional water body. Further evidence has been requested for the use of thermal avoidance thresholds and a detailed assessment of the abundance and length of impinged smelt for each month monitored against the range of tidal conditions and thermal plume from the operating Sizewell B station.

This report addresses these requests by presenting evidence of smelt impingement collected as part of the Sizewell B Comprehensive Impingement Monitoring Programme (CIMP) relative to tidal state and in relation to predicted temperatures extracted from Sizewell B thermal discharge modelling.

Data on the Sizewell B (SZB) thermal plume at the sea bed and sea surface at the location of the SZB intakes were extracted from an annual run of the validated General Estuarine Transport Model (GETM) model of Sizewell for the year 2009 (BEEMS Technical Report TR302). These data were analysed against impingement data from the BEEMS CIMP programme for that year. Four more years of impingement monitoring were also analysed to increase the sample size for the analysis of potential tidal effects to five years. The years 2014-2017 were selected as they correspond to Environment Agency WFD smelt monitoring data collected at the Alde-Ore.

This report also considers the length frequency distribution of impinged smelt to determine differences in impingement of different life stages.

Conclusions

The impingement results show that the largest number of smelt were impinged during the summer months. Approximately 5.4% of all smelt impinged were caught during the period February to April when adult smelt migrate up estuary to reproduce (131 samples). In the warmer summer months of May to September 84.8% of the total smelt impingement occurred (344 samples). Most fish impinged throughout the year were in the 1-year and 2-year age class, with some older fish between 3- and 6-years old present in very low numbers during the period February to April.

Over the five years of impingement monitoring data presented here, neither tidal state nor elevation had any significant effect on impingement rates of smelt. This remained the case when three different lag time offset scenarios were employed to represent the possible presence of a time delay between fish entering the SZB intakes and being caught in the CIMP sampling nets, as suggested by Turnpenny *et al*., (1994).

Detailed thermal modelling to investigate the influence of modelled temperature regimes on impingement rates of smelt for the year 2009 showed that significantly more fish were impinged in warmer temperatures, corresponding with the species' summer feeding pattern. Between 13 – 28.4% (range depending on offset scenarios) of smelt were impinged when absolute water temperatures at the seabed exceeded 21.5˚C. This occurs less than 3% of the time during the year at the position of the intakes, with maximum annual temperature predicted to be 22.6°C. This demonstrates that smelt did not avoid the area of the intakes when water temperatures reached their maximum. Within the temperature range predicted in the location of the intake, no effect of excess temperature on impingement was found, up to a 2.5°C uplift at the bottom of the water column (indicative of the intake) and 4°C at the surface. Periods where excess water temperatures are >3°C at the seabed are rare (1.7% per annum) and did not correspond to CIMP sampling dates. The data did not provide any evidence of thermal avoidance, within the temperature ranges observed.

Analysis of smelt length distribution shows that significantly smaller fish were impinged at warmer background and absolute temperatures- this is considered to be a seasonal effect of smaller fish utilising the offshore waters as summer feeding grounds. No effect of excess temperature, tidal elevation or tidal state on the length distribution of impinged fish was observed. Analysis of the length-frequency distribution of smelt impinged annually and each month showed no evidence that large fish are more likely to avoid the area of the intakes with increasing excess seabed temperatures up to 2.5°C.

The wider WFD assessment for Alde-Ore smelt will be addressed in a subsequent report once the full evidence is collated and the sensitivity of the Alde-Ore Transitional Fish Classification Index (TFCI) to any potential changes in smelt abundance has been assessed.

Changes in Revision 3:

Version 2 of this report was submitted to the Environment Agency in May 2020 and results were presented at a Marine Technical Forum with stakeholders on 23rd September 2020. Version 3 specifically addresses the Environment Agency query that "*it was unclear from the data provided in SPP101 if larger smelt avoid the area with an increasing background and absolute temperature".*

Section 2.2.2 investigates potential relationships between temperature (absolute, background and most relevantly for the thermal plume, excess temperature) on the length of smelt impinged, both annually and for each month sampled. Analysis of the length-frequency distribution of smelt impinged annually and each month showed no evidence that large fish are more likely to avoid the area of the intakes with increasing excess seabed temperatures up to 2.5°C.

1 Background

The cucumber smelt, *Osmerus eperlanus*, is an anadromous species, which lives primarily in the marine environment but migrates up estuary to spawn in early spring (February to April), after which adults return to sea (Maitland, 2003). Smelt was once a common species in the British Isles, however, overexploitation and pollution has contributed to its decline. Smelt occur in around 36 water courses in England and Wales, with large populations in the rivers Thames, Humber and Dee, the Wash and Great Ouse, as well as in water courses of the Norfolk Broads (Colclough and Coates, 2013). Smaller populations exist in the rivers Alde-Ore, Ribble and Conwy, and recovery of supposed extinct populations seems to be underway in the rivers Tyne and Mersey (Colclough and Coates, 2013). Smelt are listed as a Priority Species in Section 41 of the Natural Environment and Rural Communities (NERC) Act 2006.

Smelt migrations from the sea are not well studied in the UK with the exception of the population that reproduces in the River Cree which drains into the Solway Firth in Scotland (which is the best studied of all Scottish smelt populations). In that population, prior to the spawning run, adult smelt form pre-spawning shoals in the estuary from October/November. The fish then move into the lower reaches of the Cree to spawn over a few days from February to March. After hatching fry, are swept down to the upper estuary where they feed on zooplankton (Galloway Fisheries Trust, 2018). Juveniles use the estuary as a nursery ground before moving to outer estuaries in their first winter (in December/January in the lower Thames at Thurrock; Power and Attrill, 2007) and subsequently to sea for summer feeding.

In the Baltic, pre-spawning shoaling smelt concentrations form in estuaries in November with spawning taking place in the period March to early June depending upon water temperature; spawning starts and finishes earlier in the south (Shpilev *et al.,* 2005). In the Elbe, smelt are the most abundant fish species and biomass peaks in the lower estuary in autumn/winter when pre-spawning shoaling aggregations form that subsequently move upriver to spawn in March to April (Sepulveda *et al.,* 1993).

It is, therefore, reasonable to expect similar behavioural patterns for smelt in east coast English estuaries including the Alde-Ore. In which case, any migration of mature adult smelt heading along the Sizewell coast for the Alde-Ore (southwards) or for the Yare (northwards) would be expected in the period October to November of each year, with 1+ juveniles appearing in large numbers in summer. Impingement levels of mature adult smelt at Sizewell would, therefore, be expected to be very low in the period December to April when adult fish would be expected to have migrated into the estuaries.

Smelt form part of the fish component captured on the cooling water drum screens during impingement sampling at Sizewell B nuclear power station (SZB) in Suffolk and are predicted to be captured in the proposed Sizewell C (SZC) development. The origin of smelt in the coastal waters around Sizewell and in Suffolk were considered to belong to a population associated with the Norfolk Broads and the estuarine and

brackish waters around Great Yarmouth and Lowestoft (Maitland, 2003). More recent, comparative genomic analyses concluded that smelt from Sizewell and from the River Thames, Waveney, and Great Ouse are genetically homogeneous with no genetic structuring seen within the region (BEEMS Technical Report TR423). It is considered probable, but not yet proven, that the smelt impinged at SZB originate from a southern North Sea stock and very large numbers have been observed in the River Scheldt in Belgium. The nearest estuary to the north of Sizewell is the Blyth, approximately 12km to the north of the existing power station. Smelt migrate up estuary in the period from February to April. Surveying using fyke nets and kick sampling methods was carried out in the tidal and estuarine areas of the Blyth in April and May 2016 when spawning migrations would have been expected if the fish were spawning in the river. No smelt were found in the area, the absence being attributed to an absence of suitable spawning substrate and a barrier to migration (BEEMS Technical Report TR382). It is therefore considered highly unlikely that there is a spawning population in the Blyth. The nearest estuary with a known spawning smelt population is the Alde-Ore, 25 km to the south of Sizewell (BEEMS Technical Report TR382).

Under Water Framework Directive classifications, the Alde-Ore estuary is a heavily modified transitional waterbody (GB520503503800) routinely monitored by the Environment Agency in accordance with the requirements of the WFD. The WFD assesses the ecological status of transitional waterbodies, for which fish communities are a biological quality element (BQE). To assess the status of fish communities in transitional waters a multi-metric index, termed the Transitional Fish Classification Index (TFCI) has been developed (Coates *et al*., 2004; WFD-UKTAG, 2014). The status of the fish community in the Alde-Ore was classified as ‘good’ in 2015 and 2016 (Environment Agency Catchment Data Explorer). Smelt is an indicator species for the TFCI. Indicator species are ‘disturbance-sensitive species’ and smelt are selected as an indicator due to their sensitivity to dissolved oxygen (Coates *et al*., 2007).

As part of ongoing consultation on the predicted effects of Sizewell C on fish, concerns have been raised about the potential for the proposed Sizewell C development to cause deterioration of the WFD status of the Alde-Ore transitional water body. It has been suggested that due to (a) the potential for the smelt migration to the Alde-Ore to be disrupted by thermal avoidance of the Sizewell cooling water discharge plumes and the (b) the potential for smelt impingement during cooling water abstraction could significantly reduce the size of the Alde-Ore smelt population.

The purpose of this report is to address one specific concern and provide a “*detailed assessment of abundance and length frequency of impinged smelt, for each month monitored - against the range of tidal conditions assessed - including the interaction of the thermal plume with the intake location for SZB during these times*”. The report considers the following evidence:

- The seasonal abundance and length distribution of smelt impinged at SZB is presented to determine the seasonal impingement rates of smelt.
- Detailed analyses are undertaken to determine if fish size determines impingement probabilities at SZB during different tidal conditions or thermal uplifts.
- Analyses consider if thermal uplifts, absolute temperature, or tidal conditions have a bearing on the total number of fish impinged at SZB.

Whilst this report focuses on monitoring data to elucidate the implications of the tidal state and temperature on impingement it also provides directly applicable context to the relevance and suitability of thermal avoidance thresholds for smelt at Sizewell. However, wider WFD concerns will be addressed in a subsequent report collating all the evidence and assessing the sensitivity of the Alde-Ore TFCI for any potential changes in smelt abundance.

2 Implications of tide and temperature on impingement

2.1 Methodology

To analyse the impingement rates of smelt at SZB, data were compiled from the Comprehensive Impingement Monitoring Programme (CIMP). Data from five years of impingement monitoring were analysed from the years 2009 and, 2014, 2015, 2016 and 2017. The year 2009 was selected as this year corresponded to the representative year for the validated Sizewell GETM model (BEEMS Technical Report TR302), allowing impingement monitoring results to be directly compared to the modelled Sizewell B thermal plume dynamics with temperature data extracted from the seabed and sea surface at the exact location of the intakes. The second group of four years impingement monitoring was included to increase the sample size for tidal effects to five years. The years 2014-2017 were selected as they correspond to WFD smelt data collected at the Alde-Ore.

CIMP samples are gathered over a 24-hour period, which is split into six hourly samples during the day and an 18-hour bulk overnight sample. For the purposes of this study, bulk samples were discarded as it is not possible to determine the state of the tide or thermal uplifts at the point smelt were impinged. Therefore, all samples used in this analysis were collected during daylight hours. 198 hourly samples were collected in 2009, 77 samples in 2014, 118 samples in 2015, 141 samples in 2016 and 124 samples in 2017. A total of 658 one-hour samples were taken, of which 36.9% were sampled during flood tide, 46.8% were sampled during ebb tide and 16.3% were sampled during slack water. The tidal conditions and sampling effort for each year are presented in Table 1.

There is likely to be a delay between fish abstracted at the headworks being caught in impingement monitoring nets, due to transit time of fish through the intake tunnels and residence time in the forebay (Turnpenny *et al*., 1994). The exact lag between abstraction and impingement is not known precisely. To account for delays, a one-hour and two-hour offset from the point of impingement monitoring was applied, allowing the range of ambient conditions fish would have experienced at the point abstraction to be accounted for.

The station does not continuously run at full capacity and samples were normalised to the number of fish at full capacity based on the number of pumps in operation at the time of sampling. GETM assumes the station operating at full capacity and during periods of reduced pumping the thermal plume may slightly differ from the predicted plume.

2.1.1 Tidal state

In order to determine the tidal state associated with each hour-long sample, the tidal elevation was taken from the midpoint of each sample to create a tidal profile, from which the tidal state, flood, slack or ebb was established. Tidal data were taken directly from Sizewell for the years 2009, 2017, and 29th September – December 2016. The precise tidal data at Sizewell were not available for the remaining years and was taken from the Lowestoft tide gauge and adjusted by plus 75 minutes and -1.5m to account for the time difference between Lowestoft and Sizewell. The tidal dynamics between the two sites are similar, but not identical. Tidal differences due waves and weather conditions between the two sites, which may alter the tidal patterns from the modelled tides and can affect the exact translation of the tide wave along the coast on a day-to-day basis may result in small errors. However, it is assumed that the +/- error on each data point would tend toward a fixed average.

2.1.1.1 Transit time offset

As suggested by Turnpenny *et al*., (1994) a time lag may be present between fish entering the SZB intakes and being caught in the CIMP sampling nets. To address this, data were analysed using three different scenarios.

1. Impingement monitoring and abstraction at the intakes with no time offset (i.e. there is minimal delay between enter the intakes and being sampled).

2. Impingement monitoring is offset by one hour (i.e. fish enter the intakes an hour earlier than they are sampled in the net).

3. Impingement monitoring is offset by two hours (i.e. fish enter the intakes two hours earlier than they are sampled in the net).

Table 1. The percentage distribution of samples at each tidal state (flood / ebb / slack) for each year at no transit time offset, minus -1 hr time offset and minus -2 hr time offset.

Year	No time offset			-1 hr offset			-2 hr offset		
	Flood	Ebb	Slack	Flood	Ebb	Slack	Flood	Ebb	Slack
2009	42.4	41.9	15.7	46.0	37.9	16.2	46.5	37.9	15.7
2014	18.2	64.9	16.9	28.6	54.5	16.9	37.7	45.5	16.9
2015	29.7	52.5	17.8	32.2	49.2	18.6	36.4	44.9	18.6
2016	39.7	44.7	15.6	40.4	43.3	16.3	42.6	41.8	15.6
2017	43.5	40.3	16.1	46.0	36.3	17.7	47.6	34.7	17.7
5-year average	**36.9**	**46.8**	**16.3**	**40.3**	**42.7**	**17.0**	**43.0**	**40.3**	**16.7**

2.1.2 Effect of temperature on impingement

Detailed thermal modelling using GETM was undertaken for the year 2009 (BEEMS Technical Report TR302), enabling temperatures at any given time point to be matched to impingement monitoring data. The influence of modelled temperature regimes on impingement rates of smelt was investigated in detail from the impingement data from 2009. Impingement monitoring began in 2009 and consisted of 36 sample visits, the most of any year in the period 2009-207. In 2009 impingement monitoring commenced in February Sampling took place during November (7 samples), when pre-spawning shoaling smelt concentrations may occur within estuaries, and February to April (10 samples) when smelt migration upstream occurs and late arrivals may transit past the station, as such sufficient sampling to account for migratory smelt during these periods (Table 2).

Table 2. The seasonal sampling distribution for 2009, including raw and raised numbers of smelt at full pumping capacity. No visits were conducted in January or October.

Period	Number of visits 2009	Number of smelt impinged (raw)	Number of smelt impinged at full pumping capacity
January	0	0	0
February	3	0	0
March	4	0	0
Q1 total	**7**	**0**	**0**
April	3	1	1.3
May	3	32	48.07
June	3	16	41
Q2 total	**9**	**49**	**90.37**
July	4	74	126
August	4	105	195.8
September	1	23	30.6
Q3 total	**9**	**202**	**352.4**
October	0	0	0
November	7	15	19.7
December	4	15	20.3
Q4 total	**11**	**30**	**40**
Total	**36**	**281**	**482.77**

The temperature parameters used in this report were all determined from GETM and are as follows:

1. Background temperature: the ambient temperature in the absence of a power station, this is presented as it is reflective of the situation in the absence of the station operating.

2. Thermal uplift: the change in temperature (ΔT) above background resulting from thermal discharges from the Sizewell B station.

3. Absolute temperature: the background temperature + the thermal uplift from the station outfall.

Data were extracted from the model at both the seabed and sea surface, at the coordinates of the Sizewell B intake headworks. The seabed is most relevant due to the position of the intake headworks[1], but both are presented. Thermal uplifts and absolute temperature predicted in GETM assumes full pumping capacity at all times of the year, therefore there is the potential for small mismatches with the actual absolute and uplift temperatures if pumping capacity is reduced. Background temperature is independent of pumping rate.

2.1.3 Statistical methods

The relationship between length of impinged fish and environmental variables (tidal state and elevation in the full dataset encompassing 2009 to 2017, and temperature in the 2009 dataset) was analysed using linear regressions. Impacts of fish length and environmental variables on impingement were explored using generalised linear models and assuming that impingement follows a negative binomial distribution. The pluriannual nature of the data was taken into account in an interaction term between years and the explanatory variables. Normality of the residuals was verified for each model. Impacts of fish length and environmental variables on impingement were explored using generalised linear models. Impingement was

[1] The mid-depth layer temperature was also extracted from the model. Results were very similar to the seabed, which was considered to be most reflective of the thermal conditions experienced by impinged fish.

assumed to follow a negative binomial distribution, which accounts for uneven sampling during extreme conditions.

The coefficients and associated p-values of the models are reported. The exponentiated coefficients estimated by the generalised linear models can be interpreted as the multiplicative effects on the expected number of smelt impinged. In the multiannual analysis, the effect of interaction between years and explanatory variable was tested using nested models. All statistical analyses were conducted using the statistical software R (R Core Team, 2019) and the package MASS (Venables and Ripley, 2002).

2.2 Results and Interpretation

2.2.1 Seasonal Impingement rates at SZB

Impingement rates of smelt varies throughout the year, with more smelt impinged in the warmer summer months (Figure 1). August is the peak month for smelt impingement, with a mean of 1.56 $x10^{-5}$ smelt per m^3 (equivalent to 69.4 fish per day at full 51.5 cumec capacity). June and July are the next highest months with 1.24 $x10^{-5}$ m^3 and 1.34 $x10^{-5}$ m^3 impinged, respectively. Impingement rates were low during the winter months with <1 $x10^{-5}$ smelt per m^3 impinged from October to April (Table 3).

Over the years 2009, 2014, 2015, 2016 and 2017, only 5.4% of all fish impinged were caught during the smelt migratory period, February to April (131 samples). In comparison, the warmer summer months of May to September accounted for 84.8% of the total catch (344 samples).

The majority of fish impinged throughout the year were in the 1- and 2-year age class, with some older fish between 3- and 6-years old present in low numbers (Figure 2). 50% of smelt mature at year 1 and 100% mature by year 2 (BEEMS Technical Report TR383). Larger smelt are observed during the migratory period in February and March, albeit in low abundance (Table 3). A seasonal component to the length distribution of fish impinged was evident. During the winter when fewer fish were impinged the length distribution was statistically larger, whilst in the summer more smaller fish were impinged (Figure 2).

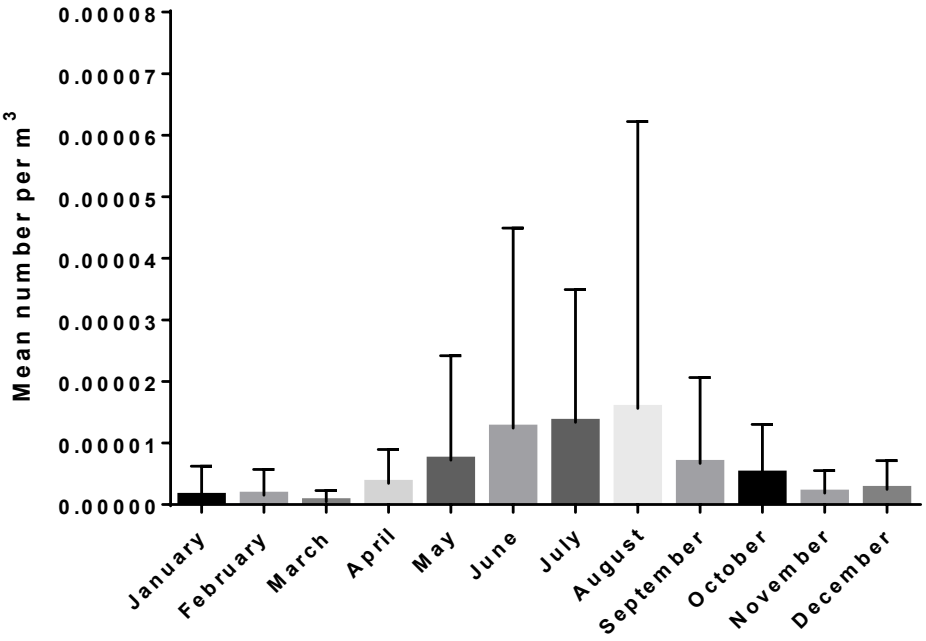

Figure 1. Seasonal smelt, *Osmerus eperlanus*, mean impingement rates per m^3 at SZB, assumes full pumping capacity equates to 51.5 cumecs (185400 m^3 per hour). over the years 2009, 2014, 2015, 2016 and 2017, mean numbers of fish normalised to numbers at full pumping capacity, error bars show standard deviation.

Table 3. Monthly impingement rates of smelt, *Osmerus eperlanus*, at SZB over the years 2009, 2014, 2015, 2016 and 2017, including total number of samples, total and mean numbers of fish impinged hourly normalised to full pumping capacity and standard deviation. Assumes no time offset between intakes and sampling nets.

Month	Measure	Flood	Ebb	Slack	Combined
January	Total samples:	13	10	4	27
	Total Fish:	4.00	0.00	2.67	6.67
	Mean (+/- s.d.)	0.31 (1.11)	0.00 (0.00)	0.67 (1.33)	0.25 (0.91)
February	Total samples:	7	20	5	32
	Total Fish:	1.33	7.62	0.00	8.95
	Mean (+/- s.d.)	0.19 (0.50)	0.38 (0.93)	0.00 (0.00)	0.23 (0.78)
March	Total samples:	19	18	8	45
	Total Fish:	1.33	2.67	0.00	4.00
	Mean (+/- s.d.)	0.07 (0.31)	0.15 (0.43)	0.00 (0.00)	0.09 (0.34)
April	Total samples:	21	24	9	54
	Total Fish:	2.67	26.66	5.24	34.57
	Mean (+/- s.d.)	0.13 (0.40)	1.11 (1.22)	0.58 (0.93)	0.64 (1.02)
May	Total samples:	22	23	9	54
	Total Fish:	33.40	25.00	14.00	72.40
	Mean (+/- s.d.)	1.53 (4.07)	1.09 (1.86)	1.56 (3.50)	1.34 (3.14)
June	Total samples:	31	31	13	75
	Total Fish:	43.44	107.38	21.67	172.50
	Mean (+/- s.d.)	1.40 (3.67)	3.46 (8.18)	1.67 (4.14)	2.30 (6.03)
July	Total samples:	33	43	14	90
	Total Fish:	81.78	79.41	62.00	223.19
	Mean (+/- s.d.)	2.48 (3.83)	1.85 (3.10)	4.43 (6.07)	2.48 (4.00)
August	Total samples:	17	48	12	77
	Total Fish:	57.80	152.61	12.50	222.91
	Mean (+/- s.d.)	3.40 (8.74)	3.18 (9.64)	1.04 (1.96)	2.89 (8.64)
September	Total samples:	15	25	8	48
	Total Fish:	23.30	22.00	14.33	59.60
	Mean (+/- s.d.)	1.55 (3.61)	0.88 (2.03)	1.79 (1.89)	1.24 (2.58)
October	Total samples:	7	24	5	36
	Total Fish:	3.00	27.00	3.00	33.00
	Mean (+/- s.d.)	0.43 (0.53)	1.13 (1.75)	0.60 (0.89)	0.92 (1.50)
November	Total samples:	34	26	12	72
	Total Fish:	9.60	10.53	4.90	25.03
	Mean (+/- s.d.)	0.28 (0.67)	0.41 (0.73)	0.41 (0.61)	0.35 (0.67)
December	Total samples:	24	16	8	48
	Total Fish:	7.08	9.70	5.30	22.08
	Mean (+/- s.d.)	0.29 (0.71)	0.61 (0.72)	0.66 (1.42)	0.46 (0.86)

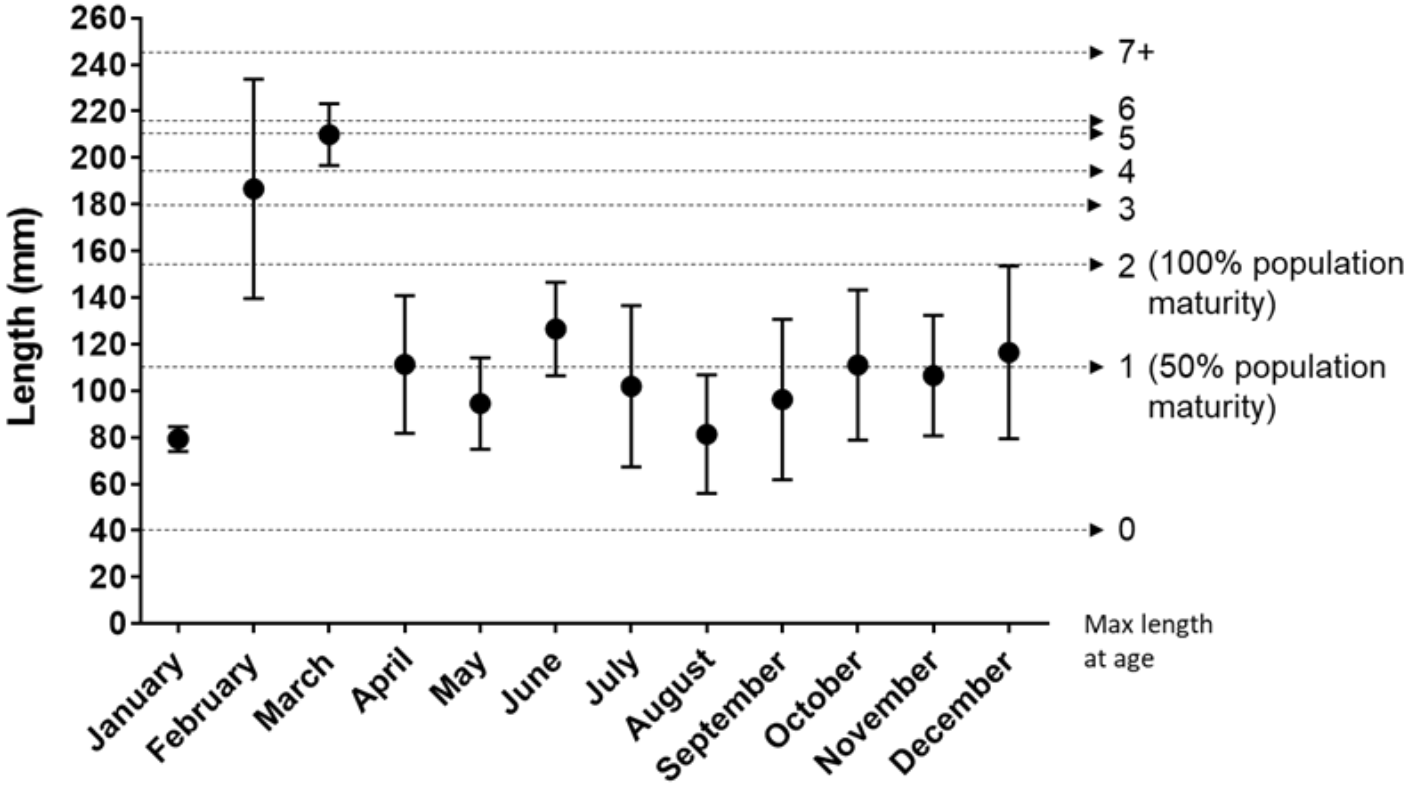

Figure 2. Seasonal length distribution of smelt, *Osmerus eperlanus*, impinged at SZB over the years 2009, 2014, 2015, 2016 and 2017. Mean lengths with error bars showing the standard deviation. Dotted arrows represent the maximum length at each age group (data from BEEMS Technical Report TR383).

2.2.2 Influence of temperature, tidal state and elevation on length of impinged smelt

There is the potential for different life-history stages to have different sensitivity to thermal uplifts or utilise the tidal phases differently. To address this, the influence of temperature (2009 data only), tidal state and elevation (all years) on length of impinged smelt was investigated.

2.2.2.1 The influence of length on impingement

The model showed statistically significant differences in impingement due to the interaction between length and years ($p = 0.00047$), indicating that the trend between length and impingement varies between years. However, the absence of significant interaction in the 2014-2017 dataset ($p = 0.055$) indicates that this effect is caused by the trend observed in 2009 during which relatively smaller fish were impinged. Models applied to 2014-2017 and 2009 datasets indicate that length had no significant effect on the impingement count ($p = 0.19$).

2.2.2.2 The influence of temperature on length

The 2009 impingement data was interpreted to determine if there was a statistically significant association between temperature and the length of impinged fish. A statistically significant effect of the surface background and absolute temperatures on length was found ($p < 0.001$ and $p < 0.001$, respectively). This was also observed in the bottom background and absolute temperature ($p < 0.001$ and $p < 0.001$, respectively). The data consistently show that smaller fish were impinged at warmer background and absolute water temperatures.

This pattern is consistent with the seasonal impingement data whereby smaller fish are impinged in the summer months, in contrast thermal uplift was not shown to have an effect. No effect of excess temperature on length was observed over the course of the year (surface: $p = 0.798$, bottom: $p = 0.351$) (Figure 3).

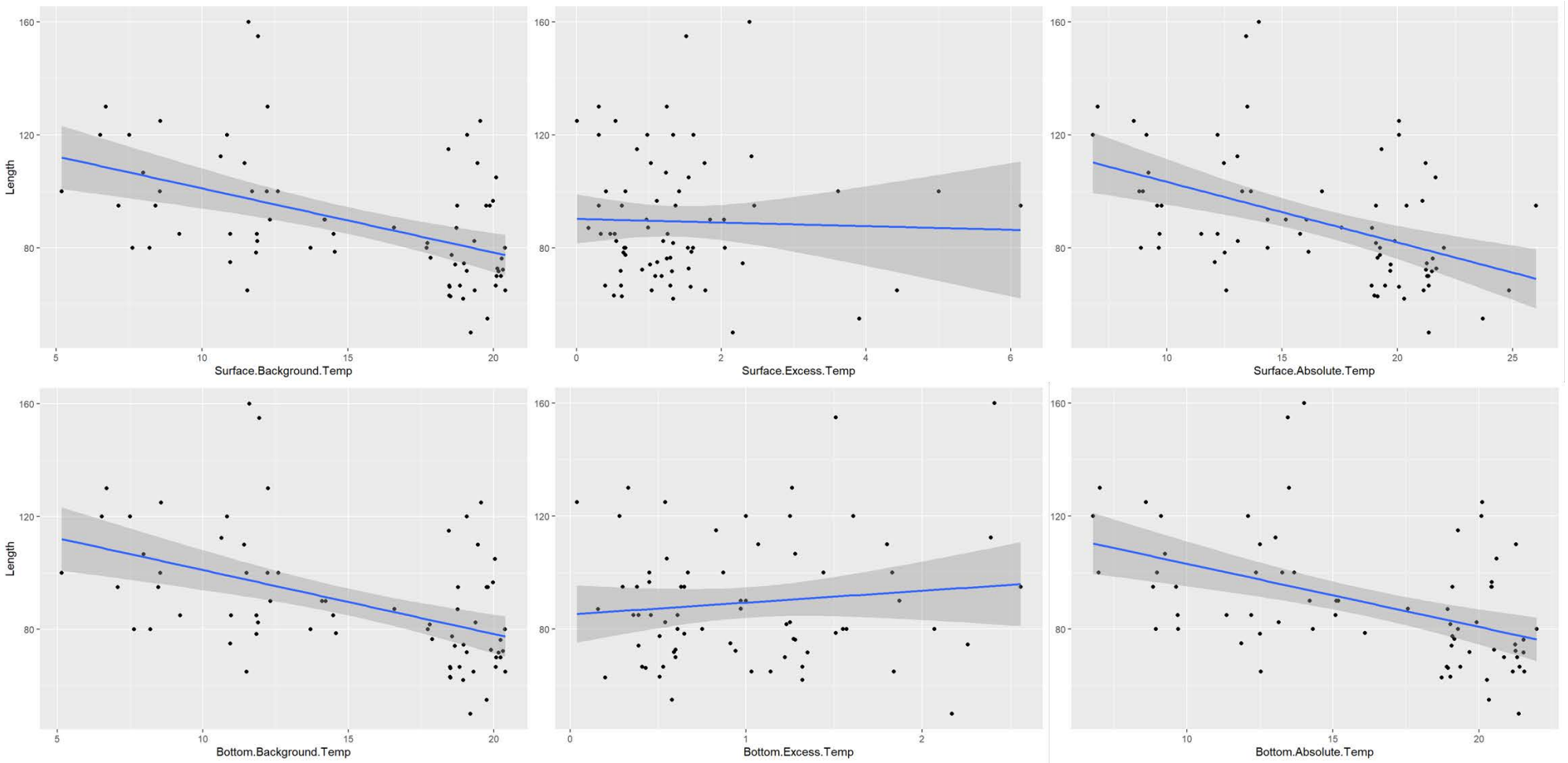

Figure 3. Smelt, *Osmerus eperlanus*, length distribution impinged at SZB according to surface and bottom background temperatures, excess temperatures and absolute temperatures with no transit time offset. Data from the year 2009.

It has been established that smaller fish are typically caught in the summer. Therefore, to remove seasonal effects, the 2009 data were interpreted on a monthly basis to determine if there was a statistically significant association between temperature and the length of impinged fish at different periods of the year. The relationship between size and temperature (background, excess, absolute) was analysed for each month. This allows seasonal effects to be removed and better enables the question of whether larger smelt avoid the area of the intakes with increasing temperatures.

In 2009, the year coincident with the thermal plume modelling, impingement sampling commenced in February. No smelt were impinged during the migratory period from February to April, this is consistent with the low numbers of smelt caught during this period in the full data set analysed 2009, 2014-2017 (Figure 1). Insufficient numbers of smelt were impinged in June to do meaningful analyses. Therefore, analyses focused on smelt impinged in May, July, August, September, November, and December (Table 4). In additional to analysing the raw data, fish were also grouped into three size classes based on the quartiles of the length distribution: small (<68.75), medium, and large (>95) and analysed using a cumulative linear model. The majority of smelt were year-1 fish, where 50% maturity occurs, however, some fish in the >95mm group would be mature year-2 fish. The largest individual caught was 160mm (Table 4).

No effect of excess temperature on fish length was observed in any month tested, except November, when an increase in bottom temperature was associated with an increase in the size of fish impinged ($p = 0.02$; Figure 4). However, the number of smelt impinged in November was low ($n = 15$). The results provide no indication of avoidance of the plume by larger fish up to 2.5°C uplifts at the seabed.

No consistent trend in the length of fish impinged with absolute or background temperatures was observed between months using the raw data or size class information.

These data does not provide any evidence that large fish are more likely to avoid the area of the intakes with increasing temperatures or thermal uplifts.

Table 4. Monthly impingement of smelt, *Osmerus eperlanus*, at SZB for 2009, including total number of samples, size (min/max/mean) and standard deviation for each month sampled.

Month	Number of smelt	Size (mm)			Stdev
		Minimum	Maximum	Mean	
May	30	65	105	81.7	9.0
July	74	40	130	74.5	19.0
August	104	50	120	73.8	11.3
September	23	65	125	82.2	12.9
November	15	65	160	103.7	28.1
December	15	80	130	104.7	17.4

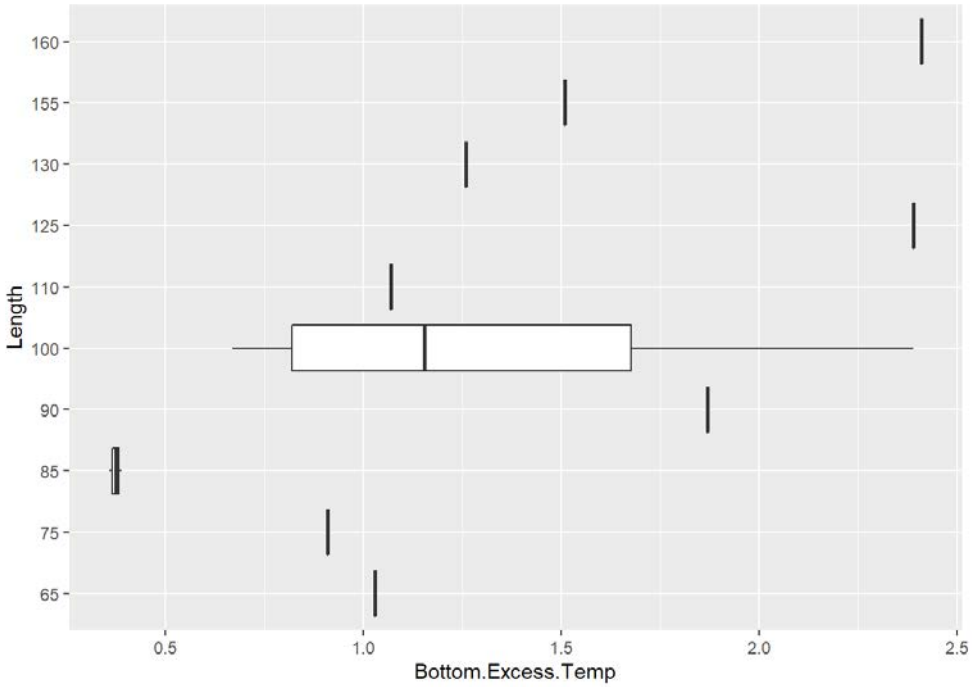

Figure 4. The length of smelt, *Osmerus eperlanus*, impinged in November 2009 relative to modelled excess bottom temperature.

2.2.2.3 The influence of tidal state and elevation on length of impinged smelt

The full data series was interpreted to determine if there was a statistically significant association between tidal state and the length of impinged fish within and between years. The models indicate a statistically significant interaction between tidal elevation and year, when considering data from 2009-2017 (p=0.0179) caused by the opposing trends in 2016 and 2017 (Figure 5). As there was no general trend, pluriannual data were pooled to determine if there were any effects of elevation and tidal state across the five years. No effect of either tidal elevation (p=0.587) or tidal state on length was observed (p=0.725) (Figure 5 and Figure 6).

Given the lack of a general trend of tidal state on the size of impinged fish the factor was removed when considering tidal state on impingement numbers.

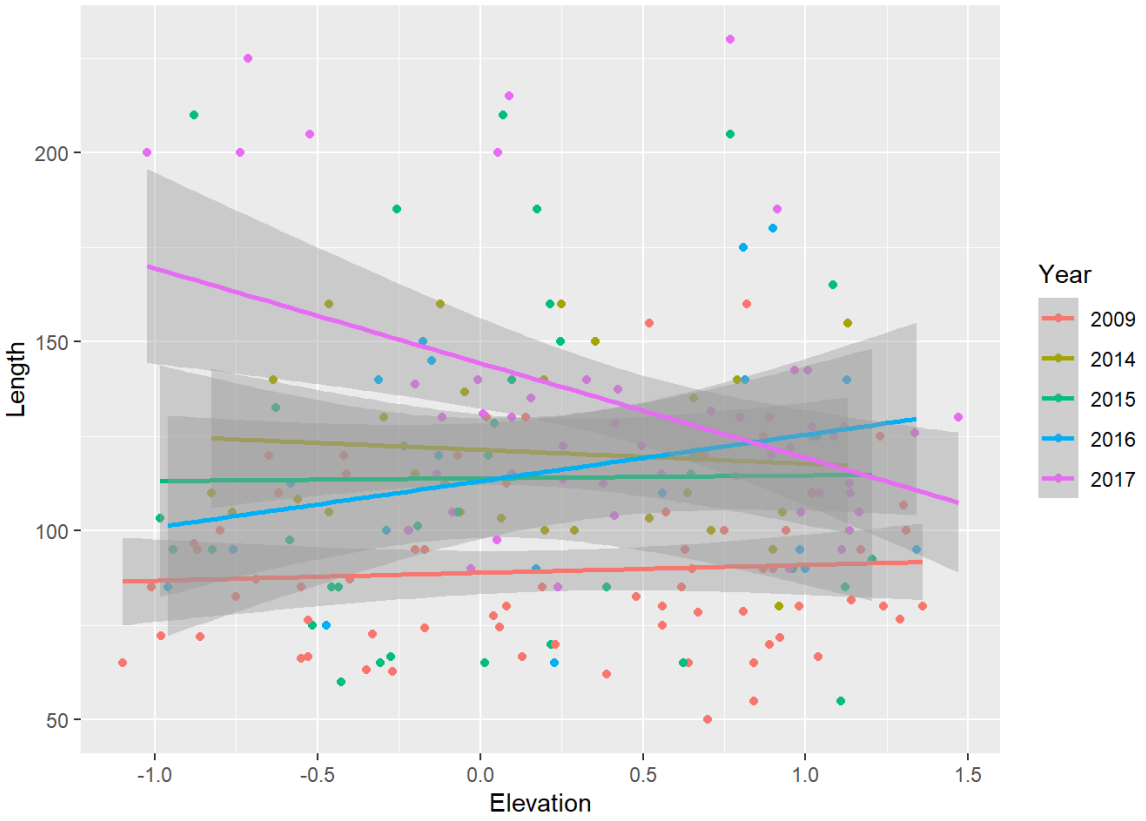

Figure 5. Smelt, Osmerus eperlanus, length distribution at SZB according to tidal elevation. Data from the years 2009 - 2017.

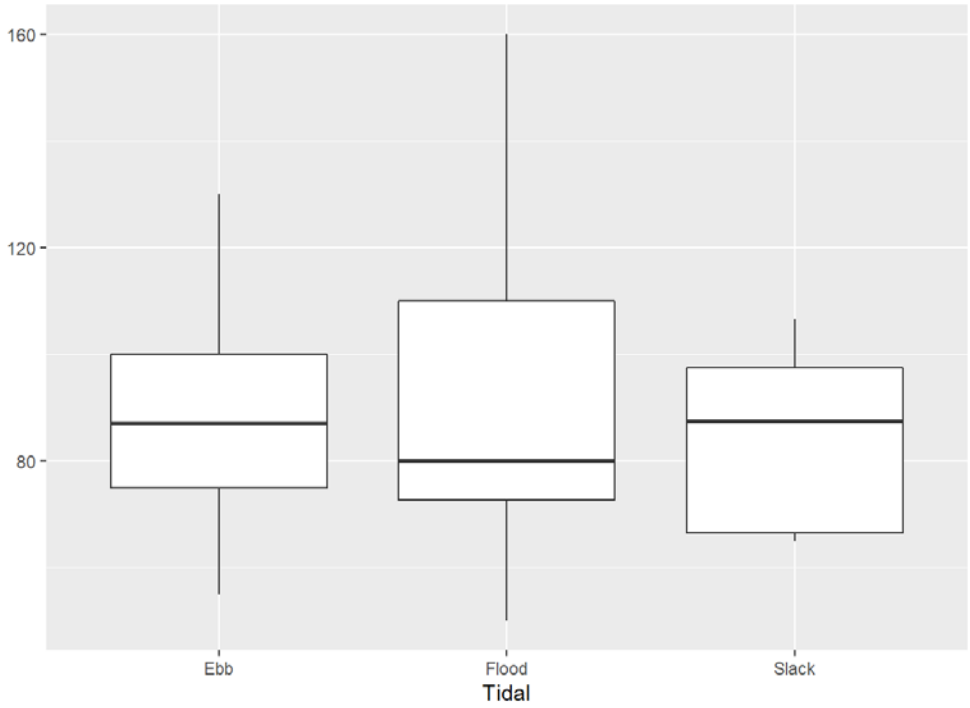

Figure 6. Smelt, *Osmerus eperlanus*, length distribution at SZB according to tidal state. Data from the years 2009, 2014, 2015, 2016 and 2017.

2.2.3 Influence of tidal state on impingement

The dataset was analysed to determine whether impingent rates had any tidal bias, according to the flood, ebb or slack tide, or if it there is an effect of tidal elevation.

Table 5. Results (p-values) of the statistical analyses on the influence of tidal state on smelt impingement rates over the years 2009, 2014, 2015, 2016 and 2017. ns = not significant; * represents significant p-values.

	No time offset	-1 hr offset	-2 hr offset
Elevation*Year	**0.00973 (*)**	0.05222 (ns)	0.3183 (ns)
Elevation	NA	0.87594 (ns)	0.4494 (ns)
Tidal State*Year	0.07011 (ns)	0.48347 (ns)	0.10893 (ns)
Tidal State	0.18952 (ns)	0.08506 (ns)	0.073812 (ns)

These results indicate statistically significant interaction between elevation and year at the 5% significance level in the no time offset dataset (p=0.00973) meaning a test for elevation was not applicable. This seems to be due to a decreasing trend with tidal elevation in 2014 and an increasing trend in 2017, the remaining years have no clear trends. No other statistical effect of tidal state or elevation is found and there is no general trend (Table 5, Appendix A, Figure 9 and Figure 10). These results suggest that both tidal state and elevation do not significantly influence impingement rates of smelt. It is possible the different trends in 2014 and 2017 could be a random effect or the result of an additional variable that changed between years, which is not captured.

Due to the requirement to use two different tidal data sets, one from Sizewell and the other adapted from the Lowestoft tide gauge, there are some caveats to this assessment of elevation. Despite the caveats, this

method of assessment is sufficient to determine the windows of flood, ebb and slack tidal state. It is clear from the data that no real trend between tidal conditions and impingement rates occurs.

2.2.4 Influence of temperature on impingement

The dataset was analysed to determine whether temperature has an influence on smelt impingement rates. Temperature data were determined by the GETM, both the model and the impingement rate assume full pumping capacity at all times.

Table 6. Results of the statistical analyses on the influence of temperature on smelt impingement rates in 2009. ns = not significant; * represents significant p-values.

Data	Analysis	Surface Excess Temperature	Surface Absolute Temperature	Bottom Excess Temperature	Bottom Absolute Temperature
No time offset	estimate	-0.2713	0.2692	-0.06724	0.28553
	p-value	0.0867 (ns)	**< 2E-16 (*)**	0.81787 (ns)	**< 2E-16 (*)**
-1h time offset	estimate	0.05054	0.27782	0.375	0.28459
	p-value	0.75171 (ns)	**< 2E-16 (*)**	0.195 (ns)	**< 2E-16 (*)**
-2h time offset	estimate	-0.3016	0.26665	0.1158	0.27979
	p-value	0.0572 (ns)	**1.6E-15 (*)**	0.6775 (ns)	**< 2E-16 (*)**

Absolute bottom and surface temperatures were found to have a significant positive effect on impingement rates at the 5% significance level, with more fish impinged in warmer water temperatures (Table 6, Figure 7). This corresponds with the seasonal impingement pattern (Figure 1). In 2009 between 13 – 28.4% (range depending on offset) of smelt were impinged when absolute water temperatures at the seabed exceeded 21.5˚C, which occurs less than 3% of the time during the year at the position of the intakes with maximum annual temperature predicted to be 22.6°C. This demonstrates that smelt did not avoid the area of the intakes when water temperatures reached their maximum.

