The Sizewell C Project

9.67 Quantifying Uncertainty in Entrapment Predictions for Sizewell C

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Executive Summary

NNB Generation Company (SZC) Ltd (hereafter SZC Co) 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. Fish and other biota may become drawn into the station in the abstracted cooling water. SZC has been designed with a suite of mitigation measures designed to reduce environmental impacts of abstraction and discharge of cooling water. Embedded mitigation measures to reduce impingement of fish and invertebrates at SZC include the installation of low velocity side entry (LVSE) intake heads and dedicated fish recovery and return (FRR) systems, coupled with a chlorination strategy that would prevent impinged biota being exposed to anti-fouling chemicals.

The cooling water filtration system, includes drum and band screens that are designed to protect the condensers and other essential cooling water systems from blockage. The cooling water system also includes a FRR system, whereby fish large enough to be impinged by the mesh would be returned to the marine environment via the FRR outfalls. Smaller life-history stages including eggs, larvae and juvenile fish of some species may be susceptible to entrainment, whereby they pass through the mesh and thus through the stations cooling water system to be discharged at the outfalls.

As part of the Development Consent Order (DCO) application for the operation of the new station, the effects of water abstraction on fish have been evaluated. As different life-history stages of fishes may be impinged or entrained, total losses consider both pathways and are termed entrapment. The majority of fish entrapped are expected to be juvenile stages, large numbers of which would not typically survive to adulthood even without the presence of Sizewell C. To determine the potential for population level effects due to entrapment of these predominantly juvenile life history stages, losses of these fish have been expressed as equivalent adults by calculating an equivalent adult value (EAV). These equivalent adult losses, expressed as an annual rate, can then be contextualised in relation to an annual comparator such as population size, spawning stock biomass (SSB) or fisheries landings.

When losses of equivalent adults as a percentage of spawning population size are low, the long-term risks to the population are also low. Values around one percent and lower pose very low risks to populations when they are known to tolerate higher rates of mortality from other sources. For example, in the case of commercially exploited species it has been well established that most populations can sustain annual losses of 10-20% or more of population size owing to fishing in addition to natural mortality. For species exploited by fisheries therefore, 1% annual losses pose an extremely low risk of detectable effects on population size and dynamics. If values exceed more than one or two percent, a more detailed analysis and consideration of risks is warranted.

Approach to addressing uncertainty

The assessment of impacts on fish populations provided as part of the DCO Application includes assessment of entrapment impacts with and without mitigation measures. However, statutory stakeholders have questioned the assumptions regarding the effectiveness of mitigation measures and the sensitivity of the assessment to such uncertainties. The aim of this report is to determine the sensitivity of entrapment assessments on fish populations to uncertainties in the operational performance of the proposed fish mitigation measures and uncertainties in sampling techniques. The sensitivity analysis also accounts for the natural fluctuations of fish stocks used as the comparator for losses.

The proposed LVSE intake head design is considered to provide some advantages over the current SZB intake head design in terms of reductions in fish entrapment per cubic metre (cumec) of seawater abstracted but estimates of the effectiveness of the LVSE heads have not been agreed. Acknowledging that the effectiveness of the LVSE intake heads is not certain, the sensitivity analyses in this report assumes no benefit of the LVSE heads.

The FRR system is designed to return robust species that are impinged onto the drum and band screens safely back to sea. As part of the Hinkley Point C Inquiry[1], the Environment Agency provided a technical report (Technical Brief: TB008 Fish Recovery and Return System Mortality Rates) to:

> "*set a FRR mortality rate for each species and a range around the FRR mortality rate for each species. The range set accounts for the uncertainty in the underlying evidence used to set the FRR mortality rate, and in the efficiency of the bespoke FRR system proposed for Hinkley Point C (HPC)*."

The Sizewell C project will replicate the design of HPC as much as possible. However, the reduced tidal range at Sizewell compared with Hinkley Point allows several beneficial design changes to incorporate site-specific improvements over the HPC design, meaning the SZC FRR should afford higher survival rates for impinged fishes. The Environment Agency FRR mortality ranges have therefore been applied to SZC to provide a conservative estimate for the perceived uncertainty in the FRR efficiency.

The entrainment and impingement monitoring at the operational SZB station provides a highly applicable and powerful tool to predict SZC entrapment. However, as with any sampling or monitoring programme the results must be considered in relation to the limitations within the dataset. Stakeholders have raised concerns relating to the potential for diurnal biases to be introduced into impingement predictions arising from incidences of overflowing overnight bulk samples at SZB[2]. The concern is that species more susceptible to impingement at night may be underestimated. A second concern raised by Interested Parties (IP) is the potential for an 'entrainment gap' whereby a fraction of fish, too large to be efficiently sampled by entrainment monitoring but too small to be efficiently impinged on the SZB 10mm drum screens, may be underrepresented in entrapment estimates.

To account for these uncertainties in sampling, a screening exercise was completed to determine the likelihood for the reduced proportion of overnight bulk samples relative to daylight hourly samples within the impingement monitoring data to result in under- or overestimation in 24-hour impingement predictions. In cases where underestimates were identified the data was considered in more detail. Correction factors ranging from 1.021 to 1.135 were applied to impingement estimates for four species of conservation interest where diurnal biases may have been introduced resulting in underestimates in impingement predictions. The four species were smelt, river lamprey, European eel and twaite shad. No correction factors were applied in the case of species where a greater proportion of daylight samples may have led to overestimates in impingement rates.

In the case of the entrainment gap, this report applies literature growth and mortality rates at age to back-calculate the expected numbers of fish within the body size window that corresponds to the 'entrainment gap'. This report specifically considers the potential for an entrainment gap in sprat, gobies (*Pomatoschistus* spp.) and herring and attempts to quantify the abundance of any missing size fraction. These species have been selected as they spawn in waters adjacent to Sizewell and are the three most abundant species in entrainment monitoring sampling and contribute to the top 95% of individuals in the impingement record (and because of their small bodies), gobies are most susceptible to the 'entrainment gap'. Entrainment gap numbers have been included in the entrapment uncertainty analyses.