The thermal uplift caused by the outfall plume has the potential to impact impingement rates if smelt avoid or are attracted to this plume during certain conditions. No effect of excess temperature on impingement was found. Periods where excess water temperatures are >3°C at the seabed are rare (1.7% per annum) and did not correspond to CIMP sampling. However, there was no indication of reduced smelt impingement when predicted thermal uplifts at the seabed exceeded 2.5°C. Whilst the data are sparse, surface uplifts of above 4°C did not appear to have any effects on smelt impingement (Figure 8).

There is a poor association between thermal uplifts and tidal state due to the position of the SZB intakes relative to the outfall, which are in an east-west configuration compared to the north-south trajectory of the local tides. As such, the tidal effects and thermal effects are largely uncoupled.

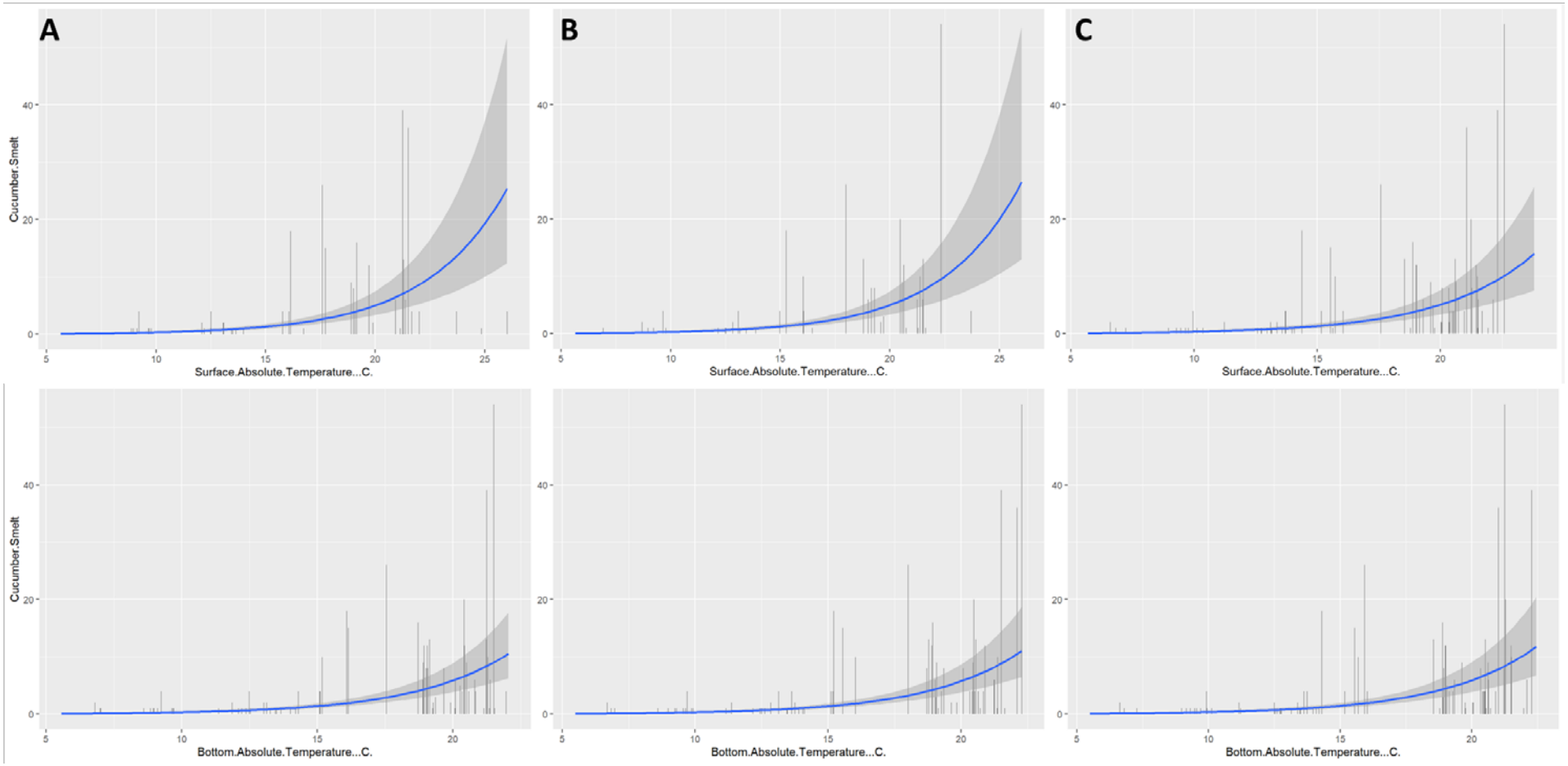

Figure 7. Smelt, *Osmerus eperlanus*, impingement at SZB according to absolute surface and bottom temperatures, with no transit time offset, A - no transit time offset, B - minus 1 hr time offset and C - minus 2 hr time offset. Data from the year 2009 mean numbers of fish normalised to numbers at full pumping capacity. Note differing temperature scales.

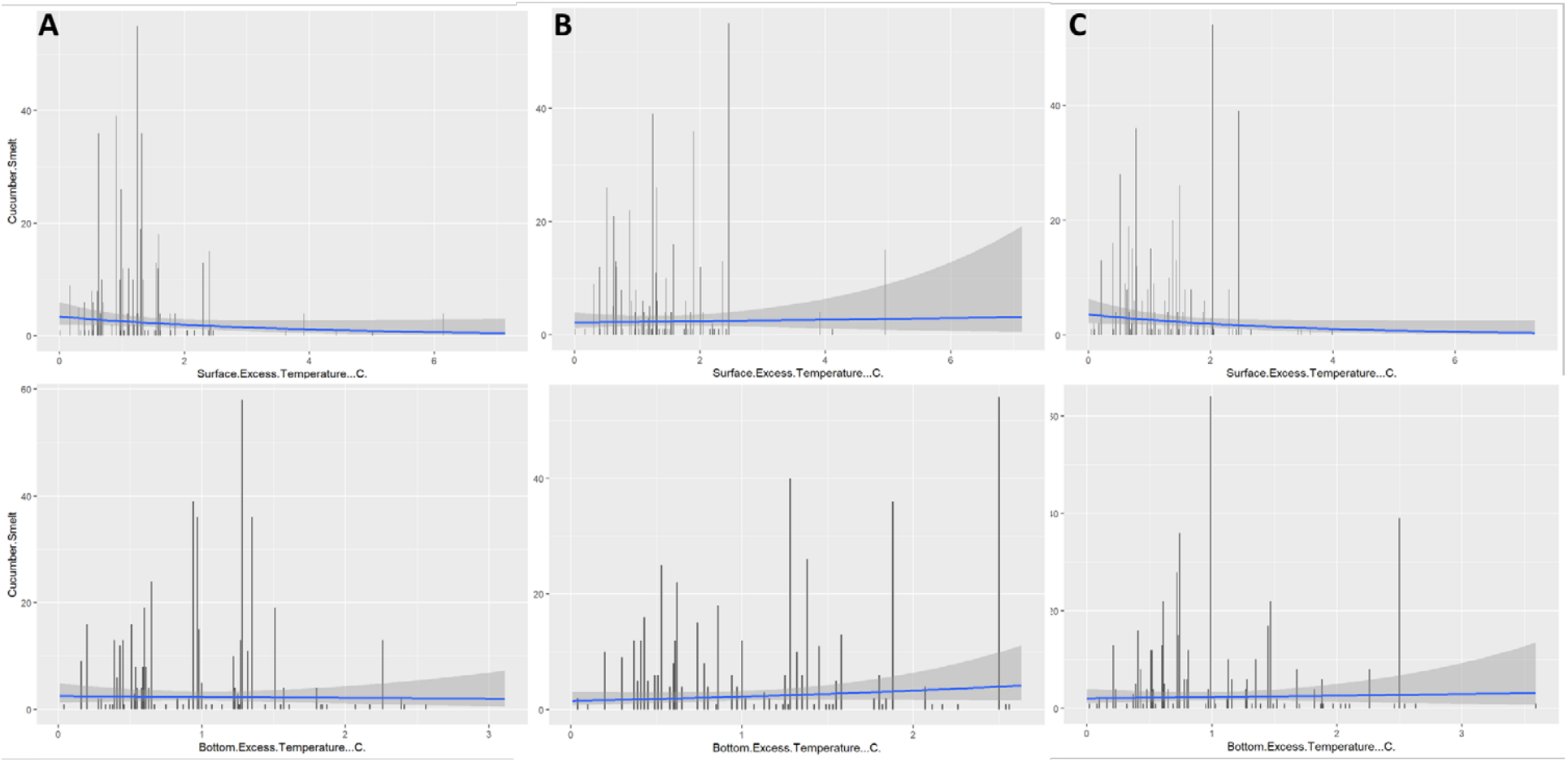

Figure 8. Smelt, *Osmerus eperlanus*, impingement at SZB according to excess surface and bottom temperatures, with no transit time offset, A - no transit time offset, B - minus 1 hr time offset and C - minus 2 hr time offset. Data from the year 2009. Note differing temperature scales.

3 Summary

The results of this study show that low numbers of smelt are present in impingement samples at SZB throughout the year. Highest periods of impingement occur in the summer months with a peak in mean impingement rate of 1.56 $x10^{-5}$ smelt per m^3 (equivalent to 69.4 fish per day at full 51.5 cumec capacity) in August. The majority of fish impinged throughout the year were in the 1- and 2-year age class, of which 50% and 100% are expected to be mature, respectively (BEEMS Technical Report TR383).
Very low numbers of older fish, between 3- and 6-years, were impinged during the period between February and April when smelt migrate up estuary.

Over the five years of impingement monitoring data presented here, neither tidal state nor elevation significantly influenced impingement rates of smelt. This remained the case when three different lag time offset scenarios were employed to represent the possible presence of a time delay between fish entering the SZB intakes and being caught in the CIMP sampling nets, as suggested by Turnpenny *et al*., (1994).

Detailed thermal modelling (GETM) to investigate the influence of modelled temperature regimes on impingement rates of smelt for the year 2009 showed that significantly more fish were impinged in warmer temperatures, corresponding with the seasonal impingement pattern. Between 13 – 28.4% (range depending on offset scenarios) of smelt were impinged when absolute water temperatures at the seabed exceeded 21.5˚C. This occurs less than 3% of the time during the year at the position of the intakes, with maximum annual temperature predicted to be 22.6°C. This demonstrates that smelt did not avoid the area of the intakes when water temperatures reached their maximum. Equally, no effect of excess temperature on impingement was found, up to a 2.5°C uplift at the bottom of the water column and 4°C at the surface. Periods where excess water temperatures are >3°C at the seabed are rare (1.7% per annum) and did not correspond to CIMP sampling. The data presented here does not provide any evidence of thermal avoidance, within the temperature ranges observed.

Analysis of smelt length distribution shows that significantly smaller fish were impinged at warmer background and absolute temperatures- this is considered to be a seasonal effect of smaller fish utilising the offshore waters as summer feeding grounds. No effect of excess temperature, tidal elevation or tidal state on the length distribution of impinged fish was observed. Analysis of the length-frequency distribution of smelt impinged annually and each month showed no evidence that large fish are more likely to avoid the area of the intakes with increasing excess seabed temperatures up to 2.5°C.

References

BEEMS. Sizewell C - Impingement predictions based upon specific cooling water system design, Version 6 Technical Report TR406. Lowestoft, UK: Cefas, 2020.

BEEMS. Sizewell- Equivalent Adult Value (EAV) metrics. Technical Report TR383. Lowestoft, UK: Cefas, 2020

BEEMS. Sizewell- Thermal Plume modelling GETM Stage 3 results with the preferred SZC cooling water configuration. Edition 3. Technical Report TR302. Lowestoft, UK: Cefas, 2020

BEEMS. Smelt Populations in the Vicinity of Sizewell. Technical Report TR382. Lowestoft, UK: Cefas, 2019.

BEEMS. The Origins of Smelt (Osmerus Eperlanus) Populations at Sizewell. Technical Report TR423. Lowestoft, UK: Cefas, 2016.

Breine J. and G. Van Thuyne. Opvolging van Het Visbestand van Het Zeeschelde-Estuarium Met Ankerkuilvisserij Resultaten Voor 2014. Brussel:2014.

Britain, G., 2006. The Natural Environment and Rural Communities Act 2006 (Commencement No. 2) Order 2006. Stationery Office.

Coates, S., Waugh, A., Anwar, A., Robson, M., 2007. Efficacy of a multi-metric fish index as an analysis tool for the transitional fish component of the Water Framework Directive. Mar. Pollut. Bull. 55, 225–240.

Coates, S.A., Colclough, S.R., Robson, M. and Harrison, T.D., 2004. Development of an Estuarine Classification Scheme for the Water Framework Directive. Phase 1&2–Transitional Fish Component. Environment Agency Technical Report E1-131/TR, p.42.

Colclough, S., Coates, S., 2013. A review of the status of smelt *Osmerus eperlanus* (L.) in England and Wales – 2013. Report EA/001 to the Environment Agency, York, UK, by SC2, Chatham, Kent.

Environment Agency, 2017. Catchment data explorer.

Gallway Fisheries Trust 2018. Assessing the sparling (*Osmerus eperlanus*) popultion of the River Cree during “Saving The Sparling” project 2018. Commissioned report No JHWAD23, Gallway Fisheries Trust, Newton Stewart.

Maitland P.S. The Status of Smelt Osmerus Eperlanus in England (ENRR516). English Nature Research Reports. English Nature Research Reports. 2003.

Power, M. and J. Attrill, M.J. 2007. Temperature-dependent temporal variation in the size and growth of Thames estuary smelt (*Osmerus eperlanus*). Mar. Ecol. Prog. Ser. 330: 213–222.

R Core Team (2019). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Sepulveda, A., Thiel, R. and Nellen, W. 1993. DISTRIBUTION PATTERNS AND PRODUCTION OF EARLY LIFE STAGES OF EUROPEAN SMELT, *Osmerus eperlanus* L., FROM THE ELBE RIVER.ICES CM 1993/M.39

Shpilev, H., Ojaveer, E., Lankov, A. 2005. Smelt (*Osmerus eperlanus*) in the Baltic Sea. Proc Estonian Acad.

Sci. Biol. 54,3,230-241.

Turnpenny, A. W. H., Wood, R. and Thatcher, K. P. (1994). Fish deterrent field trials at Hinkley Point Power Station. Fawley Aquatic Research Laboratories Ltd. Report to Energy Technology Support Unit, DTI. Report ETSU T/04/00198/REP

Venables, W. N. & Ripley, B. D. (2002) Modern Applied Statistics with S. Fourth Edition. Springer, New York. ISBN 0-387-95457-0

WF D-UKTAG (2014). UKTAG Transitional Water Assessment Method. Fish Fauna. Transitional Fish Classification Index. www.wfduk.org

Appendix A

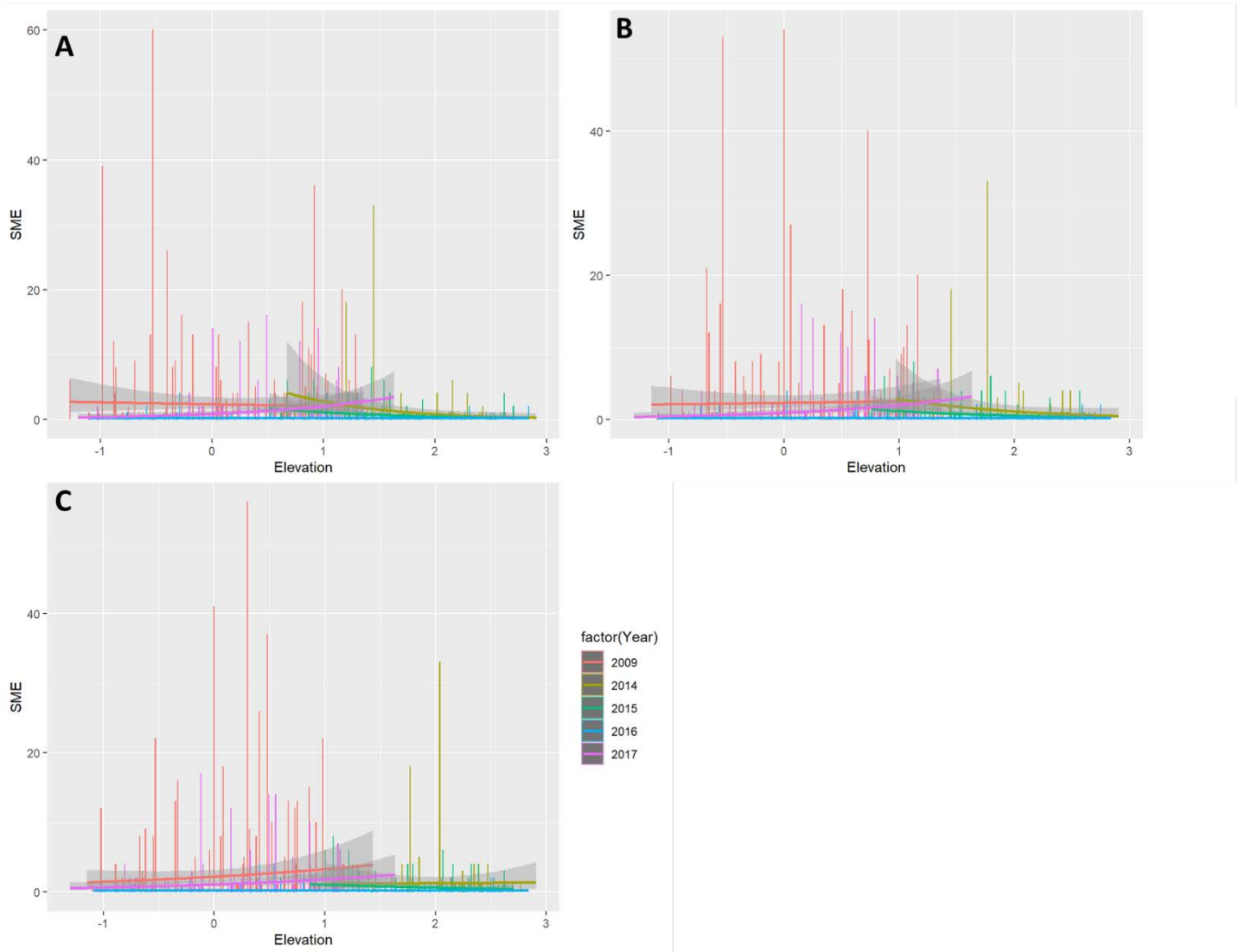

Figure 9. Smelt, *Osmerus eperlanus*, impingement rates at SZB according to tidal elevation, with A - no transit time offset, B - minus 1 hr time offset and C - minus 2 hr time offset. Data from the years 2009, 2014, 2015, 2016 and 2017, mean numbers of fish normalised to numbers at full pumping capacity.

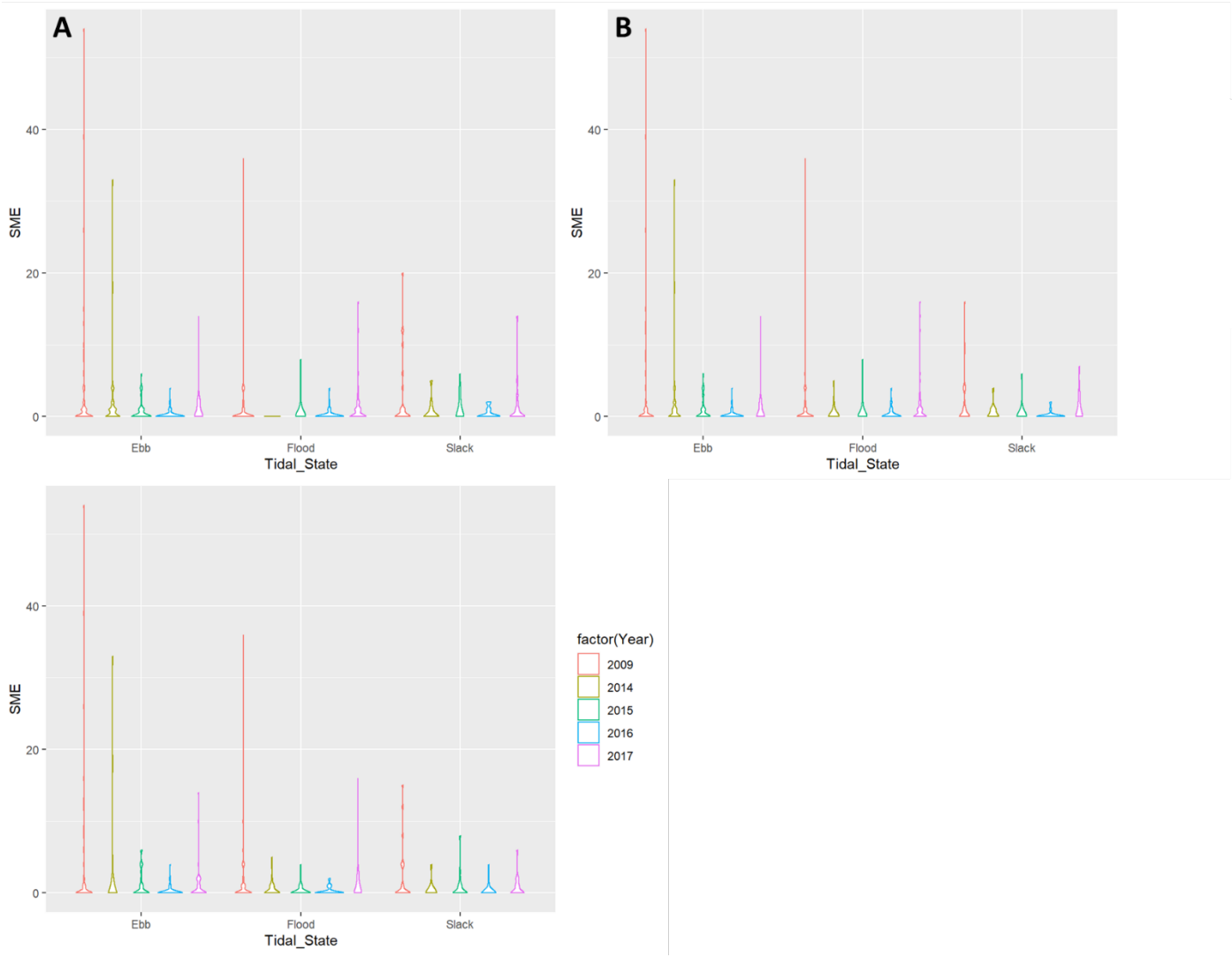

Figure 10. Smelt, *Osmerus eperlanus*, impingement rates at SZB according to tidal state, with A - no transit time offset, B - minus 1 hr time offset and C - minus 2 hr time offset. Data from the years 2009, 2014, 2015, 2016 and 2017, mean numbers of fish normalised to numbers at full pumping capacity.

Table 7. Proportion at age used to apportion smelt length classes, based on published growth parameters (BEEMS Technical Report TR383)

Length class (TL, mm)	Proportion at age								
	0	1	2	3	4	5	6	7+	Total
0	1								1
5	1								1
10	1								1
15	1								1
20	1								1
25	1								1
30	1								1
35	1								1
40	1								1
45		1							1
50		1							1
55		1							1
60		1							1
65		1							1
70		1							1
75		1							1
80		1							1
85		1							1
90		1							1
95		1							1
100		1							1
105		1							1
110		1							1
115			1						1
120			1						1
125			1						1
130			1						1
135			1						1
140			1						1
145			1						1
150			1						1
155			1						1
160				1					1
165				1					1
170				1					1
175				1					1
180				1					1
185					1				1
190					1				1
195					1				1
200						1			1
205						1			1
210						1			1
215							1		1

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP102

NOT PROTECTIVELY MARKED

SPP102 - Use of Spawning Production Foregone EAVs for impingement assessment

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1 Background

For nuclear new build sites (NNB) where there is an existing operational power station, predictions of the impingement of fish and invertebrates at the NNB are based on the observed numbers of individuals impinged by the existing station scaled up to represent the abstraction rate of the new station. To evaluate the impacts of a new build on fish stocks, the predicted losses due to the new station (by number or weight) are compared with a known population estimate, such as the annual Stock Spawning Biomass (SSB) or, in the absence of this, annual international fishery landings as a highly conservative measure of stock abundance.

*The SSB is the combined weight of all individuals in a fish stock that are capable of reproducing. To calculate the spawning stock biomass, you need to have estimates of the number of fish by age group, estimates of the average weight of the fish in each age group and an estimate of the amount of fish in each age group that are mature. For example, if there are 1,000 fish of age three that weigh on average 0.5 kg and 50% of these fish are mature, the spawning biomass of this age group is 250 kg (1000 X 0.5 X 0.5).*Source: Ghislain Chouinard, Chair of the North Sea advice drafting group (ADGNS), International Council for the Exploration of the Sea (ICES), ICES website.

ICES advises public authorities with competence for marine management including the European Commission (EC) and provides peer reviewed estimates of SSBs and compilations of international fisheries landings data**.**

Mortalities of fish and crustaceans caused by power station cooling water intakes usually involve the juvenile part of a population because it is that part that is particularly vulnerable because of their presence in inshore nursery areas and their poorer swimming capability compared with adults (Turnpenny *et al*., 1988). The natural mortality of juveniles is much higher than that of the adults of a species, as juveniles are often prey for other species, or simply have a larger proportion of their natural lives to survive. Consequently, the loss of a single juvenile does not equate to the loss of a breeding adult of the same species. The concept of the Equivalent Adult Value (EAV) factor provides a means to relate the numbers of impinged juveniles to the corresponding number of adults that would survive naturally to enter the spawning population. This allows for a realistic evaluation of the impact of the station.

A previous method of estimating EAVs is described in Turnpenny (1989) where the EAV of a juvenile fish is defined as the average lifetime fecundity of an adult that has just reached maturity which is required to replace that juvenile. However, the approach requires the input of several parameters related to female maturity and fecundity, many of which are difficult to estimate with any level of certainty. Consequently, Turnpenny (1989) only estimated EAV's for six fish species. Although the Turnpenny (1989) EAVs may be applied to other species within similar functional groups, in the absence of an appropriate EAV a precautionary value of 1 must be applied. An EAV of 1 assumes that every juvenile impinged at the station would otherwise have survived to become a breeding adult and so gives rise to an overestimate, often substantial, of the potential impacts on populations.

To overcome the limitations with the Turnpenny EAV approach, a method has been developed based upon well-established fisheries science principles. The approach assumes that it is the level of natural mortality experienced by an individual in the wild between the age at impingement/entrainment and the age at which it would have reached maturity that determines the EAV. This is determined by two biological characteristics, natural mortality and growth. The benefit of this approach is that these two parameters are more readily obtained for a given species or, if they are not available in the literature, could be collected with a moderate amount of effort. Further, the approach allows for EAVs to be calculated for specific geographic areas, reflecting the stock structures of specific populations. A description of the approach and the underlying concepts along with any assumptions of growth or natural mortality used are given in BEEMS Technical Reports TR383 and TR426 for Sizewell and Hinkley Point respectively.

The Environment Agency in their draft note TB010 (Environment Agency 2019) have proposed an alternative method of calculating EAVs based upon the Spawning Production Foregone (SPF) approach. The SPF approach calculates the total loss to the population as the sum of the equivalent adults lost from the

spawning population in a year plus the future spawning potential of lost fish that would otherwise have reproduced over a series of years until mortality).

Spawner Production Foregone is a branch of fisheries science that was developed to determine how much spawning capacity could be taken per year by fishing without affecting population sustainability. The science was adapted for use in the USA to formulate a compensation equation for entrainment and to the knowledge of the authors of this SPP, SPF EAVs have not been used in those studies to produce quantitative impingement effects calculations using SSBs or landings data.

TB010 concludes:

> *"From our review of the available methods, the **Spawning Production Foregone** method is considered by the Environment Agency to be the most appropriate to use to assess the entrapment losses at HPC over the operational lifetime of the station. It addresses many of the factors of relevance in the valuation of lost fish by incorporating natural mortality rates, proportional maturity rates, and repeat spawning potential. The Spawning Production Foregone method takes into account the value of repeat spawning fish, and produces numbers of equivalent adults which are directly comparable to Spawning Stock Biomass.*
>
> *The Spawning Production Foregone method does not consider the effect of fishing mortality. Survival both before and after maturity may be less than indicated by the method if commercial fishing is removing fish from the population with fishing mortality generally beginning at sizes less than 100% maturity. This means that the Spawning Production Foregone method may overvalue older fish to some extent by not considering fishing mortality. However, for key species of concern such as Atlantic cod, and many of the diadromous species, fishing mortality is limited given the current status of the stocks."*

All EAV calculation methods are accounting procedures and the SPF EAV is an extension of the 6 step Cefas methodology to account for future spawning potential by adding three additional steps 7-9 (TB010) to the original 6 steps. The difference in the results between the two methods is that by accounting for the future spawning potential, SPF EAVs can potentially have a value greater than 1 (i.e. a juvenile fish can be assigned a value of greater than 1 adult if the species can spawn repeatedly over multiple years; and if some of those spawners could survive numerous spawning seasons after attaining maturity). The Cefas methodology calculates the number of juvenile fish that could reach maturity and therefore has a maximum value of 1 (i.e. the impinged fish could have an EAV of 1 if the whole catch was mature at point of capture but for most species the EAV is much less than 1 due to the large number of juveniles in the catch which would not survive to maturity due to natural mortality).

By not including SPF, predictions of adult fish losses using EAVs derived via the Cefas methodology are directly comparable with annual SSBs or fish landings whereas impingement calculations based upon SPF are not (Section 6).

In principle it is possible to undertake impingement assessments using SPF EAVs, the question is how that could be done in order to derive meaningful conclusions on population effects. The aim of this report is to answer that question.

2 Purpose of this Report

For stocks where suitable data exist, quantitative assessment of the annual effects of anthropogenic mortality on fish populations (of which fishing is the largest component) is done by comparison with the annual SSB. For impingement assessments, the number of juveniles impinged must be transformed into the equivalent number of mature adults to permit a valid comparison with the annual SSB by use of EAV factors. The calculated impingement effect of an NNB is therefore directly proportional to the EAV i.e. the larger the EAV, the larger the indicated impingement effect. It therefore matters that the EAV calculation is founded on the most appropriate science and that EAVs are then used correctly in an assessment process.

The algorithm used in TB010 makes several biological assumptions that may or may not be valid to derive its calculations and to arrive at the 'population equilibrium' described in the note; however, in this SPP we have neglected these biological assumptions and focussed the study on four straightforward issues:

1. Neglecting the assumptions used to derive population equilibrium, is the algorithm for calculating SPF EAVs correct?

2. TB010 recognises that the method has not included fishing mortality (F). For commercial species F is additional to natural mortality (M) and often impacts before fish are mature and therefore the number of repeat spawning opportunities in future years will be more limited than implied by the calculations in TB010. Is not including F in the calculation of SPF EAVs significant?

3. How can SPF EAVs be used to calculate impingement effects?

4. The Cefas EAV methodology does not include the impact of fishing mortality on the assumption that fishing mortality will not substantially increase total mortality until fish become mature. For completeness we have considered the validity of this assumption as well. Should the EAVs calculated with the Cefas methodology be adjusted for fishing mortality?

To illustrate the differences in the results from the two different EAV approaches we have used impingement data for bass (Stock area: ICES divisions IVbc, VIIa, VIId-h). Bass is a long lived, slow maturing demersal species that would be expected to have very different calculated EAVs using the SPF and Cefas methodologies. Most importantly we have described the steps that would have to be taken to apply an SPF EAV to a practical impingement effect assessment in order to derive meaningful conclusions.

The report reaches some broad conclusions that are described in Section 7. These conclusions are independent of site or of fish species.

3 Data used for the assessment and calculated EAVs

The data used in the assessment are described below. For all calculated EAVs the same values for growth, maturity and natural mortality are used in order to ensure comparability of the results. The impingement dataset is common for both the Cefas and SPF EAV methodologies and uses the Sizewell B Comprehensive Impingement Monitoring Programme (CIMP) data for bass for the period 2009 – 2017 as described in BEEMS Technical Report TR406. The calculated EAV from these data using the Cefas methodology is 0.224 and reflects the predominantly juvenile composition of the impinged bass.

98% of the impinged bass were immature (not yet capable of spawning) and 2 % were mature at the point of impingement. The modal number of impinged bass were aged 2 whereas bass do not begin to mature before age 4, 50% maturity is not reached until ages 5 to 6 and 89% maturity is at age 8 followed by a slow asymptotic rise to 100% maturity at age 14. Large numbers of the impinged bass would therefore be expected to suffer natural mortality before reaching maturity, thereby leading to a small EAV value of much less than 1 with the Cefas methodology (0.224).

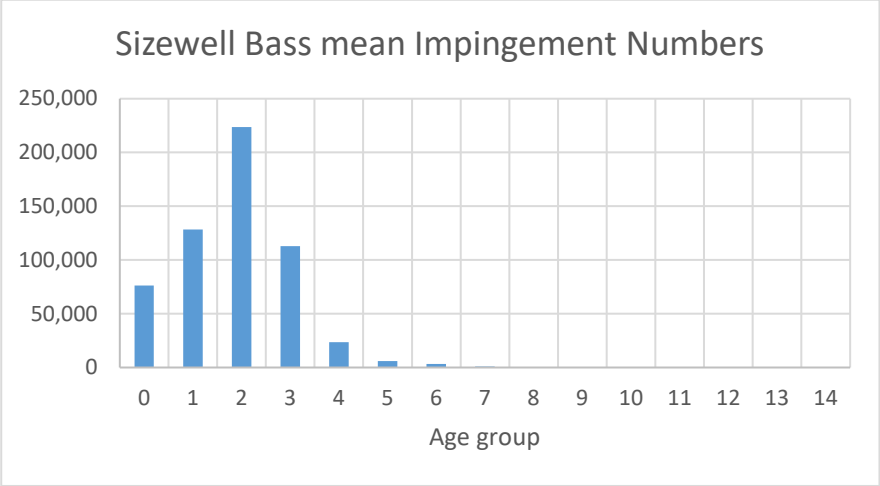

Figure 1 Distribution of mean impingement numbers by age group in the period 2009-2017 (maturity starts to occur at age 4)

The calculated SPF EAV from the original TB010 method is 0.99. Compared with the EAV calculated with the Cefas methodology, the larger SPF EAV reflects the possible contribution of future spawning and indicates that 1 impinged juvenile is nearly equivalent to 1 mature adult. As stated in TB010, the SPF EAV methodology does not include fishing mortality (F) which is additional to natural mortality.

To derive F we have used fishing mortality at age, based on the mean F for bass calculated from the years 2009-2017 (to be consistent with the SZC impingement outputs) from ICES WGCSE 2019 (French fleet which represented a mean of 57% of total bass landings in the period 2009-2017 – ICES WGCSE 2018).

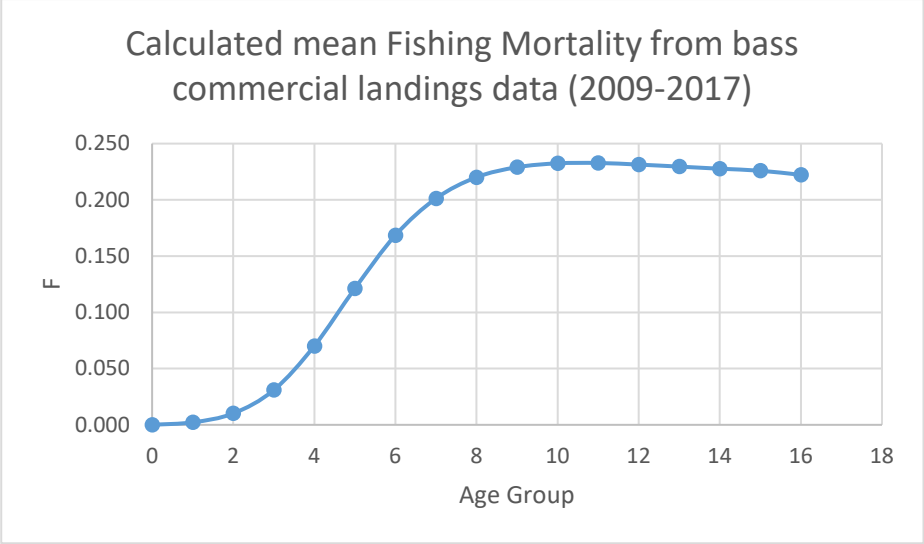

Figure 2 Fishing mortality for bass from ICES WGCSE working group report 2019

The associated landings data are shown in Figure 3.

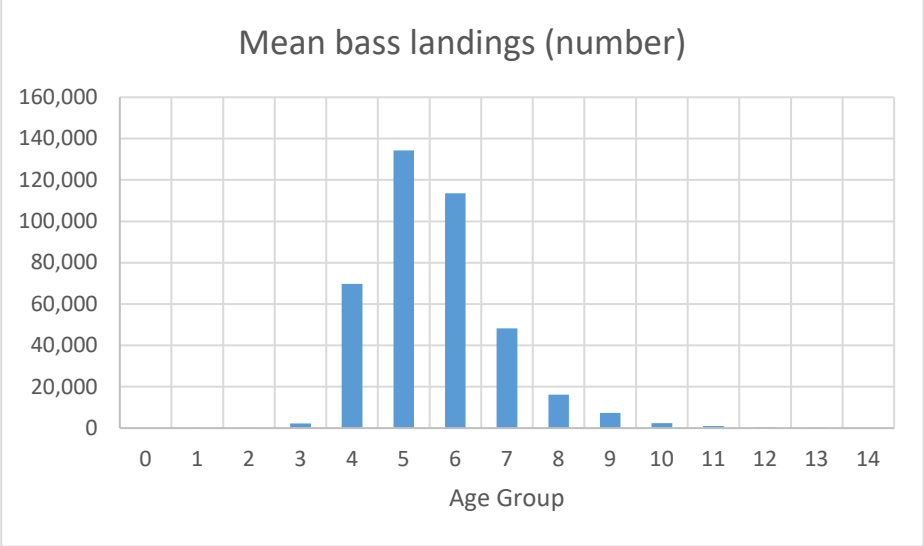

Figure 3 Bass landings number at age (mean 2009 to 2017)

From the landings data it can be seen that the peak landings are at age 5 when only 42% of bass are mature. A total of 52% of the entire commercial landings were estimated to be mature but the remaining 48% would have had no opportunity to reproduce before death. The young age at capture means that proportionately few bass live to spawn in multiple years. These landings underestimate actual fishing catch as the data do not include discards (unwanted fish) which predominantly takes place with fish of age groups 3 and 4, which are too small to land legally. From 2009 to 2014 the estimated discard rate was approximately 5% of the catch, but after 2015 the discard rate is less certain and has been estimated to be in the range 10% to 27% of the catch. The change was due to the change in minimum landing size from 36cm until 2015 and 42 cm thereafter. This had the effect of increasing the discards of fish caught as bycatch in other fisheries (ICES WGCSE 2019). Accordingly, juvenile mortality is understated by the landings data and therefore most fish caught in the commercial bass fishery are in fact immature. The discard rates are considered subject to considerable uncertainty which reduces the accuracy of F at age calculations (See section 6).

4 Are the SPF EAVs calculated correctly in TB010?

This brief discussion ignores the assumptions used to model population equilibrium in TB010 and focuses on the algorithm as described.

We have examined the methodology described in TB010 and have recreated the analysis spreadsheets. The first 6 steps of the 9 step SPF EAV methodology are identical to those described for the Cefas methodology in BEEMS Technical Reports TR383 and TR426. Steps 7-9 are new.

Step 7 contains an error whereby the same natural mortality was applied annually to each age class - Age 0 fish were subject to the same value of M each year. However, with every year, the fish surviving should have been subject to the next year's natural mortality. This has been corrected in the spreadsheets we have used (Excel spreadsheets with “sw modified” in the filename).

Using the SZB impingement data, the SPF EAV is 0.99 using the original TB010 algorithm and 1.05 in the corrected algorithm. All subsequent calculations have used the corrected SPF EAV algorithm.

5 What is the effect of omitting F from the SPF EAV calculations?

Including published values of F into the SPF EAV calculation is simply a matter of replacing all occurrences of natural mortality at length in the algorithm with total mortality at length (Total mortality is equal to fishing mortality plus natural mortality). This reduces the calculated bass SPF EAV to 0.598 from 1.050 (57% of the original SPF EAV) and reflects the fact that far fewer fish would survive to spawn on multiple occasions than estimated in the original TB010 algorithm. However, even after correcting for F, the resultant SPF EAV is still 2.67 times larger than the EAV calculated via the Cefas EAV methodology.

6 How can SPF EAV be used for impingement effect predictions?

ICES' fisheries assessment models do not account for production foregone as they simply reflect the Spawning Stock Biomass (SSB) of fish alive in a particular year. Landings data only describe the number of fish landed in a year. Neither of these two indicators include future spawning potential. As such, comparison of impingement numbers derived with SPF EAVs against SSBs or landings data is an invalid calculation as the measures are fundamentally incompatible.

However, with appropriate manipulation modified SSB and landing numbers can be created that do include future spawning potential. For example, bass landings are simply a much larger form of anthropogenic mortality than impingement losses from an NNB. As such the SPF can be calculated using exactly the same SPF EAV algorithm by just replacing the impingement numbers at age with the landing numbers at age. Similarly, from the estimated annual SSB in a year a modified SSB can be calculated that includes potential future spawning production (i.e. the modified SSB would include the spawning stock in the year of interest plus the predicted future spawning production from that annual SSB). This does, however, expose the questionable biological basis behind the SPF methodology e.g. recruitment in any year is poorly correlated with stock size.

As an example, we have computed the SPF EAV for bass landings to be 2.52 (Including the effects of F). As would be expected the landings SPF EAV is much larger than the impingement SPF EAV at 0.598 because there are far fewer juvenile fish in the landings data (with peak age 5) compared with the impingement data (with a peak age 2) and therefore much lower losses from natural mortality. It is worth reflecting on what the use of the SPF EAV implies. If the actual landings in a year are 1000 fish, the use of the SPF EAV means that the landings represent 1000 fish caught in that year plus the potential production of 1520 fish spread over several future years. This form of assessment is not compatible with ICES' assessments of the sustainability of fish stocks nor with any management regime under the Common Fisheries Policy but gives an idea of how SPF would have to be applied to fisheries management if the concept were to be used.

6.1 Summarised effect calculation – incorrect use of SPF EAVs

Using the Cefas EAV methodology, the calculated impingement effect = Impingement number*impingement EAV/landings = impingement numbers*0.224/landings

If the SPF EAVs are applied only to the impingement numbers the calculated impingement effect = impingement number *SPF EAV/landings = impingement number *0.598/landings (Corrected TB010 EAV methodology including the effects of F).

i.e. The corrected TB010 calculated effect is 2.67 times the Cefas effect value (0.598/0.224) or 4.42 times the Cefas value based on the original TB010 algorithm.

However, the effect calculation using the SPF EAV is incorrect as it has adjusted only one side of the equation for future production whereas to derive a meaningful result the landings have to be multiplied by the appropriate SPF EAV as well. On first principles we would expect the additional SPF EAV factors to approximately cancel out.

6.2 Summarised effect calculation after correcting the landings for SPF.

SPF EAV for landings data = 2.52 (including the impact of F)

So the correct effect calculation using SPF EAVS is:

Calculated impingement effect = (impingement SPF EAV * impingement number)/ (landings SPF EAV * landing number):

Calculated impingement effect = 0.598* impingement number/2.52*landings = 0.237*landings

Compared with the calculated effect using the Cefas EAV of 0.224*landings

As expected, the two results are approximately the same with the calculated SPF EAV impingement effect being 1.058 times the calculated effect using the Cefas EAV methodology (or a difference of less than 6%).

6.3 Limitations on the reliability of SPF EAVs

By adjusting both sides of the impingement effect equation (i.e. impingement and landings) by the appropriate SPF EAVs we would have expected the calculated effects from using the Cefas and SPF EAV methodologies to be approximately the same and indeed the calculated difference in impingement effects is apparently less than 6%.

That prompts the question of why the numbers are different. In short, the disparity is considered most likely to be due to the additional uncertainties introduced by requiring estimates of F in the SPF EAV.

The SPF EAV calculations are sensitive to assumptions of F at age (Section 5). F is an estimated quantity that varies annually and depends on:

- the accuracy of landings data; and
- the adequacy of biological sampling at the landing ports.

Most commercial species experience discarding and until recently bass was at the lower end of the discard spectrum with an apparent estimated discard rate of 5% (ICES WGCSE 2019). More recently this has increased into the range of approximately 10% to nearly 27%. As it is the smaller fish that are discarded, F at younger ages (3 and 4) is biased by discarding. The impingement SPF EAV is particularly sensitive to this bias which will lead to overestimated EAVs.

Effective biological sampling is required to derive Age-Length keys but this has been an area where a lack of funding in member states has resulted in under sampling in the ports leading to poorly estimated length at age values and further uncertainty in the F at age calculations.

The resulting uncertainties in the values of F at age, and also in the conversion of recorded landing weights into landing numbers, are considered to result in low precision in estimated SPF EAVs. We have not attempted to quantify potential errors in the SPV EAVs due to uncertainties in F but we do expect that uncertainty will easily exceed the indicated 6% difference between the different effect predictions in Section 6.2.

6.4 Should the EAVs calculated with the Cefas methodology be adjusted for fishing mortality?

In the Cefas EAV methodology the impact of fishing mortality is not included in the algorithm on the assumption that fishing mortality will not substantially increase total mortality until fish become mature. Based upon the work in this report that assumption is questionable for bass. The question is then, is the omission significant?

To explore this question F was inserted into the Cefas EAV methodology by replacing natural mortality with total mortality (equal to natural mortality plus fishing mortality). The calculated EAV changed from 0.224 without F to 0.197 with F, or a 12% reduction. In comparison and, as expected, the SPV EAV is much more sensitive to F, with the SPF EAV changing from 1.05 without F to 0.598 with F, or a 43% reduction. In theory it would be more accurate to include F in the Cefas methodology and by so doing the predicted impingement effects of the station would be reduced slightly. However, given the uncertainties in the derivation of F at age, it has been decided that it would be disproportionate to rework the impingement effect calculations and on a precautionary basis the existing EAV values have been retained..

7 Conclusions

Considering the aims of this study in turn:

1. **Is the algorithm for calculating SPF EAVs correct?**
   TB010 contains a small error in the methodology, which underestimates the SPF EAV. This has been corrected.

2. **Is not including F in the calculation of SPF EAVs significant?**
   Neglecting F produces a significantly overestimated SPF EAV for bass; TB010 understates the likely errors in so doing. The bass SPF EAV changed from 1.05 without F to 0.598 with F, or a 43% reduction. We conclude that F cannot be neglected in SPF EAV calculations for species subject to commercial fishing. Most species including diadromous species are subject to fishing mortality.

3. **How can SPF EAVs be used to calculate impingement effects?**
   Impingement calculated with SPF EAVs cannot be compared with annual SSBs or landings as the comparators are not compatible with each other. If SPF EAVs are used for impingement assessment, landings or SSBs also need to adjusted by specific SPF EAVs. The net effect is that the introduction of SPF EAVs makes no material difference to the calculated impingement effect and is a redundant additional calculation step. SPF EAVs rely on estimates of F at age and this additional parameterisation introduces a new set of uncertainties into the assessment.