Finally, entrapment predictions have been compared to the mean population comparator during the impingement monitoring period (2009-2017). To account for the interannual variability in the population comparator, the sensitivity assessment accounts for the variance in the comparator over the same period. This report provides further context on the uncertainty in the population estimates for twaite shad.

Statistical bootstrapping approaches have been applied to entrapment predictions relative to the baseline population allowing estimates of the mean and 95th percentile effects to be established.

Results

[1] The Hinkley Point C Water Discharge Activity (WDA) Appeal Inquiry on the Permit Variation Application Relating to Acoustic Fish Deterrent heard evidence during a 9-day hearing from 8th - 24th June 2021. Evidence included the effectiveness of mitigation measures including the FRR system.

[2] Compared with hourly samples during the day, a large single sample is taken overnight due (due to site access) overnight and on occasions this single, bulk sample overflows before its contents can be counted the following day.

The absolute annual entrapment predictions and proportions of the population comparators have been calculated. The three most commonly impinged species at Sizewell are sprat, herring and whiting, whilst gobies (*Pomatoschistus* spp.) are the most commonly entrained species. The mean annual entrapment effect for sprat is predicted to be 0.03% of the Spawning Stock Biomass (SSB, upper 95th percentile 0.04%), for herring entrapment is predicted to result in losses of 0.01% of SSB (upper 95th percentile 0.02%), and for whiting mean losses are 0.08% of SSB (upper 95th percentile 0.11%). Such losses are not significant at the population level (Table A).

Sea bass and gobies of the genus *Pomatoschistus* spp. are the only taxa where entrapment exceeds 1%.

The mean annual losses of sea bass due to SZC entrapment is predicted to be 0.99% of SSB with an upper 95th percentile estimate of 1.85%. These estimates are considered to be precautionary because sea bass are not uniformly distributed within the Greater Sizewell Bay with densities inshore of the Sizewell-Dunwich Bank, where the SZB intakes are located, higher than offshore where the SZC intakes would be located. This suggests that impingement predictions scaled-up from SZB may overestimate SZC sea bass impingement. Furthermore, the diurnal bias screening exercise indicated that bass may be more susceptible to impingement during daylight hours. The effects on sea bass are not predicted to be significant at the population level. However, to provide the highest degree of confidence in the assessment a full International Council for Exploration of the Sea (ICES) stock assessment was run for sea bass [REP8-131]. The results of the stock assessment confirmed this conclusion as it showed no discernible effects on population trends and only very minor effects on absolute SSB despite the application of highly precautionary loss estimates.

The mean annual loss of gobies (*Pomatoschistus* spp.) is 1.74% with an upper 95th percentile estimate of 1.77%. These gobies are a short lived, fast maturing, highly fecund species with high degrees of natural variability. Because gobies are a productive species with a short lifespan and early age of maturity, and because they are not fished, they will be able to sustain additional mortality rates greater than 10% of population size. The predicted level of losses of gobies (*Pomatoschistus* spp.) is not regarded as significant at the population level.

The uncertainty analysis has assumed no mitigation benefit from the LVSE intake head and considered a range of FRR effectiveness values produced by the Environment Agency for HPC (TB008). Correction factors have been applied to account for the potential diurnal bias introduced by a greater proportion of daylight samples and measures have been taken to quantify the entrainment gap for three species most likely to be subject to underestimation in entrainment predictions. The application of the entrainment gap and inclusion of correction factors to account for diurnal biases, has resulted in increases in the relative population level effects (Table A). However, in all cases where correction factors were applied due to uncertainties in the sampling procedures no material changes were observed and the conclusion of no significant population level effects due to entrapment from SZC remains.

Where residual uncertainty remains, it is necessary to consider the magnitude of such uncertainties in relation to the predicted effects, species by species, accounting for the inbuilt precaution in the entrapment assessments.

This report provides further evidence that the proposed development of Sizewell C would not have significant effects on the population sustainability of the key species assessed. That is, the size of the spawning populations increase and decrease at the same times and at almost identical rates whether or not SZC is operating.

Table A. Predicted entrapment numbers for each of the key species at SZC. Conservation species of interest in bold have been treated with a correction factor to account for diurnal biases, species underlined have been corrected for the potential 'entrainment gap'. Cells in green are below the initial 1% screening threshold. Cells in red indicate values in exceedance of the initial screening 1% threshold and are subject to further investigation.

Common name	Entrapment predictions as a % of the population Comparator				Comparator
	Lower 5%	Median	Mean	Upper 95%	
Sprat	0.016	0.027	0.028	0.043	SSB
Herring	0.008	0.012	0.012	0.017	SSB
Whiting	0.043	0.073	0.075	0.113	SSB
European sea bass	0.395	0.913	0.993	1.851	SSB
Gobies (*Pomatoschistus* spp.)	1.714	1.737	1.739	1.773	Population estimate
Dover sole	0.002	0.005	0.005	0.008	SSB
European anchovy	0.033	0.080	0.093	0.201	Landings
Dab	0.024	0.033	0.034	0.048	Landings
Thin-lipped grey mullet	0.178	0.436	0.461	0.822	Estimated SSB
Flounder	0.003	0.007	0.007	0.013	Landings
Cucumber smelt	0.332	0.542	0.572	0.918	Estimated SSB
European plaice	0.000	0.000	0.000	0.000	SSB
Atlantic cod	0.018	0.047	0.052	0.104	Landings
Thornback ray	0.117	0.223	0.232	0.376	Landings
Twaite shad	0.032	0.069	0.780*	0.218	Elbe population estimate
	2.075	4.828	9.425*	27.316	Scheldt population estimate
River lamprey	0.032	0.057	0.060	0.098	Humber population
European eel	0.131	0.201	0.209	0.308	RDB
Horse-mackerel	0.000	0.001	0.001	0.002	Landings
Mackerel	0.000	0.000	0.000	0.000	SSB
Tope	0.000	0.013	0.016	0.049	Landings
Sea Trout	0.000	0.000	0.020	0.080	Catch numbers
Sea lamprey	NA	NA	NA	NA	NA
Allis shad ‡	0.000	0.000	0.000	0.000	Population estimate

* High mean values are a statistical artefact of extreme outputs generated due to the variance in the population estimate. In such a case the median is a more reliable comparator. In the case of the Scheldt, where population recovery only occurred in 2012 effect predictions are not realistic worst-case estimates as described in Section 3.1.4. ‡ A single allis shad was impinged on the 28th May 2009 in an invalid bulk sample, meaning impingement predictions are not available for the species. However, impact assessments continue to consider the species as present and acknowledge its occurrence in the impingement record.