4. **Is the omission of F from the Cefas EAV methodology significant?**
   The inclusion of F at age into the CEFAS EAV methodology produced a small reduction of 12% in the calculated bass EAV from 0.224 without F to 0.197 with F. As would be expected the Cefas methodology is much less sensitive to the omission of F than the SPF EAV methodology. In theory it would be appropriate to include F in the Cefas methodology and by so doing the predicted impingement effects of the station would be slightly reduced. However, given the uncertainties in the derivation of F at age, it has been decided that it would be disproportionate to rework the impingement effect calculations and on a precautionary basis the existing EAV values have been retained.

It is concluded that there is no benefit from the use of SPF EAVs in impingement assessment and that their use is considered likely to increase, rather than decrease, uncertainty in assessments.

8 Accompanying spreadsheet calculations

This report is accompanied by a set of spreadsheets containing the calculations described in this report as follows:

Table 1 EAV calculation spreadsheets

Filename	Contents
EAV-SZ-bass v5a – EA method	Original TB010 SPF EAV for bass impingement.
EAV-SZ-bass v5a – EA method sw modified	Modified TB010 impingement SPF EAV as described in section 4.
EAV-SZ-bass v5a – EA method sw modified plus F SZC no mitigation2	Modified TB010 impingement SPF EAV including impact of fishing mortality (F). The same spreadsheet calculates the EAV using the Cefas methodology including the impact of F.
EAV-SZ-bass v5a – EA method sw modified plus F commercial land2	Modified TB010 SPF EAV including impact of fishing mortality (F) for commercial bass landings.

9 References

BEEMS Technical Report TR383. Sizewell Equivalent Adult Value (EAV) metrics. Cefas, Lowestoft.

BEEMS Technical Report TR406. Sizewell C – Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft

Environment Agency. 2019. Technical Brief: TB010 Draft. Converting impingement and entrainment numbers to Equivalent Adult Values and Spawning Production Foregone

ICES WGCSE 2018 Report Section 31 -Seabass (*Dicentrarchus labrax*) in divisions 4.b–c, 7.a, and 7.d–h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea)

ICES WGCSE 2019. Report Section 29 -Seabass (*Dicentrarchus labrax*) in divisions 4.b–c, 7.a, and 7.d–h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea)

Turnpenny, A.W.H., Utting, N.J., Millner, R.S., Riley, J.., 1988. The effect of fish impingement at Sizewell “A” Power Station, Suffolk on North Sea Fish stocks, CERL Report No. TPRD/L/3279/R88.

Turnpenny, A.W.H., 1989. The equivalent adult approach for assessing the value of juvenile fish kills, with reference to commercial species in British Water. CERL Report No. RD/L/3454/R89.

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP103

NOT PROTECTIVELY MARKED

SPP103 - Consideration of potential effects on selected fish stocks at Sizewell

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Executive Summary

EDF Energy plans to build a new coastal nuclear power station (Sizewell C), adjacent to the operational Sizewell B and decommissioned Sizewell A sites in Suffolk. The station will be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. Water abstraction can lead to the entrapment (entrainment + impingement) of fish at different life-history stages. As part of the Development Consent Order (DCO) application for the operation of the new station, EDF Energy is required to evaluate the effects of water abstraction on fish.

The predicted effects of fish entrapment in the SZC cooling water system are provided in BEEMS Technical Report TR406. The assessment methodology used in that report is based on well-established fisheries science principles and relies on comparison of the calculated SZC fish losses with the relevant spawning stock biomass estimates produced by the International Council for the Exploration of the Sea (ICES) for the appropriate internationally agreed stock areas. This is how the much larger environmental effects of fishing are internationally assessed and managed and represents the use of the most up to date peer reviewed methodologies and scientific evidence.

Sizewell C stakeholders have indicated that in principle they agree with the assessment methodology used to determine the effects on fish at the stock level but some stakeholders have questioned the application of ICES stock units for assessing potential effects of a coastal power station on local populations.

This report has been produced in response to these comments and provides two separate sets of evidence:

a. Additional narrative information on the appropriate scale of assessment for contextualising population level effects on fish species at Sizewell. The Environment Agency requested additional information on 12 of the 24 representative fish species found at Sizewell species and that is provided in Section 2 of this report.

b. An alternative assessment methodology that focuses on very local effect predictions is provided in Section 3. This assessment determines the predicted depletion of different fish species due to SZC acting in combination with SZB within the Greater Sizewell Bay and the tidal excursion (of SZC water abstraction), the ICES rectangle containing Sizewell (33F1) and the ICES division 4c which contains ICES rectangle 33F1. These assessment areas are much smaller than the stock areas for all assessed species. The results are presented in tables and in heat map formats.

Conclusions from the assessment of the predicted effects of fish abstraction on local fish abundance at Sizewell.

Local level depletion of fish has been considered by applying a simplified conceptual model of impingement relative to tidal replenishment. Whilst the model necessitates making assumptions relating to fish distribution and behaviour it is possible to approximate the likely size of effects at a range of scales. The table below shows the predicted annual effects for 6 important species from the pelagic, demersal and epibenthic species groups found at Sizewell (a more comprehensive list is presented in Section 3 of this report). In the local area close to the power stations the predicted losses of pelagic fish (e.g. sprat and herring) are calculated to be approximately 1.5% of the abundance expected in that area without any power station. In the adjacent rectangle 33F1 which extends form Lowestoft to just north of Felixstowe the expected reduction in pelagic species falls to 0.07% and outside of that area in ICES division 4c the expected loss falls to approximately 0.01% or less. A Figure for sprat showing how effects are concentrated within the local area and rapidly dilute with distance is provided below.

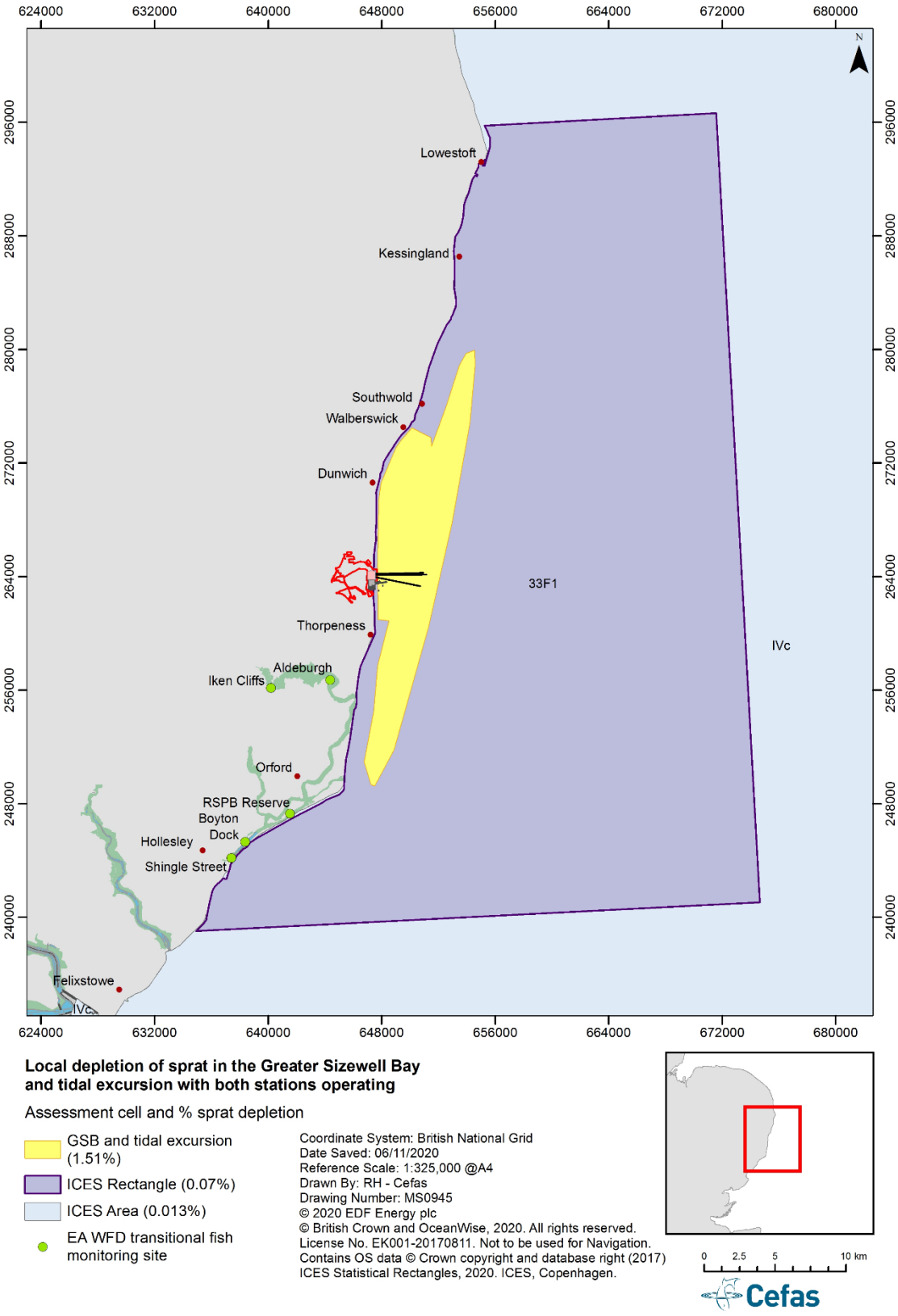
The predicted effects on demersal and epi benthic species are similarly small even in the area closest to the stations. Outside of that area the effects are minimal. In all cases the effects would not be discernible against the much larger natural variations in local fish population densities.

These three assessment areas are not the relevant stock areas for any of the Sizewell fish species; these are much larger (BEEMS TR406). The purpose of this local assessment is therefore not to assess the effects of SZC on the long-term sustainability of each stock, but to determine if there could be any local effects that

are sufficiently large to affect the conservation objectives of HRA protected species (e.g. via the prey of marine birds and marine mammals) or the WFD status of the nearest transitional water bodies (which are in 33F1) via the Transitional Fish Classification Index.

The predicted effects are very small, particularly in relation to natural variation in abundance, it is therefore concluded that impingement from Sizewell B and Sizewell C would not have any adverse food-web effects on designated features of HRA sites nor on the classification of nearby transitional water bodies under the WFD.

Species	Group	Period that the species is found in the area close to the Sizewell power stations (days)	Predicted % local fish population depletion in each area due to SZB + SZC		
			GSB + tidal extent	33F1	4c
Sprat	Pelagic	Dec. – March (121)	1.51	0.07	0.013
Herring	Pelagic	Feb. – April (89)	1.50	0.06	0.009
Whiting	Demersal	Oct. – April (221)	2.39	0.12	0.039
Seabass	Demersal	Dec. – March (121)	2.37	0.11	0.020
Sand goby	Epi-benthic	Aug. – Dec. (162)	0.88	0.04	0.010
Dover sole	Epi-benthic	March – Oct. (245)	0.88	0.05	0.016



Areas of localised depletion for sprat assessed with both stations acting in-combination during the period December to March. The GSB + tidal excursion (yellow), ICES statistical rectangle 33F1 (purple) and part of ICES Statistical Area 4c (blue) are shown.

Changes in Revision 3 of this report.

Preliminary written feedback on version 2 of this report was provided by the Environment Agency on 21st September 2020. Further comments were received when the findings of version 2 were presented to statutory stakeholders during the Marine Technical Forum (MTF) held on the 23rd September 2020. Responses to specific comments on version 2 of this report are encapsulated herein. Formal responses to each of the MTF comments will be provided by way of the MTF comments tracker.

In addition to the specific comments on version 2, this report considers the following key stakeholder concerns raised at the MTF;

- Stakeholders questioned whether the use of evidence for fisheries management purposes differs from the use of evidence for Habitats Regulations Assessment (HRA), Water Framework Directive (WFD, and Environmental Impact Assessment (EIA) purposes.
- Stakeholders reiterated the desire to see impacts from the power station considered at the 'local scale', particularly for *"locally isolated and static subpopulations residing close to the power station"*.

These two points are carefully considered in the new assessment methodology presented in Section 3.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C), adjacent to the operational Sizewell B and decommissioned Sizewell A sites in Suffolk. The station will be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. Water abstraction can lead to the entrapment (entrainment + impingement) of fish at different life-history stages. As part of the Development Consent Order (DCO) application for the operation of the new station, EDF Energy is required to evaluate the effects of water abstraction on fish.

Impingement predictions for the proposed Sizewell C development are based on the comprehensive impingement monitoring programme (CIMP) dataset collected from Sizewell B between 2009 and 2017. The results of the impingement predictions are presented in BEEMS Technical Report TR406. In addition to making impingement predictions, the report contextualises losses against the most relevant stock unit and/or population units to determine if entrapment by the station represents a significant effect on the stock.

Fish mortality due to impingement at Sizewell C can be considered as a form of harvesting. The International Council for the Exploration of the Sea (ICES) provides advice on fishing opportunities and stock status for individual stocks. For many species, annual analytical assessments are carried out that utilise information on life history, fishing effort and catches to assess the size of the stock, in particular the spawning stock biomass (SSB). In cases where a full analytical assessment is available for a species entrapped at Sizewell, and the SSB has been estimated, the predicted losses due to entrapment are compared with the ICES estimated SSB for the stock area (BEEMS Technical Report TR406). These SSB estimates provide the most robust peer reviewed scientific evidence. In the case of species where there are insufficient data for ICES to carry out a full analytical assessment to establish absolute SSB, predicted losses are compared with international landings for the stock area. Such a comparison is unrealistically conservative as landings will be much less than the stock size. For an unexploited stock, landings will typically be much less than 20% of the adult stock size and, even for a heavily exploited stock, landings will rarely exceed 50% of the stock size. A detailed description of the assessment methods is provided in BEEMS Technical Report TR406.

Population structures of marine fish species fall along a continuum from panmictic (e.g. European eel, *Anguilla anguilla*) to numerous distinct sub-populations (e.g. North Sea herring, *Clupea harengus*). The majority of species exhibit complex population structures. In the open sea, sub-populations of many species mix to a considerable extent; especially during summer feeding and on nursery areas, with harvesting affecting multiple components of the overall population simultaneously. ICES' definition of stock units integrates all of the information on site fidelity to spawning, nursery and feeding areas together with knowledge of migration patterns and the degree of intermixing that takes place between any sub populations. Stock assessment units are not static and change when the weight of evidence indicates that a change would likely lead to better assessments and management advice.

Consultation with statutory stakeholders on SZC entrapment predictions resulted in requests for additional information to expand the information in Section 7.8 of TR406 on the potential for local effects on fish species at Sizewell. Following engagement with the Environment Agency at a technical meeting on fish on the 19th March 2020 further clarification on the species of concern was provided on the 27th April 2020:

- Allis shad;
- Twaite shad;
- Herring;
- Cucumber smelt;
- Sea lamprey;
- Sprat;
- Seabass;
- Sand goby;
- Dab;

- Flounder;
- Thornback ray; and,
- Sandeel.

The intention of this Scientific Position Paper is to specifically address these concerns by summarising the key information from BEEMS Technical Report TR406 “Impingement predictions based upon specific cooling water system design” and the provision of additional evidence as requested to clarify potential effects of the proposed Sizewell C station on the selected fish stocks at Sizewell.

The evidence provided herein will be integrated into an updated version of BEEMS Technical Report TR406 (version 7) that will be submitted a part of a Supplementary Information Pack for the DCO in November 2020.

In addition to the specific comments on version 2, Sizewell C stakeholder have also requested further information on the potential for local level-effects of the proposed development and the provision of further evidence to support the Habitats Regulations Assessment (HRA) and Water Framework Directive (WFD) assessments. Section 3 of this report applies a simplified conceptual model of impingement relative to tidal replenishment to determine local depletion. Whilst the model necessitates making assumptions relating to fish distribution and behaviour it is possible to approximate the likely size of effects at a range of scales from the Greater Sizewell Bay and tidal excursion, to wider effects in relevant ICES Statistical areas.

The purpose of this local assessment is therefore not to assess the effects of Sizewell C on the long-term sustainability of each stock, but to determine if there could be any local effects that are sufficiently large to affect the conservation objectives of HRA designated features (e.g. via the prey of marine birds and marine mammals) or the WFD status of the nearest transitional water bodies (which are in 33F1) via the Transitional Fish Classification Index.

2 Species of concern for which additional information has been requested

The section reviews each of the species where further information was requested from the Environment Agency. The stock units for effect comparisons in BEEMS Technical Report TR406 v6 are provided in Table 1.

2.1 Allis Shad

Stakeholder comment: "*There are closer populations than the Garonne population, which make us question the validity of using the Garonne fish as the reference population. Can further evidence be provided".*

Further comment on version 2: The Environment Agency and Natural England acknowledged that impingement predictions amount to a single fish per annum. However, emphasised the necessity to follow the HRA processes and assign fish to specific SACs including the Tamar Estuary where Environment Agency monitoring detected 419 fish in 2015.

Response:

A single fish was impinged at Sizewell at SZB on the 28th May 2009. The specimen was 50 cm in length, at the top end of the expected length at maturity that typically varies between 30 and 50 cm (Froese, Pauly, 2019).

The Gironde-Garonne-Dordogne stock is typically used as a reference in population dynamics (Wilson, Veneranta, 2019) though it is not the closest river with a breeding population to Sizewell. The Gironde-Garonne-Dordogne stock is by far the largest in Europe and likely to provide the source for the occasional fish that are caught in the summer feeding grounds in the Celtic Sea, the English Channel and southern North Sea. Fish from the Gironde-Garonne-Dordogne stock have high site fidelity and forage near natal rivers; however, the species exhibits moderate but nevertheless sufficient straying to uniformise the Allis shad population genetic structure along French shores. The furthest recorded distance between natal and spawning river was > 700 km (Martin *et al*., 2015), which is roughly similar to the distance between Sizewell and the mouth of Loire in Brittany. However, it is acknowledged that the nearest population in relation to Sizewell is not the Garonne. Whilst allis shad exhibit high natal river fidelity, some fish do stray and that is how repopulation is known to occur. Long-distance straying also occurs though less frequently, for example, fish that were born in France were caught on their spawning migration in Portugal, and fish born in northwest France (Bay of Biscay) were caught in northern France, English Channel (Wilson and Veneranta, 2019).

Populations of *Alosa alosa* exist along the north-eastern Atlantic coasts in some large rivers of France (Loire, Gironde–Garonne–Dordogne, and Adour) and Portugal (Minho and Lima) (BEEMS Scientific Position Paper SPP071/s). A single allis shad in spawning condition was found dead on 26 April 2020, 200km up the river Scheldt in Belgium after an organic pollution event in early April 2020 more than 100km upstream in France. The shad was an exceptionally large specimen of 57cm length and confirmed as an allis shad by gill raker count; according to the Instituut Natuur-En Bosonderoek (INBO) this is the first record in 100 years of allis shad in the upper Scheldt (INBO 2020). The fish would have entered the Scheldt from the sea in early to mid-April and INBO hypothesise that it may have originated from one of a number of recent reintroduction programmes in the Oosterschelde (Eastern Scheldt). The Scheldt and the Rhine are the nearest European rivers to Sizewell (approximately 150km to the east) with records of allis shad but there is no convincing evidence of self-sustaining populations, particularly in the Scheldt. INBO have previously caught a 27cm immature fish off the coast in 2012 and this was also attributed to the reintroduction programme. Allis shad have recently been reintroduced into the Rhine, which is the nearest reproduction area, but the number of recruits is still small.

In the UK, two locations have been identified where individuals in breeding condition have been recorded: the river Tamar in SW England and the Solway Firth on the border between England and Scotland (Jolly *et al*., 2012). Individual immature adults are occasionally found in the Bristol Channel, the English Channel and

the UK east coast. The Tamar Estuary forms part of the Plymouth Sound and Estuaries SAC situated approximately 700 km from Sizewell. Allis shad are a qualifying feature, but not a primary reason for site selection. Allis shad are qualifying features for SACs in Wales such as Pembrokeshire Marine, however, these sites are further afield.

In terms of proximity, the Sizewell specimen may have been a straggler from the reintroductions in the Scheldt or the Rhine, from other self-sustaining populations in Brittany and Normandy or from the largest population in the Garonne. It is feasible, though unlikely, that the shad impinged at Sizewell was from the Tamar population. As such effects on the Plymouth Sound and Estuaries SAC will be assessed.

As the shad impinged off the Sizewell coast was impinged at the end of May, and adults gather in rivers in May for spawning having migrated from the sea typically in early to mid-April, the SZB specimen was likely to have been a mature fish which lost its homing instinct. The specimen would not have reproduced that year had it not been impinged and would have had to survive to the following year for the opportunity to reproduce.

The single impingement record of allis shad at Sizewell would, therefore, not make any material difference to whatever population it was from. No self-sustaining population could be dependent upon 1 fish.

Table 1. Data sources used to provide information on relevant stock unit, landings and SSB from BEEMS Technical Report TR406.v6. Updates following this report are provided in bold.

Species	ICES Working Group	Stock unit	Assessment type	Impingement effect comparator	Reference
Sprat	HAWG	Subarea 4 (North Sea)	Analytical assessment	SSB	ICES, 2018
Herring	HAWG	Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak & Kattegat, Eastern Channel)	Analytical assessment	SSB	ICES, 2018
Seabass	WGCSE	Divisions 4.b-c, 7.a, & 7.d-h (Central & southern N Sea, Irish Sea, English Channel, Bristol Channel & Celtic Sea)	Analytical assessment	SSB	ICES, 2018c
Sand goby	-	Not defined	Not assessed	Population abundance	Rogers and Millner (1996)
Dab	WGNSSK	Subarea 4 & Division 3.a (North Sea, Skagerrak & Kattegat). **Updated: Revised stock area Subarea 4 (North Sea).**	Trends only	Landings	ICES, 2019b
Flounder	WGNSSK	Subarea 4 & 3.a (North Sea & Skagerrak and Kattegat). **Updated: Minor revision to updated landings data.**	Trends only	Landings	ICES, 2018b
Cucumber Smelt	-	Not defined but includes the East Anglian coast and rivers on the European coast from the Elbe to the Scheldt	Estimated adult numbers migrating up-river	Elbe populations EA landings & ICES Landings	BEEMS Scientific Position Paper SPP100 EA, 2018, 2017a, 2017b, 2015, 2014, 2013a, 2013b, 2013c; ICES, 2019
Thornback ray	WGEF	Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak, Kattegat & eastern English Channel). **Updated: Revised stock area Subarea 4 & 2.a (North Sea and Norway).**	Trends only	Landings	ICES, 2019c
Twaite shad	-	Not defined but includes the River Elbe and Belgian river Scheldt. A separate spawning population on the river Weser has not been included in the assessment. (**Revised numbers from those presented in TR406 v6)**	Estimated adult numbers migrating up-river	European populations in the Elbe. ICES Landings	BEEMS Scientific Position Paper SPP100 ICES, 2019
Allis shad	-	Garonne	Analytical assessment	Adult stock in 2009, ICES Landings	BEEMS Scientific Position Paper SPP071/s) ICES, 2019
Sea lamprey	-	Not defined	Not assessed	-	

Working group acronyms:
HAWG - Herring Assessment Working Group for the Area South of 62°N
WGNSSK - Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak
WGCSE - Working Group on Celtic Seas Ecoregion
WGEF - Working Group on Elasmobranch Fishes.

2.2 Twaite Shad

Comment: "*The company should assess all losses as though they come from each of the mainland continental rivers in turn, rather than assessing against the combined stock from those rivers (evidence from Unlocking the Severn is that there is a high degree of spawning site fidelity)".*

Further comment on version 2: The Environment Agency noted that; *"While it may be unlikely that all impinged twaite shad would originate from one river, such an assessment would provide a precautionary quantitative approach, the likelihood of this outcome occurring can then be discussed qualitatively."*

Response:

Spawning site fidelity is noted, however the proposed development is hundreds of kilometres away from the main European spawning rivers of the Scheldt, Weser and Elbe in mainland Europe (BEEMS Technical Report TR406 Figure 5). There are no known UK east coast spawning sites for twaite shad. Genetic analyses of twaite shad from Sizewell demonstrate that they do not originate from the Severn catchment (Jolly et al., 2012). Sabatino and Alexandrino (2012) identified a North Sea twaite shad population with low genetic diversity between fish sampled off Belgium (Scheldt) and Denmark and also the Solway Firth. These analyses identified separation between the Baltic and North Sea populations. The twaite shad caught at Sizewell range from >1 yr old juveniles to sexually mature adults that are probably a part of the North Sea mixed population widely dispersed across feeding grounds. In contrast to allis shad, this species is iteroparous so adults may return to the sea to forage several times during their lifetime after each seasonal spawning. Given the distance of the proposed development from the spawning rivers in mainland Europe and the likelihood of population mixing during feeding in the marine environment it is not logical to associate all the fish impinged at Sizewell to a single river system. Given the geography, it is much more likely that origin of fish caught at Sizewell would be approximately in proportion to the size of the spawning populations in the European Rivers. The assessment in BEEMS Technical Report TR406 considers the two main rivers: the Elbe and the Scheldt (only) and is therefore likely to be precautionary. Between 2009 and 2017, an estimated annual average of 5.2 million adult twaite shad passed through these two river systems (BEEMS Scientific Position Paper SPP100) with the majority being in the Elbe. The estimated losses of adult fish (1,046 individuals, TR406) associated with the proposed development with mitigation measures in place are predicted to be 0.02% of the River Elbe and Scheldt average population. Sizewell C is expected to impinge fish from different European rivers on a pro-rata basis according to their abundance and it is therefore considered highly unlikely that there would be a significant effect on the population in any given river.

Between the periods 1992−1993 and 2009−2010, spawner' densities in the most important Elbe population increased significantly (at least ten-fold, though with high level of uncertainty) and spawning grounds expanded (Magath, Thiele, 2013). The Elbe population was still increasing in recent years (Tulp *et a*l., 2017). The Belgian population of the Scheldt estuary also is exhibiting signs of "genuine improvement" (Wilson, Venerata, 2019) although recovery is later than in the Elbe after action to improve water quality and more recently to improve access to spawning grounds. Very large numbers of migrating twaite shad have been reported in the Scheldt in 2020 with "vast areas of spawning in the river" (pers. comm: Pieterjan Verheist, University of Ghent, 8 May 2020). Numbers of a small population of twaite shad in the Weser Estuary are stable without clear trends in the recent years (Tulp *et al*., 2017). This process of quick stock recovery occurred during unimpeded functional activity of Sizewell B. Therefore, the proposed development is not predicted to have any meaningful effect on the twaite shad populations.

2.3 Herring

Comment: "*Detail covered in our gap analysis and discussed with EDF on 19th March, including the Blackwater herring. For herring, the way the assessment is done has changed, but this has not been explained are assessing, but never explained why to justify the scale being used. There is some inconsistency in whether herring stocks are regarded as separate or not."*

Response:

The assessment unit for herring reflects the latest ICES advice. The small Blackwater population is not included in the ICES stock assessment. Section 7 of BEEMS Technical Report submitted as part of the DCO details the potential for effects on the Blackwater population and concludes that: *"The weight of evidence*

therefore indicates that Sizewell impingement is from main North Sea stock (as assessed by ICES and used in this report) and not the Thames estuary Blackwater stock".

Whilst it is feasible that the proposed development would impinge Blackwater herring, the proportion of Blackwater herring in the mixed southern North Sea population is very small and impacts on the Blackwater SSB are likely to be minimal.

2.4 Cucumber smelt

Comment: "*Detail covered in our gap analysis and discussed with EDF on 19th March*".

Response:

The concern pertains to smelt populations within the Alde-Ore estuary particularly in relation to the potential for a derogation in the Water Framework Directive fish status of the Alde & Ore transitional waterbody. BEEMS Scientific Position Paper SPP108 specifically addresses these concerns. The potential for localised effects on smelt is provided in Section 3.5.

2.5 Sea Lamprey

Comment: *"Sea lamprey are stated as being widespread throughout the North Sea, but without supporting evidence. There appears to be a serious decline in Humber populations. Significance of sea lamprey entrapment might be more significant if stocks of this species are crashing. Please provide some description of sea lamprey stocks in the North Sea (as has been provided for river lamprey)".*

Further comment on version 2: The Environment Agency welcomed the additional context on sea lamprey populations in the North Sea and requested any further quantitative information was available on distribution, population size and at sea behaviour.

Response:

In contrast to most of anadromous fish (i.e. those that return from the sea to spawn in rivers where they were born), the sea lamprey (*Petromyzon marinus*) is deprived of homing instinct but rather exhibits regional panmixia during foraging at sea. To complete the life cycle, the species uses a 'suitable river' strategy being attracted to spawning sites by pheromones released by its larvae (Waldman *et al*., 2008). So, after the marine trophic phase, adult sea lampreys re-enter freshwater via the nearest 'suitable' river with already dwelling larvae and migrate upstream where they build nests, spawn and soon die (ICES 2015). Because of this, there is no genetic differentiation between sea lampreys sampled along European shores (Almada *et al*., 2008; Wilson, Veneranta, 2019). Therefore, there might be no discrete populations/stocks in the southern North Sea (neither a particular Humber population), as the entire European population / stock exhibits a considerable genetic homogeneity over a spatial scale of > 2000 km, from Yorkshire to the South Portugal (Genner *et al*., 2012). Increase or decrease of sea lamprey in a particular river is dictated by water quality and local human activities (e.g. dam building).

Sea lamprey has declined in British freshwaters over the last hundred years mostly due to pollution and engineering barriers, becoming extinct in many rivers (Maitland 2003). The current International Union for Conservation of Nature (IUCN) status of anadromous sea lamprey in Europe is of "least concern" (IUCN website, Natureserve 2013), however, ICES recently suggested to consider it as "vulnerable" (Wilson, Veneranta, 2019). The UK reported to the EU in the 2007-2012 reporting round that the status of the species in the UK is unknown. There is no ICES sea lamprey stock assessment for the North Sea. At sea numerical estimation is not available as adults cannot be in any reliable manner. Distinct trends of population increase have been registered across the entire distribution range of the species in European waters in recent years though more data are needed to confirm whether this trend actually reflects overall recovery of all stocks (Wilson, Veneranta, 2019).

In the UK, *P. marinus* is reasonably widespread in rivers. In some places it is still common, but it has declined in parts of its range and has become extinct in a number of water sheds. It appears to reach its

northern limit of distribution in Scotland and does not occur north of the Great Glen (JNCC website). UK SACs where sea lamprey is a primary reason for site selection are predominantly on the western and southern coasts. The nearest UK SAC to Sizewell where sea lamprey is a qualifying feature, but not a primary reason for site selection, is the Humber Estuary and the associated spawning site of the River Derwent. Sea lamprey SACs are found all along the European coast from the Netherlands to Denmark with specific concentrations in the Scheldt, Hollands Diep, Waddenzee, Ems, Weser, Elbe and Eider and any sea lamprey caught at Sizewell could have originated from any of these systems. In terms of geography the Dutch coast is nearer to Sizewell than the Humber but due to their parasitic feeding lifestyle the distances travelled by the species depend largely upon the dynamics of their prey. As sea lamprey do not home to natal rivers, mortality at Sizewell could not be attributed to any specific site of origin.

A single sea lamprey was recorded in the 2009-2017 CIMP programme in 2015. The expected annual impingement losses of sea lamprey at Sizewell C with the full mitigation is less than 1 fish (0.36 i.e. 1 fish every 2-3 years) with a maximum of 1 individual per annum (BEEMS Technical Report TR406). This is considered negligible for a stock that is widespread throughout the North Sea. There is no reason to consider that the 1 fish caught at Sizewell was from the Humber Estuary and it could have eventually arrived at Sizewell from any North Sea breeding river (or even further afield) dependent on the behaviour of its hosts and would be unlikely to have returned to that river other than by chance, if it had survived to breed.

2.6 Sprat

Comment: *"Sprat are assessed against the North Sea stock but see Section 4.5 of ICES WKSPRAT Report 2013 (attached). There are examples of local populations in the North Sea that have been depleted. It also says 'There are sprats in the Wadden Sea and in the outer Thames estuary, areas that are more closely connected to the main sprat population in the southern North Sea but which may represent populations with distinct dynamics.' This needs to be acknowledged and discussed by the applicant."*

Further comment on version 2: Further information and clarification was requested from the Environment Agency on local populations identified by ICES and the appropriateness of the scale of assessment.

Response:

Sprat has been assessed against the North Sea population incorporating Division 3.a and Subarea 4 (Table 1) based on advice from the latest Benchmark Workshop on Sprat (WKSPRAT) in 2018, which summarised the most recent scientific evidences thus superseding previous approaches e.g. ICES WKSPRAT Report 2013. At this Benchmark Workshop all sprat in Division 3.a and Subarea 4 were combined into a single stock unit due to new evidence from a genetic study carried out for the Northeast Atlantic (ICES, 2018a). Local, genetically distinguishable populations have been identified on the periphery of Division 3.a, along the Norwegian coast and likely the Swedish coast. Norwegian populations are not part of the assessment of sprat in the North Sea (ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion (ICES 2020a)

ICES keeps stock definitions under continuous review and makes adjustments to these definitions when the weight of scientific evidence indicates that a change is appropriate. We have used the most up to date evidence based upon ICES 2018 conclusions.

2.7 Seabass

Comment: *"In the latest version of TR406 a personal communication is included from Lisa Readdy. 'Previously, Pawson et al 2007 recommended amended stock units for assessment purposes based upon some UK tagging studies. However, scientific knowledge about bass has advanced since these 2000-2005 studies and ICES continues to recognise the 4b-c, 7a and 7d-h stock unit as the most appropriate for bass stock assessment purposes based upon all of the available scientific evidence.' Citations are not provided for the work which means that Pawson et al 2007 is superseded. The EA review will be helped by informing us of all the literature used to come to our conclusions on the appropriateness of the scale of assessment".*

Further comments on version 2: The Environment Agency noted that *"The extra information is useful but seems to not yet be conclusive as to whether bass in the North Sea should be considered part of a separate sub-population to the Irish Sea (or elsewhere) due to small sample sizes of tagged fish – something the*

applicant acknowledges, but argues that this as a reason for not splitting the stock. HRA/EIA may require a different response to the same evidence than fishery management. (i.e. fishery management – we can't be sure they are separate stocks so treat as one, HRA – we can't be sure they are separate stocks so treat as separate).

Natural England commented that: *"It is important to note we fully recognise the expertise of ICES as fisheries managers. We do feel that, when reviewing the best available evidence, the use of evidence for fisheries management purposes differs from the use of evidence for HRA/EIA/impact assessment purposes. In this case for this impact assessment we still query why the scale of assessment remains so large, given the work that CEFAS and Ifremer are currently undertaking to establish to what degree there is finer population structure and mixing of those within the existing stock. In this case it may be prudent, for the purpose of conducting an impact assessment for a project which will not be subject to the same adaptable management as commercial fisheries, to re-evaluate the use of ICES management units to contextualise the local impacts of SZC. While seabass may currently represent the best example of this overarching point, this query also relates to each species where the use of ICES management units has been flagged as best available evidence, and the reason for overlooking evidence of smaller sub-populations with limited/unknown rates of mixing. Overall this reduces our confidence in the findings of the assessment".*

Response:

Seabass is a highly mobile species with a homing instinct in many if not most individuals, and some 55% of tagged bass were recaptured within 16km of their original release position (Pawson et al., 2007). However, it is not clear how the authors dealt with captures of fish within the season (e.g. recaptured the week following tagging).

Summarising the history of previous tagging studies, the ICES Benchmark Workshop on Sea Bass (ICES 2018b) stated:

"*The sea bass inhabiting the Atlantic Ocean show a remarkable homogenous genetic structure although homing based on mark–recapture data suggests some level of population structure. Off the Strait of Gibraltar (9.a) there is evidence of introgression by the Mediterranean group. Sea bass inhabiting the areas Northern (4.b&c, 7.a,d–h) and Biscay (8.a&b) represent genetically one population unit. The current management in two stocks (Northern and Biscay) can be considered a conservative and correct measure*.

DSTs (Data storage tags) have been used to investigate individual behaviour patterns of adult sea bass between the North Sea and English Channel (Quayle et al., 2009) and between the Western Channel and Bay of Biscay. Movements were reconstructed using information about release and recapture locations alongside the temperature and depth of individual fish using hidden Markov models (e.g. Woillez et al., 2016) Five of 11 recaptured bass exhibited migrations of greater than 100 km between the central and southern North Sea and western English Channel, which supports previous studies (Thompson and Harrop, 1987; Pawson et al., 1987; 2007b) and provides evidence of migratory links between the North Sea and English Channel. Ifremer tagged sea bass with DSTs in the Iroise Sea from 2010–2012. Reconstructed tracks confirmed the highly migratory nature of bass, with three behavioural strategies: residency in the Iroise Sea; winter spawning migrations into the Bay of Biscay; and winter spawning migrations into the Western Channel/Celtic Sea. Site fidelity was found not only on summer feeding grounds as previously observed (Pawson et al., 2008), but also and for the first time, on winter spawning grounds. This indicated that sea bass populations may have a much finer spatial structure than supported by the genetics. However, sample sizes are very small, so it is difficult to generate robust estimates of the levels of migration between the current Northern and Biscay stocks."

To explore the degree of isolation between Northern and Biscay stocks further, ICES launched two large tagging programmes. The first programme (C-Bass) is being led by the Cefas (CEFAS, 2018) and has tagged almost 200 sea bass with electronic data storage tags (DSTs). The BARGIP study is being led by Ifremer (France) and has released 1,220 fish with DSTs at ten locations in the Channel and Bay of Biscay (ICES 2018b).

Preliminary results of these studies indicated that "*adult bass can undertake very large-scale migrations. Fish tagged in 4.c exhibited two migration patterns: one that remained in the North Sea; and a second that involved long migrations as far as the Irish Sea. Fish that were tagged in May in 4.c were in spawning*

condition, suggesting the presence of spawning ground in the North Sea. Bass tagged in area 7.e appeared to remain in the English Channel, but individuals tagged in the Irish Sea could move between the Northern and Biscay stocks" (ICES 2018b).

"Sea bass movement between ICES stock units is plausible, as evidenced from some individuals tagged with DSTs, and fidelity to both spawning and feeding grounds may provide evidence of fine population structure that was identified from genetics. However, it is not possible to quantify the proportion of fish migrating between the stocks currently due to the small numbers of fish tracks analysed. As further DSTs are analysed, it may be possible to quantify exchange between stocks, so the next benchmark should consider how exchange between stocks should be incorporated into the assessment process" (ICES 2018b).

Therefore, ICES have a sound scientific background to consider seabass of the central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea representing the single stock ("The Northern stock") and the single management unit, respectively (ICES 2019a). There is a possibility of existence of sub-populations in Irish and Welsh waters (ICES, 2118b) but if those exist they are still part of the same population and stock.

The current debate within ICES Working Groups is not whether the current stock unit is too large, but indeed whether it is too small. In addition to questions about the separateness of the Bay of Biscay bass stock unit, there are similar questions about whether the bass in Irish coastal waters are indeed functionally separate (ICES 2019e). Stock identity studies are therefore currently focussed on the whether the stock unit size for seabass should be expanded and there are no suggestions within ICES of a reduction in the bass stock unit relevant to the Sizewell C assessments.

In response to comments in relation to narrowing the assessment area in an HRA context the latest position of ICES indicates no merit in narrowing the assessment area. Tagging studies are underway to determine both fine scale sub-population structure and wide scale dispersal. However, where sub-populations exist they still form part of the wider stock (=population) as this term is defined by MMO (2020).

Seabass are not an Annex II species, therefore HRA concerns pertain to indirect effects and the availability of seabass as prey items for designated species. Section 3 considers the potential for local effects in an HRA context.

2.8 Sand Goby

Comment: "*The possibility of smaller scale populations of sand goby is not considered. Genetic differentiation has been found between populations in the Schelde estuary and offshore. Refer to Pampoulie et al. (2004) and discuss the potential for final scale population structure in sand goby and how this could be affected by entrainment (and combined) entrapment losses. Pampoulie et al. (2004) Evidence for fine-scale genetic structure and estuarine colonisation in a potential high gene flow marine goby (Pomatoschistus minutus). Heredity 92, 434–445".*

Further comments on version 2: The Environment Agency commented that **"***Dispersal potential for sand goby is stated as >10km, but the '>' is not quantified (is it 11km, 20km, 100km?). Given that the length of coastline over which the stock estimate has been calculated extends over 250 km of coastline, this large scale of assessment still does not seem appropriate. Can sand goby from the Norfolk coast and Kent both contribute to sand goby numbers in Sizewell Bay?"*

Response:

The sand goby (*Pomatoschistus minutus*) is an egg brooding species with a pelagic stage duration of about one month and larval dispersal potential greater than 10 km (Gysels et al., 2004; Riley, 2007). It should be noted that gobies are a short-lived fast maturing species. Larval dispersal distances of 10 km (each breeding event) along the distribution range would result in widespread genetic mixing. This is phenomena is reflected in genetic studies that have revealed limited genetic differentiation with a weak pattern of isolation-by-distance recorded throughout the entire distributional range of this species in the Atlantic (Gysels et al. 2004). As such the degree of genetic mixing extends well beyond the region from Norfolk to Kent.

Genetic studies of allozymes in fish collected off Belgian shores did not reveal clear differentiation between individual samples between estuarine, coastal and offshore samples (Gysels et al., 2004; Pampouille et al., 2004). However, a separate analysis of microsatellite loci revealed the existence of two spatially separated breeding units "of a complex and dynamic spatiotemporal structure" (estuarine and coastal). These breeding units support their identity by spawning in the different retention zones (estuarine and coastal) but are still connected by a low number of migrants (Pampuoile *et al*., 2004). The authors avoided to use the word "population" to define these breeding units, as "the gene flow appears to be sufficiently large to swamp any potential for large genetic differences". Therefore, both entities represent sub-populations so parts of the same stock.

In the next paper published in the same journal (Gysels et al., 2004) the authors noted that the Schelde (Scheldt in English) estuary (first breeding unit) is heavily polluted while the level of pollution decreases towards the offshore areas (where the second breeding unit was defined) and "a number of studies have revealed differential mortality of distinct genotypes when exposed to heavy metals". Therefore, there is good possibility that such a genetic differentiation between two breeding sites might be a result of the different survival of spawners with different genetic loads (low- and high fitness genotypes) after larval settlement. An analogy of Darwin's description of peppered moth evolution with higher survival of black morph in polluted areas with blackened trees, and pale morph faring better in clean areas with normal trees because of selective predation by birds. Such a problem of environmentally-induced selective mortality of particular phenotypes (so genotypes), is more likely to appear in a resident benthic species, where juveniles and adults do not carry out long-distant migrations and remain in more or less the same habitat the entire life. All connections between different parts of population/stock range occur mostly through larval transport ensuring genetic connectivity and distribution of recruits over this entire range.

Therefore, the fine-scale genetic structure revealed in this species is likely to be might be an artefact caused by differential survival in the different habitats, but even if it is not, the genetic flow due to larval transport (Gysels et al., 2004) and migrations (Pampuoille et al., 2004) is so intensive, that for assessment and management reasons such closely connected entities should be considered as parts of a single population.

In the open coastal waters of Sizewell such fine-scale genetic structuring and the presence of sub-populations is not expected.

2.9 Dab

Comment: *"The paper cited in TR406 seems to indicate a smaller scale of assessment than actually used".*

Further comments on version 2: Further comments from the Environment Agency *"Although area division 3.a is now excluded, justification for the use of landings data from all of subarea iv is weak*."

"Rijnsdorp et al. (1992), which is cited in TR406 and suggests a still smaller assessment area would be appropriate (perhaps iv.b and iv.c). This has not been answered by the applicant."

"The original version of TR318 concluded that dab were part of a population limited to the southern North Sea and compared entrainment to landings data from IVc only and we have previously asked for the change from this position to be explained to us. It still hasn't been."

Response:

The ICES advice reports (ICES, 2018b and ICES, 2019b) have been reviewed. ICES, 2019b states: *"The several spawning grounds and the wide distribution of dab indicate the presence of more than one stock. Meristic data (Lozán, 1988) corroborate the hypothesis of several stocks for dab, distinguishing significantly between populations from western British waters, the North Sea and the Baltic Sea".*

Therefore, we agree that including landings in Division 3.a overestimates landings from the relevant population for contextualising losses at Sizewell. The appropriate scale for impingement losses at Sizewell should only apply annual landings from the North Sea Subarea 4 as the comparator. Landings data for dab in Subarea 4 for the period 2009 to 2017, coinciding with impingement monitoring, ranged from 2,751t (preliminary 2017 catch statistic) to 7,240t with a mean of 5,309t (ICES, 2019b).

The change in fisheries landings comparator is displayed in Table 2. Predicted losses from the proposed development increase from 0.008% of annual fisheries landings to 0.009%. BEEMS Technical Report TR406 will be updated accordingly, however, the change in landings data comparator has no bearing on the outcome of the assessment. Furthermore, it should be noted that using landings as a comparator is not equivalent to the larger (undefined) SSB for the species, so the assessment is highly precautionary, especially when the very high level of discarding of the species (mean of 766% of landings in the period 2009-2017), which are not captured in the landings data, is considered (TR406). Stocks of dab are in sufficiently good status that in 2017, ICES advised that "*the risk of having no catch limits for the dab … was considered to be low and not inconsistent with the objectives of the Common Fisheries Policy... Dab and flounder are no longer managed under a TAC*" (ICES, 2019f). This effectively means dab is unmanaged because of no concerns over sustainability leading ICES to state that …"*this advice was valid as long as dab and flounder remained largely bycatch species, with the main fleets catching dab and flounder continuing to fish the target species (plaice and sole) sustainably … If this situation changes, or dab is no longer within safe biological limits, this advice would need to be reconsidered*." (ICES, 2019f).

No significant effects are predicted on dab from the proposed development.

Table 2. Annual mean SZC predictions of impingement for dab showing landings data applied in BEEMS Technical Report TR406 and proposed revision.

Species	Mean SZC prediction (No mitigation)*	SZC prediction with LVSE intakes	FRR mortality	EAV number	EAV weight (t)	Mean landings 2009 -2017 (t)	% of landings
Dab TR406: Subarea 4 & Division 3.a (North Sea, Skagerrak & Kattegat).	143,519	51,236	27,591	12,267	0.50	6,135	**0.01** (0.008)
Dab Revised landings data: Subarea 4 (North Sea).	143,519	51,236	27,591	12,267	0.50	5,309	**0.01** (0.009)

*Scaled impingement numbers reflect updates following release of BEEMS Technical Report TR339.v3.

The updated evidence provided herein will be integrated into version of BEEMS Technical Report TR406 (version 7) that will be submitted a part of a Supplementary Information Pack for the DCO in November 2020.

2.10 Flounder

Comment: "*The change in the scale of assessment was significant but not explained".*

Further comments on version 2: Further comments from the Environment Agency *"The applicant has not understood the questions we have over the flounder assessment and has instead provided an explanation of ICES methodology. We asked why flounder losses were compared to local landings of 12 tonnes in the original TR406 (2016) but are now compared to the whole North Sea (3,365 tonnes)."*

Response:

The first version of BEEMS Technical Report TR406 was released in 2019. A preliminary impingement assessment was produced four years ago in 2016 to allow consultation and iterative development of the methodology. The preliminary report was not based on as complete an evidence as BEEMS Technical Report TR406. The Environment Agency has been informed that the preliminary report has been superseded by BEEMS Technical Report TR406 and is no longer relevant. BEEMS Technical Report TR406 is based on the most contemporary evidence, where evidence gaps are identified, or evidence changes the report is updated accordingly.

The scale of assessment adopted in BEEMS Technical Report TR406 is based on fisheries landings from Subarea 4 & 3.a (North Sea & Skagerrak and Kattegat) presented in Table 1. Updated information on landings in the different sub-divisions of the North Sea (excluding Kattegat and Skagerrak) are provided in the Table 3. The updated mean landings from 2009-2017 are 2,321.4t opposed to 2,309t in BEEMS Technical Report TR406.v6.

Table 3. Total landings of flounder in the North Sea in tonnes.

Year	27.4.a	27.4.b	27.4.c	27.3.a	Total
2009	0.0	667.2	2,147.2	272.0	3086.4
2010	1.1	607.5	2,550.8	205.4	3364.8
2011	0.3	400.9	2,646.5	145.2	3192.9
2012	8.6	234.1	1,949.9	118.6	2311.2
2013	0.7	237.5	1,465.6	172.7	1876.5
2014	0.3	169.0	1,703.8	194.7	2067.8
2015	0.1	189.9	1,645.6	76.9	1912.5
2016	0.2	136.5	1,491.2	109.7	1737.6
2017	0.7	124.2	1,059.2	159.2	1343.3
Mean 2009-2017					**2,321.4**

Flounder was last benchmarked by ICES in 2018 and the SPiCT model was successfully used for assessment of this stock. However, updating the SPiCT with new data led to increased uncertainties and in 2019 the model was abandoned. The stock status was assessed using length-based indicators, as had been the case prior to the incorporation of the SPiCT model, with survey indices taken into account. This change in methods of assessment has limited implications for stock management and no effect on the assessment of the proposed development which is contextualised against landings.

The current stock of flounder in the North Sea are sufficiently high leading to ICES stating that flounder is no longer needed to be managed under a TAC and this advice was valid as long as dab and flounder remained largely bycatch species, with the main fleets catching dab and flounder continuing to fish the target species (plaice and sole) sustainably (ICES, 2019f).

The proposed development is predicted to result in losses of 0.12t of flounder per annum with full mitigation. This represents 0.005% of the 2,321t (mean for 2009-2017) fishery (Table 3). During the 2009-2017 impingement monitoring there has not been a significant trend in the mean number of flounder impinged each year (n = 8, tau = 0.571, p = 0.063, Theil-Sen slope = 571 [1]). Flounder is a bycatch species with a high discard rate (some 12-49% of landings in the North Sea) in the period 2002-2018 (ICES, 2019d), and discard survival in the North Sea and English Channel is of 42-64% in otter trawls and ~37% in beam trawls (Catchpole et al., 2015). The stock is estimated to be in the safe limits to the extent that ICES was not requested to provide advice on fishing opportunities for this stock in 2019 because the risk of having no catch limits for flounder was considered to be low and not inconsistent with the objectives of the Common Fisheries Policy (ICES 2019d).

2.11 Thornback ray

Comment: *"There is some evidence of more localised population in the North Sea that should be considered".*

[1] Statistics report the Kendall's tau statistic, which measures the rank correlation between impingement and year (with a range from -1 to +1, with 0 when there is no correlation), and the 2-sided p-value from the Mann-Kendall test. The Theil-Sen slope estimate is a median slope estimate for a linear change in impingement per year.

Response: The population structure of thornback ray was explored in the UK waters using a reliable tool of microsatellite loci, which demonstrated absence of genetic differentiation even between distant locations. Immature rays might travel to winter foraging areas and intermix there, whereas spring and summer foraging grounds are relatively isolated due to partial philopatry (Hunter *et al*., 2006; Chevolot *et al*., 2006). However, some immature rays might migrate into the different areas thus making genetic structure uniform all around the U.K., and defining population limits becomes challenging (Chevolot *et al*., 2006). Therefore, the respective definition of stock units becomes to some extent subjective and driven by expert judgement.

Based on the latest ICES advice the assessment area applied in BEEMS Technical Report TR406 (Subarea 4 & Divisions 3.a & 7.d (North Sea, Skagerrak, Kattegat & eastern English Channel)) may be too extensive. An alternative, more precautionary approach uses the assessment/TAC-allocation Subarea 4 and Division 2a (North Sea and Norway) as defined by ICES (ICES 2019c). During the period 2010-2017 landings of thornback ray in the Division 2a were negligible (< 1 t. per annum), whereas landings in the North Sea (Subarea 4) were of 491-905t, mean 695t. (ICES, 2020b). The stock size indicator demonstrated a strong increase in ray abundance from around 2012 onwards and the agreed TAC (798-1,404t.) was not taken by fishermen (ICES, 2019c).

The change is fisheries landings comparator is displayed in Table 4. Predicted losses from the proposed development increase from 0.02% of annual fisheries landings to 0.05%. It should be noted that landings have consistently remained within the TAC set annually since 2012 (798-1,404t.). The predicted annual losses from the proposed development account for between 0.04 and 0.07% of the TAC. Furthermore, it should be noted that using landings as a comparator is not equivalent to the larger (undefined) SSB for the species, so the assessment is highly precautionary. BEEMS Technical Report TR406 will be updated to incorporate this change. The change in landings data comparator has no bearing on the outcome of the assessment. No significant effects are predicted on thornback ray from the proposed development.

Table 4. Annual mean SZC predictions of impingement for thornback ray showing landings data applied in BEEMS Technical Report TR406 and proposed revision.

Species	Mean SZC prediction (No mitigation)*	SZC prediction with LVSE intakes	FRR mortality	EAV number	EAV weight (t)	Mean landings 2009-2017 (t)	% of landings
Thornback ray (TR406: Subarea 4 & Divisions 3.a & 7.d)	7,460	2,663	549	106	0.34	1,573	**0.02**
Thornback ray Revised landings data: Subarea 4 and Division 2.a (North Sea and Norway).	7,460	2,663	549	106	0.34	695	**0.05**

*Scaled impingement numbers reflect updates following release of BEEMS Technical Report TR339.v3.

The updated evidence provided herein will be integrated into version of BEEMS Technical Report TR406 (version 7) that will be submitted a part of a Supplementary Information Pack for the DCO in November 2020.

2.12 Sand eel

Comment: *"A prey species for relevant bird and mammal features (TR431) which needs suitable consideration".*

Response:

Sand eel are present in the waters off Sizewell but in low biomass (BEEMS Technical Report TR345) Common sand eel are impinged at Sizewell B at very low numbers with annual estimates from the CIMP predicted to be 3,175 individuals of the common sand eel (*Ammodytes tobianus*) and 570 greater sand eel (*Hyperoplus lanceolatus*) over the period 2014-2017 (BEEMS Technical Report TR339). Entrainment estimates are negligible, predicted to be 0.00010% of SSB (BEEMS Technical Report TR318). As stated in

BEEMS Technical Report TR431, sand eel form a part of the diet of many piscivorous seabirds and marine mammal including designated SPA and SAC species of relevance to Sizewell. However, many seabirds and marine mammals are opportunistic foragers and display plasticity in their diet depending on prey availability. For example, ICES benchmark workshop on sand eel (WKSand) analysed the importance of the species in the diet of the different seabirds and their respective vulnerability. Among seabirds occurring in Suffolk, the Sandwich tern was found to be highly vulnerable to change in the diet, but the species is seemingly more specialised on clupeids (ICES 2017)

The following is an extract from BEEMS Technical Report TR431 (revision 2):

"The most important fish families taken by breeding piscivorous seabirds in the North Sea are sand eels and clupeids with diets varying geographically and seasonally depending on the site-specific food availability (Tasker and Furness, 1996). The scientific literature frequently indicates that sand eels form a major part of the diet of terns (Common, Sandwich and Little Tern) and lesser black backed gulls in the North Sea. For example, Furness and Tasker (2000) estimate that sand eels form 40%, 60% and 20% respectively of the diet of the above three tern species in the southern and south-eastern North Sea. For seabirds in the north-western North Sea (such as Shetland and Orkney), there are no food-fish other than sand eels or adult herring and adult mackerel Scomber scombrus*. These adult fish are too big for most seabirds to eat, and so most seabirds in this region feed predominantly on sand eels.*

However, in the southern and eastern North Sea, the fact that sand eels form only a small part of the diet of many seabirds and that clupeids are predominant in many diets, suggests that clupeid abundance may be more important in determining breeding success than sand eel abundance in these areas. This is supported by detailed studies of Common Tern breeding success in the southeastern North Sea, where it has been found that chick growth rate and fledging success are closely correlated with abundance of young herring (Greenstreet et al., 1999).

The weight of evidence from BEEMS fishing, plankton, impingement and entrainment sampling is that sand eels are present at Sizewell but in small numbers (BEEMS Technical Report TR345). Sand eels spend most of their time buried in the sediment and are only found in the water column for a proportion of daylight hours. Due to their morphology they can pass through coarse mesh nets. For example, the 10mm drum screen mesh at Sizewell B would retain some sand eels with a variable proportion being entrained dependent upon the size of the fish. Impingement sampling data with a 10 mm mesh therefore provides a relative index of sand eel abundance not an absolute measure. In the BEEMS entrainment sampling (using pumped sampling from the Sizewell B forebay with 500 and 270 µm mesh nets) sand eel larvae only represented 2% [revised numbers <1%[2]] *of the number of fish entrained compared with 31% for clupeids (BEEMS Technical Report TR318).*

The dominant fish species at Sizewell are sprat and herring which represent 64% of the total annual impingement numbers. The evidence from surveys is that only two of the five North Sea sand eel species are present at Sizewell:

- *Lesser sand eel (*Ammodytes tobianus*). This is a small, common inshore species which reaches a maximum length of 20 cm, and is found along sandy shores from the mid-tide level to 30 m water depth. The species spawns from late March to early April throughout its range, depositing its eggs on the sandy substrate. This species represented 0.03% (See Table 5) of the annual Sizewell impingement numbers with 79% being caught in the December- January period.*

- *Greater sand eel (*Hyperoplus lanceolatus*) attains a length of approximately 32 cm. It is found in sand from the inter-tidal to 150 m depth. The species spawns in April and May at depths of 20-100 m. This species represented 0.026% (see Table 5) of the annual Sizewell impingement numbers with 71% being caught in the June -September period.*

To put the relative abundances in context, the total annual sand eel impingement catch at Sizewell was 0.1% of the sprat catch and in the seabird breeding season of May to August sand eel only represented 0.5% of

[2] The values quoted from TR431.v2 are based on previous versions of BEEMS Technical Reports TR318 and TR406. These reports have subsequently been modified in response to regulatory comments. As such, the exact figures are subject to minor changes, however, the scale of assessment and conclusions remain valid.

the sprat catch (indicative results from an interim analysis of BEEMS impingement surveys from 2009-2013). In the BEEMS pelagic fishing surveys in April and May 2015 (with 2mm mesh nets) at three sites along Sizewell Bay, the near surface catch composition was 81-98% sprat and herring with sand eel never exceeding 1% of catch (BEEMS Technical Report TR356). From the available evidence it is concluded that sand eel form a negligible part of the fish assemblage at Sizewell and are, therefore, not an important component of SPA/SAC protected species in the region of the site".

As a result of the very low predicted losses of sandeel, and the small relative contribution to the diets of marine mammals and seabirds locally to Sizewell, sand eel have not been described further in BEEMS Technical Report TR406. However, impingement rates are contextualised against SSB and landings in BEEMS Technical Report TR318.

3 Local effects assessment

Sizewell C stakeholders have indicated that in principle they agree with this assessment methodology for effects on fish at the stock level but questioned the application of ICES stock units for assessing potential effects of a coastal power station on local populations. The concern pertains to the geographical scale of ICES stock units resulting in impacts from the station having small perceived effects on SSB and an inability to determine local effects should they occur.

Cefas' position remains that ICES stock units are the most relevant scale of assessment for contextualising effects in relation to the sustainability of fish within safe biological limits. This position is consistent with that of the Marine Management Organisation (MMO). The MMO defines a stock as:

"*A fish stock refers to a fish population that is isolated from other stocks of the same species*"

(MMO, 2020).

In the Relevant Representation document of 30th September 2020, the MMO agreed in principle with the use of ICES stock areas to contextualise effects; *"The MMO conclude that the use of ICES stock areas for commercial fish species represents the current best scientific evidence available. There is currently no robust information that would support use of more local stock areas in the assessment".*

There are no isolated fish populations at Sizewell, fish live and move in an unconstrained coastal environment with most species undertaking wide spatial migrations throughout the year. As such, ICES stock units represent the best available evidence for assessing the impacts of the proposed development in relation to stock sustainability. However, it is acknowledged that assessments of effects on fish must consider aspects beyond stock sustainability. Such issues have been previously been described in BEEMS Technical Report TR406 but are explored in more detail herein, and summarised in the BEEMS Technical Report TR406 v7.

The potential for localised effects is particularly pertinent in an HRA context where impacts leading to direct or indirect likely significant effects (LSE) need to be considered. For example, the effects on any fish species listed under Annex II of the Habitats Directive must be considered in relation to the Special Area of Conservation (SAC) they are a designated feature of. There are no Annex II fish species designated in the SACs adjacent to the proposed development. Assessments do consider the SAC from which wide-ranging or migratory Annex II species predicted to be impinged are most likely to be associated to (e.g. allis shad in Section 2.1). However, assessments of remote effects on wide ranging species is not of immediate relevance to the request for further information on local-level effects. The most pertinent HRA consideration of local effects of the proposed development is that of indirect effects on fish as prey items for designated species. Prey species of designated SPA, Ramsar and SAC species of relevance to the proposed development are detailed in BEEMS Technical Report TR431.

In addition to HRA considerations, the Environment Agency has raised concerns regarding the potential for the proposed development, acting in-combination with Sizewell B, to cause a deterioration in the WFD Transitional Fish Classification Index of the Alde & Ore water body. The concern is primarily in relation to smelt but also identified the potential for changes in the abundance of herring, seabass, Dover Sole, thin-lipped grey mullet and Twaite shad to influence the status of the fish classification in the water body. BEEMS Scientific Position Paper SPP108 specifically addresses these concerns. In this section the potential for the proposed development to cause localised depletion of available prey items and species of importance for the WFD is considered using a simplified conceptual model. The model considered effects at a range of scales.

3.1 Scale of local effects assessment

In an open coastal environment, the first question to address regards the appropriate scale of assessment and how to delineate the system. Sizewell B cooling water infrastructure is located inshore of the Sizewell-Dunwich Bank within the Greater Sizewell Bay (GSB). Sizewell C intakes and outfalls would be located approximately 3km offshore seaward of the Sizewell-Dunwich Bank in deeper water. The smallest scale of

assessment is the Greater Sizewell Bay from which Sizewell B abstracts water, and the tidal excursion beyond the Sizewell-Dunwich Bank from where Sizewell C would abstract water.

Water from the GSB and tidal excursion exchanges with the wider area first in ICES Statistical Rectangle 33F1 and, at a wider scale in ICES Statistical Area 4c (Figure 1). ICES rectangle 33F1 extends from Lowestoft to the north of Sizewell to just above Felixstowe to the south (Figure 2).

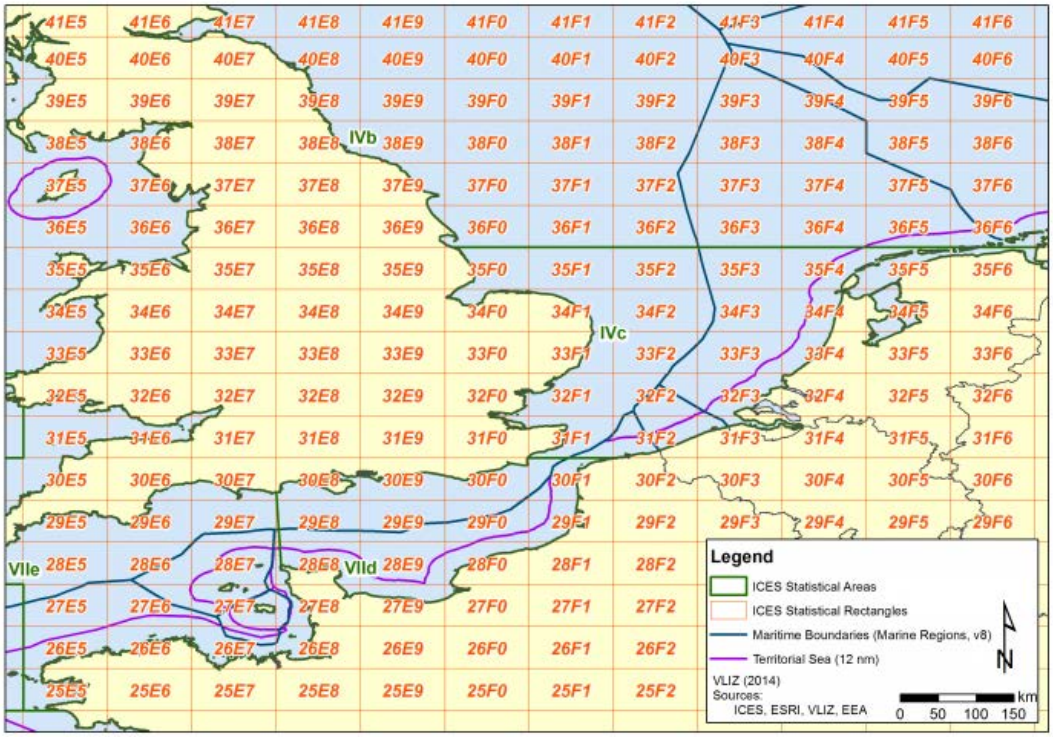


Figure 1 ICES statistical areas.

To determine the effects of entrainment on phytoplankton populations, from Sizewell B and C, BEEMS Technical Report TR385, determined the approximate volume of water within the influence of the power station during a tidal cycle. Based upon a current meter (S2) deployed near the proposed Sizewell C intake locations, a progressive vector diagram (PVD) method indicated that the north – south tidal excursion is approximately 15.9 km in each direction, and 1.4km east – west during spring tides. The trajectory of the tide flows both north and south, thus the tidal volume represents a body of water 31.8 km long and approximately 2.8km wide. For comparison, a harmonic analysis was conducted on the same current meter (BEEMS Technical Report TR233) and provided similar results to the PVD method. The surface area, average depth and estimated volume of the assessment cells is provided in Table 5.

Sizewell B forms part of the baseline against which effects on fish from the proposed Sizewell C are assessed (and against which the current WFD fish status is established). However, this assessment includes both stations acting in isolation and cumulatively thereby providing the most precautionary and transparent assessment of the impacts of Sizewell B and Sizewell C acting alone and in-combination. The potential for localised depletion of fish is considered here at the scale of the GSB and tidal excursion, 33F1 and 4c.

Table 5. Spatial area, average depth, and approximated volume of the assessment cells for local effects.

Assessment cells	Effect assessment	Surface area (ha)	Average depth (m)	Volume (x10^8 m^3)
GSB	SZB	4,120	8.8	3.64
Outer tidal excursion	SZC	7,081	13.9	9.85
GSB + tidal excursion	SZB + SZC	9,670	12.5	12.1
ICES 33F1	SZB + SZC	140,595*	23.2	326.8
ICES 4c	SZB + SZC	6,421,433	24.6	15,816

*33F1 is truncated as it intersects the east coast, a full ICES rectangle is approximately 380,000 ha.

3.2 Parameterising local effects assessments

Having defined the assessment cells, the next step is to attempt to parameterise a conceptual model for assessing local effects on fish.

The conceptual model makes the following assumptions:

- Fish are evenly distributed throughout the domain.
- Fish density (in the absence of abstraction) remains constant through the assessment period i.e. no mortality, recruitment, or migration into/out of the domain.
- Fish behaviour is not considered.
- Immigration and emigration rates are proportional to tidal exchange.
- ICES statistical Area 4c is considered a hard boundary i.e. there is no exchange/ replenishment of fish beyond this area.
- Assessment cells are not overlapping, i.e. the effects of impingement are concentrated at the local level with effects on the wider area a result of net losses due to exchange across the boundaries.

The validity of these assumptions is considered in Section 3.4.

3.2.1 Volumetric assessment

In this simplified assessment, local effects are based on volumetric assessments of abstraction relative to processes of tidal exchange. In the absence of behaviour, fish immigration and emigration from the assessment cells is proportional to the rate of tidal exchange. Tidal exchange is therefore a key parameter in the assessment. A typical value for exchange rate coefficients in partially mixed estuaries is 5% volume exchange on each tide (Dyer, 1979). As a general rule, the exchange volume at coastal sites on the east coast is approximately 10% over a 12-hour tidal period (Environment Agency, 2011). Assessments of effects on zooplankton (BEEMS Technical Report TR318) and phytoplankton (BEEMS Technical Report TR385) at Sizewell have incorporated a conservative 10% daily volumetric exchange rate. The smaller the volumetric replenishment the more conservative the assessment.

A 10% daily exchange of fish has been incorporated as the starting point for assessments between the GSB and tidal excursion and 33F1, and between 33F1 and 4c. The sensitivity of the results to changes in exchange rate is provided in Section 3.4.

It should be noted that the greater the exchange rate the lower the local effects due to dilution. However, as greater exchange rates maintain the local density, total impingement also increases with higher exchange rates. This assessment considers local level depletion, quantified losses compared to the relevant stock unit form the main part of the assessment.

Sizewell B operating at full capacity abstracts 51.5 m^3 s^{-1}. Over the course of 24 hours the station would abstract 4.45x10^6 m^3 of water from the GSB. Sizewell C operating at full capacity would abstract 131.86 m^3 s^{-1} resulting in a daily abstraction of 1.14x10^7 m^3 of water from the outer tidal excursion. The combined daily abstraction volume is 1.58x10^7 m^3 from the GSB + tidal excursion. Daily abstraction as a percentage of the assessment cell volumes (Table 5) and tidal exchange volume is provided in Table 6.

Local effects are considered within each assessment cell. The effects of abstraction are concentrated in the smallest spatial scale, wider effects are the result of reduced net exchange of fish between cells due to impingement losses. Effects in 33F1 are considered as the area beyond the GSB and tidal excursion (i.e. the volume of 33F1 – the volume of GSB and tidal excursion), likewise the effects on 4c subtract the area of overlap with 33F1. This approach allows the dilution of effects away from the impact to be observed. ICES 4a has been set as the final delineation, with no wider exchange, therefore all effects are concentrated within this area. For many species, this is conservative (Table 1). The assessment also shows localised depletion at a fine spatial scale.

Table 6. Daily abstraction volumes as a percentage of the assessment cell and daily volumetric exchange, under different operating scenarios.

Station	GSB		Tidal excursion		GSB + tidal excursion	
	% of cell	% of exchange	% of cell	% of exchange	% of cell	% of exchange
SZB	1.22	12.2				
SZC			1.16	11.6		
SZB +SZC					1.31	13.1

3.2.2 Fish density and mitigation

The initial density of fish is set at 0.001 ind. m^3, however, the assessment is independent of the starting density.

3.2.3 Mitigation

The assessment considers the proposed, and active mitigation measures for each site. Sizewell B is fitted with capped intake heads and a fish recovery and return (FRR) system. Sizewell C intakes would also be capped and adopt a low velocity side entry (LVSE) design with an FRR system. These mitigation measures would have differential effectiveness for different species groups (Table 7). The assessment herein, considers hypothetical species from representative pelagic, demersal, and epi-benthic taxa.

The sensitivity of impingement predictions to mitigation efficiency of LVSE and FRR systems associated with Sizewell C is reviewed in BEEMS Technical Report TR406 v7. A review of the evidence base undertaken by the Environment Agency as part of the Hinkley Point C project demonstrated that capped intakes reduce the abstraction of pelagic fish by a factor of 0.23 compared with uncapped heads. It should be noted that this mitigation only applies in the volumetric assessment and is not incorporated in impingement predictions as estimated impingement rates at Sizewell C are scaled up from Sizewell B (which has capped heads).

Table 7. Mitigation efficiency for different species groups in the volumetric assessment of impingement effects.

Station	Pelagic		Demersal		Epi-benthic	
	SZB	SZC	SZB	SZC	SZB	SZC
Capped intake	0.23	0.23				
LVSE	0.357	0.357	0.357	0.357	0.357	0.357
FRR	1	1	0.55	0.55*	0.2	0.2

* based on estimated seabass survival through the SZC drum and band screens.

3.2.4 Model duration

The local effects assessment was computed for 365 days, however, in almost all cases tested the system reaches equilibria within 50 days as immigration from the wider area balances losses through emigration and abstraction. The relevant duration for assessments is based on seasonal impingement rates for key species based on data in BEEMS Technical Report TR345 and TR339.

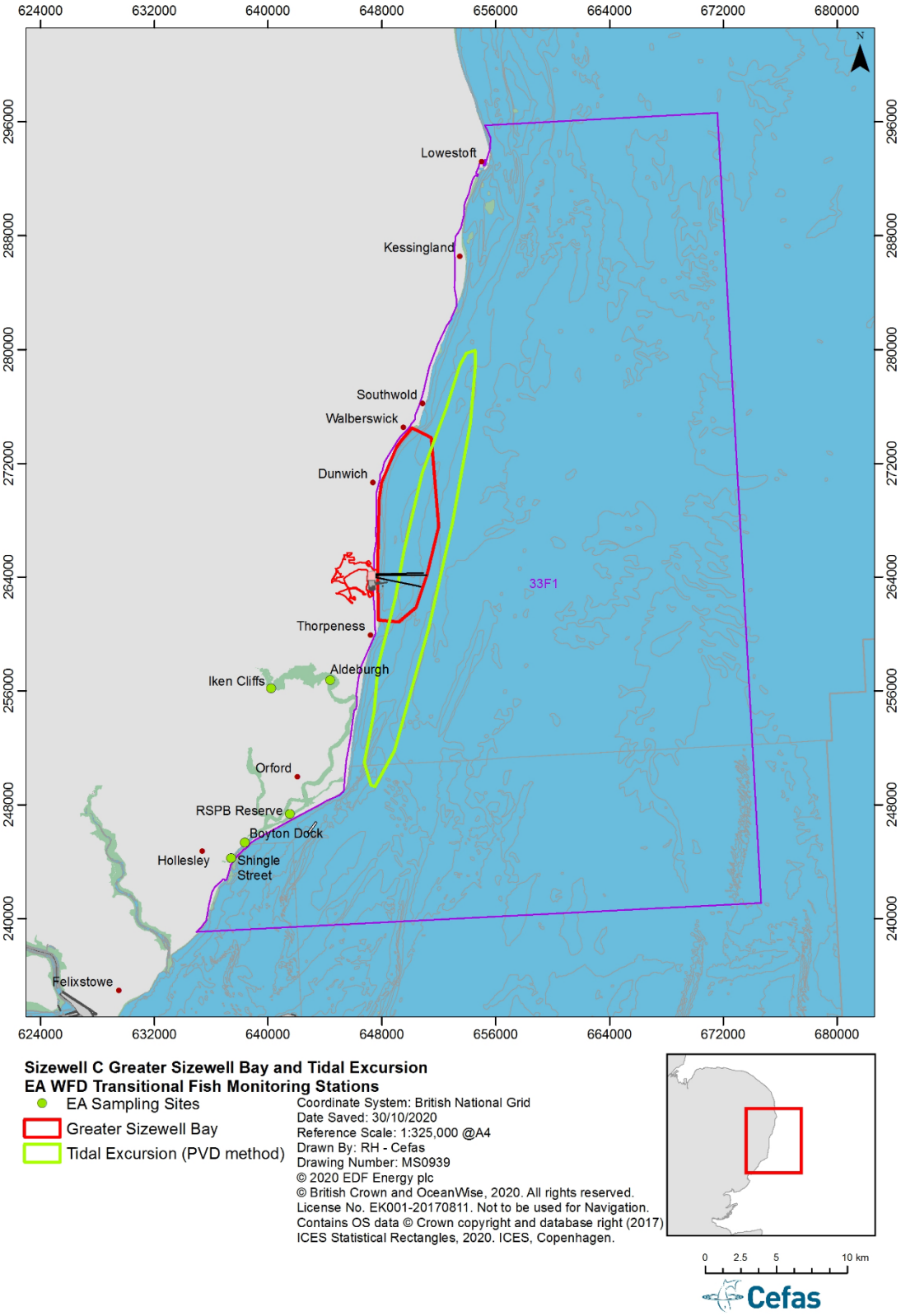


Figure 2. Scales of assessment for localised effects showing the GSB (red), the outer tidal excursion (green) and ICES Statistical Rectangle 33F1. ICES Statistical Area 4c extends beyond 33F1. The location of the Environment Agency TFCI monitoring stations in the Alde & Ore water body are also indicated.

3.3 Estimated local depletion

The potential for impingement to cause localised depletion of pelagic, demersal and epi-benthic species was considered in the case of Sizewell B and Sizewell C individually and in-combination at the scale of the Greater Sizewell Bay, tidal excursion, ICES Statistical Rectangle 33F1 and Statistical Area 4c. The assessment cells for in-combination depletion are illustrated in Figure 4.

3.3.1 Pelagic species

The two most dominant species impinged at Sizewell B are herring and sprat, together contributing over 70% of annual impingement by numbers. Both species are seasonally abundant at Sizewell with 94.9% of herring and 78.9% of sprat impinged in the period from Q4 to Q1 (BEEMS Technical Report TR339). Localised depletion was computed for the course of a year and depletion values were selected based on the period of time the species are most abundant at Sizewell. Exchange rates between the GSB and 33F1 and 33F1 and 4c were both set at 10% with mitigation.

Figure 3 shows that with both stations operating together, local depletion within the GSB + tidal excursion reaches a plateau after approximately 37 days by which point 95% of annual depletion is achieved. After 37 days, abstraction losses and emigration are balanced by immigration into the cell and effects on pelagic species spending longer than 37 days at Sizewell will therefore have very similar estimates for local depletion. Beyond 37 days depletion continues to rise at the scale of 33F1 and, to a lesser extent within ICES 4c, as the wider area dilutes impingement losses. Colour systems in the following graphs are intended to mirror the assessment cells in Figure 4.

With the stations acting in isolation, Sizewell B abstracts less than half the volume of the proposed Sizewell C station, however, Sizewell B intakes are inshore of the Sizewell-Dunwich Bank acting on a smaller volume of water. Impingement by Sizewell B results in depletion of approximately 2.5% of sprat within the GSB in the period December to March when they are most abundant, whereas herring would reduce by a similar margin in the period February to April. Sizewell C, with an LVSE intake would result in depletion of sprat and herring in the outer tidal excursion by approximately 0.88%, respectively (Appendix A).

Acting in combination, Sizewell B and a fully mitigated Sizewell C may result in reductions of 1.51 and 1.50% for sprat and herring, respectively during their period of peak abundance within the GSB + tidal excursion. Exchange with the wider area results in 0.07% depletion of sprat and 0.60% depletion of herring in the area of 33F1 beyond the GSB and tidal excursion. Within the area of 4c beyond 33F1, losses are diluted further and depletion after 121 days for sprat is 0.031% and depletion at the level of 4c is 0.009% in the case of herring (Table 8; Figure 3). Figure 4 provides a spatial illustration of the areas of depletion for sprat and dilution of effects in the wider area. In the absence of LVSE mitigation at Sizewell C, localised depletion within the GSB + tidal excursion increases to 2.76% for herring and 2.77% for sprat with both stations operating.

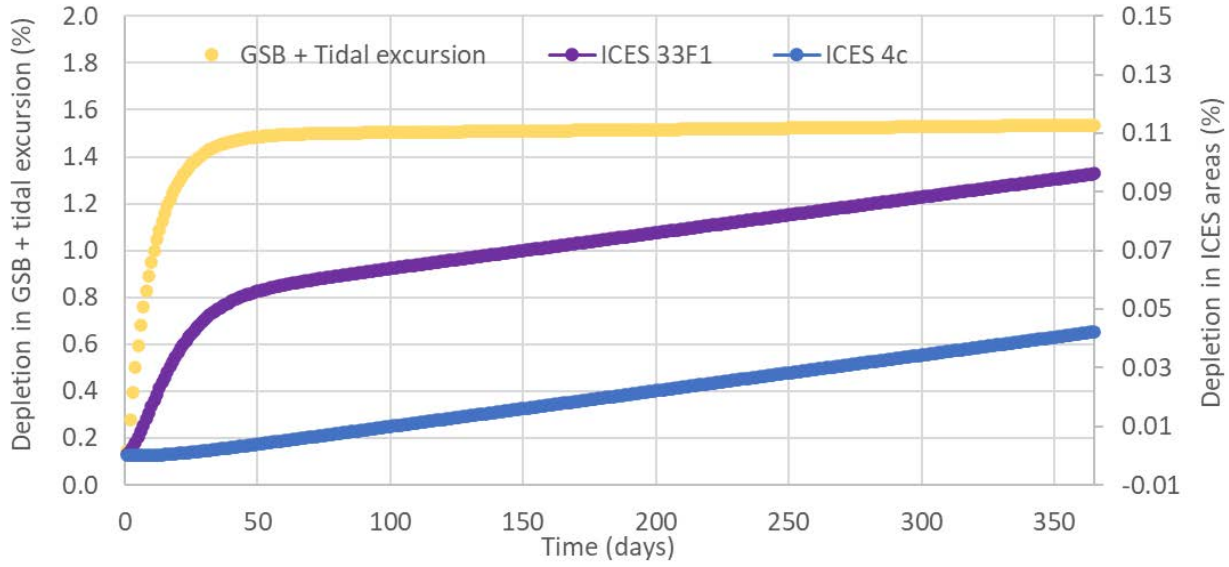


Figure 3 Localised depletion of pelagic fish due to impingement from Sizewell B and Sizewell C with full mitigation. The in-combination effects of Sizewell B and C at the scale of the GSB + tidal excursion (yellow), and the wider effects on ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue) are displayed.

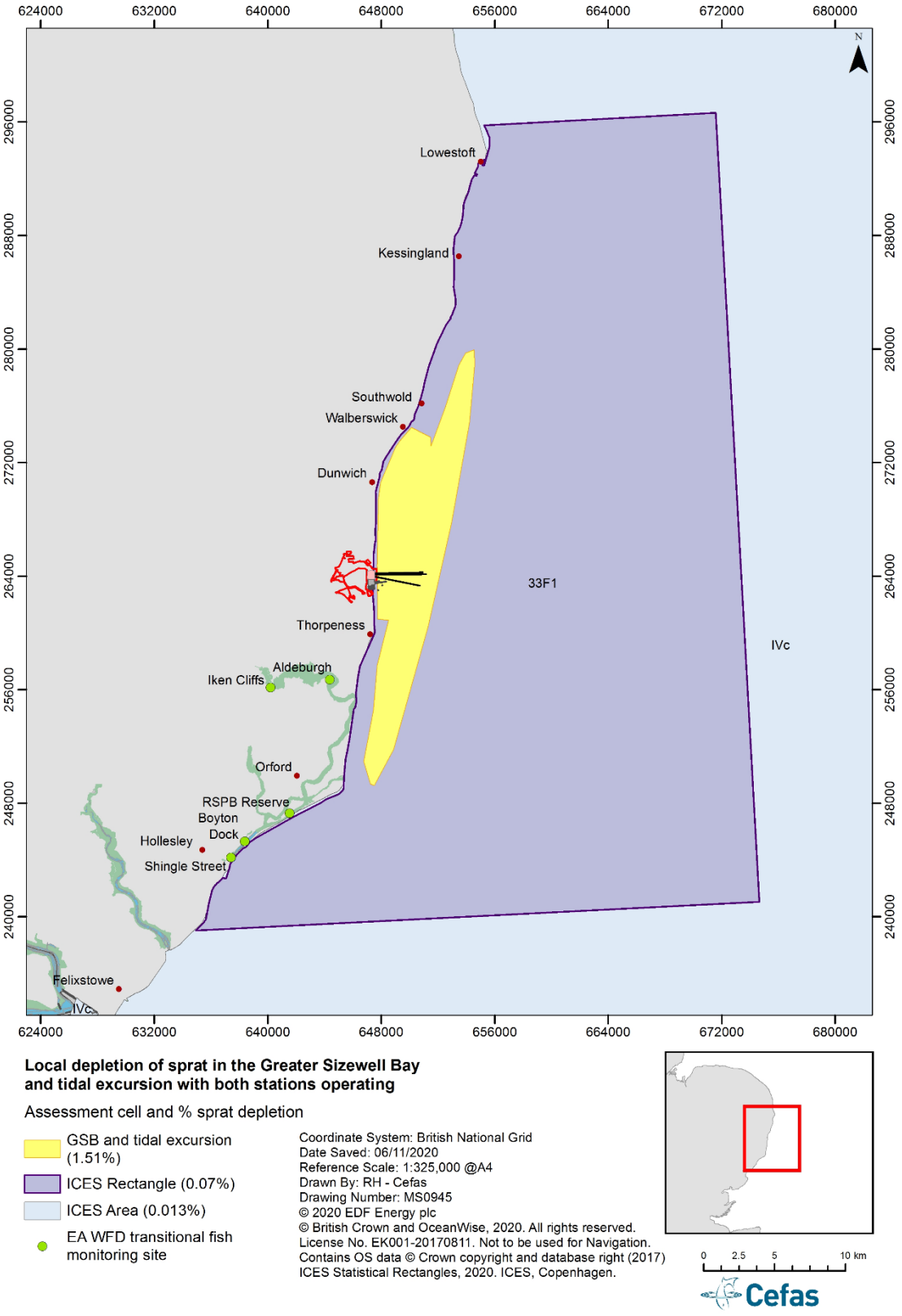


Figure 4 Areas of localised depletion for sprat assessed with both stations acting in-combination during the period December to March. The GSB + tidal excursion (yellow), ICES statistical rectangle 33F1 (purple) and part of ICES Statistical Area 4c (blue) are shown. Depletion in each assessment cell is depicted by shading relative to epi-benthic and demersal species.

3.3.2 Demersal species

Demersal species losses are mitigated by the FRR systems allowing approximately 50% survival (Environment Agency, 2005). Whiting and seabass as commonly impinged in winter at Sizewell B and together account for 17.5% of total annual impingement numbers and 27.3% by mass. Seabass are highly seasonal with 99.5% impinged in Q4 and Q1, equally 77.7% of whiting are impinged in the winter period (BEEMS Technical Report TR339).

FRR survival rates for demersal species are based on the estimated 45% survival of seabass through the trash racks, drum screens and band screens at Sizewell C. Both Sizewell B and Sizewell C have an FRR system. Exchange rates between the GSB and 33F1 and 33F1 and 4c were both set at 10%.

Similar rates of local depletion occur in the GSB and tidal extent due to the operation of Sizewell B and Sizewell C for seabass and whiting during their period of seasonal abundance off Sizewell. Sizewell B alone is predicted to result in 2.15% local depletion in seabass and whiting in the GSB, and approximately 2.1% in the tidal excursion during the operation of Sizewell C alone (Appendix A).

Acting in-combination, Sizewell B and C, result in less than 2.4% reductions in the demersal species in the GSB + tidal excursion. Exchange with the wider area results in 0.12% depletion of whiting and 0.11% depletion of seabass in the area of 33F1 beyond the GSB and tidal excursion, and 0.039% depletion of whiting and 0.020% for seabass in ICES 4c beyond 33F1 (Table 8; Figure 5). Figure 6 provides a spatial illustration of the areas of depletion for whiting and dilution of effects in the wider area.

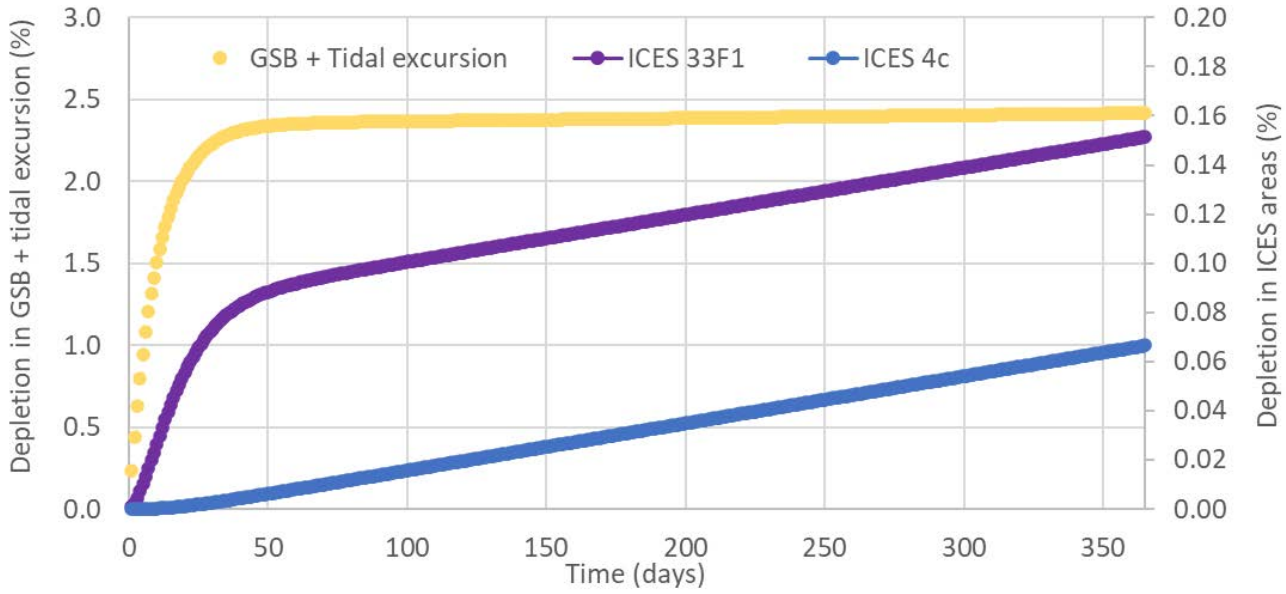


Figure 5 Localised depletion of demersal fish due to impingement from Sizewell B and Sizewell C with full mitigation. The in-combination effects of Sizewell B and C at the scale of the GSB + tidal excursion (yellow), and the wider effects on ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue) and displayed.

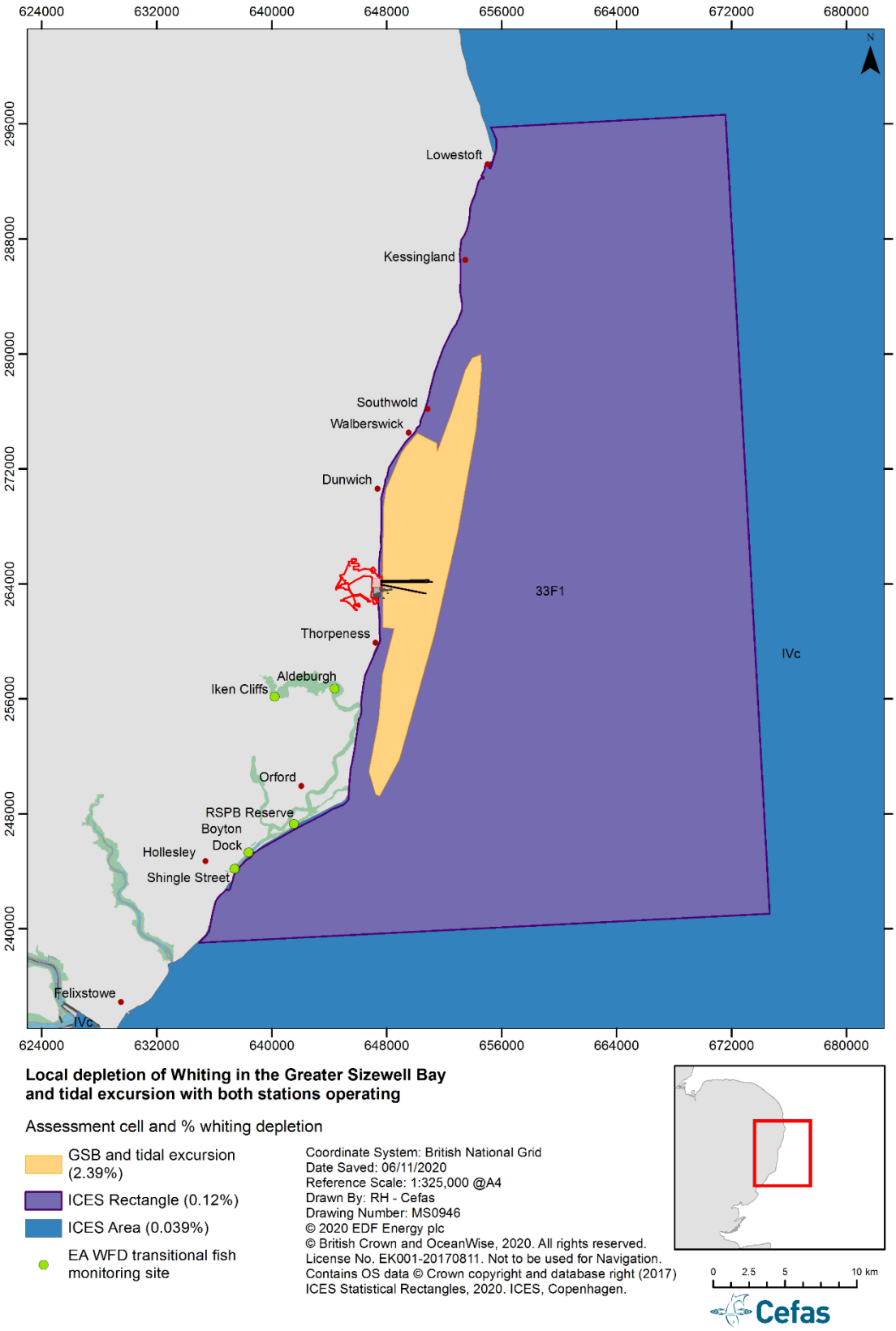


Figure 6 Areas of localised depletion for whiting between assessed with both stations acting in-combination during the period October to April. The GSB + tidal excursion (yellow), ICES statistical rectangle 33F1 (purple) and ICES Statistical Area 4c (blue) are shown. Depletion in each assessment cell is depicted by shading relative to epi-benthic and pelagic species.

3.3.3 Epi-benthic species

Epi-benthic species typically incur the least mortality through the FRR system, and 80% survival is assumed (Environment Agency, 2005). Dab, Dover Sole and sand gobies all contribute towards the top 95% of species impinged at Sizewell B. Sand gobies are predominantly impinged in Q3 and Q4, however Dover sole and dab are impinged for longer periods throughout the year (BEEMS Technical Report TR339). Impingement assessments were run for 365 days, however, equilibria within the GSB + tidal excursion was achieved by 37 days (95% of annual depletion achieved) after which effects increase in the wider area. Exchange rates between the GSB and 33F1 and 33F1 and 4c were both set at 10%.

Similar rates of local depletion occur in the GSB and tidal excursion for epi-benthic species due to the operation of Sizewell B and Sizewell C of approximately 0.8% (Appendix A). Acting in-combination, Sizewell B and C, may result in a 0.9% reduction in epi-benthic species in the GSB + tidal excursion. In the worst-case annual dab assessment, exchange with the wider area results in 0.06% depletion of fish in the area of 33F1 beyond the GSB and tidal excursion, and 0.025% in ICES 4c beyond 33F1 (Table 8; Figure 7). Figure 8 provides a spatial illustration of the areas of depletion for Dover sole and dilution of effects in the wider area.