1 Introduction

NNB Generation Company (SZC) Ltd (hereafter SZC Co) 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. Fish and other biota may become drawn into the station in the abstracted cooling water. Biota large enough to be impinged on the fine mesh filtration systems (drum and/or band screens), that are designed to protect the condensers and other essential cooling water systems from blockage, would be returned to the marine environment via the fish recovery and return (FRR) system. Smaller life-history stages including eggs, larvae and juvenile fish of some species may be susceptible to entrainment, whereby they pass through the fine filtrations systems and passage through the stations cooling water system to be discharged at the outfalls.

As part of the Development Consent Order (DCO) application for the operation of the new station, the effects of water abstraction on fish populations have been evaluated. As different life-history stages of fish may be exposed to either impingement or entrainment, total losses include both components which is herein termed entrapment. The basis for predictions of impingement by SZC is data collected at the operational SZB station. Impingement monitoring at that station consisted of a total of 205 sample visits in the period February 2009 to March 2013, and April 2014 to October 2017 (BEEMS Technical Report TR406.v7 [AS-238]). Entrainment predictions are derived from fish and invertebrate samples from the SZB forebay, taken on 40 occasions between May 2010 and May 2011 (BEEMS Technical Report TR318 [APP-324]). Due to the extremely high natural mortality rates of the very early life-history stages of fish, impingement rather than entrainment represents the primary route of impact for most fish species at the population level.

To determine the effects of entrapment of fish, two assessment approaches have been undertaken:

1. **Population level effects:** Annual losses of equivalent adult fish due to entrapment are estimated and compared with the size of the relevant population to assess whether entrapment poses any risk to population sustainability.

2. **Local level effects:** Assessments consider the potential for the station to cause localised depletion in fish numbers at the scale of the Sizewell Bay. Local depletion assessments are independent but complement the assessment of population level effects. They can be completed for the impingement and entrainment fraction separately to assess the potential for food-web effects mediated through reductions in prey availability at the most localised scale. Local effects assessments can consider both the entrainment and impingement size fractions and are independent of EAV calculations and stock sizes.

This paper considers population level effects and the sensitivity of the predicted impacts to uncertainty in the assessment parameters. Local depletion is considered in detail in BEEMS Scientific Position Paper SPP103 (Rev 5) [REP6-016].

SZC has been designed with a suite of cooling water mitigation measures designed to reduce environmental impacts of abstraction and discharge of cooling water, summarised in the **Marine Ecology and Fisheries Environmental Statement** (starting at paragraph **22.5.15** [APP-317]). Embedded mitigation measures proposed for SZC are the primary means to reduce impingement of fish and invertebrates and include the installation of low velocity side entry (LVSE) intake heads and dedicated FRR systems coupled with a chlorination strategy that would prevent impinged biota from being exposed to anti-fouling chemicals. The **Acoustic Fish Deterrent Report,** provided as a Supplementary Submission at Deadline 5 [REP5-123], explained the fish protection measures proposed for SZC and, in particular, why an Acoustic Fish Deterrent (AFD) system is not part of the suite of mitigation measures.

Impingement predictions in the DCO submissions were structured to show the effects of no mitigation, the individual effect of each mitigation measure, and the effects of the mitigation measures in combination. This approach provided a full illustration of the consequences of different components of mitigation and their predicted implications (BEEMS Technical Report TR406.v7 [AS-238]).

The Marine Management Organisation (MMO) in its **Written Representation** at Deadline 2 [REP2-140] indicated at paragraph 3.2.6:

> *"The assessment makes assumptions about the effectiveness of the LVSE system and FRR system. There is a lack of good evidence to support these assumptions and thus the scale of benefit is uncertain, however, the MMO understands that there isn't any further work that can sensibly be done to reduce this uncertainty*".

It is noteworthy that the MMO goes on to state at paragraph 3.2.7 [REP2-140]:

> *"Notwithstanding these uncertainties, the entrapment estimates indicate that even in the absence of LVSE and FRR mitigation measures, only 4 species exceed the 1% threshold* [for adult equivalent entrapment as a proportion of spawning population size]*: bass, for which density adjustment substantially reduces assessment of impact; sand goby, for which mortality rate >1% Spawning Stock Biomass (SSB) is not a concern at population level; thin-lipped mullet, for which value is an artefact of the low level of landings and absence of SSB; and eel, for which the applied Equivalent Adult Value (EAV) of 1 is unrealistically high, and is a species most likely to benefit from the FRR. On this basis, the MMO consider there is a good level of confidence that actual impacts to all fish species will not be significant. Therefore, the MMO support the conclusions of the ES."*

The aim of this report is to determine the sensitivity of entrapment assessments on fish populations to uncertainties in the operational performance of the proposed fish mitigation measures and uncertainties in sampling techniques. The sensitivity analysis also accounts for the natural fluctuations of fish stocks used as the comparator for losses.

1.1.1 Operational performance of the mitigation techniques

Predicted values of FRR mortality applied in the impingement assessments and reported in the DCO (**BEEMS Technical Report TR406.v7** [AS-238]), were based on Environment Agency (2005) values. These figures were modified for passage through the SZC trash racks, band screens and drum screens by applying species-specific the morphometrics of the fish sampled at SZB.

As part of the Hinkley Point C WDA Appeal Inquiry[3], the Environment Agency produced a technical report (Technical Brief: TB008 Fish Recovery and Return System Mortality Rates). The Environment Agency report detailed a range of FRR mortality rates for different species in the context of the Hinkley Point FRR system. For species where data are available, this report applies the Environment Agency best case and worst-case FRR mortality values. Details of the FRR efficiency ranges applied in the assessment is provided in Section 2.1.6.

Acknowledging that the effectiveness of the LVSE intake heads is not certain, the sensitivity analyses in this report assumes no benefit of the LVSE heads. Impingement per cubic metre of water (cumec) extracted is therefore assumed to be the same as for SZB. The position in relation to LVSE effectiveness is detailed in Section 2.1.5.