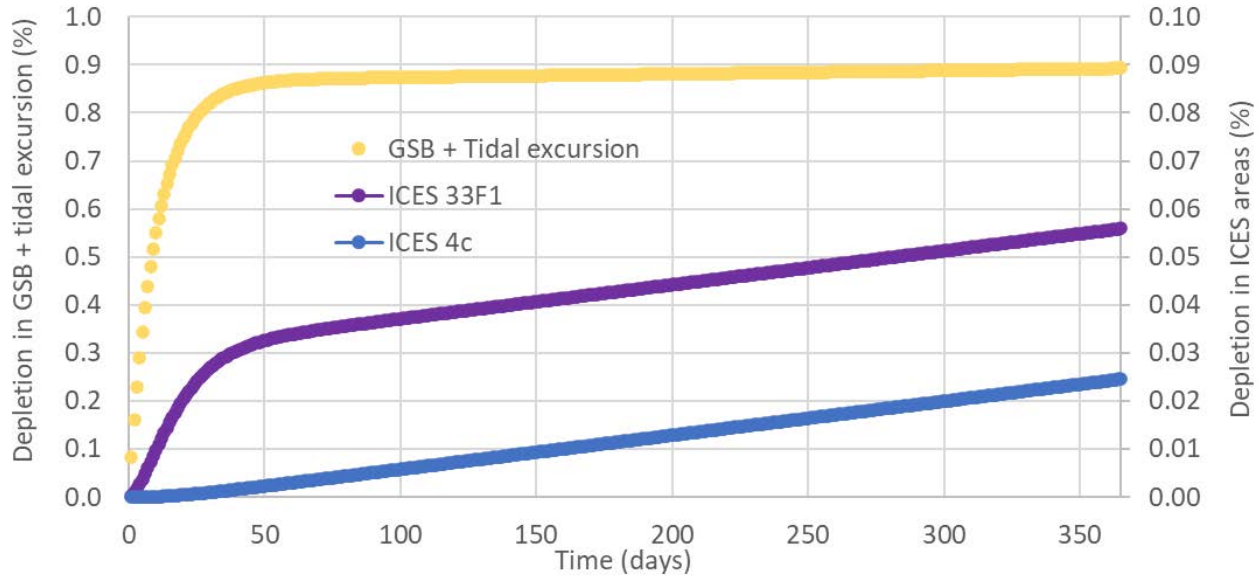


Figure 7 Localised depletion of epi-benthic fish due to impingement from Sizewell B and Sizewell C with full mitigation. The in-combination effects of Sizewell B and C at the scale of the GSB + tidal excursion (yellow), and the wider effects on ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue) and displayed.

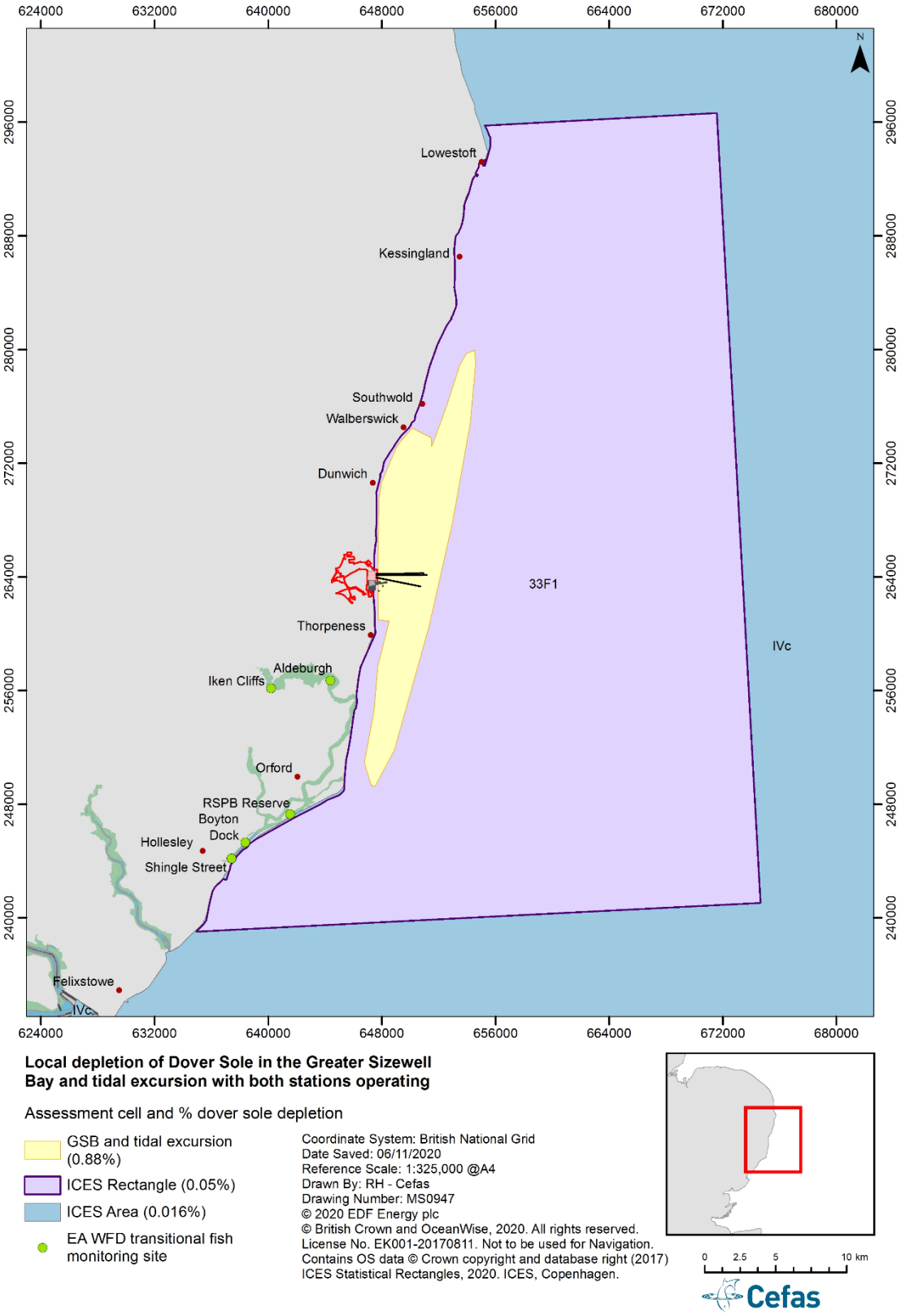


Figure 8 Areas of localised depletion for Dover sole assessed with both stations acting in-combination during the period March to October. The GSB + tidal excursion (yellow), ICES statistical rectangle 33F1 (purple) and ICES Statistical Area 4c (blue) are shown. Depletion in each assessment cell is depicted by shading relative to pelagic and demersal species.

Table 8 Seasonal abundance of selected key species impinged at Sizewell B and predicted depletion during those periods at a range of scales. Values provided show local effects of Sizewell B and Sizewell C operating in-combination and are based on 10% exchange rates. Green shaded species have been identified as potential prey items for HRA designated species at Sizewell. Underline species have been identified by the Environment Agency as important for the Alde & Ore WFD fish classification.

Species	Group	Period that the species is found in the area close to the Sizewell power stations (days)	Predicted % depletion in each area due to SZB + SZC		
			GSB + tidal excursion	33F1	4c
Sprat	Pelagic	Dec. – March (121)	1.51	0.07	0.013
Herring	Pelagic	Feb. – April (89)	1.50	0.06	0.009
Whiting	Demersal	Oct. – April (221)	2.39	0.124	0.039
Seabass	Demersal	Dec. – March (121)	2.37	0.105	0.020
Sand goby	Epi-benthic	Aug. – Dec. (162)	0.88	0.042	0.010
Dover sole	Epi-benthic	March – Oct. (245)	0.88	0.047	0.016
Anchovy	Pelagic	June (30)	1.42	0.05	0.002
Dab	Epi-benthic	Throughout (365)	0.89	0.056	0.025
Thin-lipped grey mullet	Demersal	Jan. – March (90)	2.37	0.099	0.014
Flounder	Epi-benthic	Oct. – June (282)	0.89	0.050	0.019
Plaice	Epi-benthic	Throughout (365)	0.89	0.056	0.025
Smelt	Pelagic	April – Nov. (253)	1.52	0.08	0.029
Cod	Demersal	Dec. – March (121)	2.37	0.105	0.020
Thornback ray	Epi-benthic	March – May (92)	0.87	0.037	0.005
Eel	Epi-benthic	Aug. – Nov. (131)	0.88	0.039	0.008
Twaite shad	Pelagic	March – Aug. (184)	1.51	0.07	0.20

Seasonal source data BEEMS Technical Report TR345 and TR339.
Key prey species for HRA designated Annex II species are detailed in BEEMS Technical Report TR431.

3.4 Sensitivity to parameters

The values for local depletion are presented as an average value over the relevant assessment scale. Localised depletion would be concentrated downstream of the intakes and as such would be tidally dynamic. Processes of mixing and fish behaviour would reduce depletion further from the intakes. However, what the results clearly show is how the effects of impingement are rapidly diluted.

The results presented herein, are subject to the validity of the assumptions described in Section 3.43.2. Two key assumptions are considered in further detail 1) the replenishment rate (tidal exchange), and 2) the assumption of equal density.

3.4.1 Sensitivity to replenishment rates

The underlying factor determining the potential for local depletion is the rate at which impinged fish are replenished from the wider area by immigration. Quantifying immigration and emigration rates for a range of species in an open coastal site is simply not feasible. This point was reflected in the Relevant Representations from the MMO that stated *"it is the MMO's view that in the absence of information on the rate at which fish may migrate into the GSB area (the replenishment rate), it is difficult to quantitatively assess localised impacts on abundance".* This report attempts to put depletion in a local context by applying tidal exchange rates as a proxy for fish replenishment.

Whilst many species use tidal transport as a means to conserve energy and move within coastal environments, assuming fish have no inherent behaviour is clearly an oversimplification. Different species will have different retention times within the GSB, some will occur within the GSB year-round whilst others will be seasonal migrants. Shoaling behaviour and changes in density are considered in the following section.

Within this assessment there are effectively two boundaries; 1) the boundary of the GSB and tidal excursion with 33F1, and 2) the boundary between 33F1 and 4c. Exchange rates (as a proxy for fish replenishment) were tested across an range or levels from 0.1% to 25% for epi-benthic species with the lowest mortality due to impingement, and for highly mobile but impingement sensitive pelagic species using parameters in Table 7.

The greater the exchange rate the lower the local effects due to dilution. However, as greater exchange rates maintain the local density, total impingement also increases. This point is illustrated in Figure 9, which shows the relationship between local depletion in the GSB + tidal excursion and impingement from Sizewell C and Sizewell B. By fixing impingement numbers based on 10% daily exchange it is apparent that annual impingement has low sensitivity to variations in exchange parameters between 5 and 25%, with total impingement numbers changing less than ±2%. However, annual impingement drops by 9.4% as exchange rates fall to 1% whilst local level depletion increases from 1.5% with 10% exchange rates to 14% with 1% exchange (Figure 9).

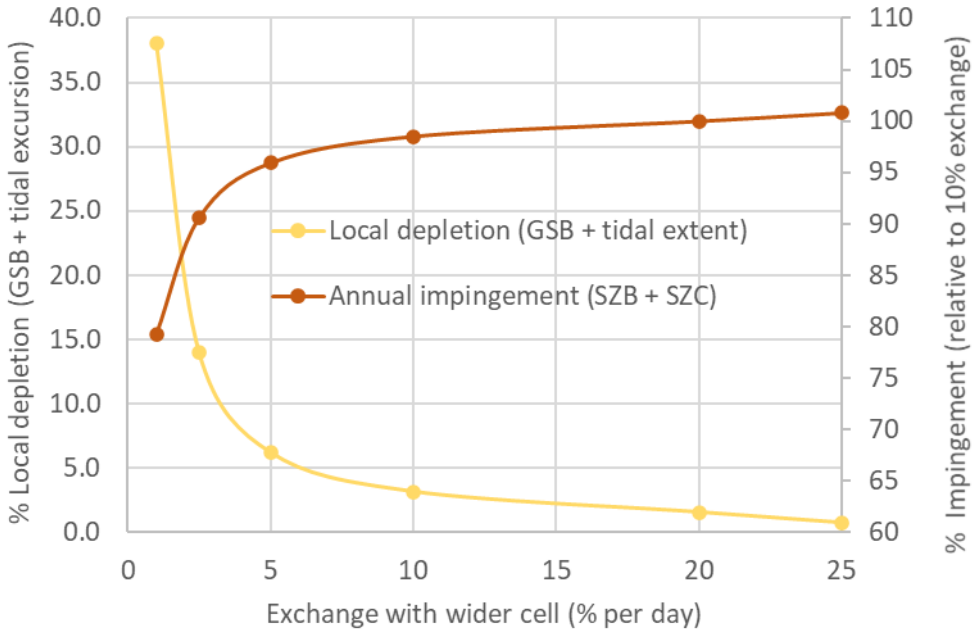


Figure 9. The relationship between local depletion and impingement of pelagic species within the GSB + tidal excursion with both Sizewell B and Sizewell C operating. Impingement rate percentages are calculated based on annual impingement using 10% daily exchange.

3.4.1.1 Epi-benthic species

With Sizewell C operating in combination with Sizewell B, exchange rates of just 1% between adjoining assessment cells result in localised depletion of 8.5% of epi-benthic species after 365 days. The greater the exchange rate the more rapidly local effects are diluted within the wider area. Localised effects at the scale of the GSB + tidal excursion reduce dramatically between 0.1% and 5% after which they level off. At 5% daily exchange across the boundaries, effects within the GSB + tidal excursion are reduced to 1.83% compared to 0.89% with a 10% exchange (Figure 10). This demonstrates that local depletion of epi-benthic species is minimal even with modest tidal exchange rates.

The effects of the exchange between 33F1 and 4c was tested by fixing exchange between 33F1 and the GSB + tidal excursion at 10% and changing the exchange with the wider ICES areas from 0.1 to 10%. Local effects at the GSB + tidal excursion scale increase 2-fold as exchange rates decrease from 10% to 0.1%, between 33F1 and 4c, however, local effects are relatively insensitive to changes in wider exchange beyond 2.5%. As would be expected, the change in exchange rates between 33F1 and the wider 4c area has the greatest effects on 33F1. However, as little as 1% exchange per day results in just 0.35% depletion in abundance of epi-benthic fish in 33F1 after 1 year (Figure 11).

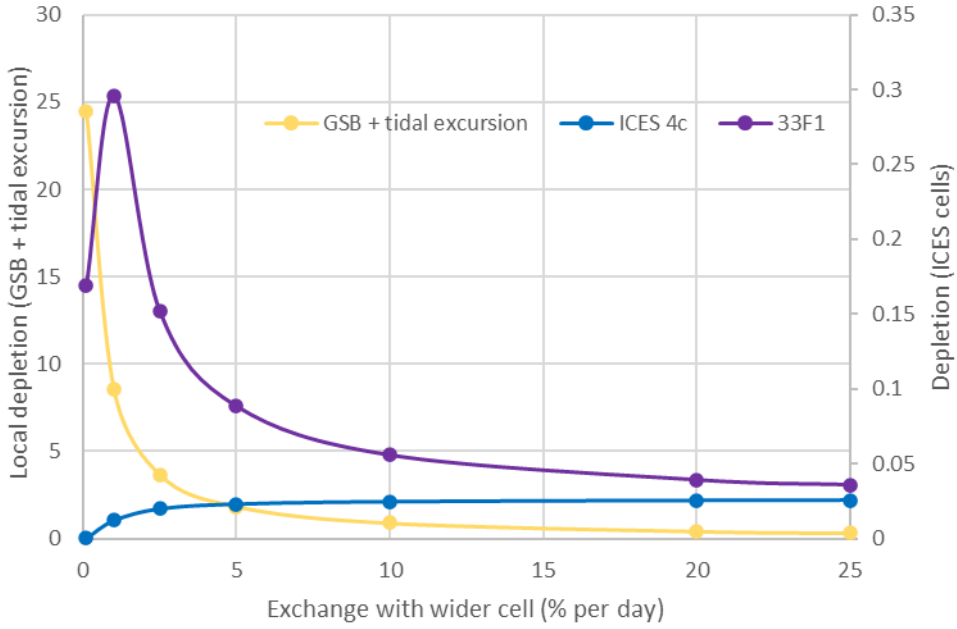


Figure 10. The sensitivity of localised effects on epi-benthic species to changes in daily tidal exchange after 365 days due to Sizewell B and Sizewell C operating with full mitigation acting at the scale of the GSB + tidal excursion (yellow), ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue). Exchanges rates; 0.1%, 1%, 5%, 10%, 20%, and 25% across all boundaries.

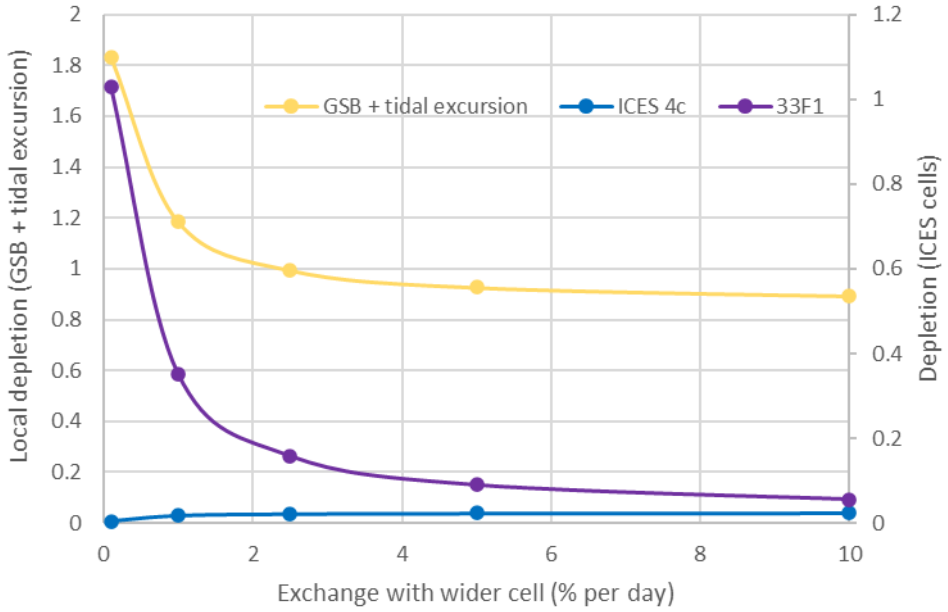


Figure 11. The sensitivity of localised effects on epi-benthic species to changes in daily tidal exchange after 365 days due to Sizewell B and Sizewell C operating with full mitigation acting at the scale of the GSB + tidal excursion (yellow), ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue). Exchanges rates fixed at 10% between the GSB + tidal excursion and ranging from 0.1%, 1%, 5%, 10%, across the 33F1 and 4c boundary.

3.4.1.2 Pelagic species

In the case of mobile pelagic species, a 10% per day replenishment rate applied in the original assessment appears suitably precautionary for most species. To simulate the case of species that spend longer periods in the coastal waters off Sizewell, for example smelt, the sensitivity to exchange rates following 253 days was assessed. With Sizewell C operating in combination with Sizewell B, exchange rates of just 1% of fish between adjoining assessment cells resulted in localised fish depletion of 13.3% after 253 days. At 5% daily exchange across the boundaries, effects within the GSB + tidal excursion are reduced to 3.1% compared to 1.5% local depletion with a 10% exchange (Figure 12). Local depletion is therefore modest at 3% or below if there is greater than a 5% exchange rate of smelt within the GSB + tidal excursion each day from the wider area.

Smelt are relatively common on the East Anglian coast. Comparative genomic analyses (BEEMS Technical Report TR423) have concluded that smelt from Sizewell and the Rivers Thames, Waveney, and Great Ouse are genetically homogeneous with no genetic structuring seen within the region indicating a single stock unit from at least the Greater Ouse to the Thames. It is also considered likely, although currently unproven, that there is a single population along the European coast from at least the Elbe to the Scheldt (BEEMS Technical Report TR406). The Elbe and the Scheldt have very large breeding populations of smelt and it is likely that they have extensive summer feeding grounds in the southern North Sea that may overlap areas used by UK sub populations. The regional, and wider distribution of smelt indicates that replenishment rates would be sufficient to limit localised depletion to modest levels.

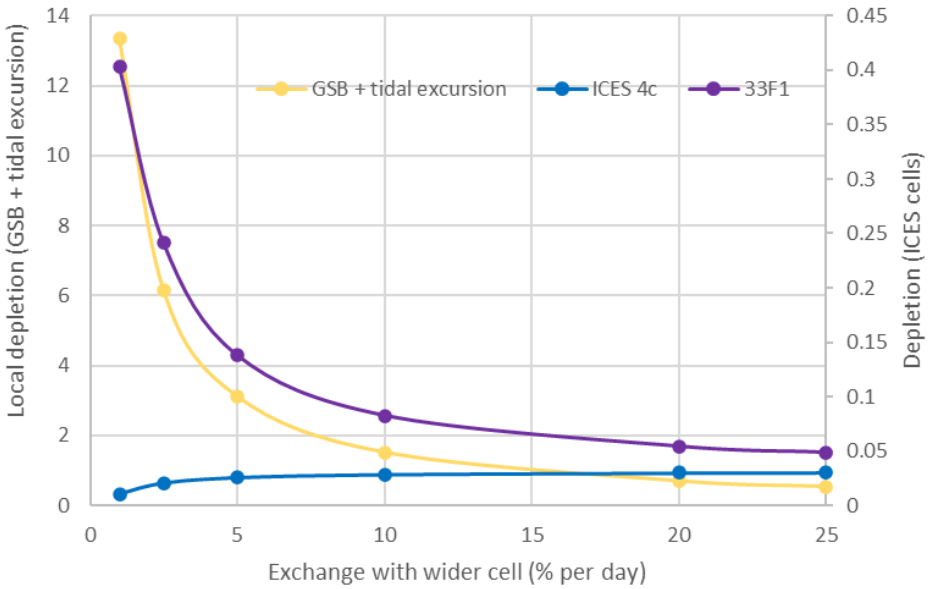


Figure 12. The sensitivity of localised effects on pelagic species (smelt) to changes in daily tidal exchange after 253 days due to Sizewell B and Sizewell C operating with full mitigation acting at the scale of the GSB + tidal excursion (yellow), ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue). Exchanges rates; 0.1%, 1%, 5%, 10%, 20%, and 25% across all boundaries.

3.4.2 Sensitivity to density assumptions

The assumption of uniform density across the domain is another simplification of the conceptual model. Many species exhibit seasonal migrations and shoaling behaviour. Shoaling species result in highly variable impingement rates, as such local depletion would equally be variable. Should a shoal pass directly within the small abstraction zone for the intakes, the local depletion would be higher than predicted. Equally should a large shoal within the GSB or tidal excursion not encounter the intakes there would be negligible depletion of the species.

As the scale of the assessment increases the assumption of uniform density is increasingly violated. For example, juvenile herring and sprat utilise inshore nursery grounds overwinter and whilst adult sprat spawn inshore they move further offshore post-spawning to forage mostly in the southern and western North Sea (BEEM Technical Report TR406). However, both 33F1 and 4c extend to the coast areas and impingement numbers are small relative to the stock allowing replenishment of fish.

The conceptual model is does not incorporate mortality and recruitment. To account for this the concept of localised depletion was challenged with an extreme example using sand goby entrapment as a trial. The model was run for 365 days with Sizewell B acting alone on the GSB and assuming no mitigation benefits (i.e. 100% mortality or entrainment). Exchange rates were maintained at 10% per day at the boundary of the assessment cells.

In this example, local depletion increases substantially in the GSB reaching the asymptote after approximately 30 days. Over a full year local depletion within the GSB would reduce initial densities by 10.1% (Figure 13). Sizewell B has been operational for 35 years and sand gobies were the second most abundant species in Sizewell baseline 2 m beam trawl samples and were found across the whole area. Between-site variability at a given sampling site was generally high or very high and between-survey abundance was also variable. The mean numbers recorded in June in Sizewell fish surveys are comparable to densities reported in Region 2 (Winterton to North Foreland) during Cefas Young Fish Surveys (BEEMS Technical Report TR345).

Sand gobies, *Pomatoschistus minutus* exhibit a rare spawning strategy with only one spawning season, during which females breeds repeatedly in multiple brood cycles. The spawning season extends from

January to June (Froese, Pauly, 2019). Egg are laid in bivalve shells and development takes 2-10 days before larvae emerge and have a pelagic lifestyle for at least one month (Fonds, 1973) being transported by currents across wide areas (Section 2.8). Gobies are small, short-lived species that mature within their first year. A long recruitment period with wide larval dispersal coupled with seasonal migrations into/out of the bay means replenishment rates and natural variability swamp the impacts from entrapment.

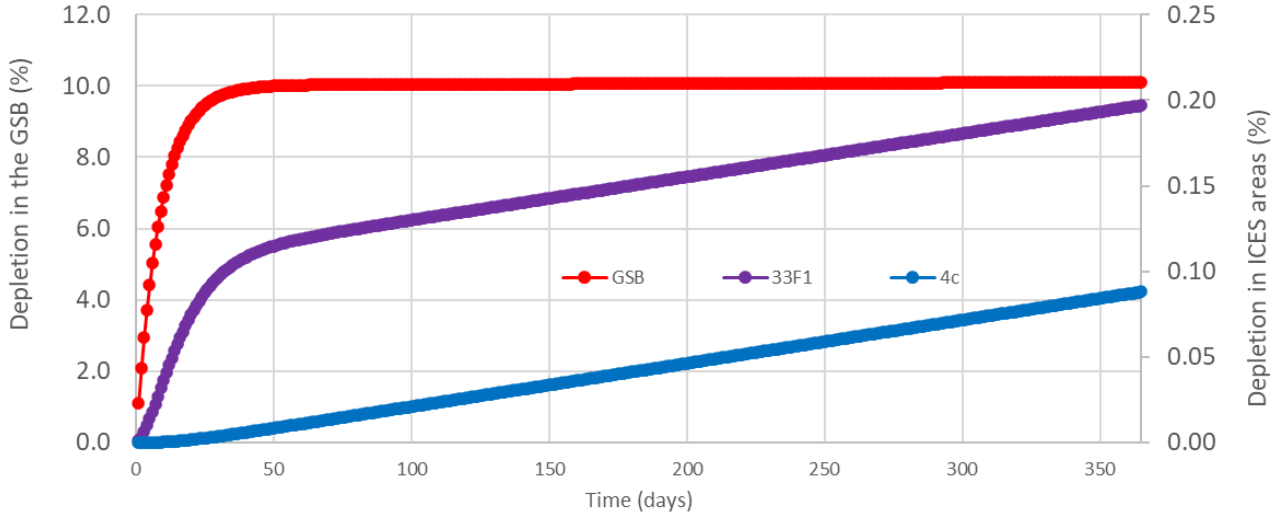


Figure 13. The sensitivity of localised effects on an entrained species (sand goby) due to Sizewell B operating with no mitigation at the scale of the GSB (red), ICES Statistical Rectangle 33F1 (purple) and Statistical Area 4c (blue). Exchanges rate 10% across all boundaries.

3.5 Local depletion in an HRA and WFD context

3.5.1 Food webs

The most seasonally abundant species at Sizewell that are considered important prey items for designated Annex II species include the pelagic species sprat, herring and anchovy, epi-benthic species such as plaice, Dover sole, flounder and gobies and demersal whiting and juvenile seabass.

Assuming 10% daily exchange of fish, the maximum levels of local depletion due to Sizewell C and Sizewell B acting in-combination is less than 2.5% for demersal species on average during periods of seasonal presence (Table 8). It should be noted that this represents and average value across the GSB + tidal excursion and the effects would be tidally dynamic and concentrated around the intakes. Even at exchange rates as low as 5% per day local depletion of pelagic species is expected to be in the order of 3% with epibenthic species reduce by less than 2% and demersal species by less than 5% with both stations operating.

The foraging ranges of designated sandwich terns, lesser black-backed gulls, and common terns during the breeding season extends beyond the GSB + tidal excursions allowing them to access prey resources from a wider area. Marine mammals and overwintering red throated diver have greater foraging ranges still without the restriction of having to return to local breeding colonies near Sizewell. Little terns have the most area restricted foraging ranges and during the breeding season little tern forage close to their colonies out to a maximum distance of approximately 2.4 km offshore (BEEMS Technical Report TR431). Breeding colonies at Minsmere, Dingle marshes and Slaughden would forage to a large extent within the GSB + tidal extent, however, foraging would primarily be within the Sizewell-Dunwich Bank and more subject to the immediate effects of Sizewell B.

Local depletion in prey resources must be considered in the context of natural variability in resource availability. Table 9 illustrated the year-to-year variability impingement rates at Sizewell B, a proxy for natural abundance, in prey species. The large interannual variability clearly demonstrates that local depletion would be dwarft in comparison. For example, local depletion of sprat is predicted to be approximately 1.5% within

the GSB + tidal extent, whilst mean interannual variability is a factor of 2 with maximum between year variation a factor of 3.6. Local depletion of anchovy during its seasonal occurrence at Sizewell is predicted to be 1.4% for both stations operating (Table 8) whilst interannual variability is on average orders of magnitude greater at 5.02.

The scale of local depletion of prey resources is well within the bounds of natural variability, which predator-prey relationships are adapted to. Furthermore, the FRR systems means that biomass is retained within the system resulting in bottom-up effects stimulating secondary production and, in some cases, affording opportunistic feeding opportunities for seabirds. Whilst the majority of FRR discards sink and would therefore not be accessible to surface feeding seabirds, floating discards would represent a potential foraging opportunity to scavenging seabirds (BEEMS Technical Report TR501). Experiments have shown seabirds can take up to 71%, 8%, 12% and 4% of discarded roundfish, flatfish, elasmobranchs and invertebrates, respectively, in the southern North Sea (Garthe *et al.,* 1996).

Local depletion of prey availability, notwithstanding the retention of biomass from the FRR systems, is predicted to be minimal in comparison to natural variability. As such, no significant reduction in the prey availability of designated HRA species is anticipated.

Table 9 Year to year variations in annual predicted SZB impingement numbers (2009-2017) for selected species.

Species	The largest year-year changes in annual numbers from the Sizewell B CIMP dataset 2009-2017[1] (shown as the ratio of predicted impingement numbers in adjacent years)	
	Mean interannual variability (2009-2017)	Maximum interannual variability between any two adjacent years
Sprat	2.02	3.64
Herring	1.33	1.70
Whiting	1.56	2.04
Bass	2.36	6.11
Sand goby	4.20	15.32
Sole	1.53	2.26
Dab	2.40	7.88
Anchovy	5.02	8.85
Flounder	1.35	1.74
Plaice	1.78	2.74

[1] Data from the year 2013 is not available, therefore interannual variability looks at changes from 2012 to 2014.

3.5.2 WFD fish status

The Environment Agency raised a concern pertaining to the WFD in regards to the impact of Sizewell B and Sizewell C acting in-combination causing local depletion of smelt potentially resulting in a deterioration in the status of the Alde & Ore water body Transitional Fish Classification Index (TFCI). This issue is specifically addressed in BEEMS Scientific Position Paper SPP108 where the sensitivity of the Alde & Ore was tested to

To determine the sensitivity of the TFCI to smelt abundance, smelt numbers in samples were manipulated at a range of levels including complete smelt absence. At the resuest of the Environment Agency, a further test considered absence of smelt and Twaite shad and 50% reductions in herring and seabass. The sensitivity of the TFCI was also tested through manipulated removals of thin-lipped mullet and Dover sole, as well as a scenario whereby smelt, thin-lipped mullet and Dover sole were all simultaneously reduced by 50%.

1. The calculated Ecological Quality Ratio (EQR) was insensitive to manipulated reductions in smelt abundance of 25% and 50%.

2. Total absence of smelt reduced the EQR by 11% but 'good' status remained (when fyke net data is not included in the TFCI calculation).

3. The extreme case requested by the Environment Agency including absence of shad (1 individual caught between 2013-2018) and smelt along with 50% reductions in herring and bass reduced the EQR by 10.3%, however, 'good' status remained (TFCI calculated without the inclusion of fyke net data).

4. Total absence of thin-lipped grey mullet and Dover sole reduced the EQR by less than 4% in each case and 'good' status remained. The status also remained 'good' following the combined 50% reduction of smelt, Dover sole and thin-lipped grey mullet.

Under all of the scenarios trested for fish manipulations, there was no deterioration of 'good' status when the 2019 TFCI was calculated without fyke net data[3] (BEEMS Scientific Position Paper SPP108).

The mouth of the Alde-Ore Estuary is 25km to the south of Sizewell C (near Shingle Street in Figure 2) beyond the GSB + tidal excursion and would therefore be subject to lower levels of local depletion predicted here. Even in the conservative case of 5% daily replenishment rates of smelt within the GSB + tidal excursion the localised reduction of smelt during summer feeding is predicted to be 3.1%, opposed to 1.5% depletion assuming 10% daily exchange (Section 3.4.1.2). Given the TFCI is insensitive to 25 and 50% reductions in smelt abundance, the Alde & Ore water body is removed from the GSB + tidal extent and local level depletion is predicted to minor it is highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore.

[3] The TFCI calculated with fyke data causes the water body EQR to fall by 18.8% to 'moderate' status. BEEMS Scientific Position Paper SPP108 concluded that low sampling efficiency and statistical artefacts relating to the small fyke net sample size and low catches caused the apparent deterioration of the water body classification to 'moderate' (Section 12). Fish manipulations as described with the inclusion of fyke nets did not result in deterioration below 'moderate' status.

3.6 Local depletion conclusions

Local level depletion of fish has been considered by applying a simplified conceptual model of impingement relative to tidal replenishment. Whilst the model necessitates making assumptions relating to fish distribution and behaviour it is possible to approximate the likely size of effects at a range of scales. The main conclusions are:

1. With both stations operating together, local depletion reaches a plateau after approximately 50 days, after which point abstraction losses and emigration are balanced by immigration into the local area.

2. Replenishment rates of 10% per day of fish within the GSB + tidal excursion result in minor reductions in pelagic species such as sprat and herring of 1.5% per day during periods of seasonal abundance with both Sizewell C and Sizewell B operating with full mitigation.

3. Demersal and epi-benthic species incur lower impingement mortality and species such as whiting and Dover sole are predicted to incur local depletion of 2.4% and 0.9%, respectively with both Sizewell C and Sizewell B operating with full mitigation.

4. Local depletion is sensitive to the replenishment rate. However, even modest rates of replenishment of 5% per day result in modest local depletion at the scale of the GSB + tidal excursion.

5. Pelagic species such as smelt spend long periods foraging in the inshore waters off Sizewell. At 5% daily exchange across the boundaries, effects within the GSB + tidal excursion resulted in 3.1% depletion of smelt compared to 1.5% local depletion with a 10% exchange.

6. Local depletion due to impingement is orders of magnitude below natural variability in abundance to which predator-prey relationships are adapted to. Therefore, impingement from Sizewell B and Sizewell C would not have any adverse food-web effects on designated features of HRA sites.

7. Local effects are not predicted to cause a deterioration in the WFD fish status of the Alde & Ore water body as local depletion is well below the sensitivity of the TFCI.

4 References

Almada, V. C., Pereira, A. M., Robalo, J. I., Fonseca, J. P., Levy, A., Maia, C., and Valente, A. (2008). Mitochondrial DNA fails to reveal genetic structure in sea-lampreys along European shores. Molecular Phylogenetics and Evolution 46, 391–396.

Cefas, 2018. Project MF1233: Population studies in support of the conservation of the European sea bass. Available at: http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=2&ProjectID=18963).

BEEMS Scientific Position Paper SPP071/s. Shad (*Alosa fallax* and *Alosa alosa*) impingement predictions for HPC, Edition 3. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP100. Estimates of European populations of twaite shad and cucumber smelt of relevance to Sizewell. Cefas, Lowestoft.

BEEMS Technical Report TR318. Predictions of entrainment by Sizewell C in relation to adjacent fish and crustacean populations and their fisheries. Cefas, Lowestoft.

BEEMS Technical Report TR339. Cefas Comprehensive Impingement Monitoring Programme 2014-2017. Cefas, Lowestoft.

BEEMS Technical Report TR345. Sizewell characterisation report - fish. Cefas, Lowestoft.

BEEMS Technical Report TR423. The origins of smelt (*Osmerus eperlanus*) populations at Sizewell. Cefas, Lowestoft.

BEEMS Technical report TR431. Sizewell SPA/SAC features and associated marine prey species. Cefas, Lowestoft.

BEEMS Technical report TR501. Food web interactions and potential for indirect effects at Sizewell. Cefas, Lowestoft.

Catchpole, T., Randall, P., Forster, R., Smith, S., Ribeiro Santos, A., Armstrong, F., Hetherington, S., Bendall, V., Maxwell, D. 2015. Estimating the discard survival rates of selected commercial fish species (plaice - *Pleuronectes platessa*) in four English fisheries (MF1234), Cefas report, pp108.

Chevolot, M., Ellis, J.R., Hoarau, G., Rijnsdorp, A.D., Stam, W.T., Olsen, J.L. 2006. Population structure of the thornback ray (*Raja clavata* L.) in British waters. Journal of Sea Research, 56: 305-316.

Coscia I., Mariani S. 2011. Phylogeography and population structure of European sea bass in the north-east Atlantic. Biological Journal of the Linnean Society, 2011, 104, 364–377.

Froese, R., Pauly, D. Editors. 2019. FishBase. World Wide Web electronic publication. Available at: www.fishbase.org, version (12/2019).

Garthe, S., Camphuysen, K., & Furness, R. W. (1996). Amounts of discards by commercial fisheries and their significance as food for seabirds in the North Sea. Marine Ecology Progress Series, 136(1–3), 1–11.

Genner, M.J., Hillman, R. McHugh, M., Hawkins, S.J., Lucas, M.C. 2012. Contrasting demographic

histories of European and North American sea lamprey (*Petromyzon marinus*) populations inferred from mitochondrial DNA sequence variation.

Fonds, M. 1973. Sand gobies in the Dutch Wadden Sea (*Pomatoschistus*, Gobiidae, Pisces). Netherlands Journal of Sea Research, 6 (4) : 417-478.

Froese, R. and D. Pauly. Editors. 2019. FishBase. World Wide Web electronic publication. www.fishbase.org, version (12/2019).

Gysels, E.S., Leentjes, V. Volckaert, F.A.M. 2004. Small-scale clinal variation, genetic diversity and environmental heterogeneity in the marine gobies *Pomatoschistus minutus* and *P. lozanoi* (Gobiidae, Teleostei). Heredity, 93, 208–214.

Hinter, E., Berry, F., Buckley, A.A.,Stewart, C., Metcalf, J.D. 2006. Seasonal migration of thornback rays and implications for closure management. Journal of Applied Ecology 43: 710–720.

ICES. 2015. Report of the Workshop on Lampreys and Shads (WKLS), 27–29 November 2014, Lisbon, Portugal. ICES CM 2014/SSGEF:13. 206 pp.

ICES. 2017. Report of the Benchmark on Sandeel (WKSand 2016), 31 October - 4 No-vember 2016, Bergen, Norway. ICES CM 2016/ACOM:33. 319 pp.

ICES. 2018a. Report of the Benchmark Workshop on Sprat (WKSPRAT 2018), 5–9 November 2018, ICES HQ, Copenhagen, Denmark. ICES CM 2018/ACOM:35. 60 pp.

ICES. 2018b. Report of the Benchmark Workshop on Seabass (WKBASS), 20–24 February 2017 and 21–23 February 2018, Copenhagen, Denmark. ICES CM 2018/ACOM:44. 283 pp.

ICES 2019a. ICES Advice on fishing opportunities, catch, and effort Celtic Seas and Greater North Sea ecoregions Published 28 June 2019. Sea bass (*Dicentrarchus labrax*) in divisions 4.b–c, 7.a, and 7.d–h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea). Available at: http://www.ices.dk/sites/pub/Publication%20Reports/Advice/2019/2019/bss.27.4bc7ad-h.pdf

ICES. 2019b. Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK). ICES Scientific Reports. 1:7. 1271 pp. Available at: http://doi.org/10.17895/ices.pub.5402

ICES 2019c. Thornback ray (*Raja clavata*) in Subarea 4 and in divisions 3.a and 7.d (North Sea, Skagerrak, Kattegat, and eastern English Channel). ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion. Published 4 October 2019. Available at: http://ices.dk/sites/pub/Publication%20Reports/Advice/2019/2019/rjc.27.3a47d.pdf

ICES 2019d. Flounder (*Platichthys flesus*) in Subarea 4 and Division 3.a (North Sea, Skagerrak and Kattegat). ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion Published 28 June 2019.

ICES 2019e. Report Section 29 -Seabass (*Dicentrarchus labrax*) in divisions 4.b–c, 7.a, and 7.d–h (central and southern North Sea, Irish Sea, English Channel, Bristol Channel, and Celtic Sea).

ICES 2019f. Dab (*Limanda limanda*) in Subarea 4 and Division 3.a (North Sea, Skagerrak and Kattegat). ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion Published 8 November 2019. Available at: http://ices.dk/sites/pub/Publication%20Reports/Advice/2019/2019/dab.27.3a4.pdf

ICES 2020a. Sprat (*Sprattus sprattus*) in Division 3.a and Subarea 4 (Skagerrak, Kattegat, and North Sea). ICES Advice on fishing opportunities, catch, and effort Greater North Sea ecoregion Published 14 April 2020.

ICES 2020b.Official Nominal Catches 2006-2017. Accessed 16/04/2020 via http://ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx

INBO 2020. Elft terug in de Schelde na 100 jaar afwezigheid. INBO Website, downloaded 11 May 2020.

Jolly, M.T., Aprahamian, M.W., Hawkins, S.J., Henderson, P.A., Hillman, R., O Maoileidigh, N., Maitland, P.S., Piper, R., M.J., G., 2012. Population genetic structure of protected Allis shad (*Alosa alosa*) and twaite shad (*Alosa fallax*). Mar. Biol. 159, 675–687.

Libungan, L.A., Óskarsson, G.J., Slotte, A., Jacobsen, J.A., Pálsson, S. 2015. Otolith shape: a population marker for Atlantic herring *Clupea harengus*. J Fish Biol. 86(4):1377-95.

Magath, V., Thiel, R. 2013. Stock recovery, spawning period and spawning area expansion of the twaite shad *Alosa fallax* in the Elbe estuary, southern North Sea. Endangered Species Research, 20: 109-119.

Maitland P.S. 2003. Ecology of the River, Brook and Sea lamprey *Lampetra fluviatilis*, *Lampetra planeri* and *Petromyzon marinus.* Conserving Natura 2000 Rivers Ecology Series No. 5. English Nature, Peterborough

MMO 2020. Marine Management Organisation. UK sea fisheries statistics 2019. Available at: https://www.gov.uk/government/statistics/uk-sea-fisheries-annual-statistics-report-2019. Accessed on 16/10/2020.

Martin, J., Rougemont, Q, Drouineau, H., Launey, S., Jatteau, P., Bareille, G., Berail, S., Pécheyran, C., Feunteun, E., Roques, S., Clavé, D., Nachón, D.J., Antunes, C., Mota, M., Réveillac, E., Daverat, F. 2015. Dispersal capacities of anadromous Allis shad population 1 inferred from a coupled genetic and otolith approach. Can. J. Fish. Aquat. Sci. 72(7): 991-1003, https://doi.org/10.1139/cjfas-2014-0510

NatureServe. 2013. *Petromyzon marinus*. The IUCN Red List of Threatened Species 2013: e.T16781A18229984. Available at: https://dx.doi.org/10.2305/IUCN.UK.2013-1.RLTS.T16781A18229984.en.

Pampoulie, C. Gysels, E.S. Maes, G.E., Hellemans, B., Leentjes, V., Jones, A.G., Volckaert, F.A.M. 2004. Evidence for fine-scale genetic structure and estuarine colonisation in a potential high gene flow marine goby (*Pomatoschistus minutus*). Heredity, 92, 434–445

Riley, K. 2007. *Pomatoschistus minutus* Sand goby. In Tyler-Walters H. and Hiscock K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews, [on-line]. Plymouth: Marine Biological Association of the United Kingdom. Available at: https://dx.doi.org/10.17031/marlinsp.1204.1

Sabatino, S., Alexandrino, P., 2012. Genetic diversity and population structure of the Eurasian shads *Alosa alosa* and *Alosa fallax*. CIBIO report AARC project - Activity 4, December 2012.

Tulp, I., Bolle, L. J., Dänhardt, A., de Vries, P., Haslob, H., Jepsen, N., van der Veer, H. W. 2017. Fish. Wadden Sea Quality Status Report 2017. Wilhelmshaven: Common Wadden Sea

Secretariat.

Vanderputte, M., Gagnaire, P-A., Allal, F. 2019. The European sea bass: a key marine fish model in the wild and in aquaculture. Animal Genetics., 50, 195–206.

Waldman, J., Grunwald, S., Wirgin, I. 2008. Sea lamprey *Petromyzon marinus*: an exception to the rule of homing in anadromous fishes. Biology Letters, 4: 659-662.

Wilson, K., and Veneranta, L. (Eds). 2019. Data-limited diadromous species – review of European status. ICES Cooperative Research Report No. 348. 273 pp. Available at: http://doi.org/10.17895/ices.pub.5253

Appendix A

Table 10 Seasonal abundance of selected key species impinged at Sizewell B and predicted depletion during those periods at a range of scales. Values provided show local effects of Sizewell B operating alone and are based on 10% exchange rates. Green shaded species have been identified as potential prey items for HRA designated species at Sizewell. Underline species have been identified by the Environment Agency as important for the Alde & Ore WFD fish classification.

Species	Group	Period (days)	SZB		
			GSB	33F1	4c
Sprat	Pelagic	Dec. – March (121)	2.50	0.03	0.007
Herring	Pelagic	Feb. – April (89)	2.50	0.03	0.003
Whiting	Demersal	Oct. – April (221)	2.15	0.03	0.011
Seabass	Demersal	Dec. – March (121)	2.15	0.03	0.06
Sand goby	Epi-benthic	Aug. – Dec. (162)	0.79	0.01	0.003
Dover sole	Epi-benthic	March – Oct. (245)	0.79	0.01	0.005
Anchovy	Pelagic	June (30)	2.40	0.024	0.001
Dab	Epi-benthic	Throughout (365)	0.80	0.02	0.007
Thin-lipped grey mullet	Demersal	Jan. – March (90)	2.14	0.03	0.011
Flounder	Epi-benthic	Oct. – June (282)	0.79	0.01	0.005
Plaice	Epi-benthic	Throughout (365)	0.80	0.02	0.007
Smelt	Pelagic	April – Nov. (253)	2.51	0.04	0.015
Cod	Demersal	Dec. – March (121)	2.15	0.03	0.003
Thornback ray	Epi-benthic	March – May (92)	0.79	0.01	0.001
Eel	Epi-benthic	Aug. – Nov. (131)	0.79	0.01	0.002
Twaite shad	Pelagic	March – Aug. (184)	2.51	0.04	0.010

Table 11 Seasonal abundance of selected key species impinged at Sizewell B and predicted depletion during those periods at a range of scales. Values provided show local effects of Sizewell C operating alone and are based on 10% exchange rates. Green shaded species have been identified as potential prey items for HRA designated species at Sizewell. Underline species have been identified by the Environment Agency as important for the Alde & Ore WFD fish classification.