1.1.2 Uncertainty in sampling techniques

The results of any monitoring programme or sampling campaign must be considered within the bounds of the limitations and assumptions of sampling. Whilst the entrainment and impingement monitoring at the operational SZB station provides a highly applicable and powerful tool to predict SZC entrapment limitations within the dataset remain. However, as discussed in Section 3.3, residual uncertainties must be considered in relation to the inbuilt precaution in the assessments and the low predicted level of effects.

Annual impingement rates are calculated by a statistical approach termed bootstrapping, which resamples the monitoring data to recalculate impingement rates based on 5,000 permutations. The uncertainty analysis applies the full distribution of impingement permutations for each species thereby accounting for variability in the predicted rates of annual impingement (Section 2.1.1). Furthermore, Revision 2 of this report applies only

[3] The Hinkley Point C Water Discharge Activity (WDA) Appeal Inquiry on the Permit Variation Application Relating to Acoustic Fish Deterrent heard evidence during a 9-day hearing from 8th - 24th June 2021. Evidence included the effectiveness of mitigation measures including the FRR system.

the upper entrainment predictions (rather than the range of values incorporated in the assessment in Revision 1, see Section 2.1.3).

Stakeholders have raised concerns relating to the potential for diurnal biases to be introduced into impingement predictions arising from incidences of overflowing overnight bulk samples. This is a result of the inability to staff overnight bulk samples due to security restrictions at the operational SZB nuclear facility. In summer months, overflow typically arises due to large numbers of ctenophores and gelatinous zooplankton clogging the nets. Overflows may also result due to ingress of weed and/or mud, or in the winter months due to inundation with pelagic species, primarily sprat and herring, and demersal whiting. Following the incidence of bulk overflows, samples collected during the day are extrapolated to estimate 24-hour impingement. The concern is that species more susceptible to impingement at night may be underestimated. A second concern raised by Interested Parties (IP) is the potential for an 'entrainment gap' whereby a fraction of fish, too large to be efficiently sampled by entrainment monitoring but too small to be efficiently impinged on the SZB 10mm drum screens, may be underrepresented in entrapment estimates.

Revision 2 of this report seeks to address the uncertainties raised by Regulators and IPs during the Examination period and respond to comments on Revision 1 of this report including those of the RSPB/Suffolk Wildlife Trust (SWT) **Deadline 7 Submission** [REP7-154]. Specifically, this report aims to address the sampling uncertainties in the following ways:

- **CIMP bulk overflow:** The Environment Agency in their **Written Representations** at Deadline 2 [REP2-135] and in their comments on Revision 1 of this report [REP7-132], outlined concern pertaining to the overflow of the bulk samples during the Comprehensive Impingement Monitoring Programme (CIMP) at SZB and the potential for introducing diurnal bias leading to over- or underestimations of 24-hour impingement rates. This report quantifies the potential diurnal bias. For species where impingement estimates may have led to underestimations, a correction factor has been applied. Section 2.1.2 considers the potential for diurnal bias focusing on four species of conservation importance (smelt, river lamprey, twaite shad and European eel), full technical details of the approach are provided in Appendix A.

- **Entrainment Gap:** Concerns were raised by IPs regarding the potential for an 'entrainment gap' primarily in relation to sprat and sand gobies[4]. This report applies literature growth and mortality rates at age to back-calculate the expected numbers of fish within the body size window that corresponds to the 'entrainment gap'. Predicted entrapment numbers with and without the 'entrainment' gap are then compared. The effects of the entrainment gap were assessed sprat, sand gobies and herring. Entrainment gap numbers have been included in the entrapment uncertainty analyses presented in this report. The entrainment gap is considered further in Section 2.1.3.2, full technical details of the approach are provided in Appendix B.

The term 'sand gobies' has been applied within DCO documents as a shorthand to describe a taxa comprising gobies of the genus *Pomatoschistus* spp. of which the sand goby (*P. minutus*) is the dominant species (see Section 3.2.1.2 for further details).

1.1.3 Contextualising losses

Most fish predicted to be entrapped at SZC will be juveniles. High natural mortality of the young age classes impinged means that most of the impinged fishes would not survive to contribute to the adult spawning population even if they had not been impinged. To determine the impact of losses of these fish from the adult population the losses are converted into equivalent adults (that is, the number of those juveniles impinged that would normally be expected to survive to maturity taking into account predation, disease etc). Equivalent adult value (EAV) factors are used to convert an annual rate of loss due to entrapment of predominantly juvenile fish into an annual rate of loss of fish that would mature and join the spawning population.

The risks posed by entrapment from SZC in terms of annual EAV losses are expressed in relation to population comparators. Comparators might be fish numbers in the population or spawning stock biomass

[4] Responses to Together Against Sizewell C Deadline 8 Post Hearing Submissions [REP8-284] have been provided at Deadline 10 (Doc. Ref. 9.120).

(SSB). Where direct population comparators are not available, losses are contextualised relative to commercial catches (landings). To account for the fact that the baseline population is not static, the uncertainty analysis considered the variability in the assessment comparator during the period of 2009-2017, coincident with the CIMP (Section 2.1.7). The variability in the population comparator was considered from the mean value and standard error, using bootstrapping of 5,000 iterations assuming a normal distribution.

Revision 2 of this report provides further context on the uncertainty in the population estimates for twaite shad. No twaite shad spawning rivers occur on the east coast of the UK. Cefas estimated the population size of the Elbe and Scheldt populations based on European monitoring data to provide a comparator for entrapment estimates. There are inherent limitations in the population estimates. Natural England [REP2-153] and the Environment Agency [REP2-135] in their **Written Representations** at Deadline 2 requested further uncertainty analyses based on the underlying assumptions in the population estimate and the estimation of confidence intervals to be applied. These uncertainties have been considered further and updated figures applied as the population comparators (Section 2.1.7.1). Full technical details of the approach are provided in Appendix C.