Species	Group	Period (days)	SZC		
			Tidal extent	33F1	4c
Sprat	Pelagic	Dec. – March (121)	0.88	0.03	0.006
Herring	Pelagic	Feb. – April (89)	0.88	0.03	0.004
Whiting	Demersal	Oct. – April (221)	2.10	0.09	0.028
Seabass	Demersal	Dec. – March (121)	2.08	0.08	0.014
Sand goby	Epi-benthic	Aug. – Dec. (162)	0.77	0.03	0.07
Dover sole	Epi-benthic	March – Oct. (245)	0.77	0.03	0.012
Anchovy	Pelagic	June (30)	0.83	0.02	0.001
Dab	Epi-benthic	Throughout (365)	0.78	0.04	0.018
Thin-lipped grey mullet	Demersal	Jan. – March (90)	2.08	0.07	0.010
Flounder	Epi-benthic	Oct. – June (282)	0.78	0.04	0.013
Plaice	Epi-benthic	Throughout (365)	0.78	0.04	0.018
Smelt	Pelagic	April – Nov. (253)	0.89	0.04	0.014
Cod	Demersal	Dec. – March (121)	2.10	0.08	0.015
Thornback ray	Epi-benthic	March – May (92)	0.77	0.03	0.004
Eel	Epi-benthic	Aug. – Nov. (131)	0.77	0.03	0.006
Twaite shad	Pelagic	March – Aug. (184)	0.88	0.04	0.010

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP104

NOT PROTECTIVELY MARKED

Worst case glass eel entrainment assessment for Sizewell C

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Executive summary

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture Science (Cefas) has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC DCO application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

The proposed Sizewell C (SZC) cooling water system would be fitted with four low velocity side entry (LVSE) intake heads mounted on the seabed approximately 3 km offshore. The proposed intake heads would be capped structures with the intake surfaces orthogonal to the direction of the tidal flows. These state-of-the-art intakes are specifically designed to reduce the cross-sectional area available to intercept any fish being transported in the tidal flows. The reduction in cross-sectional area combined with their low intake velocity is predicted to substantially reduce the number of fish abstracted per cumec (cubic metre per second) of seawater compared with Sizewell B (SZB) which is fitted with two capped omnidirectional intake heads with intake velocities that exceed the swimming ability of most fish to avoid being abstracted.

In discussions with stakeholders about the Sizewell C fish assessment contained in BEEMS Technical Report TR406 v6 issues were raised about:

- Confidence in the assessment of potential entrainment effects on European eel, in particular the adequacy of the BEEMS survey data.
- The possibility that night-time glass eel density is higher than during day-time and the effects this could have on entrainment estimates.
- The need to compare results with trends from the Flatford glass eel monitoring data.
- Whether the glass eel entrainment mortality figure was realistic and accounted for the full range of entrainment conditions.

This Scientific Position Paper addresses all of these issues and provides what is considered to be a worst-case entrainment assessment together with the results of associated sensitivity analyses.

Conclusions

The following conclusions have been reached:

i. The BEEMS glass eel survey tows had a limit of detection of 0.000387 eels m^{-3} per tow. Assuming that eels are only at risk of capture during one tidal direction, the Sizewell survey detected a mean density of 7.44 10^{-6} eels m^{-3} in 52 flood tide tows. The same survey methodology and sampling equipment at Hinkley Point, detected mean densities of 0.0058 m^{-3} and a peak density of 0.0457 m^{-3}. From these data it is concluded that the density of glass eels at Sizewell is at least two orders of magnitude lower than at Hinkley Point. Based upon the predicted Sizewell glass eel density, the amount of effort required to increase the predicted Sizewell glass eel catch from 1 eel to 10 to 15 eels would be infeasible and would require impractically long working days and increasing the number of survey days from 11 to most days during the migration period. This amount of survey effort is disproportionate when a worst-case estimate can be derived from the data provided by the existing surveys.

ii. The BEEMS entrainment survey at Sizewell in 2011 would not have been sensitive enough to detect glass eels at the expected density in that year. The necessary increase in sampling to detect one eel would have been infeasible based upon the 2015 glass eel survey results.

iii. The calculated glass eel entrainment loss at SZC would be 0.0089% Anglian RBD SSB based upon 2015 survey data.

iv. Considering the cumulative effect of the sensitivity analyses described in this report, the SZC entrainment losses could have been underestimated by a factor of 2.66, assuming a hypothetical increase in night time density of a factor of 10 (noting that there are no measurements to support this figure). Using the calculated Gironde day: night ratio the SZC entrainment losses may have been overestimated by a factor of 1.25. On that basis the SZC entrainment losses are estimated to be between 0.0071% to 0.024% Anglia RBD SSB.

v. To put the SZC entrainment estimate into context, the estimated SZC mean eel impingement loss is 0.045% Anglian RBD SSB. i.e. the predicted SZC entrapment loss is in the range 0.052% to 0.069% Anglian RBD. One percent of the Anglian RBD SSB is equivalent to an estimated 0.005% of the European eel stock.

1 Background

EDF Energy plans to build a new coastal nuclear power station (Sizewell C, SZC), adjacent to the operational Sizewell B (SZB) and decommissioned Sizewell A (SZA) sites in Suffolk. The station would be of a once-through design, abstracting large volumes of seawater for cooling the condenser steam. As part of the application for the building and operation of the new station, EDF Energy is required to evaluate the effects that the abstraction of seawater may have on the marine environment. The Centre for Environment, Fisheries and Aquaculture Science (Cefas) has been contracted by EDF Energy to undertake the necessary marine studies to provide the evidence base for the SZC Development Consent Order (DCO) application via a comprehensive set of studies known collectively as the BEEMS programme for SZC.

SZC would need to abstract approximately 132 cumecs (m^3 s^{-1}) compared with approximately 51.5 cumecs for the existing SZB station.

The cooling water intakes for SZC would be protected by widely spaced bars to prevent the intake of cetaceans, seals and large items of debris, but a significant number of small organisms (small fish and crustaceans, and plankton) will inevitably enter the cooling water intakes. The larger organisms must be removed before the water enters the power station cooling system to prevent them blocking the condenser tubes. These organisms are removed through impingement on rotating fine-mesh drum screens which protect the main cooling water supply to the station condensers and band screens that protect the essential and auxiliary cooling water systems. The smaller organisms (mostly fish eggs and larvae and other plankton) that pass through the drum screens are entrained and pass through the power station cooling system without causing significant blockages.

The proposed Sizewell C (SZC) cooling water system would be fitted with four low velocity side entry (LVSE) intake heads mounted on the seabed approximately 3 km offshore. The proposed intake heads would be capped structures with the intake surfaces orthogonal to the direction of the tidal flows. These state-of-the-art intakes are specifically designed to reduce the cross-sectional area available to intercept any fish being transported in the tidal flows. The reduction in cross-sectional area combined with their low intake velocity is predicted to substantially reduce the number of fish abstracted per cumec (cubic metre per second) of seawater compared with Sizewell B (SZB) which is fitted with two capped omnidirectional intake heads with intake velocities that exceed the swimming ability of most fish to avoid being abstracted.

BEEMS Technical Report TR406 provides an assessment of the predicted losses for fish and crustacea due to entrapment (impingement plus entrainment) in the SZC cooling water system. For assessment purposes 24 key fish taxa in the Greater Sizewell Bay were selected as representative of the assemblage at Sizewell based on either their commercial value, their ecological importance, or their conservation status. The European eel (*Anguilla anguilla*) is one of those species, selected due to its conservation status. In principle, the following eel life stages could be at risk from entrapment in SZC:

- Glass eels migrating along the coast to find a suitable estuarine/freshwater habitat;
- yellow eels moving between different river systems along the East Anglian coast or living in coastal waters; and
- silver eels migrating to the Sargasso Sea that after leaving estuaries along the East Anglian coast may transit past Sizewell.

SZC Impingement of yellow and silver eels is already assessed in TR406. This paper provides an estimate of the potential entrainment losses for glass eel.

1.1 Summary of the SZC entrapment assessment process

The SZC entrapment process is described in detail in TR406. There are no formal UK regulatory guidelines for assessing the significance of fish mortality levels caused by entrapment in coastal power stations and therefore any assessment must be based on expert judgment.

In TR406, two screening thresholds were selected such that entrapment losses lower than the appropriate threshold would have negligible effects on the year to year sustainability of a fish population. Effects above the appropriate threshold would not necessarily indicate a significant adverse effect but require further investigation to determine whether significant effects were, in fact, present.

The thresholds were selected based upon internationally accepted scientific best practice for the sustainability of fish stocks under anthropogenic pressures:

a. For commercially exploited stocks and conservation species (which includes stocks that are not currently exploited): 1% of the Spawning Stock Biomass (SSB) or, as a highly conservative proxy, 1% of international landings of the stock.

b. For unexploited stocks: 10% of the SSB or, as a highly conservative proxy, 10% of international landings of the stock.

The scientific rationale for the selection of these screening thresholds is detailed in Section 6 of TR406.

For some conservation species, a more precautionary approach was adopted of comparing SZC effects with 1% of a geographically limited subset of the entire stock. In particular, a highly precautionary approach was adopted for European eel whereby SZC entrapment losses were compared to the Anglian River Basin District (RBD) Silver Eel Escapement Biomass. This is equivalent to adopting a highly precautionary threshold of approximately 0.005% of an estimated SSB for the eel stock (TR406). (In practice the European eel SSB cannot currently be measured as no surveys have taken place of silver eel at the spawning area in the Sargasso Sea). In this paper the Anglian RBD Silver Eel Escapement Biomass is referred to as the Anglian RBD SSB as a shorthand description for compatibility with the other indicators described in TR406. Comparison of entrapment with the Anglian RBD SSB overestimates SZC effects as only some of the Anglian escapement biomass would survive due to natural mortality on the 5000km return journey to spawn in the Sargasso Sea.

The European eel is distributed across the majority of coastal countries in Europe and North Africa, with its southern limit in Mauritania (30°N) and its northern limit situated in the Barents Sea (72°N) and spanning the entire Mediterranean basin (ICES 2019).

The European eel life history is complex, being a long-lived semelparous (mature adults die after spawning) and widely dispersed stock. The shared single stock is genetically panmictic and data indicate the spawning area is in the southwestern part of the Sargasso Sea. The newly hatched leptocephalus larvae drift for two to three years with the ocean currents for more than 5000km to the continental shelf of Europe and North Africa, and enter continental waters. There, they metamorphose into the post-larval transparent glass eel. At this stage, glass eels migrate across the continental shelf to the coast. After reaching the coast, glass eels enter estuaries (in the UK Severn starting from about January/February). Glass eels metamorphose into pigmented elvers which either remain and feed in coastal marine or estuarine waters or begin active upstream migration to freshwater. The growth stage, known as yellow eel, may take place in marine, brackish (transitional), or freshwaters. This stage may last typically from two to 25 years (and could exceed 50 years) prior to metamorphosis to the “silver eel” stage and maturation. At this stage silver eels migrate 5000 km+ back to their spawning grounds. Age-at-maturity varies according to temperature (latitude and longitude), ecosystem characteristics, and density-dependent processes. The European eel life cycle is shorter for populations in the southern part of their range compared to the north. Age at maturity ranges from 10 to 20+ years in northern temperate waters (Vøllested, 1992), and is earlier for males than for females. (ICES 2019, McCleave 1993, Tesch 2003, Harrison *et al* 2014).

1.2 Sizewell glass eel surveys

An extensive set of surveys has been undertaken in an attempt to determine the density of glass eels off Sizewell. The terms glass eel and particularly elver are often used loosely and confusingly in the literature and the ICES eel working group (WGEEL) encourages authors to define clearly what development stage they are describing. In this paper we have used the term glass eels to represent either unpigmented or pigmented juvenile eels of less than or equal to 80mm length that are actively migrating in UK coastal waters in their search for freshwater rivers. This is to distinguish between these new recruits and elvers defined here as the group of small eels in the size range >80mm to 120mm length. The BEEMS glass eel surveys are described in BEEMS Technical Reports TR406 and TR318 and summarised below:

a. The BEEMS targeted glass eel surveys in April and May 2015 were conducted at 3 stations (the SZB intake location at approximately 640m offshore, the Sizewell-Dunwich Bank at approximately 2km offshore and at the proposed SZC intakes at 3km offshore). The surveys detected only 1 glass eel (of length 70mm) which was found in April 2015 at the surface, on the flood tide at the location of the proposed Sizewell C intake locations (BEEMS Technical Report TR356). The survey completed 105 tows in 11 sampling days using the same methodology that successfully sampled 2,781 glass eels in 321 tows over 37 sampling days in two years in the Bristol Channel.
b. No glass eels were detected in water drawn from the SZB forebay during the 12-month BEEMS Comprehensive Entrainment Monitoring Programme (CEMP) at Sizewell in 2011 (BEEMS Technical Report TR318). This programme consisted of 40 * 24h samples taken directly from the Sizewell B forebay and represented SZB abstraction at 640m offshore.
c. No glass eels were detected in the 620 plankton tows conducted in Sizewell Bay between 2008 to 2017.
d. The BEEMS Comprehensive Impingement Programme (CIMP) programme at Sizewell B from 2009 to 2017 sampled only two glass eels; one in March 2013 and one in January 2017. Both glass eels had a length of 67.5mm (BEEMS Technical Report TR406 v7)

Glass eels are too narrow to be impinged reliably at Sizewell B (or at the proposed Sizewell C) and would normally be entrained through the power station cooling water system. Impingement is not, therefore, a quantitative means of glass eel density measurement. Nevertheless, more glass eels were impinged at Hinkley Point B than at Sizewell B in comparative CIMP programmes indicating the higher eel density at Hinkley Point.

From the Sizewell survey evidence, and as would be expected, it is clear that glass eels do transit past Sizewell seeking a freshwater signal from an estuary. Surveys a. and b. above would have been expected to detect glass eels if they were present in numbers that could indicate the possibility of adverse entrainment effects at Sizewell C.

The totality of data from this sampling programme led to the conclusion in BEEMS Technical Report TR318 that whilst glass eels are present in Sizewell coastal waters, their density was very low at this location. Studies using an entrainment mimic unit (EMU) that closely reproduces the time profile of exposure to the pressures that entrained glass eels would experience inside of Sizewell C and Hinkley Point C (mechanical, temperature, pressure and concentrations of anti-biofouling biocides) have demonstrated that glass eel survival is approximately 80% (BEEMS Technical Report TR318). Given the very low density of glass eels at Sizewell determined by surveys and the measured high entrainment survival, the potential effect of glass eel entrainment in SZC was therefore assessed as negligible in BEEMS Technical Report TR318.

Due to only one glass eel being detected during the targeted surveys, no attempt was made to quantify SZC entrainment effects on glass eel. It has been suggested that the surveys may not have been adequate to detect glass eels and that the solution to this problem is to undertake more surveys. For example, it has been suggested that glass eels preferentially migrate by night in upper estuaries.

1.3 Summary of what is known about glass eel migration behaviour

Due to the extreme logistical challenges of conducting quantitative surveys on low densities of glass eels in large bodies of water, there is little direct evidence on glass eel migration behaviour in coastal waters and almost all published work is of behaviour in upper estuaries and rivers where monitoring is more tractable. McCleave and Wippelhauser (1987) reported that juvenile eels use Selective Tidal Stream Transport (STST) as far as the tidal limit, but no circa-tidal rhythm was observed in glass eels collected just above the head of the tide. Edeline *et al*. (2007) studied dispersal in metamorphosing juvenile eels by sampling glass eels from the tidal limit area. The authors reported that glass eels migrated toward the freshwater area of the upper estuary using STST and accumulated at the break point of tidal streams. Feunteun *et al*. (2003) reported that very few glass eels and elvers less than 120 mm (<1 year) were observed above the tidal limit, indicating that the older, larger stages, and not glass eels, migrate upstream to fresh water. In another study conducted above the tidal limit, White and Knights (1997a) captured 156 000 elvers and 189 000 juveniles of yellow eels, but no unpigmented glass eels. These results showed that counter-current migration was not practised by young-of-the-year (YOY) eels.

Harrison *et al* (2014) provided a synthesis of behavioural evidence but this is from upper estuaries and rivers. More recently ICES (2020) describes the temporal pattern of glass eel arrivals on the East Anglian coast and Cresci (2020) has attempted to provide a comprehensive hypothesis on how glass eel migration takes place using the most up to date available evidence. Cresci (2020) found that the behaviour between glass eel arrival on the continental slope and coastal waters is the least understood phase of the migration. The author concludes that whilst glass eels "*have the sensory capacity to use multiple cues at any given time, their migration is based on a hierarchical succession of orientation mechanisms dictated by the physical properties of the environments that they occupy: (i) lunar and magnetic cues in the pelagic water of the continental slope; (ii) chemical and magnetic cues in coastal areas; and (iii) odours, salinity, water current and magnetic cues in estuaries. Thus, there is not one single behavioural strategy underlying the migration of glass eels, but rather they have the ability and plasticity to switch orientation mechanisms depending on which cues are available at each specific step*."

There is scientific consensus that glass eels employ STST up to the tidal limit in estuaries, migrating up estuary on the flood and settling on the bed during the ebb. In practice the efficiency of migration using STST (the observed migration speed divided by the flood tide current speed) has been found to be much less than 100%. Beaulaton and Castelnaud (2005) measured efficiencies of 15-19% for glass eels in the Gironde estuary, resulting in mean up estuary migration speeds of 3 to 4 km/day. Lambert *et al.,* (2007) found that 30% of glass eels migrate on the flood by day and 70% by night in the lower section of the Gironde estuary in France

The BEEMS programme found no evidence of preferential migration by night in the Bristol Channel at Hinkley Point (BEEMS Technical Report TR274) where the estuary is more than 20km wide. This challenging 2-year monitoring programme in 2012/2013 was jointly funded by EDF Energy and the Environment Agency and consisted of 321 tows spread over 3 surveys (February/March 2012, February/March 2013 and April 2013). The surveys were conducted at 3 depths; nominally at 0m, 4m, and 7m where the depth refers to the top of the 1.4 m vertical opening of the square MIK net. It was the first time such a programme has been undertaken at a wide lower estuary site in the UK. The programme found that glass eel catches were not significantly different between day and night and concluded that:

i. glass eels used the full width of the Severn Estuary to migrate upriver;

ii. the greatest abundance of eels was consistently found in shallow, inshore sites on the southern and northern sides of the estuary;

iii. there is evidence that eel densities are greater at the surface than at deeper depths; particularly than at depths of 7m.

iv. the density of eels at the location of the HPC intakes (3km offshore) was less than at further inshore sites (0.5km – 1km offshore). Pooling the data from the 3 surveys, the density at the offshore HPC intakes was 59% of the density at the inshore HPB intake.

Glass eels have poor swimming abilities and STST is the most energy efficient means of migration in tidal waters. Based upon the relative timings of eel arrivals in UK estuaries, it is considered likely that glass eels employ STST in coastal waters during day and night, particularly where underwater light levels are low due to high suspended sediment concentrations such as on the UK east coast. The waters off Sizewell have high levels of turbidity which significantly limit light penetration and it is considered likely that a substantial proportion of glass eels transiting past Sizewell could do so during the daytime but the ratio of day:night migration is unknown. As part of sensitivity testing conducted for this SPP, the possibility of migration occurring in the ratio of 30% day:70% night as measured by Lambert *et al* (2007) has been examined. In response to stakeholder comments that the ratio of night-time to daytime ratio could be even higher a further scenario was modelled that assumed that night-time density was ten times that during daylight hours.

ICES (2020) provides evidence that the glass eel migration in UK waters has a duration of 4-5 months with largest number arriving in a 3-month period that varies with latitude. Along the East Anglian coast, the hypothesis is that glass eel migration proceeds from south to north. Support for this hypothesis is provided by the timing of glass eel arrivals as shown by European monitoring data. ICES (2020) summarises East Anglian rivers as belonging to a cluster where the main glass eel arrivals occur between April and July with a peak in April. Note that this timing relates to glass eel arrivals at upper estuary counting stations, not in the coastal zone.

In considering the value of additional surveys at Sizewell it is necessary to consider the survey power in detecting glass eels i.e. the limit of detection of the surveys needs to be considered. In addition, surveys with null results can be used to infer maximum densities from which quantitative entrainment estimates can be made. This paper uses the available data from Sizewell to:

1. Quantify the limit of detection of the surveys and to estimate the number of eels transiting past Sizewell; and
2. Calculate worst case predictions for glass eel entrainment at Sizewell C. i.e. the highest loss of eel mortality expressed relative to local production.

2 Trends in glass eel monitoring data

Before extrapolating from the Sizewell data, it is instructive to consider patterns of glass eel monitoring data from other East Anglian sites. Routine glass eel monitoring is undertaken by the Environment Agency at three fixed locations during the glass eel migration season. Annual summaries are reported in ICES (2019) and monthly data has been obtained directly from the Environment Agency.

The number of glass eel arriving in continental waters declined dramatically in the early 1980s to a low point in 2011. The reasons for this decline are uncertain but may include commercial over-exploitation, pollution, non-native parasites, diseases, migratory barriers and other habitat loss, mortality during passage through turbines or pumps, and/or oceanic factors affecting migrations. Statistical analyses of time-series from 1980–2019 show that there was a change in the trend of glass eel recruitment indices in 2011; the recruitment has stopped decreasing and has been increasing in the period 2011–2019 with a rate statistically significantly different from zero. The highest point during the period in glass eel indices across Europe from 2011–2019 was in 2014. It is not yet clear whether this change indicates a sustained recovery in eel recruitment (ICES 2019).

Figure 1 shows the results of eel monitoring (glass eel less than 80mm in length) by the Environment Agency at Beeleigh weir on the River Chelmer that drains into the Blackwater Estuary and from Flatford on the River Stour, both to the south of Sizewell. The Blackwater estuary is approximately 76km south of Sizewell and the Stour is approximately 44 km south of Sizewell. Both of the Environment Agency monitoring stations are approximately 20km upstream of the mouths of the Blackwater and Stour respectively.

The Beeleigh data show a large increase in glass eel recruitment in 2013 that has been sustained but at a lower level thereafter. The same large increase in glass eel recruitment in 2013 is also shown in the Environment Agency's monitoring data from Brownshill on the Great Ouse (Figure 2). The Brownshill data are included to demonstrate recruitment trends but no further use is made of the data in this paper because the Great Ouse drains into the Wash and is, therefore, not local to Sizewell. No data from Flatford were recorded in 2013 and whilst the trends in other years are broadly similar to those at Beeleigh, the increase in recruitment numbers is less pronounced than at Beeleigh.

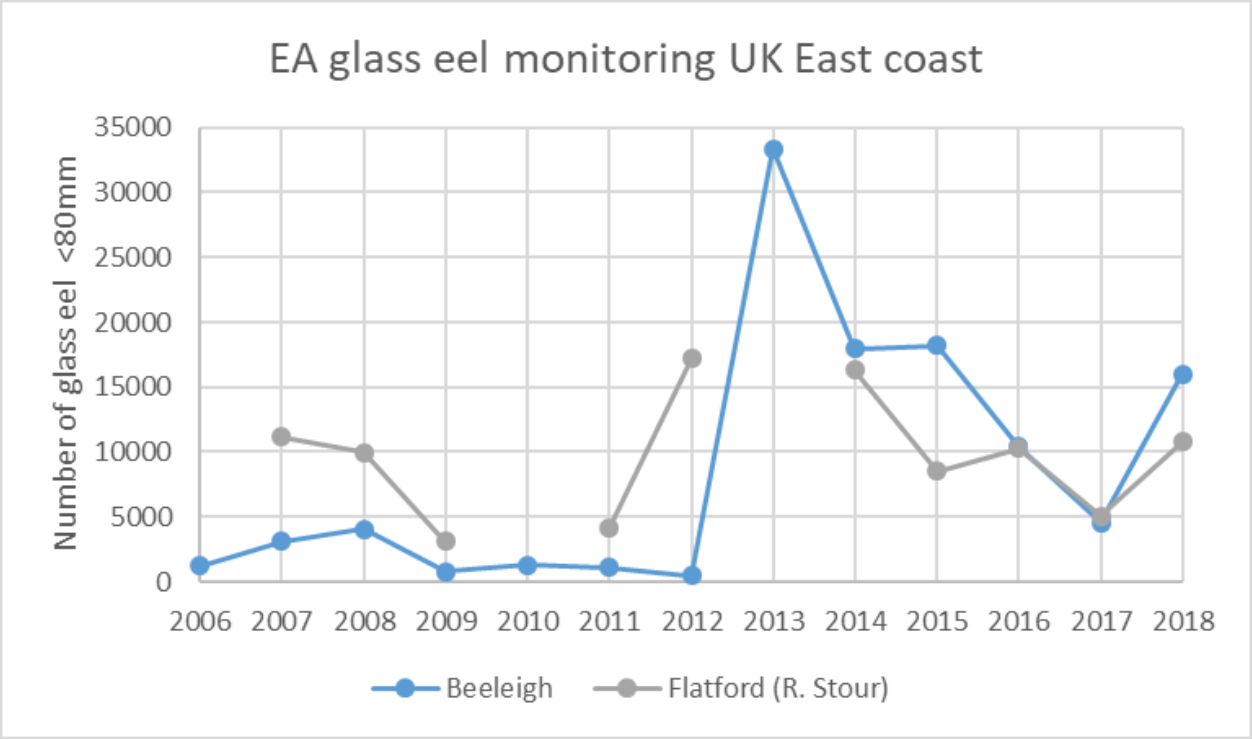

Figure 1 Environment Agency glass eel monitoring data from Beeleigh in Essex and Flatford in Suffolk, both to the south of Sizewell in East Anglia.

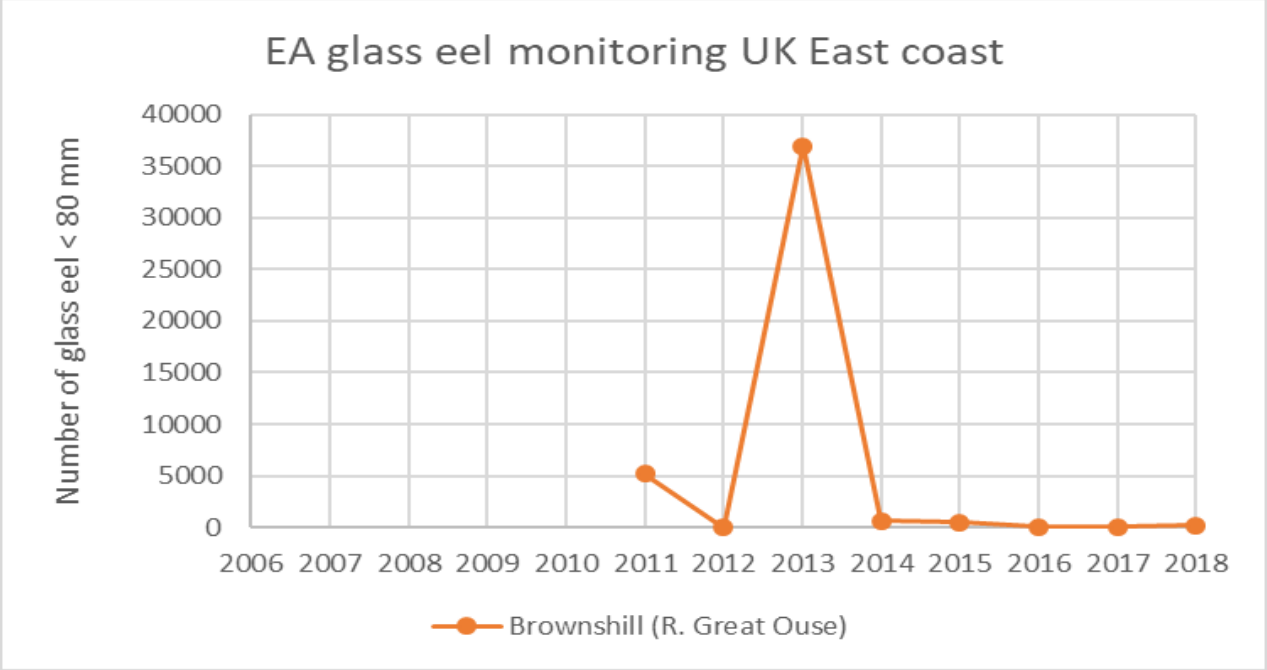

Figure 2 Environment Agency glass eel monitoring data from Brownshill approximately 60km inland of the mouth of the Great Ouse in the Wash. 2012 was only a partial count record due to extended site downtime and the data for that year are therefore not reliable.

Within year, the peak months for the glass eel migration run at Beeleigh were April – June with peak numbers in April or May with the exception of 2016 when the peak was in June. These 3 months accounted for a mean of 83% of the migrating eels in the period 2010-2018 (2016 was excluded from this calculation due to trap damage from July -September inclusive) with a range of 62% to 95%. In 2015, the peak month was May with 88% of the migrating eels occurring in April to June.

At Flatford the peak months for the glass eel migration in the period 2010 -2018 were May – July with peak numbers in May or June with the exception of 2012 when the peak was in July. The lack of granularity in the data (Cefas only has access to monthly data points) makes it difficult to characterise the relationship between the Beeleigh and Flatford data series. In general, the Flatford count peak is recorded as one month later than that at Beeleigh which fits the hypothesis that glass eels are migrating south to north along the East Anglian coast. However, in 2016 the peaks at the two sites were recorded in the same month (June) and in 2017 the peak at Beeleigh (June) was apparently 1 month earlier than at Flatford (May). In reality the recruitment peaks may only be weeks or days apart, but this detail is lost by the recording interval.

Whilst the fixed monitoring locations provide indicators of the annual trend in glass eel numbers, drawing conclusions from these stations about the timing of eel migrations into the estuaries is fraught with difficulties. There are two phases to glass eel migration. To migrate up the coast, glass eels are most likely using STST on the ebb tide but in estuaries the eels have to switch migration modes to move up estuary on the flood. The efficiency of STST has been studied in Gironde estuary and found to be 15% to 19% corresponding to an up-estuary migration speed of 3-4 km/day (Lambert *et al* 2007). The migration speed in East Anglian estuaries is unknown and may vary from month to month and from year to year. On the coast tidal velocities are different than in estuaries and glass eel migration efficiencies may be different than in estuaries. If we assume the mean migration speed up estuary of 3.5 km/day from the Gironde from Lambert *et al* (2007) for East Anglian estuaries and for the coast, the peak in Sizewell numbers should occur approximately 1 week after Flatford and 16 days after that at Beeleigh. In 2015 this implies that the Sizewell peak glass eel numbers would have occurred in May or June based upon the Beeleigh data or June based upon the Flatford data. Due to the lack of granularity in the data series and the unknown migration speeds on the coast and the estuaries, there is uncertainty in this assessment. For example, in 2015 at Beeleigh the glass eel numbers were spread fairly evenly over a 3-month period centred on May (counts of 3876, 8284, 4424) but at Flatford the migration numbers were very tightly centred on June with very few eels apparently migrating in the adjacent months (counts of 144, 7892, 345). There are no data to explain the difference in the pattern of counts for these two estuaries that are only 32 km apart.

However, there is a significant problem with using these estuarine count data to infer timings at Sizewell. It is known from monitoring in other estuaries that glass eels frequently congregate at the head of estuaries waiting for suitable environmental conditions to migrate upriver against the flow. A hypothesis is that this active migration phase is linked to water temperature (Harrison *et al* 2014). The Environment Agency monitoring stations at Beeleigh and Flatford are located near the limit of saline influence and the timing of glass eel counts at these locations could be subject to delays as the eels wait for an uncertain number of weeks for appropriate environmental conditions. Under such circumstances the measured timing of peak migration will be expected to be flattened and not strongly linked to the timing of eel arrival at the estuary mouth at all. Similarly, relative timings of measured upriver migrations may reflect different arrival times at estuary mouths but also differences in environmental conditions at the Environment Agency monitoring locations. Drawing conclusions on glass eel migration timing at Sizewell from inland monitoring stations is therefore problematical and potentially misleading

More recently glass eel monitoring has taken place off the Bradwell A power station in Essex near to the mouth of the Blackwater estuary using the same fishing gear and sampling methodology as deployed during the Sizewell glass eel surveys. Starting from February 2020, four sampling days were conducted in each month broken down into 2 days daytime sampling and 2 days nighttime sampling. For 2020 the glass eel catches are shown in Table 1.

Table 1 Glass eel catches during monitoring surveys off Bradwell in the Blackwater estuary in 2020

Month in 2020	Number of glass eels captured in 4 fishing days/month
February	1
March	9
April	0
May	0
June	0

Note: Bradwell MIK net data provided by permission of BRB GenCo.

This dataset from one year indicates peak glass eel migration numbers near to the estuary mouth in March at Bradwell which is earlier than in any year from the Beeleigh monitoring data. In 2020 the Beeleigh monitoring was from 16/3/2020 to 8/11/2020 but was interrupted by COVID-19 for the period 24/3/2020 to 6/4/2020. No glass eels were detected from March 16 to March 23 and the first glass eels were detected on 9/4/2020 with peak numbers from 25/4/2020 to 5/5/2020. The last glass eel was captured on 10/6/2020. The Bradwell glass eel monitoring detected peak migration between 11-12 March i.e. 6-8 weeks before the Beeleigh monitoring whereas extrapolation of Gironde up estuary migration speed estimates (3-4km/day) would imply that the gap should be no more than 5-7 days. The Bradwell monitoring is a limited dataset but it does provide an indication of the time lag between coastal migration peaks and the measured peaks at the tidal limit. Assuming a coastal migration speed of 3.5km/day, glass eels migrating past the Blackwater Estuary would be expected to arrive at Sizewell (76km north) in about 22 days or by the first week in April in 2020.

Considering other data collected at Sizewell, single glass eels have been detected in March 2013, April 2015 and January 2017. Considering these events together with the Bradwell monitoring data with a peak in March 2020 would indicate that the peak migration past Sizewell could be expected in the period March to April which virtually overlaps with the BEEMS glass eel surveys in April and May 2015. This timing appears more credible than the May to June that extrapolation of timings from the Environment Agency inland monitoring data might imply. So rather than too late, the Bradwell data indicates that there is a possibility that the BEEMS Sizewell glass eel surveys may have been a little early to capture the peak migration at Sizewell which could have occurred in late March to early April. This is explored in the sensitivity analyses in Section 4.2.

The relevant part of the BEEMS SZB CEMP monitoring programme took place in 2011 and the BEEMS Sizewell glass eel surveys took place in 2015. The two Environment Agency East Anglian monitoring sites indicate that glass eel numbers were greater in 2015 than in 2011; by a factor of 2.0 at Flatford and 16.8 at Beeleigh. Both river systems have experienced extensive water management engineering in the past and none of the monitoring stations intercept the entire flow from the relevant estuary. It is not, therefore, evident whether the measured interannual (and within year) differences reflect real changes in migration rates, changes in the efficiency of monitoring programmes, changes in water management policies by the local water companies or changes in eel behavior in the river systems with different water temperatures. From the data, and without more information, it is reasonable to assume that the glass eel migration numbers were greater in 2015 but it is not possible to know by how much. Beeleigh could have integrated glass eel numbers over a longer period than at Flatford judging by the extended duration of peak counts. Variable amounts of water abstraction at Beeleigh by the local water company are also considered to affect count numbers by varying amounts both in year and from year to year depending upon their operational needs. Flatford is nearer is to Sizewell and had a much more distinct migration peak lasting only one month and may therefore be more representative of the density change at Sizewell. Alternatively, there may have been different water quality parameters at the two sites in different years which made one or the other site more attractive to up estuary migrating eels. From the data all that can be said is that the East Anglian eel migration was probably higher in 2015 when the BEEMS Sizewell glass eel surveys took place than in the CEMP year of 2011, possibly by a factor of 2 based upon the nearest Flatford monitoring site data. The uncertain influence of water abstraction at Beeleigh means that there is less confidence in the interannual variability statistics derived from that site and it is also much further way than Flatford is from Sizewell.

3 Limit of detection and derived number of migrating glass eels at Sizewell

The two BEEMS surveys at Sizewell that were expected to most effectively detect glass eel were the offshore glass eel surveys in April (5 days) and May 2015 (6 days) and the SZB entrainment sampling in 2011 (40 24h samples spread over 1 year). The respective survey reports do not detail the limits of detection for these programmes, so these are reported here.

Only the 2015 glass eel survey demonstrated that glass eels are present at Sizewell. However, 11 days of survey effort only detected one glass eel on 20th April, on the flood tide, at the surface and at the SZC intake location. This corresponds to a measured density of 0.000387 eels m^{-3} (1 eel in a tow of 2581.4 m^3 of seawater sampled, BEEMS Technical Report TR356). As no other eels were detected in the 105 15-minute tows, this represents the maximum density in the survey period and represents the limit of detection of a single tow (the volume sampled from tow to tow changes slightly with the tidal/vessel speed). Assuming that eels are only at risk of capture during one tidal direction, the Sizewell survey detected a mean density of 1 eel in 52 flood tide tows or 7.44 10^{-6} eels m^{-3}.

By way of comparison, the Hinkley Point glass eel surveys used the same 15-minute tow length, the same net and approximately the same volume of water sampled. Out of 193 tows in 2013 only 9 contained no glass eels, with the largest catch being 125 at a density of 0.0457 m^{-3} (BEEMS Technical Report TR274) or 118 times the peak density at Sizewell. In 2012 only 12 out of 130 tows contained no glass eels, with the largest catch being 43 at a density of 0.016 m^{-3} (BEEMS Technical Report TR-S 211) or 41 times the peak density at Sizewell.

The 105 Sizewell tows represent 26.25 hours of surveying. Glass eels are poor swimmers and on energetic grounds they are considered to migrate using STST in one tidal direction and rest on or within the seabed on the opposing tide. Based on this conservative assumption that glass eels are only present in the water column during one tidal direction the measured density was 1 eel in 13.125 h of sampling (the amount of sampling on the ebb and flood tides was the same) or 0.0762 eels/hour in the volume intercepted by the sampling net.

3.1 Derivation of number of migrating eels at Sizewell

The key survey parameters and associated assumptions were:

1. The glass eel sampling net (MIK net) was square with a side length of 1.4m.
2. The SZC intakes are approximately 3 km offshore
3. The mean water depth at the SZB intakes is 9m and at the SZC intakes 13.5m (BEEMS Technical Report TR302)
4. Both the SZB and SZC intakes are capped head designs and across the tidal cycle will take only a small proportion of the water that they abstract from above or below of the height of the intake surfaces (intake surface heights 3m and 2m, respectively).
5. Eels actively migrate when the tidal current is flowing in the required direction.

The single glass eel was detected at the Sizewell C intake locations approximately 3km offshore. Based upon the extensive survey programme at Hinkley Point it would be expected that the density of glass eels would be highest nearest the coast and also highest in surface waters (BEEMS Technical report TR274). This behaviour would give the eels the best chance of detecting a freshwater signal. In the deeper water at the offshore SZC intakes it would expected that the eel density would be lower than at SZB. However, for the purpose of this worst case SZC assessment we have assumed the following:

i. that glass eels are uniformly distributed along a transect from the coast to the SZC intakes.
ii. that glass eels are uniformly distributed in depth from the surface to the seabed at any measurement location when the eels are migrating.

These assumptions are considered unlikely based upon behavioural evidence from Hinkley Point. If, as expected, eel density is greatest closer to the coast and in the surface waters, the predicted densities at the SZC intakes and the consequent SZC entrainment predictions made herein will be unrealistically high.

The glass eel surveys towed the sampling net into the tide. The tidal flow at Sizewell is shore parallel in an approximate north-south direction. Assuming that glass eels are uniformly distributed along the 3000m easterly transect from the coast to the Sizewell C intakes, the 1.4m net sampled (1.4/3000) of the width of the transect or 0.047%. At the SZC intake locations the net sampled (1.4/13.5) of the water column = 10.37%.

I.e. at the Sizewell C intakes, the net sampled 0.047%*10.37% or 0.00484% (4.84×10^{-5}) of the glass eels in the 3000m channel between the coast and the intakes.

Equation 1.

$$\text{Number of eels per hour migrating in the channel} = \frac{\text{Numbers of eels caught per hour at SZC location}}{\text{proportion of the coastal transect sampled by the net}}$$

= 0.0762/0.0000484 = 1574 per hour (on one tidal direction)

Or = 566,764 eels per month (assuming 30d per month)

These numbers can then be used to determine the limit of detection of the SZB entrainment programme (CEMP).

3.2 Limit of detection of the Sizewell B CEMP programme

The calculated total Sizewell B intake intercept area is approximately 118.4m^2 (BEEMS Scientific Position Paper SPP099 v4) i.e. the width of the channel that is intercepted by the SZB intakes is 118.4/3 m (where the height of intake surface is 3m) = approximately 39.47m.

Assuming that the eel density in 2011 was the same as that in 2015, the number of eels per hour on one tide migrating in the channel = 1574 per hour (Equation 1).

Number of migrating fish per day in the channel =12 * 1574 = 18,892

Equation 2.

$$\text{Predicted Sizewell B eel entrainment per 24h} = \text{eels migrating per 24h} \times \frac{\text{intercept width of intake}}{\text{channel width}} \times \frac{\text{height of intake surface}}{\text{water depth}}$$

= 18,892 * 39.47/3000 * 3/9

= 82.85 eels per 24 h

The SZB CEMP programme abstracted 0.035% of the total station cooling water flow in the period of the CEMP from March 2011 to May 2011 (typically with 3 pumps operational, BEEMS Technical Report TR235)

i.e. the best case expected catch rate for the CEMP in the period March 2011 - May 2011 = 0.00035 * 82.85 = 0.029 fish per 24h sample i.e. the chance of detecting glass eels at this density was negligible. Even over the 8 days of the CEMP survey in March to May the expected catch would only have been a best case of 0.23 eels.

This calculation assumes that eels are uniformly distributed across the channel and with depth which is considered unlikely based upon behavioural evidence from Hinkley Point. Even so the density would have had to be approximately 4 to 5 times greater than predicted to have a reasonable chance of detecting eels in the CEMP programme.

In practice, based upon Environment Agency glass eel monitoring data, the 2011 eel numbers could have been a factor of 2 (based upon Flatford data) lower during the CEMP programme than in 2015 (Section 2). As such, the predicted CEMP catch over the 8 sampling 24 hour periods would have been expected to be no more than 0.12 eels i.e., no glass eels would have been expected to be caught in 2011 unless the sampling intensity was at least 9 times higher than executed which would have required an infeasibly and disproportionately large monitoring programme to catch a single individual.

4 Calculated worst case eel entrainment at SZC

The proposed SZC LVSE intakes are designed to have a smaller intercept area for fish being transported in the tidal flow than for conventional omnidirectional intakes such as those at SZB. The calculated intercept area of the four SZC intakes is 42.3m^2 (BEEMS Scientific Position Paper SPP099 v4) or 10.58m^2 per head. Each intake surface is 2m high i.e. the calculated intercept width of each head is 5.29m. All four intakes have a combined intercept width of 21.16m.

The calculated number of migrating glass eels per 30-day month was 566,764 (Equation 1, Section 3.1).

The predicted number of glass eels abstracted at SZC is given by:

Equation 3.

$$\text{Predicted monthly Sizewell C eel entrainment} = \text{eels migrating per month} \times \frac{\text{intercept width of intakes (4)}}{\text{channel width}} \times \frac{\text{height of intake surface}}{\text{water depth}}$$

$$= 566{,}764 * 21.16/3000 * 2/13.5$$

$$= 592 \text{ eels per month}$$

Measured glass eel entrainment mortality with a simulated SZC profile (Section 1.2) = 20% (BEEMS Technical Report TR318) i.e. the expected glass eel mortality/month in SZC = 118.5 eels per month based upon the estimated maximum density in April 2015 (Section 2). Rates of entrainment mortality are considered further in Section 5.1.6 below.

Glass eels weigh approximately 0.33g/fish and therefore the weight of lost glass eels per month at SZC = 0.039kg per month. The conversion factor between glass eels and adult silver eels is given by 1kg glass eels is equivalent to 59.4 kg silver eels (Defra 2018). The equivalent silver eel mortality is therefore 2.32 kg per month or 6.97 kg (0.00697 t) in the expected 3-month period when the majority of glass eel migration takes place at Sizewell (Section 2).

The estimated Anglian River Basin District (RBD) SSB is 78.6 t (BEEMS Technical Report TR406 v7). SZC entrainment of glass eels is therefore given by:

0.00697/78.6 = 0.0089% Anglia RBD SSB

In TR406 v7 SZC impingement losses of silver eels with mitigation measures fitted to the station was calculated to be equivalent to a mean of approximately 0.045% of the Anglian RBD SSB (range 0.03% to 0.07% SSB). The addition of the entrainment losses calculated above would increase the total entrapment losses to 0.054% SSB (range 0.039% to 0.079% SSB).

5 SZC Entrainment losses - Sensitivity considerations

The predicted glass eel entrainment losses in Section 4 are based upon the single glass eel caught in the targeted glass eel study in late April 2015. There are a range of assumptions used to derive the estimate glass eel entrainment mortality and it is necessary to examine each of those in turn in sensitivity tests.

5.1 Key assumptions used in this assessment

1) Glass eel density is constant spatially and with depth in the 3000m channel between the coast and the SZC intakes.
2) Density is constant at the peak measured daily density for 3 months.
3) Density is constant day and night.
4) The BEEMS glass eel survey timing in April/May 2015 was appropriate to capture peak density.
5) The density during the BEEMS glass eel surveys in 2015 was not abnormally low.
6) Entrainment mortality in SZC will be 20%.

5.1.1 Are the assumptions of uniform glass eel spatial density at Sizewell and uniform vertical density reasonable?

The prevailing scientific hypothesis is that glass eels predominantly migrate close to the coast and near to the surface. This appears to be reasonable based upon considerations of the likelihood of detecting buoyant freshwater signals at estuary mouths. However, there is limited experimental evidence to substantiate or quantify this behaviour. The best data that are currently available are from the Bristol Channel at Hinkley Point (BEEMS Technical Report TR274) where, after pooling, the data for glass eel catches at each of the locations of Hinkley point C intakes (3km offshore) and at Hinkley Point B intake (640 metres offshore):

- the density at the offshore HPC intakes was 59% of the density at the inshore HPB intake.
- The density at 7m depth was 68% of the density at 0 m.

These factors are indicative as there were insufficient data (from 321 tows) to determine statistical significance at these specific locations. There was a statistically greater density inshore than offshore but the depth variability was only indicative and was only significant for 2 out 3 surveys. The centre of the SZC intakes surfaces would be located 11.5m depth where the glass eel density would be expected to be lower than at 7 m.

The glass eel caught at Sizewell was at the location of the SZC intakes at 3km offshore and therefore only the depth variation is relevant to the SZC entrainment calculation. Applying the Hinkley Point 68% factor, SZC entrainment may have been exaggerated by a factor of at least 1.47 as the SZC intakes would abstract at 11.5m depth rather than the surface.

Conclusion: On the evidence from an estuarine location (Hinkley Point) the constant density at depth assumption would exaggerate SZC entrainment losses by a factor of 1.47 and possibly more as the intakes are in deeper water.

5.1.2 Is the assumption of constant glass eel density over the main 3-month migration season reasonable?

The assessment assumes that the glass eel numbers were uniformly distributed across the 3-month migration period. However, the Flatford glass eel monitoring data were not distributed in that manner. Correcting for the Flatford count distribution requires dividing the entrainment result by 2.82 (Section 4).

Conclusion: On the evidence from the Flatford monitoring station it is not appropriate to assume a constant density over the main 3-month migration period and in so doing entrainment may have been over estimated by a factor of 2.82.

5.1.3 Is the assumption that density is constant during daylight hours and at night reasonable?

As far as Cefas is aware, the Sizewell monitoring was the first targeted attempt to measure glass eel density in coastal waters. That survey detected one glass eel during daylight only surveys. Such measurements are extremely difficult to undertake because the lack of spatial boundaries means that eel densities are not concentrated as they are in upper estuaries. To study spatial, depth and temporal variability at the expected low eel densities at Sizewell would require an unfeasibly large survey effort with multiple vessels. It is not, therefore, surprising that there are no data in coastal waters on glass eel behaviour over a daily cycle. The only substantial datasets are from estuarine locations; from the Hinkley point to the Welsh coast where glass eels were migrating up estuary and from transitional waters in the Gironde. The Hinkley point monitoring detected no differences between day and night density. In the Gironde estuary (Section 1) the night time density was estimated be 2.3 times higher than during the day. The 2020 BRB monitoring at Bradwell was inside the Blackwater Estuary where glass eels would already have changed their behaviour to migrate on the flood tide and only detected a total of 10 glass eels in one year but all of these were caught at night.

Applying the Gironde factor to the Sizewell results, the daytime density would be 1574 eels per hour and the night-time density 3621 eels per hour. Neglecting daylength this then translates into a monthly estimate of 935,160 eels per 30-day month: ((1574 * 6 hours+ 3621 * 6 hours) * 30 days) or an increase in expected eel density at Sizewell of approximately 1.65 times.

As a sensitivity test, a hypothetical glass eel density at night of 10 times that during the day was assumed. Ie 15,743 eels per hour at night implying a total of 3,117,201 per 30-day month or an increase in eel density at Sizewell of approximately 5.5 times.

Conclusion: There is no evidence on the diurnal variability of glass eel density in coastal waters. The limited evidence from estuarine locations may indicate a higher night-time density. Using the Gironde data the SZC entrainment would increase by a factor of 1.65. Using a hypothetical test case of night-ime density being 10 times the daytime density, SZC entrainment would increase by a factor of 5.5.

5.1.4 Was the BEEMS glass eel survey timing in April/May 2015 appropriate to capture the peak glass eel density in that year?

From comparison of the relative timings of the peak numbers of glass eel recorded in 2015 at Flatford and Beeleigh it would be possible to initially consider that the 2015 BEEMS glass eel surveys were too early to capture the peak glass eel migration period. However, this is not an easy judgement to make because the Environment Agency monitoring stations are approximately 20km inland from the estuary mouth and there are no published data from which to reliably estimate how long glass eels would take to migrate to the monitoring locations from the estuary mouth. Based upon recent monitoring at Bradwell undertaken as part of the BRB project, the Beeleigh monitoring station records peak glass eel counts approximately 6 to 8 weeks after the measured peak at the estuary mouth (Section 2). Based upon the Bradwell monitoring data peak glass eel density at Sizewell could be expected during late March-early April which virtually overlaps with the BEEMS survey timing of April -May. It is therefore conceivable that the BEEMS survey could have been slightly too late to capture the peak glass eel numbers, possibly by approximately 2 weeks.

Attempting to use the timing of the glass eel count data at the Environment Agency monitoring stations in Flatford and Beeleigh is, as explained in Section 2, fraught with difficulties due the poorly understood delay in making the transition across the tidal limit and the effect this would have on relative timings. In 2015 at Beeleigh the ratio of the eel counts in the peak month to the prior month was 2.1 but at Flatford it was 54.8. However, as discussed previously in this report, these counts near to the tidal limit cannot currently be translated into counts in the coastal zone. The BRB monitoring indicates that being a month early could result in a factor of nine difference in peak counts but the data from that survey are too sparse to draw firm conclusions.

Conclusions: The BEEMS 2015 survey may have been slightly too late to capture the peak migration density; possibly by 2 weeks but this timing is uncertain. There are insufficient data to quantify what the difference, if any, this would have made to the number of glass eels caught during the survey.

5.1.5 Was the glass eel density in 2015 (the year that in which the BEEMS surveys took place) abnormally low?

The BEEMS glass eel survey was conducted in 2015 and if that had been a year of anomalously low glass eel recruitment SZC entrainment could have been underestimated in this assessment. The count of glass eels in 2015 at the two East Anglian monitoring locations was not abnormally low for the period post 2013 (Section 2, Figure 1). Based on Environment Agency data 2013 was the peak year in recent times for glass eel recruitment. There are no data for 2013 in the Flatford dataset. The measured numbers in 2015 were 54% of the peak at Beeleigh in 2013. i.e. the glass eel density could have been a factor of approximately 2 higher than measured in the 2015 BEEMS surveys.

Conclusion: 2015 was not an abnormal year for recruitment post the 2013 rebound in glass eel counts. Recruitment numbers have been up to a factor of 2 higher at Beeleigh in 2013 but as discussed before interpretation of absolute numbers at these monitoring sites is problematical.

5.1.6 Is the assumed glass eel entrainment mortality of 20% reasonable?

It is not possible to directly measure entrainment mortality of glass eels at an operational power station. In the past experimental animals have been subject to a series of laboratory experiments that simulated individual aspects of the entrainment process e.g. temperature, pressure and chlorination. Such work was useful but was always vulnerable to criticism that the experiments failed to capture the cumulative effects of entrainment. In order to overcome this perceived problem entrainment mimic units (EMU) have been built which more closely simulate the conditions in a power station. The BEEMS project built on these early experiments and constructed a second-generation system which could repeatedly simulate entrainment conditions in specific power stations using user specified exposure profiles (BEEMS Technical Report TR318).

These experiments simulated the exposure of glass eels to mechanical, thermal, pressure and chemical impacts from abstraction to discharge for the same time intervals that would be experienced in a real station. The simulations included filtration through a 5mm mesh, the complete time of exposure in the HPC cooling water system, the profile of temperature (delta T +11.6°C) and pressure (peak change 4bar) as the eels pass through the station and also the effects of chlorination at a TRO level of 0.2mg/l from the precise time that it would be applied in the station to glass eel discharge in the power station outfalls.

In addition to chlorination, small quantities of process waste hydrazine would be discharged into the cooling water flow at the seal pit in a single daily pulse of 2.32h per day resulting in an initial hydrazine concentration of 69 ng l-1 in the cooling water flow. (This discharge scenario is the worst case as far as entrainment risk is concerned and there is alternative daily discharge scenario of 4.6h of 34 ng l-1, BEEMS Technical Report TR193). However, when hydrazine is added to chlorinated seawater the hydrazine is oxidized to nontoxic nitrogen, sodium chloride and water and the hydrazine concentration immediately decreases by approximately 90%. For example, in experiments described in BEEMS Technical Report TR363, an initial hydrazine concentration of 69 l^{-1} fell to 8.4 ng l^{-1} in the presence of chlorinated seawater at the planned TRO concentrations for SZC. To put these concentrations into an environmental risk context, the Canadian Federal Water Quality Guidelines for hydrazine indicate that concentrations below 200 ng l^{-1} have a low likelihood for adverse effects for marine life (Environment Canada 2013). Even at the planned SZC initial concentration of 69 l^{-1} the concentration of hydrazine for only 2.3h per day would be considered to present a very low risk for entrained organisms; at 8 ng l^{-1} the additional risk is considered negligible and was not necessary to simulate in the EMU system.

Based upon the EMU experiments, the predicted entrainment mortality for glass eels in SZC was found to be approximately 20%. Glass eels were found to be relatively insensitive to everything except mechanical damage passing through the system, particularly the 5mm filter. The only issue that is missing from the EMU simulations is estimated losses in the cooling water pumps. These pumps are very large and estimates

prepared for the HPC DCO were of losses between 1.6% to 1.8% based upon modelling using the STRIKER programme that has been widely applied to other pump mortality calculations (the SZC pumps would be the same as those planned for HPC). However, the EMU system was not specifically designed for glass eel experiments, in particular, the radius of bends in the EMU pipework were too small for glass eel experiments and subjected the experimental animals to far more mechanical stress than would be expected in SZC. This additional mechanical stress is considered to be greater than the predicted losses from the cooling water pumps and therefore the EMU derived entrainment mortality of 20% is considered to be appropriate.

Table 2 Summary of sensitivity analysis conclusions

Test	Description	Conclusion	Effect on Assessment
1	Are the assumptions of uniform glass eel spatial density at Sizewell and uniform vertical density reasonable?	On the evidence from an estuarine location (Hinkley Point) the constant density at depth assumption would exaggerate SZC entrainment.	Entrainment losses may have been overestimated by a factor of **1.47** and possibly more
2	Is the assumption of constant glass eel density over the main 3-month migration season reasonable?	On the evidence from the Flatford monitoring station it is not appropriate to assume a constant density over the main 3-month migration period.	Entrainment losses may have been overestimated by a factor of **2.8**.
3	Is the assumption that density is constant during daylight hours and at night reasonable?	There is no evidence on the diurnal variability of glass eel density in coastal waters. The limited evidence from estuarine locations may indicate a higher night-time density.	Using the Gironde data the SZC entrainment may have been underestimated by a factor of **1.65**. Using a hypothetical test case of night-time density being 10 times the daytime density, entrainment losses would have been underestimated by a factor of **5.5**.
4	Was the BEEMS glass eel survey timing in April/May 2015 appropriate to capture the peak glass eel density in that year?	The 2015 survey may have been slightly too late to capture the peak migration density; possibly by 2 weeks but this timing is uncertain.	There are insufficient data to quantify what the difference, if any, this would have made to the number of glass eels caught during the survey.
5	Was the glass eel density in 2015 (the year that in which the BEEMS surveys took place) abnormally low?	2015 was not an abnormal year for recruitment post the 2013 rebound in glass eel counts. Recruitment numbers have been up to a factor of 2 higher in2013 at Beeleigh but as discussed in this report, interpretation of absolute numbers at these monitoring sites is problematical.	Entrainment losses could be up to a factor of **2** higher in a peak recruitment year
6	Is the assumed glass eel entrainment mortality of 20% reasonable?	A SZC entrainment mortality of 20% is considered to be precautionary.	None
Net sensitivity change			Based on all considered factors, entrainment losses could be a between a factor of **1.25** lower to **2.67** higher

In Section 4 the estimated SZC glass eel entrainment loss was calculated as 0.0089% of the Anglia RBD SSB. Considering the cumulative effect of the sensitivity analyses, the SZC entrainment losses could have been underestimated by a factor of 2.67, assuming a hypothetical increase in night time density of a factor of 10 (noting that there are no measurements to support this figure). Using the calculated Gironde day: night ratio the SZC entrainment losses may have been overestimated by a factor of 1.25. On that basis the SZC entrainment losses are estimated to be between 0.0071% to 0.024% Anglia RBD SSB.

5.2 Do the predicted results look reasonable in context?

a. In comparison with Hinkley Point results

In the glass eel surveys at Hinkley Point (BEEMS Technical Report TR274), that sampled the whole width of the Severn Estuary (apart from the shallow margins nearest to the coast), the estimated mean density between 2012 to 2013 varied from 0.0017 glass eels m^{-3} (95% CI 0.0038) to 0.0058 m^{-3} (95% CI 0.001), and within 2013 from February to April from 0.00393 m^{-3} (95% CI 0.00077) to 0.00216 m^{-3} (95% CI 0.00052). At Sizewell the single measurement in April 2015 was 0.00039 m^{-3} with a mean density over 52 tows of 7.44 10^{-6} m^{-3} implying that the mean density measured at Hinkley Point was between 230 times (2012) and 780 times (2013) higher than the mean density at Sizewell in 2015. Given the unknown relationship between measurements of glass eel recruitment on the east and west coast of the UK it is not possible to scale these estimates to a common year.

However, given that, by virtue of its relative geography in relation to the glass eel migration route from the Atlantic, the glass eel migration run in the Severn is the largest in the UK, the relative ratio of the measured eel densities off Sizewell and in the Severn does not seem disproportionate at more than two orders of magnitude difference.

b. In comparison with Environment Agency monitoring counts

In Section 2 the total glass eel counts at Flatford and Beeleigh for 2015 were 8,529 and 18,189, respectively. Three months at the predicted Sizewell flux of 566,764 eels per month equals 1,700,292 eels (Section 3.1). The two estuary counts therefore represented 0.5% and 1.07% of the predicted Sizewell total. Neither counter captured all eel immigration into their respective estuaries, but the totals seem reasonable as intuitively it would be expected that a small percentage of the eel flux would be extracted at each river system dependent upon the size of the estuary and the freshwater flow rate.

6 Conclusions

BEEMS Technical report TR406 v6 did not provide a quantitative estimate for predicted SZC entrainment of glass eels due to the negligible densities determined during the survey programmes at Sizewell (e.g. 1 glass eel caught in 105 tows in targeted glass eels surveys). Consultees have stated that their confidence in predictions of eel entrapment is low. In particular, they are unable to conclude the potential level of entrainment to glass eels and the suggested solution was to undertake further surveys.

Null or low catches are useful in that they can be used to set ceilings for the density of fish in an area. We have used the existing Sizewell data to calculate the limit of detection for the two main survey techniques used to measure glass eel density at Sizewell in the BEEMS programme and to determine the predicted SZC entrainment level. The measured glass eel density at Sizewell has been set in the context of data from Environment Agency glass eel monitoring stations in two East Anglian sites (Flatford and Beeleigh) and from results from the Blackwater monitoring programme conducted by the Bradwell B project.

The following conclusions have been reached:

i. The BEEMS glass eel survey tows had a limit of detection of 0.000387 eels m^{-3} per tow. Assuming that eels are only at risk of capture during one tidal direction, the Sizewell survey detected a mean density of 7.44 10^{-6} eels m^{-3}. The same survey methodology and sampling equipment at Hinkley Point, detected mean densities of 0.0058 m^{-3} and a peak density of 0.0457 m^{-3}. From these data it is concluded that the density of glass eels at Sizewell is at least two orders of magnitude lower than at Hinkley Point. Based upon the predicted Sizewell glass eel density, the amount of effort required to increase the predicted Sizewell glass eel catch from 1 eel to 10 to 15 eels would be infeasible and would require impractically long working days and increasing the number of survey days from 11 to most days during the migration period. This amount of survey effort is disproportionate when a worst-case estimate can be derived from the data provided by the existing surveys.

ii. The BEEMS entrainment survey at Sizewell in 2011 would not have been sensitive enough to detect glass eels at the expected density in that year. The necessary increase in sampling to detect one eel would have been infeasible based upon the 2015 glass eel survey results.

iii. The calculated glass eel entrainment loss at SZC would be 0.0089% Anglian RBD SSB based upon 2015 survey data.

iv. Considering the cumulative effect of the sensitivity analyses described in this report, the SZC entrainment losses could have been underestimated by a factor of 2.67, assuming a hypothetical increase in night time density of a factor of 10 (noting that there are no measurements to support this figure). Using the calculated Gironde day: night ratio the SZC entrainment losses may have been overestimated by a factor of 1.25. On that basis the SZC entrainment losses are estimated to be between 0.0071% to 0.024% Anglia RBD SSB.

v. To put the SZC entrainment estimate into context, the estimated SZC mean eel impingement loss is 0.045% Anglian RBD SSB. i.e. the predicted SZC entrapment loss is in the range 0.052% to 0.0.069% Anglian RBD. One percent of the Anglian RBD SSB is equivalent to an estimated 0.005% of the European eel stock.

References

Beaulaton,L., Castelnaud, G. 2005. THE EFFICIENCY OF SELECTIVE TIDAL STREAM TRANSPORT IN GLASS EEL ENTERING THE GIRONDE (FRANCE). Bull. Fr. Pêche Piscic. (2005) 378-379: 5-21.

BEEMS Scientific Position Paper SPP099. Predicted impingement performance of the SZC LVSE intake heads compared with the SZB intakes. Cefas, Lowestoft.

BEEMS Technical Report TR193. Sizewell C Discharges H1 type assessment – supporting data report, Edition 5. Cefas, Lowestoft.

BEEMS Technical Report TR-S 211. Glass eel distribution in the inner Bristol Channel / lower Severn Estuary. Cefas, Lowestoft.

BEEMS Technical Report TR235. Comprehensive Entrainment Monitoring Programme (CEMP) 2010/2011 at Sizewell B: Final report. Pisces Conservation Limited.

BEEMS Technical Report TR274. Dynamics of glass eels in the Bristol Channel 2012-2013. Cefas, Lowestoft.

BEEMS Technical Report TR302 Ed. 3. Sizewell Thermal Plume Modelling: GETM Stage 3 results with the preferred SZC cooling water configuration. Cefas, Lowestoft

BEEMS Technical Report TR318. Predictions of entrainment by Sizewell C in relation to adjacent fish and invertebrate populations. Cefas, Lowestoft.

BEEMS Technical Report TR356 Sizewell Glass Eel Surveys. Cefas, Lowestoft.

BEEMS Technical Report TR363. Thermal and chemical effects of power station cooling water on biomass, abundance and physiology of phytoplankton communities at Sizewell. Cefas, Lowestoft.

BEEMS Technical Report TR406 v7. Sizewell C – Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft

Cresci, A. 2020. A comprehensive hypothesis on the migration of European glass eels (*Anguilla anguilla*). Biol. Rev. 2020.

Defra, 2018. Report to the European Commission in line with Article 9 of the Eel Regulation 1100/2007 Implementation of UK Eel Management Plans - June 2018.

Edeline E., Beaulaton L., Le Barh R. and Elie P., 2007. Dispersal in metamorphosing juvenile eel Anguilla anguilla. Mar. Ecol. Prog. Ser., 344.

Environment Canada 2013. Canadian Environmental Protection Act, 1999 Federal Environmental Quality Guidelines. Hydrazine. 2013 Available from: http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=D66353C2-1.

Feunteun E., Laffaille P., Robinet T., Briand C., Baisez A., Olivier J.M. and Acou A., 2003. A review of upstream migration and movements in inland waters by anguillid eels. Towards a general theory. In: Aida K., Tsukamoto K. and Yamauchi K. (eds.), Eel Biology. Springer-Verlag, Tokyo, 191–213

Harrison, A.J., Walker, A.M., Pinder, A.C., Briand, C., Aprahamian. M. W. 2014. A review of glass eel migratory behaviour, sampling techniques and abundance estimates in estuaries implications for assessing recruitment, local production and exploitation. Rev. Fish Biol. Fisheries V24, Issue 1, pp 967-983.

ICES. 2019. Joint EIFAAC/ICES/GFCM Working Group on Eels (WGEEL). ICES Scientific Reports. 1:50. 177 pp. http://doi.org/10.17895/ices.pub.5545CES.

ICES 2020. Workshop on the temporal migration patterns of European eel (WKEELMIGRATION). ICES Scientific Reports. 2:25. 108 pp. http://doi.org/10.17895/ices.pub.5993

Lambert P, Durozoi B, Beaulaton L. 2007. Validation des hypotheses comportementales utilisees dans la quantification du flux d'anguille (Anguilla anguilla) dans l'Isle. Contrat Indicang. Rapport Cemagref no. 115.

McCleave, J.D., 1993. Physical and behavioural controls on the oceanic distribution and migration of leptocephali. J. Fish Biol. 40 Suppl., 243–273.

Tesch, F.W., 2003. The Eel, 5th ed. John Wiley & Sons.

Vøllested, L.A., 1992. Geographic variation in age and length at metamorphosis of maturing European eel: environmental effects and phenotypic plasticity. J. Anim. Ecol. 61, 41–48.

APPENDIX 2.17.A SUPPLEMENTARY INFORMATION ON FISH ASSESSMENTS

REPORT NO. SPP108

NOT PROTECTIVELY MARKED

SPP108 - Sensitivity of the Alde & Ore Transitional Fish Classification Index (TFCI) to changes in smelt, *Osmerus eperlanus*, abundance.

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Executive Summary

Under the Water Framework Directive, the Environment Agency monitors the ecological status of the Alde-Ore Estuary as a single water body (Alde & Ore). To assess the status of fish communities in transitional waters a multi-metric index, termed the Transitional Fish Classification Index (TFCI) is applied. The TFCI is designed to detect impacts on the fish status due to a wide range of anthropogenic pressures. A decline in the status of the water body triggers further investigation and potentially additional monitoring.

The status of the fish community in the Alde & Ore was classified as 'Good' in 2015 (EQR: 0.63) and 'Good' in 2016 (EQR: 0.61). Derivation of these EQR values is assumed to be based on 3 years of data in 2015 (2013-2015) and based on 4 years of data in 2016 (2013-2016). During this period seine nets and beam trawls were the predominant sampling method but fyke nets were also deployed and used in the EQR calculation. No classification was produced in 2017 or 2018. The recently published 2019 results show the water body transitional fish status apparently fell to 'Moderate' in 2019 (Environment Agency Catchment data Explorer, 2020). The 2019 EQR provided by the Environment Agency is 0.56 and was calculated using a combination of seine nets, beam trawls and fyke nets using data from 2013-2018.

This report calculates the TFCI baseline Ecological Quality Ratio (EQR) by applying Environment Agency fish monitoring data for the Alde & Ore water body. The baseline EQR is calculated using six years TFCI sampling data from 2013 to 2018. Sampling data from the period includes 75 seine nets, 34 beam trawls and 8 fyke nets (fykes sampled in 2013 and 2014 only). Seine samples returned 26 species and the highest catches, whilst beam trawls caught 21 species in total. Fyke nets only caught a single seabass and one ballan wrasse.

The Environment Agency algorithms used to derive EQR values now incorporate bootstrapping approaches. The bootstrapping method permutes samples from each gear type 1,000 times to determine the mean result and confidence intervals that are averaged between gears to calculate an overall EQR. By putting equal emphasis on all sample gears, the bootstrapping approach differs from the weighted mean approach and produces different results particularly when sample numbers and catch efficiency is low. It is noteworthy that if the 2019 EQR is calculated with fyke nets using the weighted mean approach, as in 2015 (0.63) and 2016 (0.61), the water body status is 'good' (EQR 0.63). Conversely if the 2015 and 2016 data are recalculated using the same bootstrapping method applied in 2019, the status falls to 'moderate' with EQRs of just 0.53 and 0.50, respectively.

This report was first commissioned prior to the release of the 2019 classification to determine the potential for smelt reductions to cause a deterioration in the status of the Alde & Ore water body. It has evolved following comments from the Environment Agency and in response to the latest water body classification to look at two aspects of the Alde & Ore TFCI:

I. the appropriateness of using fyke nets in the calculated Alde & Ore EQR, and

II. the sensitivity of the TFCI to reductions in fish abundance.

Calculations of the EQR for Alde & Ore water body were carried out using data provided by the Environment Agency and using their analysis software (R-code).

Cefas identified the following conclusions, which are supported following technical review by Steve Coates of SLR Consulting who was one of the original authors of the Transitional Fish Classification Index methodology and associated sensitivity analysis.

1.1 Effect of Fyke net data on the TFCI results

1. Recalculaton of the 2019 water body classification using data from 2013-2018 without the use of fyke net data, results in an Alde & Ore EQR of 0.69 representing 'good' status. The confidence of class for good status or above is 0.92.

2. Repeating the 2019 calculation but with inclusion of the 8 fyke net sample results in the Alde & Ore EQR falling by 18.8% to 0.56 corresponding to 'moderate' status.

3. In this situation, the TFCI calculation and status of the Alde & Ore water body is compromised by low sampling efficiency and statistical artefacts relating to the small fyke net sample size and low catches. It is concluded that inclusion of fyke nets has caused the apparent deterioration of the water body classification to 'moderate'.

4. It is concluded that fyke net results should not be included in the Alde & Ore TFCI calculation.

1.2 Effect of reductions in fish numbers on the water body classification:

The Environment Agency has raised a number of concerns relating to the potential for the proposed Sizewell C development to impact fish species of relevance to the Ade & Ore TFCI. Following establishment of the baseline EQR with and without fyke net data, the sensitivity of the TFCI was established in response to manipulated removal of those species that the Environment Agency has identified as important for the Alde & Ore TFCI including; smelt, shad, herring, seabass, thin-lipped grey mullet and Dover sole.

To determine the sensitivity of the TFCI to smelt abundance, smelt numbers in samples were manipulated to four levels representing declines of 25%, 50%, a single smelt caught per annum (in seine samples), and total smelt absence. A further test considered absence of smelt and Twaite shad and 50% reductions in herring and seabass (multispecies manipulation). These tests were run with and without fyke nets. The sensitivity of the TFCI was also tested further through manipulated removals of thin-lipped mullet and Dover sole, as well as a scenario whereby smelt, thin-lipped mullet and sole were all simultaneously reduced by 50%. These manipulations were calculated with no fyke nets included.

5. The EQR was insensitive to manipulated reductions in smelt abundance of 25% and 50%. Such reductions are considerably beyond the expected local level effects of the proposed Sizewell C development (BEEMS SPP103).

6. Total absence of smelt reduced the EQR by 11% but 'good' status remained. (Fyke net data removed from the TFCI calculation)

7. The Environment Agency requested a scenario of total absence of both shad (1 individual caught between 2013-2018) and smelt along with 50% reductions in herring and bass. In this extreme scenario the EQR was reduced by 10.3%, however, 'good' status remains (TFCI calculated without the inclusion of fyke net data).

8. Total absence of thin-lipped grey mullet and Dover sole reduced the EQR by less than 4% in each case and 'good' status remained (TFCI calculated without the inclusion of fyke net data). The status also remained 'good' following the combined 50% reduction of smelt, Dover sole and thin-lipped grey mullet.

9. Under all of the scenarios trested for fish manipulations, there was no deterioration of 'good' status when the TFCI was calculated without fyke net data, or no deterioration below 'moderate' status when the TFCI was calculated with fyke net data.

1.3 Likelihood of thermal occlusion or SZC impingement to affect smelt in the Alde & Ore

The potential for the proposed development to affect smelt, and other migratory fish (e.g. Dover sole and grey mullet), in the Alde & Ore by means of thermal occlusion and impingement was investigated and it was concluded that:

10. There is no evidence for thermal occlusion either at the mouth of the Alde & Ore where temperatures are ≤1°C as a 98th percentile, or across a hypothetical corridor at the frontage of Sizewell C, 25km to the north of the estuary mouth.

11. Impingement from Sizewell C is predicted to have a negligible effect on regional smelt stocks (0.18% of the conservatively estimated 'Anglian' SSB) and would not result in significant local reductions in smelt abundance entering the estuary (BEEMS Technical Report TR406; BEEMS SPP103).

Based on the evidence of the Alde & Ore TFCI sensitivity assessments in this report to effects far greater than those anticipated from the proposed development, it is considered highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore water body.

1 Background

The European smelt, *Osmerus eperlanus*, is an anadromous species, which lives primarily in the inshore marine environment but migrates up estuaries to spawn in early spring (February to April), after which adults return to sea (Maitland, 2003). Smelt was once a common species in the British Isles, however, overexploitation and pollution has contributed to its decline (Wheeler, 1979). Smelt occur in around 36 water courses in England and Wales, with large populations in the Humber and Dee, the Wash and Great Ouse, as well as in water courses of the Norfolk Broads (Colclough and Coates, 2013). Smaller populations exist in the rivers Alde and Ore, Ribble and Conwy, and recovery of supposed extinct populations seems to be underway in the rivers Thames, Tyne and Mersey (Colclough and Coates, 2013; Coates, pers.comm). Smelt is listed as a Priority Species in Section 41 of the Natural Environment and Rural Communities (NERC) Act 2006. Since 2010, smelt are afforded protection within the Salmon and Freshwater Fisheries Act 1975 (as amended). Within Scotland the Lower River Cree has been designated as a Site of Special Scientific Interest (SSSI) as this is the only known spawning site of smelt in Scotland.

Through ongoing consultation with the Environment Agency, a concern has been raised relating to the potential for the proposed Sizewell C development to negatively impact smelt populations potentially causing a deterioration in the status of the Alde & Ore water body.

The Alde & Ore water body is a heavily modified transitional waterbody (GB520503503800) routinely monitored by the Environment Agency in accordance with the requirements of the Water Framework Directive (WFD). The WFD assesses the ecological status of transitional water bodies, for which fish communities are a Biological Quality Element (BQE). To assess the status of fish communities in transitional waters a multi-metric index, termed the Transitional Fish Classification Index (TFCI) has been developed (Coates *et al*., 2007; WFD-UKTAG, 2014).

The TFCI is used to determine and report the WFD status of the fish community as an Ecological Quality Ratio (EQR). An EQR with a value of one represents reference conditions and a value tending to zero represents a severely impacted site. The TFCI is designed to detect impacts on the fish status due to a wide range of anthropogenic pressures.

The TFCI is based on the metrics listed in Table 1.

Table 1 WFD Transitional Fish Classification Index metrics

Number	Metric	Community characteristic
1	Species composition	Species diversity and composition
2	Presence of indicator species	
3	Species relative abundance	Species abundance
4	Number of taxa that make up 90% of the abundance	
5	Number of estuarine resident taxa	Nursery function
6	Number of estuarine-dependent marine taxa	
7	Functional guild composition	
8	Number of benthic invertebrate feeding taxa	Trophic integrity
9	Number of piscivorous taxa	
10	Feeding guild composition.	

The TFCI is calculated as the sum of all metric scores and converted into an EQR operating over a range from zero (a severe impact) to one (reference/minimally disturbed). The four class boundaries are:

High/Good = 0.81 Good/Moderate = 0.58 Moderate/Poor = 0.4 Poor/Bad = 0.2 (WFD-UKTAG, 2014).

Monitoring of the Alde & Ore by the Environment Agency is completed through annual (spring and autumn) surveys using 1.5m beam trawls and seine netting, in some years fyke nets have been deployed. The Alde & Ore was classified as having 'Good' status in 2011 and 'Moderate' in 2012 but the calculated EQR values were built on outdated methods applying 'old building blocks' (OBB) used in assessments at the time. From 2015 onwards, the TFCI applied the new building blocks (NBB), whereby the TFCI tool reference conditions are calculated differently, and assessments account for sampling effort. The status of the fish community in the Alde & Ore was classified as 'Good' in 2015 (EQR: 0.63) and 'Good' in 2016 (EQR: 0.61). No classification was produced in 2017 or 2018.

The recently published 2019[1] results show the water body transitional fish status apparently fell to 'Moderate' in 2019 (Environment Agency Catchment data Explorer, 2020). The 2019 EQR provided by the Environment Agency is 0.56 and was calculated using a combination of data from seine nets, beam trawls and fyke nets (a summary of how the EQR was calculated is provided in Section 2.1.1.3). The process used to calculate each of the above EQRs was different i.e. due to changes in one of more of: number of years in assessment, fyke net data in or out, analysis methodology)

1.1 Environment Agency comments on the Alde & Ore

1.1.1 Concerns pertaining to smelt

As part of the 'EA Sizewell C - Marine Ecology Gap Analysis' document (12 March 2020) and following the Marine Ecology Meeting (19 March 2020), the Environment Agency raised a number of concerns regarding the potential for the proposed Sizewell C development to impact smelt in the Alde & Ore water body, thus affecting the water body status. Smelt is an indicator species (Metric 2; Table 1) for the TFCI. Indicator species are 'disturbance-sensitive species' and smelt are selected due to their sensitivity to low dissolved oxygen (Coates *et al.,* 2007).

In summary, the Environment Agency requested further information on:

- the structure of the smelt population and whether impinged smelt are from a single North Sea population or from local, sub-populations;
- the at-sea behaviour of smelt and the potential for thermal plumes to cause occlusion of migratory routes; and
- the potential for a deterioration in fish status of the Alde & Ore water body.

In the context of this Scientific Position Paper (SPP108), the original concern addressed relates to the potential for a deterioration in the status of the Alde & Ore TFCI and responds to the specific Environment Agency comment:

> *"**Issue:** SZC impingement is presented as being about the same as SZB - this means that when both power stations are operating, impingement rates are considered to be almost double what they are now.*
>
> ***Impact:** Smelt are an important feature of the fish assemblage in the Alde Ore waterbody and the significant reduction of this species from the assemblage would lead to a deterioration in this waterbody.*
>
> ***Proposed solutions:** Include further consideration of whether an impact on smelt in the Alde Ore can be ruled out, particularly when SZB and SZC are operating in combination."*

In previous email correspondence (20 February 2020) the Environment Agency state:

> *"Initial assessment has shown that removing smelt from the Ore & Alde fish classification causes a 12% deterioration in Ecological Quality Ratio (EQR)".*

[1] WFD results released on 17th September 2020.

1.1.2 Inclusion of fyke net data and concerns pertaining to thin-lipped grey mullet

The Environment Agency provided TFCI code and data for the Alde & Ore in June 2020 for the period 2013 to 2018. The Environment Agency advised at the time that the data was still undergoing data QA and the fyke data may not be used in the final 2019 classification.

Cefas completed analyses to determine the sensitivity of the TFCI to changes in smelt catches including complete removal of smelt (following methods detailed in Section 2.2). The analyses looked at different sampling periods (i.e. including data from 2, 3- and 4-year assessment periods) and discounted fyke nets due to the very low catches in 2013 and 2014 (Section 2.1).

The results of Cefas analyses were presented at the Marine Technical Forum on 23 September 2020. During the meeting the Environment Agency advised that the 2019 Alde & Ore EQR was calculated using the previous six-years of available data, and fyke net samples had been included in the calculation.

The Environment Agency also expressed concerns about the impacts of the proposed development on thin-lipped grey mullet and the potential for implications on the TFCI.

1.1.3 SZC DCO Relevant Representation concerns pertaining to migratory species (smelt, Dover sole and grey mullet)

On the 29 September 2020, NNB GenCo (SZC ltd) received the Relevant Representations from the Environment Agency on the Sizewell C DCO submission, including the WFD Assessment. The Environment Agency reiterated their prior concerns pertaining to the impacts of the proposed development on smelt and expressed concerns relating to "*other marine migrants, including dover sole and grey mullet*". Additional correspondence (Section 1.1.2) provided concerns regarding golden mullet and the estuarine species thin-lipped grey mullet. It should be noted that within the TFCI, Dover sole are marine juveniles, not migrants (S. Coates pers. comms.).

Pertinent comments for this report include (full responses to all comments will be provided in due course):

"No consideration has been given in the WFD assessment to the risk of such marine migrants avoiding the thermal plume and fewer entering the Alde/Ore and Blyth estuaries."

"No evidence has been provided to support the position that smelt in the Ore/Alde water body are from a wider stock covering the North Sea. Smelt are an important species in the Ore/Alde water body and removing the species from the classification would result in an 11% deterioration in the fish classification. Given the predicted numbers of smelt to be impinged for the period when both SZB and SZC are operational, we are not able to conclude that a potential for deterioration to the Ore/Alde and possibly the Blyth water bodies does not exist."

1.1.4 Concerns pertaining to smelt, shad, herring seabass

Following ongoing consultation, further advice was provided by the Environment Agency in regard to the Alde & Ore classification in their email of 16 October 2020. The Environment Agency provided clarification that an investigation will be undertaken to determine the deterioration in the water body status from 'good' to 'moderate'.

The Environment Agency also expressed concerns relating to other species which make up part of the Alde & Ore fish assemblage.

"To progress with the SZC TFCI deterioration assessment for the Ore & Alde waterbody we request that the assessment includes the species listed below with the scenarios stated. We are requesting these species be included as they have been identified as being of importance in the Ore & Alde TFCI. Please provide these assessments with and without the inclusion of the fyke net method. The appropriateness of including the fyke net method in the 2019 classification is being investigated.

"Species present in the Ore & Alde waterbody identified as requiring a deterioration assessment as a result of entrapment from SZC. Please provide the assessment with the removal of smelt and shad, and include the reduction of herring and bass using the scenarios stated."

Latin Name	Common Name	Require deterioration assessment for Ore & Alde	Assessment scenario
Alosa fallax	Twaite shad	Yes	Presence/absence
Clupea harengus	Herring	Yes	25,50,75% reductions
Dicentrarchus labrax	European seabass	Yes	25,50,75% reductions
Osmerus eperlanus	European smelt	Yes	Presence/absence

That email contained no further reference to Dover sole but noted that golden grey mullet are caught in Alde & Ore monitoring and requested information on impingement records of the species in the coastal waters off Sizewell B. There are no records of golden grey mullet impingement between 2009 and 2017 (Section 2.1.1.2).

This SPP aims to consider the primary concerns from the Environment Agency and addresses the following points:

- Is the inclusion of fyke net data appropriate in the calculation of the 2019 Alde & Ore TFCI?
- Following the original request, the report determines the sensitivity of the TFCI to changes in smelt abundance.
- The sensitivity of the TFCI to changes in thin-lipped mullet and Dover sole.
- The sensitivity of the TFCI to changes in smelt, shad, herring seabass.

The discussion of this report re-visits the assessment of effects including the potential for thermal occlusion and impingement by the proposed development acting cumulatively with Sizewell B on fish populations in relation to the Alde & Ore.

Noting the Environment Agency request within their Relevant Representations that in order *"to assess a potential impact to a WFD waterbody the scale of assessment may need to be done at a smaller scale"* [than ICES stock units], we consider the potential for local effects and the appropriateness of assessing significant reductions in fish abundance is examined (Section 4.3.2).

A technical review of version 3 of this report was completed by Steve Coates of SLR Consulting who was one of the original authors of the transitional Fish Classification Index methodology and associated sensitivity analysis. Steve Coates has agreed with the analyses and conclusions presented in this report.

2 Source data and methodology

2.1 Environment Agency Alde & Ore data

Data on ecological status of fish communities in the Alde & Ore water body have been collected by the Environment Agency during spring and autumn annual surveys. This report is based on R code and data provided by the Environment Agency (5 June 2020).

The survey data file used in the subsequent analysis '*Data_Trac_Fish_FinalR_All_Years'* from 2013 to 2018 inclusive. These are the most up to survey date data as of October 2020. TFCI survey methods used to sample the Alde & Ore include a 1.5m beam trawl (4-8 samples per annum) and 43m seine nets (12-16 samples per annum). In 2013- 2014, fyke nets were deployed.

This report calculates EQRs for the six-year sampling period 2013-2018 with and without fyke nets.

2.1.1 Alde & Ore TFCI sampling

The location of the sampling sites in the Alde & Ore between 2013 and 2014 is provided in Figure 1. Seine nets, beam trawls and fyke nets have been deployed at Shingle Street - Lower, RSPB reserve and Boyton Docks. Seine and beam trawls have also been completed at Iken cliffs.

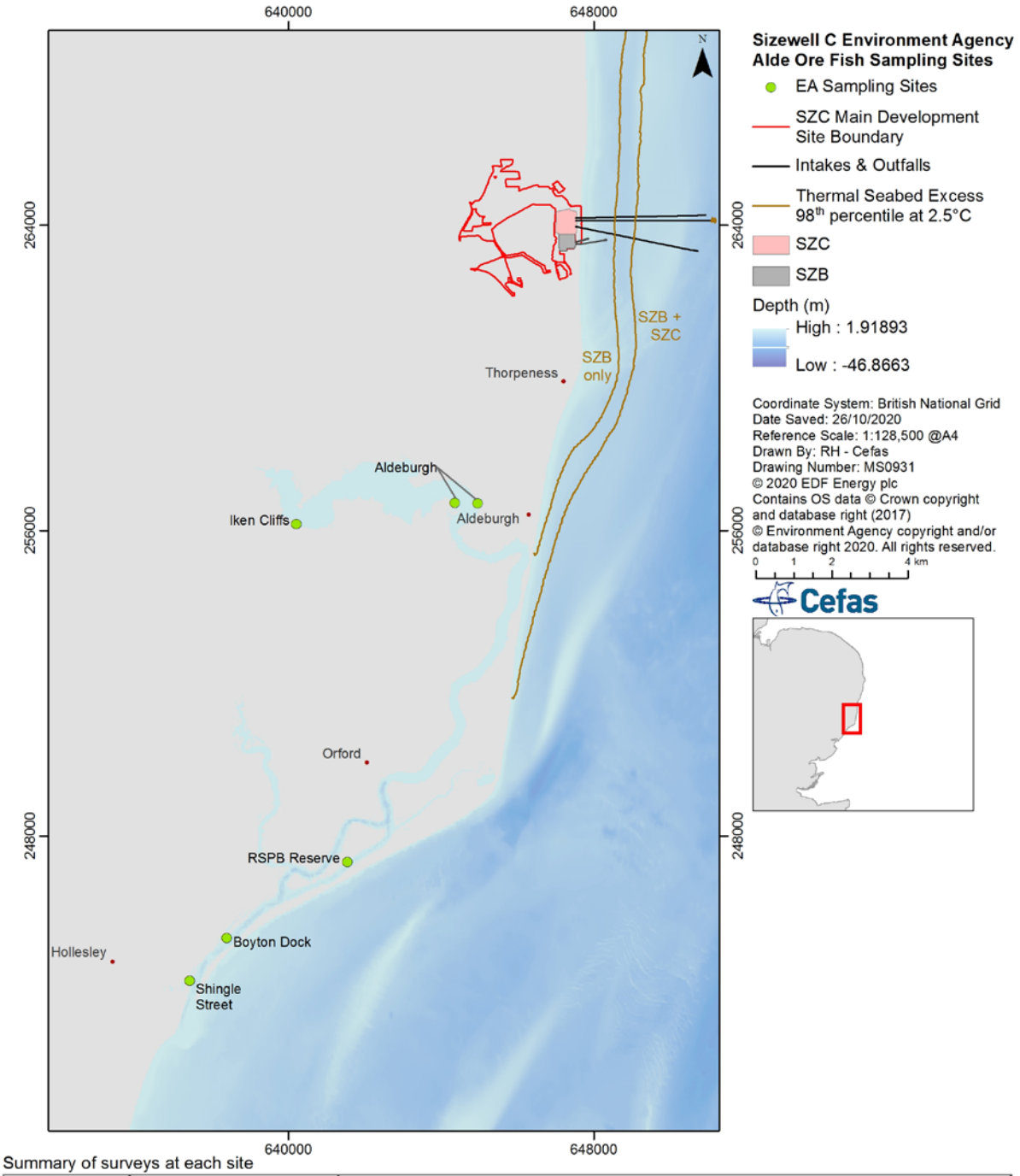
2.1.1.1 Smelt samples

Numerically, seine netting was the most effective gear to sample smelt within the Alde & Ore during the Environment Agency surveys, however, in terms of annual presence/absence beam trawling was nearly as effective. A total of 206 smelt were captured between 2013-2018 in seine nets, whilst only 16 were captured by beam trawls (Table 2). Fyke nets were deployed on 8 occasions in 2013/14 and captured no smelt.

Table 2. Smelt numbers in Environment Agency seine and beam trawl samples between 2013 and 2018 in the Alde & Ore.

Year	Number of seine samples	Number of beam trawls	Number of fyke samples[2]	Number of smelt captured by seine	Number of smelt captured by trawl	Number of smelt captured by fyke
2013	11	7	2	0	0	0
2014	16	8	6	123	9	0
2015	14	5	0	54	3	0
2016	8	4	0	0	0	0
2017	14	6	0	28	3	0
2018	12	4	0	1	0	0
Total	**75**	**34**	**8**	**206**	**15**	**0**

[2] In 2005 and 2006 fyke nets were successfully deployed at Boyton Dock, RSPB Reserve and Shingle St Lower (Section 4.1).



Summary of surveys at each site

Site Name	Sampling Cover	Survey Dates
Aldeburgh	Beam Trawl	May 2013
Boyton Dock	Beam Trawl	May 2013, Oct 2013, Jun 2014, Oct 2014, Jun 2015, Nov 2015, Sep 2016, Jun 2017, May 2018
Boyton Dock	Fyke Netting	Jun 2014, Oct 2014
Boyton Dock	Seine Netting	May 2013, Oct 2013, Jun 2014, Oct 2014, Jun 2015, Nov 2015, Sep 2016, Jun 2017, May 2018
Iken Cliffs	Beam Trawl	May 2014, Sep 2014, May 2015, Sep 2016, May 2017, May 2018
Iken Cliffs	Seine Netting	May 2014, Sep 2014, May 2015, Oct 2015, Sep 2016, May 2017, Oct 2017, May 2018, Sep 2018
RSPB Reserve	Beam Trawl	May 2013, Oct 2013, May 2014, Sep 2014, Jun 2015, Sep 2016, May 2017, Sep 2017, May 2018
RSPB Reserve	Seine Netting	May 2013, Oct 2013, May 2014, Sep 2014, Jun 2015, Oct 2015, Sep 2016, May 2017, Sep 2017, May 2018
Shingle Street	Beam Trawl	May 2013, Jun 2014, Oct 2014, May 2015, Sep 2016, May 2017, Sep 2017, May 2018
Shingle Street	Fyke Netting	May 2013, Oct 2014
Shingle Street	Seine Netting	May 2013, Jun 2014, Oct 2014, May 2015, Sep 2016, May 2017, Sep 2017, May 2018, Sep 2018

Figure 1 Environment Agency TFCI sample locations and gear deployments from 2013 to 2018 in the Alde & Ore water body.

2.1.1.2 Fish community

Between 2013 and 2018 a total of 117 samples were collected; these included 75 seine nets (64%) at five sites, 34 beam trawls (29%) at five sites, and eight fyke nets samples (7%) at three different sites (Figure 1). Seine nets were the most effective method for sampling the fish community resulting in 26 of 30 species in total collected and 18,228 individuals. Fewer beam trawls accounted for nearly as many species (21) and 505 individuals (Table 3). Only 8 fyke net samples were deployed and these resulted in only 2 fish captured, an individual seabass and one ballan wrasse, both in 2013 (Table 3). Their efficiency in respect of fish catch per station was about 60 times less than in beam trawls and ~1,000 times less than seine samples. TFCI monitoring in the Alde & Ore recorded a total of 393 golden grey mullet (*Chelon aurartus* ex. *Liza aurata*; Table 3), 97% of which were captured up estuary at Iken cliffs in 2017, some individuals were caught at Shingle Street (Figure 1). The Environment Agency have requested information on impingement records of the species in the coastal waters off Sizewell. There are no records of golden grey mullet impingement at Sizewell between 2009 and 2017.

Table 3. Number of fish captured by the different gears between 2013 and 2018 in the Alde & Ore.

Species	Beam trawl, 34 samples	Fyke Net, 8 samples	Seine, 75 samples
Agonus cataphractus	2		1
Alosa fallax			1
Anguilla anguilla			1
Aphia minuta	5		8
Atherina presbyter	1		915
Callionymus lyra			5
Chelon labrosus			3
Ciliata mustela	1		
Clupea harengus	10		8860
Dicentrarchus labrax	6	1	1043
Gasterosteus aculeatus			39
Gobius niger	2		
Labrus bergylta		1	
Liparis liparis	1		
Liza aurata			393
Liza ramada			201
Myoxocephalus scorpius	2		5
Osmerus eperlanus	15		206
Pholis gunnellus	1		2
Platichthys flesus	15		68
Pleuronectes platessa	1		17
Pollachius pollachius			1
Pomatoschistus microps	3		10
Pomatoschistus minutus	384		5882
Solea solea	25		69
Sprattus sprattus	22		475
Syngnathus acus	1		9
Syngnathus rostellatus	4		10
Taurulus bubalis	3		1
Zoarces viviparus	1		3
Total (Mean number of fish per sample)	**505 (14.85)**	**2 (0.25)**	**18,228 (243.04)**

2.1.1.3 TFCI analyses

The methodology for sampling and calculating the TFCI and respective EQRs is explained in the UKTAG Guide to the Transitional Fish Classification Index (WFD-UKTAG, 2014). The TFCI multi-metric includes measures of species diversity and composition, species abundance, nursery function and trophic integrity. Each metric is assessed by comparing the observed metric values with metric values under reference conditions. The TFCI is calculated as the sum of all metric scores and converted into an EQR with a range from zero (a severe impact) to one (reference/minimally disturbed).

From Environment Agency guidance the minimum number of samples required to calculate a TFCI score is 30 (Environment Agency, 2011). With greater sampling, the potential to detect rarer species increases thereby increasing the EQR. Because of the increase in EQR with increasing sampling effort, some empirical relationships between the number of samples and observed EQR have been derived to allow the 'true' EQR to be estimated for any number of hauls in a particular estuary in England or Wales (not Scotland) (WFD-UKTAG 2014). These relationships are incorporated into R-code to estimate the EQR from survey data.