1.1.4 Entrapment uncertainty summary

The full uncertainty analysis quantifies uncertainty in the entrapment predictions and incorporates the following parameters to predict annual entrapment rates with associated confidence intervals:

- Upper rate of entrainment.
- The potential entrainment gap for sprat, gobies (*Pomatoschistus* spp.) and herring.
- The full distribution of impingement rates.
- Application of a correction factor to account for potential diurnal bias in smelt, river lamprey, twaite shad and European eel.
- A worst-case of zero benefit has been applied to the LVSE mitigation.
- The effectiveness of the FRR system (a range of values proposed by the Environment Agency for the similar, albeit more complicated[5], FRR design at HPC (TB008) is applied).
- Variation in the baseline population comparator.

The parameters have been bootstrapped with 5,000 iterations, drawing on all the uncertainty variables. From this distribution a mean, median, lower 5th percentile and upper 95th percentile population impact has been derived.

[5] The Hinkley Point C FRR system has an additional ‘handling’ element due to an Archimedes’ screw which carries the fish to a sufficient elevation to drain back to see under gravity, larger drum screens resulting in longer retention times and a longer route of return to the sea.

2 Methodology of estimation of uncertainty

2.1 Sources of uncertainty

This section considers the methodologies applied to quantify the uncertainty associated with each of the input parameters relating to sampling uncertainties, operational performance of the mitigation measures and the population comparators, as described in Section 1, and how these have been applied in the assessment of annual entrapment rates.

2.1.1 Variability in impingement predictions.

Fish abundance and distribution is heterogeneous in space and time and many species show highly seasonal patterns of impingement. Impingement monitoring at SZB was designed in a pseudo-random fashion to eliminate tidal biases whilst sampling the full year to capture seasonal patterns. Samples consisted of six 1-hour samples collected during the daylight hours and an overnight bulk sample, thus providing a 24-hour impingement record. A total of 205 site visits were undertaken in the 8 years from 2009-2012, and 2014-2017 thereby incorporating interannual variability into the data set.

Impingement monitoring methodologies at SZB and the statistical methodology to estimate an annual mean impingement prediction with 95% confidence intervals for SZB is detailed in BEEMS Technical Report TR339 [AS-238]. Statistical bootstrapping approaches were applied to resample the monitoring data. Bootstrapping randomly selects samples from the data before recalculating impingement rates, this process is repeated for 5,000 iterations allowing a mean estimate and confidence intervals to be derived. Impingement estimates from SZB are then scaled up to account for the greater flow rates to predict impingement rates at SZC following approaches detailed in BEEMS Technical Report TR406.v7 [AS-238]. Impingement predictions for SZC provide a mean value along with upper and lower 95% confidence intervals.

Following comments from the Environment Agency during consultation on the WDA environmental permit, relating to treatment of invalid bulk samples and raising factors, revised impingement estimates for SZB and SZC were provided. The changes and the associated raw data were submitted to the Environment Agency detailed within BEEMS Scientific Position Paper SPP111.v2. The refinements in the impingement predictions resulted in minor changes in the absolute impingement numbers that were well within the confidence intervals reported in the **Environmental Statement** [APP-317]. The changes did not alter the original conclusions that there would be no significant effects of impingement in relation to relevant population comparisons. For full transparency, changes in the absolute numbers impinged, along with a table comparing the latest figures with the DCO figures, was submitted to the Examining Authority in response to Examining Authority Questions BIO.1.242 and BIO.1.243 (see **Appendix 7L** of [REP2-110][6]).

The unmitigated annual impingement rates along with 95th confidence intervals generated by 5,000 bootstrapped iterations for each of the key species at SZC are provided in Table 1. The uncertainty analyses randomly resample from the full distribution of annual impingement predictions.

[6] SZC Co. Responses to Examining Authority's Written Questions. Appendix 7L Detailed response to questions ExA Ref. Bio 1.242 and 1.243 [REP2-110].

Table 1. Mean and 95% confidence intervals of predicted annual unmitigated impingement numbers at SZC. Numbers are not adjusted for equivalent adult values (EAV).

Common name	Sizewell C Unmitigated Impingement numbers (SPP111.v2 and [REP2-110])		
	Mean	Lower	Upper
Sprat	6,153,906	3,173,989	10,415,898
Herring	2,211,750	1,310,172	3,352,700
Whiting	1,495,192	1,095,717	1,954,416
European sea bass	641,398	296,862	1,113,750
Gobies (*Pomatoschistus* spp.)	483,487	205,548	916,287
Dover sole	211,083	146,474	290,806
European anchovy	148,332	43,495	356,894
Dab	128,476	76,309	214,481
Thin-lipped grey mullet	107,602	33,386	207,685
Flounder	32,149	24,367	42,211
Cucumber smelt (UK EA)	22,165	13,867	32,370
European plaice	21,956	14,135	32,723
Atlantic cod	16,505	5,716	30,807
Thornback ray	6,700	4,172	9,833
Twaite shad	2,693	1,340	4,691
River lamprey	2,607	1,430	4,393
European eel	2,463	1,530	3,628
Horse-mackerel	1,560	488	1,560
Mackerel	277	14	277
Tope	55	0	55
Sea Trout	8	0	8
Sea lamprey	4	0	4
Allis shad	0	0	0

2.1.2 Potential for diurnal bias

Restricted site access at operational nuclear power stations due to site security restrictions means it has not been feasible to collect hourly samples or monitor the collection of overnight bulk samples as impingement monitoring personnel cannot remain on the SZB nuclear facility site outside normal working hours. The inability to monitor the overnight samples means bulk samples may be subject to overflow if the sample net becomes clogged.

In summer months, overflow typically arises due to large numbers of ctenophores clogging the nets. Overflows may also result due to ingress of weed and/or mud, or in the winter months due to inundation with pelagic species, primarily sprat and herring, and demersal whiting (see Appendix A.6). A bulk sample is considered invalid if water overflows the top of the trash bins, as this could potentially result in underestimates of impingement. When bulk samples are considered to be invalid, the six hourly samples collected during daylight hours are extrapolated to estimate 24-h impingement.

Of the 205 impingement samples collected seasonally over 8 years, there were a total of 100 valid bulk samples. As such, there are 105 occasions where the bulk sample has been removed from the analysis and daytime hourly samples have been extrapolated to establish a 24-hour estimate of impingement. Depending on the species-specific diurnal behaviour there is the potential that the greater proportion of daytime samples used in the 24-h impingement predictions could result in over- or underestimates of impingement.