The observed values of metrics are compared with predicted reference metric values (*MVpred*) for England and Wales, which depend on sampling effort and are defined as (WFD-UKTAG 2014):

$$MV_{pred} = a\{1 - (1 - b)^s\}$$

Where *a* and *b* are constants determined from regression analysis of the pooled reference dataset, and *s* is the number of samples. Thus "*a*" is a coefficient showing the actual value of the reference metric that would be known with a near infinite amount of sampling, and "*b*" is a gear specific coefficient accounting for improving of knowledge with increasing sample effort. As the number of samples is finite and not so high, *MVpred* would be always lower than the particular value in the ideal reference ecosystem.

Amongst the metrics, Metric 1 (species composition) is the most important, as it accounts for completeness of the recorded list of species and therefore also directly impacts Metric 2 (presence of indicator species), Metric 5 (number of estuarine resident taxa), Metric 8 (number of benthic invertebrate feeding taxa), Metric 9 (number of piscivorous taxa) and indirectly – all other metrics but Metric 3.

Following the aforementioned statistical assumptions, the predicted Metric 1 value (total number of species that may potentially be recorded for a particular gear for a particular effort) attains 95% of "catchable" species relative to the reference list after 38 hauls by 1.5 m beam trawl (b = 0.08) and 59 hauls by seine (b = 0.05) and 19 samples for fyke nets (b = 0.15, Table 21 in TW Fish UKTAG). As such the 2013-2018 data series has sufficient seine (75) and beam trawl (35) samples so not to be heavily influenced by correction factors for these gear types. The 8 fyke samples would be anticipated to achieve 73% of the species richness of the reference value for Metric 1 and is subject to the correction coefficient.

Because of the influence of correction coefficients at low sampling effort, the TFCI test scores in this report are calculated with and without fyke nets

2.1.2 TFCI code and data

For the task of this particular study, the *'tcfi.v1'* program written in R language (R Core Team, 2019) and RStudio Version 1.2.5019 was used to analyse original survey data as provided by the Estuarine & Coastal Monitoring & Assessment Service (ECMAS), Environmental Agency. The data file analysed is *'Data_Trac_Fish_FinalR_All_Years'.* The period analysed is 2013-2018 with, and without fyke net data.

2.2 TFCI sensitivity analysis

The first step of the assessment calculates the baseline TFCI with and without fyke net samples. Once the baseline TFCI has been established, the sensitivity of the multi-metric to changes in smelt abundance can be tested. The original Environment Agency concern relates to *"significant reductions"* or complete *"removal"* of smelt from the system (Section 1.1.1). As such, a range of scenarios were considered for each test:

1. **Baseline conditions:** Smelt numbers maintained as observed.

2. **25% decrease in abundance:** Smelt numbers in all samples where they are present are reduced by 25%, rounded down to the nearest whole value, so stations where only one fish was captured are considered as not recording any smelt (0.75 of a smelt = no fish).

3. **50% decrease in abundance:** Smelt numbers in all samples where they are present are reduced by 50% rounded down to the nearest whole value, so stations where only one fish was captured are considered as not recording any smelt (0.5 of smelt = no fish).

4. **Single annual presence:** A single smelt was captured by seine netting each year during the test period.

5. **Smelt Absent:** No smelt recorded in any sample across the six-year test period.

In 2013 and 2016 no smelt were captured (Table 2). To prevent artificially inflating the TFCI 'single annual presence measures did not add smelt to these years.

In 2018, only 1 smelt was captured, as such 25% and 50% reductions result in smelt being absent from that survey year.

To address the Environment Agency concern detailed in Section 1.1.4 a single multispecies reduction was considered with and without fyke nets:

6. **Multispecies reduction:** A hypothetical scenario whereby smelt, and shad were absent, and herring and seabass reduced by 50% during the 6-year assessment period.

Finally, the TFCI sensitivity to decreased numbers of other important species was also tested in four variants, in this case fyke nets were excluded:

7. **Sole absent**. No sole recorded in any sample across the test period.

8. **Thin-lipped grey mullet absent**. No mullet recorded in any sample across the test period.

9. **50% decrease in multispecies abundance:** Smelt, sole and thin-lipped mullet numbers simultaneously reduced by 50%.

Reducing the number of individuals of commonly occurring species by orders of 25% or more in the Alde & Ore due to the proposed development is considered an unrealistic scenario. Section 4.3.2 summarises results from a local effects simulation. With full mitigation Sizewell C acting in combination with Sizewell B is expected to result in 3% or less local depletion of sensitive pelagic species such as smelt at the scale of the Greater Sizewell Bay and tidal excursion (BEEMS SPP103). The Alde-Ore Estuary is distant from the area of localised depletion and as such any effects would be diluted further still.

3 Results

3.1 Baseline TFCI with and without fyke nets

The first step of the assessment determined the EQR and status of the water body with and without fyke nets. The TFCI assessment approaches calculate the EQR and confidence of class based on a method of bootstrapping, whereby samples for each gear type are permutated 1,000 times. The EQR is also provided as a weighted mean between the gear types. The results in the tables provide the bootstrapped values based, the weighted average EQR is also provided for context.

When fyke nets are removed from the TFCI calculation for the six-year period from 2013-2018 the baseline EQR is 0.69 equating to 'good' status. The associated confidence of class for good status is 0.90 showing a strong association to the classification of good status.

When fyke nets are included in the TFCI calculation, the EQR drops dramatically to 0.56 equating to a classification of 'moderate' status and is consistent with the EQR attained by the Environment Agency for their 2019 classification. Despite the bootstrapped EQR falling to 0.56 coincident with 'moderate' status, the confidence of class for 'good' status or above was 0.69. It is noteworthy that the weighted mean EQR, which puts less emphasis on the fyke net samples, remains within the boundaries of 'good' at 0.63.

This dramatic drop in the baseline EQR and associated status demonstrates the sensitivity of the TFCI in this specific case to survey gears, statistical correction factors, and the method used to compute the EQR.

Table 4. Baseline TFCI scores using 6 years of survey data with and without fyke nets.

Year	2013-2018 without fyke nets	2013-2018 with fyke nets
Baseline EQR (weighted mean in parentheses)	0.69 ± 0.041 (0.66)	0.56 ± 0.044 (0.63)
Status	Good	Moderate
Probability of 'Good' status (CofC G)	0.901	0.688
Combined probability of either 'Good' or 'High' status	0.920	0.698

3.2 Smelt manipulations with and without fyke nets

Survey data were manipulated to reduce the numbers of smelt as described in Section 2.2, to determine the sensitivity of the TFCI. Sensitivity analyses were completed with and without fyke nets.

3.2.1 25% reduction in smelt abundance

Testing the TFCI sensitivity to a 25% reduction in smelt abundance in samples showed a negligible effect in the case of the calculation with no fyke nets. The EQR remained at 'good' status.

A minimal -2.7% effect on the calculated EQR value was observed in for the TFCI with fyke nets compared to the baseline (Table 5), in each case the status was 'moderate'.

3.2.2 50% reduction in smelt abundance

Testing the TFCI sensitivity to a 50% reduction in smelt abundance in samples again showed a negligible effect in the case of the calculation with no fyke nets and the status remaining good.

In the TFCI calculated using fyke nets a minimal -2.5% effect on the calculated EQR value compared to the baseline was observed (Table 5).

3.2.3 Single annual presence

When the TFCI data was manipulated such that only a single smelt was recorded per annum in seine net samples the EQR score reduced by 7.4% to 0.64 in the TFCI with no fyke nets. However, the status remained 'good' with a confidence of class for good status of 0.84.

In the case of the TFCI calculated with fyke nets the EQR dropped by a similar percentage (6.9%) to 0.52 remaining in the 'moderate' classification (Table 5).

3.2.4 Smelt absence

When total loss of smelt from all samples was simulated the EQR in the case of no fyke nets fell 11%. However, the EQR value remained within the 'good' status boundary with a confidence of class for good of 0.68.

In the case of the TFCI calculated with fyke nets the EQR also dropped by 11% to 0.50 when smelt were removed from all six years of sample data, the status remained at 'moderate' (Table 5).

The 11% reduction is EQR mirrors the Environment Agency statement about the removal of smelt, however, despite the reduction in EQR the status remains the same in both cases with or without fyke nets.

In this worst-case smelt scenario (total absence) the weighted mean EQR calculated with fyke nets was 0.56 reflecting a 11% reduction from the baseline (0.63) and a deterioration into 'moderate' status, whereas the weighted mean EQR without fyke nets fell from 0.66 to 0.59 i.e. 11.9% reduction but remained as 'good' status boundaries (Table 5).

In summary total removal of smelt only caused a change in the status of the water body in the case whereby fyke nets are included and a weighted mean EQR is applied.

Table 5. Sensitivity of the TFCI to changes in smelt abundance with tests calculated using a 6 yeas of survey data with and without fyke nets.

Year	**2013-2018 without fyke nets**	**2013-2018 with fyke nets**
Baseline EQR (weighted mean in parentheses)	0.69 ± 0.041 (0.66)	0.56 ± 0.044 (0.63)
Status	Good	Moderate
Probability of 'Good' status (CofC G)	0.901	0.688
Combined probability of either 'Good' or 'High' status	0.920	0.698
25% decrease in abundance	0.694±0.040	0.547±0.045
Status (and % decline from baseline)	**Good (+0.2%)**	**Moderate (-2.7%)**
Probability of 'Good' status (CofC G)	0.901	0.669
Combined probability of either 'Good' or 'High' status	0.916	0.678
50% decrease in abundance	0.691±0.041	0.548±0.044
Status (and % decline from baseline)	**Good (-0.2%)**	**Moderate (-2.5%)**
Probability of 'Good' status (CofC G)	0.902	0.615

Year	2013-2018 without fyke nets	2013-2018 with fyke nets
Combined probability of either 'Good' or 'High' status	0.915	0.625
Single annual presence	0.641±0.041	0.523±0.044
Status (and % decline from baseline)	**Good (-7.4%)**	**Moderate (-6.9%)**
Probability of 'Good' status (CofC G)	0.838	0.633
Combined probability of either 'Good' or 'High' status	0.838	0.633
Total absence (weighted mean in parentheses)	0.616±0.040 (0.588)	0.501±0.042 (0.558)
Status (and % decline from baseline)	**Good (-11.0%)**	**Moderate (-10.9%)**
Probability of 'Good' status (CofC G)	0.675	0.504
Combined probability of either 'Good' or 'High' status	0.675	0.504

3.3 Sensitivity of the TFCI to a multispecies manipulation

To address the Environment Agency concern regarding a reduction in multiple species of importance to the Alde & Ore classification a hypothetical scenario was simulated. The scenario involved the removal of all 221 smelt from samples, removal of the single shad specimen and 50% reductions in herring and seabass during the 6-year assessment period. This scenario was assessed with and without fyke net samples included (one of the two specimens caught by fyke nets was a seabass, as such this halves the total fyke net catches; Table 3).

The multispecies manipulation resulted in a 10% reduction in the EQR in the calculation without fyke nets but the EQR value remained within the 'good' status boundary with a confidence of class for good of 0.76 (Table 6). At first sight it may appear strange the reduction in EQR is less than in the absence of smelt alone. This is in part due to the fact that more individuals of a given species does not necessarily infer better status. In this case herring account for 47% of the total species contribution. As such, reducing herring by 50% increases the number of taxa that contribute to the top 90% (Metric 4; Table 1) thereby increasing the score for that metric. The same is true but to a lesser degree for Metric 7 (Functional guild composition).

In the case of the TFCI calculated without fyke nets the EQR also dropped by 7.7% to 0.52, the status remained at 'moderate' (Table 6). In this hypothetical scenario the weighted mean EQR calculated with and without fyke nets was 0.58 on the boundary of moderate/good status, and 0.61 'good' respectively.

Given that a hypothetical scenario of a 50% reduction in seabass and herring and absence of smelt and shad did not reduce the water body classification, no further assessments were made in relation to concerns outlined in Section 1.1.4.

Table 6. Sensitivity of the TFCI to multispecies changes in abundance with tests calculated using a 6 yeas of survey data with and without fyke nets.

Year	2013-2018 without fyke nets	2013-2018 with fyke nets
Multispecies manipulation: absence of smelt and twaite shad, 50% reduction in numbers of herring and bass. (weighted mean in parentheses)	0.621±0.036 (0.606)	0.519±0.039 (0.575)

Year	2013-2018 without fyke nets	2013-2018 with fyke nets
Status	Good (-10.3%)	Moderate (-7.7%)
Probability of 'Good' status (CofC G)	0.759	0.588
Combined probability of either 'Good' or 'High' status	0.759	0.588

3.4 Sensitivity of the TFCI to manipulations in grey mullet and Dover sole

The sensitivity of the TFCI was tested further through manipulated removals of thin-lipped grey mullet and sole, as well as a scenario whereby smelt, thin-lipped mullet and sole were all reduced by 50%. These manipulations were only calculated without fyke nets included.

The total absence of thin-lipped mullet resulted in an approximate 4% decline in the six-year EQR value. However, the status remained 'good' with a confidence of class for good status of over 90% (Table 7).

The total absence of sole resulted in an approximate 3% decline in the six-year EQR value. Again, the status remained 'good' with a confidence of class for good status of 92% (Table 7).

In the case where all three species of smelt, thin-lipped mullet and sole were reduced by 50% across all samples in all years, the EQR was reduced by approximately 3% from the baseline. **The status remained 'good' with an 80% confidence of class for good or high status** (Table 7).

Table 7. Sensitivity of the TFCI to changes in smelt abundance with tests calculated using a 6 yeas of survey data with and without fyke nets.

Year	2013-2018 without fyke nets
Baseline conditions	0.692±0.041
Status	Good
Probability of 'Good' status (CofC G)	0.901
Combined probability of either 'Good' or 'High' status	0.920
Complete absence of sole	0.669±0.036
Status (and % decline from baseline)	Good (-3.3%)
Probability of 'Good' status (CofC G)	0.920
Combined probability of either 'Good' or 'High' status	0.921
Complete absence of thin-lipped mullet	0.666±0.36
Status (and % decline from baseline)	Good (-3.8%)
Probability of 'Good' status (CofC G)	0.919
Combined probability of either 'Good' or 'High' status	0.919
50% decrease in smelt, sole and mullet	0.672±0.044
Status (and % decline from baseline)	Good (-2.9%)

Year	2013-2018 without fyke nets
Probability of 'Good' status (CofC G)	0.764
Combined probability of either 'Good' or 'High' status	0.801

4 Discussion

4.1 How sensitive is the Alde & Ore TFCI to inclusion of fyke net samples?

The results reported herein clearly demonstrate the effect of the sampling and statistical methods to the outcome of the TFCI calculation and therefore the status of the Alde & Ore water body. The inclusion, or not, of fyke net data in the TFCI calculation has a fundamental bearing on the outcome of the EQR and water body classification. So much so that by including fyke nets the six-year EQR (2013-2018) drops from 'good' status (0.69) by 18.8% achieving an EQR of 0.56 or 'moderate' status.

When a BQE fails to achieve 'good' status further investigations are implemented, which may include investigative monitoring. It is noted that the Environment Agency are currently undertaking such investigations on the 2019 classification of the Alde & Ore; "*The appropriateness of including the fyke net method in the 2019 classification is being investigated."*

Fyke nets were deployed only twice in 2013 and 6 times in 2014 (Figure 1). In total four samples were deployed at Shingle Street (Table 3), which is subject to shingle movement and is a challenging location to deploy fyke nets (S. Coates pers. comms). Based on only 8 samples fyke nets would be expected to reveal 73% of faunistic diversity (Metric 1) based on the reference list for this gear type (Section 2.1.1.3). However, the 8 deployments resulted in just two fish, a single bass and a single ballan wrasse (Table 3), captured in 2013. The small sample size and low catch means the EQR calculated for fyke nets is subject to two statistical issues:

Firstly, a correction coefficient is applied to account for sample size. The correction coefficients result in changes to the observed value. The correction coefficient does not account for existing habitat and faunistic differences between estuaries. The Environment Agency (2011) states that *"… the adjustment factors derived in our study are highly uncertain and relate only to particular ecotypes. Furthermore, employing different monitoring strategies in different estuaries not only biases the observed EQR, but also affects the precision of the observed EQR, which cannot easily be eliminated using adjustment factors.*

Secondly, and potentially of greater concern in this specific instance is the use of bootstrapping to calculate the EQR. The bootstrapping method permutes the samples 1,000 times to determine the mean result and confidence intervals. However, with just two samples from 8 containing fish, this approach is subject to statistical violations. This point is well illustrated when the 2015 (2013-2015 data including fyke nets) and 2016 classification (2013-2016 data including fyke nets) is recalculated using bootstrapping methods resulting in the EQRs falling to 0.53 and 0.50, respectively. The reported 2015 and 2016 EQRs of 0.63 and 0.61 ('good' status) including fyke nets can be replicated based on the weighted mean EQR result which puts less emphasis on fyke samples. It is noteworthy that when the 2019 EQR is computed with fyke nets using the weighted mean the EQR is 0.63, indicative of 'good' status.

The Environment Agency indicated in June 2020 that the inclusion of fyke nets in the 2019 classification was being reviewed and provided clarification that an investigation will be undertaken to determine the apparent deterioration in the water body status from 'good' to 'moderate'. The question becomes, should fyke net data be included in the 2019 classification?

Fyke nets are not highly species-specific, and both scientific and commercial variants capture a variety of fish in saline habitats (Bergström *et al.,* 2013; Seafish, 2020). In the Environment Agency (2011) report 'Quantifying uncertainty in the Transitional Fish Classification Index tool' the sole use of fyke nets to calculate the TFCI typically results in a 20-50% reduction in EQR. When fyke nets are included with beam trawl and seine nets, the EQRs are typically reduced by a factor of 4-15% (Environment Agency, 2011).

However, previous TFCI data from the Alde & Ore in 2005 and 2006 indicates that fyke nets have been successfully deployed and in the case of European eel have had the greatest capture rates. 2005 was the first year the water body was monitored. Ten fyke net samples in 2005 and 2006 recorded a total of 91 individuals from 11 species including European eel, seabass, cod, dab, plaice, sand goby, sprat, pouting/bib, flounder Dover sole and lesser weaver from 10 samples. Deployed correctly, fykes can add to the species contribution. In 2013 and 2014, 8 further samples caught only two more individuals; a single seabass and a new species, the ballan wrasse (Table 8). With the exception of cod and dab the other 9 species that were captured by fyke nets in 2005 and 2006 were sampled in seine nets and beam trawls within the Alde-Ore Estuary in 2013 and 2014, indicating they were present. Therefore, the low fyke net catches may have resulted from mis-deployment or simply the position within the estuary was not suited for their use. In the Solway Estuary, for example, fyke nets were scoured into the bed and tended to fill up with sand and silt (Environment Agency, 2011).

The Environment Agency (2011) report concluded that *"sensitivity analysis showed that the observed EQR is influenced by the combination of gear types deployed: Seine nets tended to be the most efficient way of sampling the fish community, yielding a higher EQR for a given number of samples than beam trawls or fyke nets,* ***while these last appear to be redundant when seine nets and beam trawls are also deployed****"* (Environment Agency, 2011). It appears evident that the fyke samples collected in 2013 and 2014 add very little or no information to the TFCI and artificially deflate the EQR. This statistical artefact is exacerbated by applying bootstrapping rather than a weighted average approach to determine the EQR. It seems logical that if fyke nets are to be included in the 2013-2018 TFCI classification, the EQR should be calculated by weighted averages of the EQR for each gear type. In such a case, the water body remains as 'good' status with an EQR of 0.63 (Table 4). However, this still represents a notable reduction to the EQR of 0.69 calculated by bootstrapping, or the EQR of 0.66 using the weighted average score if fyke nets are not included (Table 4).

Table 8. Environment Agency fyke net samples collected within the Alde & Ore.

Year	Season	Site	Samples	Species	Total count
2005	Autumn	Boyton Dock	2	7	25
2005	Spring	RSPB Reserve	4	3	6
2006	Autumn	Shingle Street - Lower	2	4	12
2006	Spring	Shingle Street - Lower	2	6	48
2013	Spring	Shingle Street - Lower	2	2	2
2014	Autumn	Boyton Dock	2	0	0
2014	Autumn	Shingle Street - Lower	2	0	0
2014	Spring	Boyton Dock	2	0	0
Total			**18**	**12 (unique)**	**93**

4.2 Sensitive of the Alde & Ore TFCI to smelt and other species

In response to Environment Agency concerns the sensitivity of the TFCI was tested against te changes in fish abundance including total absence of Dover sole, thin-lipped grey mullet, and smelt. An additional test also considered absence of smelt and Twaite shad and 50% reductions in herring and seabass (multispecies manipulation). In all cases tested, the EQR declined with the greatest reductions resulting from total smelt removal -11%, and in the multispecies manipulation (-10.3%) in the calculation with no fyke nets. In this case the multispecies manipulation produced a higher EQR than the removal of smelt alone. This is in part due to herring accounting for 47% of the total species contribution. As such, reducing herring by 50% increases the

number of taxa that contribute to the top 90% (Metric 4; Table 1) thereby increasing the score for that metric. The same is true but to a lesser degree for Metric 7 (Functional guild composition).

The Alde & Ore TFCI has very limited sensitivity to smelt density in samples. Complete removal of smelt from all samples did reduce the EQR by 11% but had no effect on the water body status when fyke nets are not included in the assessment and the status remains as 'good'.

It is important to make the distinction between a reduction in the population of fish versus the ability of sampling to detect changes in the population. In the case of smelt, a high degree of interannual variability in the Environment Agency catch rates in the Alde & Ore is observed. Such variability would be expected with shoaling species. Numbers of smelt sampled ranged from 0 to 132 individuals per annum despite similar sampling effort (Table 2). Interannual variability in smelt numbers is observed in impingement estimations from 2009 to 2019 recorded during impingement monitoring at Sizewell B (Figure 2). The mean interannual variability (2009-2019, noting the absence of 2013 data) was a factor of 2.5 (maximum of 4.4), indicating natural interannual fluctuations in the mean predicted abundance. However, despite interannual variability in impingement rates at Sizewell B, there is no trend in the mean number of fish impinged over time (n = 10, tau = -0.02, p = 1, Theil-Sen slope = -41 [3]). There is also no statistically significant correlation between annual numbers impinged at Sizewell B and numbers sampled in TFCI surveys (Pearson's product-moment correlation, r = 0.507, p = 0.383).

It is likely that the variability in the numbers of smelt sampled during TFCI surveys arises from the probability or not of encountering a school of smelt. Across all survey gears and sampling years 206 smelt were captured in 12 out of 75 seine samples (mean 17.2, ± 22.1 s.d). This level of variability indicates the survey methods are sensitive to the behaviour and spatial/temporal distribution of smelt within the estuary rather than changes in population size. This variability is demonstrated when the mean numbers of smelt captured in seine samples is permuted with bootstrapping bias-corrected accelerated percentile (Bca), bootstrapped 10,000 times); Figure 2 presents the mean and upper and lower 95th confidence intervals of smelt caught in TFCI seine samples each year.

The water body typology and variations in freshwater inputs may influence smelt catches in the Alde-Ore. The Alde & Ore is Type 4 transitional water body characterised a fully mixed, polyhaline or euhaline, and mesotidal. Smelt favour macrotidal systems with large freshwater inputs providing a signal to spawn. Anthropogenic pressures of drought conditions could limit smelt occurrence in the water body (S. Coates pers. comm.).

Changes in smelt numbers alone, either real through changes in abundance or perceived through sampling not detecting these species, are highly unlikely to cause a deterioration in the status of the Alde & Ore water body. The following section considers the coastal and estuarine behaviour of smelt and the potential for the proposed development to effect Alde & Ore smelt populations.

[3] Smelt statistics report the Kendall's tau statistic, which measures the rank correlation between impingement and year (with a range from -1 to +1, with 0 when there is no correlation), and the 2-sided p-value from the Mann-Kendall test. The Theil-Sen slope estimate is a median slope estimate for a linear change in impingement per year.

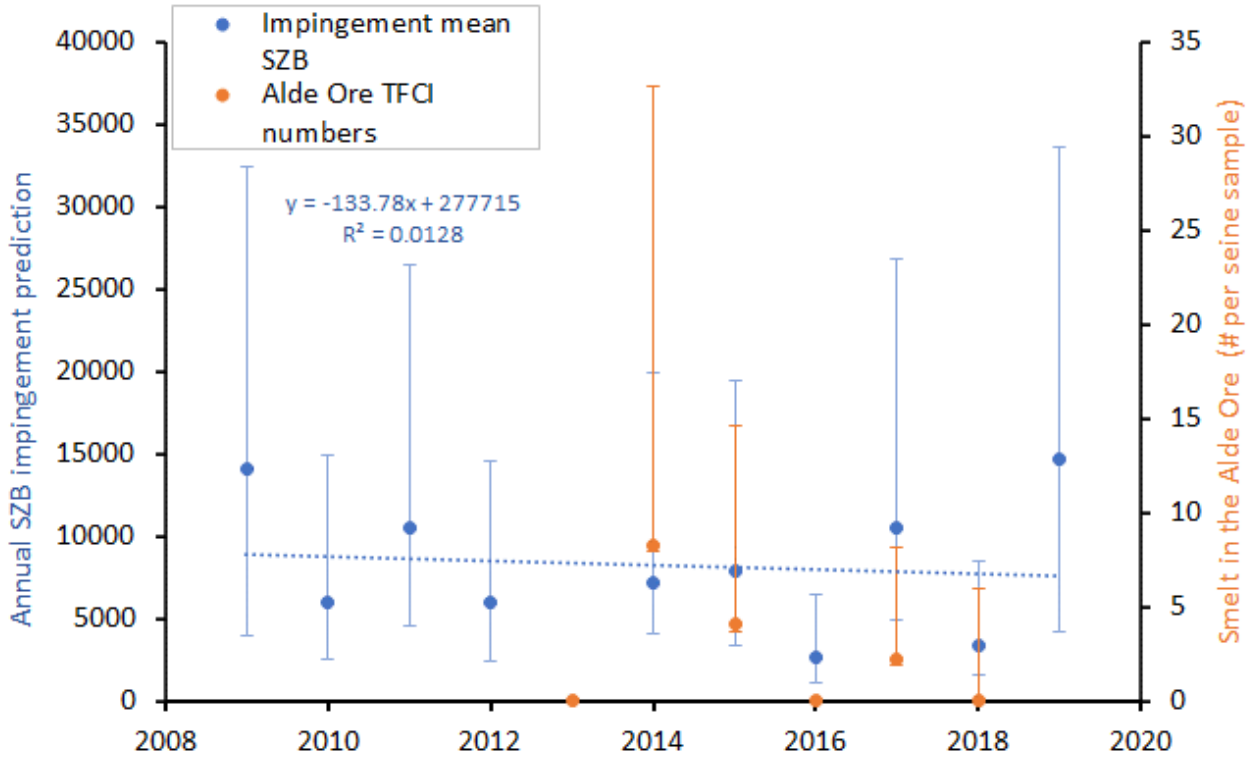


Figure 2. Smelt, *Osmerus eperlanus*, annual impingement (mean ± 95th upper and lower percentile) at SZB over the years 2009-2019 (note: 2018 and 2019 results are provisional) in blue, and mean number of smelt captured during annual Alde & Ore surveys with bootstrapped 95th percentile confidence intervals in orange (survey effort is provided in Table 2).

4.3 What are the potential effects of the proposed development on Alde & Ore smelt?

Smelt migrations within estuaries are not well studied in the UK except for the population that reproduces in the River Cree which drains into the Solway Firth in Scotland. In that population, prior to the spawning run, adult smelt form pre-spawning shoals in the estuary from October/November. The fish then move into the lower reaches of the Cree to spawn over a few days from February to March. After hatching, fry are swept down from the freshwater environment to the brackish upper estuary where they feed on zooplankton (Galloway Fisheries Trust, 2018). Juvenile smelt use estuaries as nursery grounds before moving to outer estuaries in their first winter (in December/January in the lower Thames at Thurrock; Power and Attrill, 2007) and subsequently to sea for summer feeding.

The proposed development, acting in-combination with Sizewell B, has the potential to impact smelt abundance in the Alde & Ore through two main impact pathways:

- **impingement** of smelt during summer feeding at sea or during migration past the station to the estuary 25km to the south; and
- **thermal occlusion** potentially affecting migration into the estuary or movements past the power station.

4.3.1 Thermal occlusion

Existing thermal standards for transitional waters specify that an estuary's cross section should not have an area larger than 25% with a temperature uplift above 2°C, for more than 5% of the time (95th percentile). During the operation of Sizewell C and Sizewell B, excess temperatures at the mouth of the Alde-Ore Estuary are ≤1°C as a 98th percentile (BEEMS Technical Report TR302). Therefore, there would be no thermal occlusion of the mouth of the Alde-Ore.

Unlike in estuarine systems, thermal standards for occlusion along open coastal systems do not exist. However, if fish have to pass through a coastal plume on their migration route to or from an estuary there is a possibility of the plume acting as a barrier to migration depending upon the behaviour of the fish and their migration routes. As a precautionary stance a hypothetical migration corridor 3km wide from the coast to the Sizewell C outfalls was considered as a potential migration route and estuarine thresholds were applied. Uplifts of 2°C are precited to exceed 25% of the corridor for 18.7% of the year when both stations are operating, with the main effect resulting from the existing Sizewell B station. Temperatures of 3°C exceed 25% of the corridor for 1.5% of the year. As an additional step the period of smelt migration into freshwater was considered in February to April migratory period when both Sizewell C and Sizewell B are in operation. During this period temperatures of 3°C exceed 25% of the cross-sectional area of the coastal corridor for less than 5% of the time.

In response to specific Environment Agency requests, BEEMS SPP101 'Implications of tidal elevation and temperature on smelt, *Osmerus eperlanus*, impingement at Sizewell' determined the length-frequency distribution of impinged smelt at Sizewell B in relation to seasonality, tidal state and the modelled thermal plume (for the year 2009). Over the five years investigated in detail (2009, 2014, 2015, 2016 and 2017) only 5.4% of all smelt impinged were caught during the February to April migratory period (131 samples)[4]. In comparison, the warmer summer months of May to September when smelt are feeding in the coastal waters, accounted for 85.2% of the total smelt impinged (344 samples). Neither tidal state nor elevation significantly influenced impingement rates of smelt. There was a seasonal trend in the length distribution of smelt impinged with fewer but larger smelt caught in February to April and higher densities of younger fish caught in the summer (BEEMS SPP101).

Impingement monitoring showed that smelt did not avoid the area of the intakes when modelling predicted water temperatures were at their maximum. Between 13 – 28.4%[5] of smelt were impinged when absolute water temperatures at the seabed exceeded 21.5˚C despite this occurring less than 3% of the time at the position of the intakes. Within the temperature range predicted in the location of the intakes, no effect of excess temperature on impingement was found, up to a 2.5°C uplift[6] at the bottom of the water column (indicative of the intake position) and up to 4°C at the surface. Furthermore, there was no relationship between thermal uplifts and the size of smelt impinged. The data did not provide any evidence of thermal avoidance, within the temperature ranges observed (BEEMS SPP101). Figure 1 shows the contour for the 2.5°C thermal uplift, as a 98th percentile bottom temperature, for both stations operating is well over 10km distant from the mouth of the Alde-Ore Estuary.

It is therefore very unlikely that the thermal plumes caused by Sizewell C and Sizewell B operating would result in thermal occlusion across a hypothetical coastal corridor. The potential for thermal occlusion at Sizewell, 25km north of the Ade Ore estuary mouth influencing fish migrating to the estuary is extremely remote. In the case of the other migratory species the Environment Agency raised concerns over; Dover sole[7] and thin-lipped grey mullet, the fact that these species are impinged by Sizewell B when the intakes are in relative proximity to the outfalls suggests that thermal occlusion of fish transiting past the station on route to the Alde-Ore Estuary is remote. Furthermore, even removal of these species from the TFCI calculation only resulted in a 3-4% reduction in the EQR (Table 7) and as such no deterioration in the status of the water body.

4.3.2 Impingement effects on smelt in the Alde & Ore

The Environment Agency raised the concern that *"SZC impingement is presented as being about the same as SZB - this means that when both power stations are operating, impingement rates are considered to be almost double what they are now*".

[4] It is likely that spawning adults would have congregated in the estuary mouth prior to the migratory period when they may be migrating up estuary (Section 4.1 and Galloway Fisheries Trust, 2018).
[5] The range in values accounts for the different time offset considered for fish being abstracted at the intakes and transiting to the forebay.
[6] Periods where excess water temperatures exceed 3°C at the seabed are rare (1.7% per annum) and did not correspond to impingement monitoring sampling dates.
[7] Within the TFCI Dover sole are marine juveniles, not migrants (S. Coates pers. comms.).

BEEMS Technical Report TR406.v7 (November 2020) estimates very similar impingement rates for Sizewell C (7,424 smelt per annum[8]) and Sizewell B (8,122). Sizewell B has been operational since 1995 and forms part of the baseline against which stock level effects are assessed and for establishment of the WFD fish status i.e. any fish mortality due to Sizewell B already exists as part of the baseline against which the TFCI metric for the Alde & Ore is derived. The effects of Sizewell C would represent an additional pressure to that baseline.

Colclough and Coates (2013) concluded that the smelt found in the Wash are probably from a common stock which may access some or all of the tributaries that flow into the Wash, and Maitland (2003) reported that it is likely that stocks in Suffolk belonged to a population associated with the Norfolk Broads and the estuarine and brackish waters around Great Yarmouth and Lowestoft. More recent genetic analysis of 215 smelt collected from the Sizewell B CIMP programme and from the Thames, Waveney, Great Ouse and Tamar estuaries showed that East Anglian smelt are genetically homogeneous with no genetic structuring seen within the region. Smelt from the Tamar were clearly distinct from the East Anglian collections (BEEMS Technical Report TR423).

Given the genetic information of smelt at Sizewell, it is probable that the smelt impinged are from multiple locations on the east coast of the UK and, based on the comparable distances, from European estuaries of at least the Scheldt (Belgium) and the Elbe in Germany (this hypothesis is considered reasonable but it is recognised that it has not yet been proven). Following the extinction of smelt in the Thames in (Wheeler 1979) recovery of Thames smelt is considered most likely from the Medway demonstrating elasticity of the species in its inshore coastal and outer estuarine behaviour (S. Coates pers. comms).

Taking a conservative stance that all smelt impinged at Sizewell are solely from a discrete ‘Anglian’ stock sub-population, EAV corrected losses can be compared to the restricted regional fishery with a mean of 8t per annum (>11t in 2012) as 1.11%. The effects on the conservatively estimated SSB is therefore in the order of 0.18% (BEEMS Technical Report TR406). The proposed development, acting in-combination with Sizewell B, would have a negligible effect on regional smelt stocks.

Furthermore, local effects simulations have been completed at the scale of the Greater Sizewell Bay (GSB) and tidal excursion, and the coastally truncated ICES Statistical Rectangle 33F1 which extends from Lowestoft to the north of Sizewell to just above Felixstowe to the south. The simplified model is not appropriate for determining sustainability of fish stocks, which follows ICES best practice. However, by applying a set of simplifying assumptions regarding fish distribution and behaviour it is possible to provide a contextualised assessment of local depletion in relation to pre-defined spatial scales, in this case, the smallest unit incorporating the GSB and tidal excursion where local depletion would be greatest. With both stations operating together, local depletion reaches a plateau after approximately 50 days, after which point abstraction losses and emigration are balanced by immigration into the local area. The conceptual model considers daily depletion of fish in relation to replenishment from the wider area. With full mitigation and daily replenishment rates equivalent to tidal exchange (10%), Sizewell C acting in combination with Sizewell B may result in local depletion in the GSB and tidal excursion of approximately 1.5% for sprat and herring, and 0.9% for species such as Dover sole (BEEMS SPP103). Juvenile smelt are impinged at Sizewell B throughout the summer months. Assuming 10% exchange of fish with the wider area each day, depletion of smelt in the GSB and tidal excursion is predicted to be 1.52%, even with low rates of exchange (5% per day) the local depletion by the end of the summer period would remain at just 3.1% (BEEMS SPP103). The Alde-Ore Estuary is distant from the area of localised depletion and as such any effects would be diluted further still to 0.08% in ICES rectangle 33F1 which includes the mouth of the Alde-Ore Estuary. Accordingly, the >25% reduction in fish abundance applied here for sensitivity testing of the Alde & Ore water body TFCI due to the additional impacts of Sizewell C is unrealistically high, particularly noting Sizewell B impacts are already within the baseline and the Alde & Ore water body.

The lack of evidence for thermal occlusion either at the mouth of the Alde-Ore Estuary or across a hypothetical corridor at the frontage of Sizewell C, some 25km to the north, and the prediction for negligible effects on regional smelt stocks indicates that the proposed development would not result in significant reductions in smelt abundance entering the estuary. No trend in smelt impingement rates at Sizewell B has been observed for the decade 2009-2019, indicating a stable population that displays interannual variability.

[8] Total smelt impingement accounting for mitigation but not EAV equivalent.

During the same time period (2013-2018) the number of smelt caught during WFD sampling in the Alde & Ore has fluctuated Figure 2. Given that total smelt absent from all 6 years of TFCI samples did not cause a deterioration of Alde & Ore 'good' fish status (when fyke samples are excluded), it is considered highly unlikely that the Alde & Ore fish EQR would be sensitive to impacts from the proposed development on smelt.

5 Summary

The status of the fish community in the Alde & Ore was classified as 'Good' in 2015 (EQR: 0.63) and 'Good' in 2016 (EQR: 0.61). Derivation of these EQR values is assumed to be based on 3 years of data in 2015 (2013-2015) and based on 4 years of data in 2016 (2013-2016). During this period seine nets and beam trawls were the predominant sampling method but fyke nets were also deployed and used in the EQR calculation. No classification was produced in 2017 or 2018. The recently published 2019 results show the water body transitional fish status apparently fell to 'Moderate' in 2019 (Environment Agency Catchment data Explorer, 2020). The 2019 EQR provided by the Environment Agency is 0.56 and was calculated using a combination of seine nets, beam trawls and fyke nets using data from 2013-2018.

This report calculates the TFCI baseline Ecological Quality Ratio (EQR) by applying Environment Agency fish monitoring data for the Alde & Ore water body. The baseline EQR is calculated using six years TFCI sampling data from 2013 to 2018. Sampling data from the period includes 75 seine nets, 34 beam trawls and 8 fyke nets (fykes sampled in 2013 and 2014 only). Seine samples returned 26 species and the highest catches, whilst beam trawls caught 21 species in total. Fyke nets only caught a single seabass and one ballan wrasse.

The Environment Agency algorithms used to derive EQR values now incorporate bootstrapping approaches. The bootstrapping method permutes samples from each gear type 1,000 times to determine the mean result and confidence intervals that are averaged between gears to calculate an overall EQR. By putting equal emphasis on all sample gears, the bootstrapping approach differs from the weighted mean approach and produces different results particularly when sample numbers and catch efficiency is low. It is noteworthy that if the 2019 EQR is calculated with fyke nets using the weighted mean approach, as in 2015 (0.63) and 2016 (0.61), the water body status is 'good' (EQR 0.63). Conversely if the 2015 and 2016 data is recalculated using the same bootstrapping method applied in 2019, the status falls to 'moderate' with EQRs of just 0.53 and 0.50, respectively.

This report was first commissioned prior to the release of the 2019 classification to determine the potential for smelt reductions to cause a deterioration in the status of the Alde & Ore water body. It has evolved following comments from the Environment Agency and in response to the latest water body classification to look at two aspects of the Alde & Ore TFCI:

I. the appropriateness of using fyke nets in the calculated Alde & Ore EQR, and

II. the sensitivity of the TFCI to reductions in fish abundance.

Calculations of the EQR for Alde & Ore water body were carried out using data provided by the Environment Agency and using their analysis software (R-code).

Cefas identified the following conclusions, which are supported following technical review by Steve Coates of SLR Consulting who was one of the original authors of the Transitional Fish Classification Index methodology and associated sensitivity analysis.

5.1 Effect of Fyke net data on the TFCI results

1. Recalculaton of the 2019 water body classification using data from 2013-2018 without the use of fyke net data, results in an Alde & Ore EQR of 0.69 representing 'good' status. The confidence of class for good status or above is 0.92.

2. Repeating the 2019 calculation but with inclusion of the 8 fyke net sample results in the Alde & Ore EQR falling by 18.8% to 0.56 corresponding to 'moderate' status.

3. In this situation, the TFCI calculation and status of the Alde & Ore water body is compromised by low sampling efficiency and statistical artefacts relating to the small fyke net sample size and low catches. It is concluded that inclusion of fyke nets has caused the apparent deterioration of the water body classification to 'moderate'.

4. It is concluded that fyke net results should not be included in the Alde & Ore TFCI calculation.

5.2 Effect of reductions in fish numbers on the water body classification:

The Environment Agency has raised a number of concerns relating to the potential for the proposed Sizewell C development to impact fish species of relevance to the Ade & Ore TFCI. Following establishment of the baseline EQR with and without fyke net data, the sensitivity of the TFCI was established in response to the manipulated removal of species that the Environment Agency has identified as important for the Alde & Ore TFCI including; smelt, shad, herring, seabass, thin-lipped grey mullet and Dover sole.

To determine the sensitivity of the TFCI to smelt abundance, smelt numbers in samples were manipulated to four levels representing declines of 25%, 50%, a single smelt caught per annum (in seine samples), and total smelt absence. A further test considered absence of smelt and Twaite shad and 50% reductions in herring and seabass (multispecies manipulation). These tests were run with and without fyke nets. The sensitivity of the TFCI was also tested further through manipulated removals of thin-lipped mullet and Dover sole, as well as a scenario whereby smelt, thin-lipped mullet and sole were all simultaneously reduced by 50%. These manipulations were calculated with no fyke nets included.

5. The EQR was insensitive to manipulated reductions in smelt abundance of 25% and 50%. Such reductions are considerably beyond the expected local level effects of the proposed Sizewell C development (BEEMS SPP103).

6. Total absence of smelt reduced the EQR by 11% but 'good' status remained. (Fyke net data removed from the TFCI calculation)

7. The Environment Agency requested a scenario of total absence of both shad (1 individual caught between 2013-2018) and smelt along with 50% reductions in herring and bass. In this extreme scenario the EQR was reduced by 10.3%, however, 'good' status remains (TFCI calculated without the inclusion of fyke net data).

8. Total absence of thin-lipped grey mullet and Dover sole reduced the EQR by less than 4% in each case and 'good' status remained (TFCI calculated without the inclusion of fyke net data). The status also remained 'good' following the combined 50% reduction of smelt, Dover sole and thin-lipped grey mullet.

9. Under all of the scenarios trested for fish manipulations, there was no deterioration of 'good' status when the TFCI was calculated without fyke net data, or no deterioration below 'moderate' status when the TFCI was calculated with fyke net data.

5.3 Likelihood of thermal occlusion or SZC impingement to affect smelt in the Alde & Ore

The potential for the proposed development to affect smelt, and other migratory fish (e.g. Dover sole and grey mullet), in the Alde & Ore by means of thermal occlusion and impingement was investigated and it was concluded that:

1. There is no evidence for thermal occlusion either at the mouth of the Alde & Ore where temperatures are ≤1°C as a 98th percentile, or across a hypothetical corridor at the frontage of Sizewell C, 25km to the north of the estuary mouth.

2. Impingement from Sizewell C is predicted to have a negligible effect on regional smelt stocks (0.18% of the conservatively estimated 'Anglian' SSB) and would not result in significant local reductions in smelt abundance entering the estuary (BEEMS Technical Report TR406; BEEMS SPP103).

Based on the evidence of the Alde & Ore TFCI sensitivity assessments in this report to effects far greater than those anticipated from the proposed development, it is considered highly unlikely that the proposed development would cause a deterioration in the fish status of the Alde & Ore water body.

References

BEEMS. Implications of tidal elevation and temperature on smelt, Osmerus eperlanus, impingement at Sizewell. Scientific Position Paper SPP101. Lowestoft, UK: Cefas, 2020.

BEEMS. Consideration of potential effects on selected fish stocks at Sizewell. Scientific Position Paper SPP103. Lowestoft, UK: Cefas, 2020.

BEEMS. Sizewell C - Impingement predictions based upon specific cooling water system design, Version 6 Technical Report TR406. Lowestoft, UK: Cefas, 2020.

BEEMS. Sizewell- Equivalent Adult Value (EAV) metrics. Technical Report TR383. Lowestoft, UK: Cefas, 2020

BEEMS. Sizewell- Thermal Plume modelling GETM Stage 3 results with the preferred SZC cooling water configuration. Edition 3. Technical Report TR302. Lowestoft, UK: Cefas, 2020

BEEMS. The Origins of Smelt (*Osmerus Eperlanus*) Populations at Sizewell. Technical Report TR423. Lowestoft, UK: Cefas, 2016.

Bergström, L., Karlsson, M., Pihl, L. 2013. Comparison of gill nets and fyke nets for the status assessment of coastal fish communities. WATERS Report no. 2013:7. 55 pp. Available at: https://www.researchgate.net/publication/325813159_Comparison_of_gill_nets_and_fyke_nets_for_the_status_assessment_of_coastal_fish_communities. Accessed on 13/10/2020.

Coates, S., Waugh, A., Anwar, A., Robson, M., 2007. Efficacy of a multi-metric fish index as an analysis tool for the transitional fish component of the Water Framework Directive. Mar. Pollut. Bull. 55, 225–240.

Colclough, S., Coates, S., 2013. A review of the status of smelt Osmerus eperlanus (L.) in England and Wales – 2013. Report EA/001 to the Environment Agency, York, UK, by SC2, Chatham, Kent.

Environment Agency. 2011. Quantifying uncertainty in the Transitional Fish Classification Index tool. Report. Environment Agency. 62 pp.

Environmental Agency Catchment Data Explorer 2020. Available at: https://environment.data.gov.uk/catchment-planning/WaterBody/GB520503503800 Accessed on 16/10/2020

Galloway Fishing Trust 2018. Assessing the sparling Osmerus eperlanus (L.) population of River Cree during the “Saving The Sparling” project 2018. Commissioned Report No.: Report No. – JHWAD23. Available at: https://www.nathonjones.com/files/Sparling%202018%20reportVS.pdf

Power, M. Attril M. 2007. Temperature-dependent temporal variation in the size and growth of Thames estuary smelt Osmerus eperlanus. Marine Ecology Progress Series 330:213-222

R Core Team (2019). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

WFD-UKTAG 2014. UKTAG transitional water assessment method fish fauna transitional fish classification index. Water Framework Directive – United Kingdom Advisory Group (WFD-UKTAG), Stirling, 37 pp.

Maitland P.S. The Status of Smelt Osmerus Eperlanus in England (ENRR516). English Nature Research Reports. English Nature Research Reports. 2003.

Power, M. and J. Attrill, M.J. 2007. Temperature-dependent temporal variation in the size and growth of Thames estuary smelt (*Osmerus eperlanus*). Mar. Ecol. Prog. Ser. 330: 213–222.

Seafish 2020. Fyke net. https://www.seafish.org/responsible-sourcing/fishing-gear-database/gear/fyke-net/ Accessed on 13.10.2020.

Wheeler, A., 1979. The tidal Thames. The history of a river and its fishes. London. Routledge & Kegan Paul.