The Environment Agency in their **Written Representations** at Deadline 2 [REP2-135] point to data collected from Sizewell A (SZA) that indicates greater impingement rates during periods of darkness (Turnpenny, 1988). Periods of darkness occur during the period when bulk samples were collected (15:00 until 09:00) at SZB, although the length of periods of darkness will vary throughout the year. Whilst the Turnpenny (1988) data show increased impingement at night it is not clear from Turnpenny (1988) what the underlying factors driving the observed night-time increases in impingement rates at SZA are. The data, as presented, is normalised, and averages both species and the 41 seasonal samples. Therefore, it is not possible to determine the degree to which the data are influenced by seasonal sporadic impingement events of shoaling species such as herring and sprat, which are highly abundant and show seasonal peaks in impingement.

The design and inshore location of the SZA intakes and the species impinged need to be considered when drawing comparisons to SZB. The Environment Agency explicitly make this point in their guidance documents. In the 'Screening for intake and outfalls: a best practice guide', the Environment Agency provide a table from Turnpenny and Taylor (2000) that demonstrates the reduction in per cumec fish impingement due to the location and design of the Sizewell A and Sizewell B. The reduction in impingement by SZB is particularly marked in comparison to SZA for flat fish such as plaice, sole and dab and to a lesser extent pelagic species sprat and herring (Table 2.3 of Environment Agency, 2005). The report describes the improvements in intake technologies that have led to reductions in fish impingement at SZB as:

- reduction of intake velocities;
- fitting a velocity cap to the intake to eliminate vertical flow components;
- elimination of any intake superstructure (which tend to act as artificial reefs that attract fish)[7];
- location of the intakes further offshore where juvenile densities are lower;
- installation of a fish return system.

The 'Nuclear power station cooling waters: protecting biota' evidence repot (Environment Agency, 2020) specifically commented that;

> *"Turnpenny and Taylor (2000) outlined the major differences between the intakes. The intakes at A and B stations at Sizewell are 300 m and 600 m offshore, respectively. The older A station intake is a simple vertical shaft protected by a horizontal grill, which therefore draws water vertically down. The B station has a pair of intakes which are capped. This results in lower entrance velocities with a more horizontal inflow pattern, which fish are more able to avoid.*
>
> *So, in the case of case Sizewell A/B comparison, we do not know if it is the position, the volume, the intake velocity or the velocity cap which might be important"*

Environment Agency (2020).

Given the differences between the SZA and SZB sites, this report focuses on determining the potential for diurnal bias based on the substantial data series available from the SZB CIMP.

Revision 1 of this report detailed previous analyses of the potential for diurnal bias to propagate into the impingement predictions. This included results presented in BEEMS Technical Report TR339: Appendix F [AS-238] where 22 sampling visits with full 6 hourly samples and a valid bulk sample in the period 2014 – 2017 were compared. This sub-sample of data was analysed to determine if there were differences in the hourly impingement rates between the daytime hourly and overnight bulk samples. The species investigated included herring *Clupea harengus*, sea bass *Dicentrarchus labrax*, cucumber smelt *Osmerus eperlanus* and gobies (*Pomatoschistus* spp). The results showed no significant difference between hourly and bulk impingement rates and no consistent sampling bias between the hourly and bulk samples. Thus, it was concluded that there was no significant misrepresentation of impingement rates as a result of any sampling bias (BEEMS Technical Report TR339: Appendix F [AS-238]). The Environment Agency in their **Written Representations** at Deadline 2 [REP2-135] noted that, by only including valid bulk samples, the analyses in BEEMS Technical Report TR339 [AS-238] did not consider periods of maximum abundance when diurnal behaviour may be different. However, it should be noted that in summer months overflows are typically

[7] The intakes at Sizewell A were fitted with large, surface piercing jetty structure.

driven by ingress of gelatinous zooplankton. Furthermore, during periods of high gelatinous zooplankton biomass, impingement of fish is low (see Figure 4 in Appendix A).

Revision 1 of this report also commented on the treatment of bulk samples following a Schedule 5 request from the Environment Agency on the Sizewell C WDA environmental permit application. Following comments received from the Environment Agency relating to the incidence of bulk sample overflows, all the bulk samples were reviewed, and 18 additional samples were removed where there was an indication that overflowing may have occurred (resulting in 100 bulk samples in the data set, down from 118). Analyses of the effects of removing and including these additional samples were provided in BEEMS Scientific Position Paper SPP111.v2 provided to the Environment Agency as part of the WDA environmental permit documentation. The difference in predicted annual impingement numbers at SZC following the removal of the 18 bulk samples was very minor and bi-directional i.e., some species saw minor increases whilst others had minor decreases. The mean change was a 0.7% increase for the eight species contributing to the top 95% of impingement. The only key species where impingement rates changed more than 2.5% in either direction was cucumber smelt, with an 8.4% increase in predicted impingement following the removal of the 18 bulks (the case of cucumber smelt is considered in more detail in Section 2.1.2.1 below).

It is recognised that the analyses described above considered a sub-sample of the data, and in their comments on Revision 1 of this report, the Environment Agency express continued concerns relating to overflowing bulk samples. The solution proposed by the Environment Agency is the application of a precautionary correction factor [REP7-132]. This proposed solution has been adopted herein and is detailed in Appendix A. A summary is provided in the following section.

2.1.2.1 Correcting for potential diurnal bias

To account for the potential diurnal bias, a straightforward methodology was applied allowing screening of the species potentially impacted by diurnal biases was undertaken. The first step involved removing all bulk samples from the CIMP data set and recalculating impingement predictions based on extrapolation of the hourly samples collected during daylight hours. This data was then treated in the same manner as the full CIMP data set with valid bulks and bootstrapping approaches applied to estimate annual rates of impingement.

This provides two data sets; the first is based on 205 sample visits including 100 valid bulk samples (48.8%), termed the full CIMP dataset. The second is based on a total of 203 sample visits but with no bulk overnight samples, the daytime only dataset (two samples had to be removed due to only the collection of a bulk sample or low numbers of hourly samples, see Appendix A). By comparing the two data sets it is possible to screen species where diurnal bias may occur.

The full CIMP data set (CIMP with 48.8% valid bulks) was compared with the situation where bulks were removed (the daytime only dataset). It was then possible to examine the relative effect the bulk samples have on impingement predictions. An increase in impingement estimates when bulks are removed in the daytime dataset, compared to the full CIMP dataset indicates that proportionally more individuals are caught in the daytime samples (or extrapolating the daylight hourly samples results in higher impingement predictions than observed with a 24-h sample). In the case where species numbers increase following the removal of all bulk samples it can be assumed that impingement estimates have been overestimated during incidents of bulk overflow. This remains an assumption as it implies that the patterns of impingement rates during the daytime time and overnight are consistent between incidences of bulk overflows and during collection of valid samples. However, given there are 100 valid bulk samples throughout the seasonal dataset, such an assumption is reasonable. Nevertheless, reflecting the potential uncertainty relating to this assumption, no attempt has been made to correct impingement estimates where daytime only samples indicate impingement rates may have been overestimated in the full CIMP dataset. This introduces a degree of precaution in the assessment in the case of species that are more likely to be impinged during the day.

The focus of the Environment Agency concerns is the species that show a decrease in impingement estimates when bulk samples are removed. In case of these species, incidences of overflowing bulk samples could under-estimate their total impingement. Again, the analysis makes the reasonable assumption that patterns of impingement rates during the daytime time and overnight are consistent between incidence of bulk overflows and during collection of valid samples.

Changes in impingement estimates following the removal of overnight bulk samples are species specific (Table 2). For the eight most commonly impinged species, the removal of overnight bulk samples in the daytime dataset resulted in impingement estimates increasing. The mean increase for the top eight species was 3%. Sprat, herring, and sea bass increased between 6 and 10% with the removal of the bulk samples, indicating that impingement of these species may have been overestimated by a small margin. Whiting, gobies (*Pomatoschistus* spp.), Dover sole and dab showed minor differences of less than 2% in either direction, whereas anchovy decreased by approximately 4% when the bulk samples were removed. This indicated a small underestimate of the impingement rate (Table 2; Appendix A.3).

The most commonly impinged species at SZB are sprat and herring, accounting for 69% of impingement by numbers. The removal of the bulks and extrapolation of daytime hourly samples resulted in an approximate 10% increase in sprat and a 6% increase in herring (Table 2 and Appendix A.3). This shows that when bulk samples are valid there is higher relative impingement rate during the daytime samples. This observation is supported by the known behaviour of pelagic species. In offshore waters with depths > 40m herring undertake diurnal vertical migrations, moving closer to the sea surface at night, which is linked to the changing distribution of their prey (Munk *et al.,* 1989; Heath *et al.,* 1991; Huse and Korneliussen, 2000; Beare *et al.,* 2009). Herring also tend to school more tightly by day and to disperse at dusk (Cardinale *et al.,* 2003; Nilsson *et al.,* 2003). Since the capacity for vertical migration is constrained in the shallow inshore waters at Sizewell, an element of offshore movement at night and/or less marked day to night changes in abundance in the upper few metres of the water column are expected than would be observed further offshore. In the shallow (10m) and turbid waters of the Zeeschelde, there was no statistical difference between day and night herring catches, and during both day and night juvenile herring were aggregated in the upper water layer without diel migrations (Maes *et al.,* 1999). In the deeper waters of SZC it is likely that (per volume) herring impingement at night would be lower than at SZB. Evidence from the literature on herring behaviour supports the observations of higher impingement during daylight hours therefore it is highly unlikely that impingement underestimates pelagic herring sampling due to bulk overflows. Conversely, it is likely herring impingement is overestimated to a small degree.

In the case of sea bass, removal of bulk samples resulted in a 10% increase in annual impingement predictions (Table 2; Appendix A.3), suggesting the greater proportion of daytime samples in the CIMP may lead to an overestimate of annual impingement. Adult sea bass using offshore areas are known to spend the day in deeper water and ascend at night, but this behaviour is not so pronounced or consistent inshore and in the summer months (Quayle *et al.,* 2009; de Pontual *et al.,* 2019). Experimental (tank) studies have indicated that sea bass occupy the surface layer at night and swim deeper in the water column during the day (Schurmann *et al.,* 1989). Limited data from tracking with acoustic tags during periods of summer inshore residency in Ireland show sea bass are most active at dawn and dusk (O'Neill, 2017). Another study for juvenile sea bass showed that detection probabilities varied among estuaries, with more daytime detections in one location and more night-time detections in another. The effects were relatively small, however (changes less than 20%) (Stamp, 2021). Literature evidence and impingement data therefore points to the fact that impingement predictions may overestimate sea bass impingement due to a higher incidence of sea bass collected during daytime sampling.

In total, impingement estimates for nine of the key species decreased following the removal of bulk samples, these were Dover sole, anchovy, dab, smelt, European eel, twaite shad, river lamprey, mackerel, tope and sea lamprey (Table 2). Dover sole, anchovy, dab, and tope all decreased by <5%, such small changes do not significantly influence the results or our conclusions, since in all cases the population level effects from impingement were below 0.1% of the relative population comparator in Revision 1 of this report. Sea lamprey and mackerel deceased by 100% and 48.6%, respectively. However only one sea lamprey was caught in the SZB CIMP in an overnight bulk sample, whilst mackerel are caught inconsistently and in low numbers, occurring in 9 out of 205 impingement samples. The estimated impingement numbers of these species are therefore too low to determine the potential for diurnal bias as impingement is negligible.

Four species have been identified as requiring further attention to account for potential underestimates in impingement rates, these are:

- smelt;
- river lamprey;

- European eel; and,
- twaite shad.

All four have been selected as they are species of conservation interest and, with the exception of twaite shad, impingement numbers decreased by >10% when bulk samples were removed compared to the full CIMP dataset. Whilst twaite shad has a percentage decrease of just 2% we have included it here on a precautionary basis, as it is a species of conservation interest (Table 2; Appendix A.3).

Table 2. Difference in predicted mean impingement rates between the full data including 48.8% valid bulk samples and no bulk samples. Values in green represent species where overestimates in impingement rates may occur. Species in red represent instances where underestimates may have occurred. When larger overestimates were identified, these were addressed by applying correction factors in subsequent analyses.

Common name	Difference in mean (% change)	Comment
Sprat	9.8	Potential overestimate in impingement rates
Herring	6.0	Potential overestimate in impingement rates
Whiting	1.8	Minor change
European seabass	10.3	Potential overestimate in impingement rates
Gobies (*Pomatoschistus* spp.)	1.4	Minor change
Dover sole	-0.2	Minor change
Anchovy	-4.1	Potential underestimate: No significant bearing on effect predictions < 0.1% of landings comparator.
Dab	-1.2	Minor change
Thin-lipped grey mullet	4.1	Potential overestimate in impingement rates
Flounder	3.0	
Plaice	1.4	Minor change
Smelt	-10.5	**Correction Factor applied**
Cod	9.2	Potential overestimate in impingement rates
Thornback ray	2.1	Minor change
Eel	-12.4	**Correction Factor applied**
Twaite shad	-2.0	**Correction Factor applied**
River lamprey	-12.9	**Correction Factor applied**
Horse mackerel	4.1	Potential overestimate in impingement rates
Mackerel	-48.6	Change driven by very low impingement rates
Tope	-3.3	Minor change and low impingement rates
Sea trout	292.7	Change driven by very low impingement rates
Sea lamprey	-100.0	Change driven by very low impingement rates
Allis shad	NA	NA
Salmon		NA

2.1.2.2 *European eel*

The Environment Agency raised concerns relating to the potential for higher rates of impingement of eel at night, and that impingement estimates may be underestimated due to the greater proportion of daily samples. The results of the removal of all bulk samples reduced eel impingement predictions by 12.4% individuals per annum. Consequently, a correction factor has been applied to account for the greater proportion of eels that may have been missed in the overnight bulk samples.

Eels are impinged throughout the year in relatively low numbers and the correction factor applies a proportional raising coefficient based on the proportion of valid and invalid samples (Appendix A.5).

A correction factor of 1.130 (Appendix A.5) has been applied resulting in an increase in mean SZC impingement from 2,463 (Table 1) to 2,783 eels per annum. For the uncertainty analysis the correction factor has been applied to the full distribution of the data.

There is scarce evidence from the literature to support or refute the diurnal bias in impingement rates of yellow eels at Sizewell. During their main spawning migration over deeper water, silver eels exhibit diel vertical migrations, moving from deeper water during the day into shallower water at night (Righton *et al.,* 2016). Tracking of yellow and silver eels in the southern North Sea show that selective tidal stream transport was used day and night when it occurred. Some evidence of the use of shallower water by silver eels during the night exists, but during the study one yellow eel was in shallower water during the day (McCleave and Arnold, 1999). Other studies have shown midwater movements by night and low levels of movement on the seabed by day (Westerberg, 1979; Westerberg *et al.,* 2007). In a Baltic study, most silver eel swimming at night was within just 1m of the water surface (Westerberg *et al.,* 2007). Use of the seabed by day in the early stages of silver eel migration may be a predator avoidance strategy (Lennox *et al.,* 2018).

It is feasible, as the data suggests, that a greater proportion of daylight samples may result in an underestimate of impingement of yellow eels. This has been addressed with the application of the correction factor.

2.1.2.3 Smelt

The removal of bulk samples resulted in a decrease in impingement predictions of 10.5% for smelt (Table 2). This is the opposite trend to that observed in Revision 1 of this report when analyses on a sub-sample of the data were completed (further details in Appendix A.4.1). There is no evidence of smelt vertical movements in marine waters to support or refute the observation in the impingement data. Some information on diurnal patterns of distribution is available for fresh and brackish waters, but results are not consistent. In one shallow freshwater lake in the Netherlands, smelt were found in the upper parts of the water column by day and tended to disperse to deeper water at night (Gastauer *et al.,* 2013). In an entirely freshwater Finnish lake the reverse was seen whereby smelt migrated from deeper areas in the day to the surface at night (Horppila *et al.,* 2000). In freshwater and estuaries smelt are more active and migrate at night, but it is not known if this is also true in the sea (Moore, 2016; Moore *et al.,* 2016). In inshore waters of the Baltic Sea (depths ~ 15-20 m) smelt was found to feed in the water column and on the same food as herring (Mustamäki *et al*., 2016). Herring and sprat in the Baltic Sea disperse during the night at the surface and aggregate during the day at the bottom; it is possible smelt follow a similar behavioural pattern as herring and sprat do, following their prey (Cardinale *et al.,* 2003; Nilsson *et al.,* 2003). However, removal of all the bulk samples and comparison with the full CIMP dataset provides evidence that impingement rates may have been underestimated. The opposite trend was reported when a sub-sample of the data was analysed indicating that predictions of this species are variable and driven by a few high impingement events rather than a consistent bias. As the full data set indicates the potential for underestimation, it is appropriate to apply a correction factor.

A correction factor of 1.111 (Appendix A.5) has been applied for smelt resulting in an increase in mean SZC impingement from 22,165 (Table 1) to 24,625 smelt per annum. For the uncertainty analysis the correction factor has been applied to the full distribution of the data.

2.1.2.4 River lamprey and twaite shad

Twaite shad and river lamprey are impinged in relatively low numbers throughout the year. Twaite shad are impinged in higher numbers in Q2 and Q3 whilst river lamprey are impinged relatively consistently without large seasonal fluctuations in abundance between Q2 and Q4 (BEEMS Technical Report TR339 [AS-238]). Therefore, impingement of these species will be susceptible to both the effects of diurnal biases and the greater probability of encountering a scarcely impinged species in the longer bulk samples. In either instance the reduction in the proportion of bulk samples may lead to underestimates.

The removal of bulk samples resulted in a decrease in impingement predictions of 12.9% for river lamprey and 2.0% for twaite shad (Table 2). A correction factor of 1.135 has been applied for river lamprey resulting in an increase in mean SZC impingement from 2,607 (Table 1) to 2,959 river lamprey per annum. A correction factor for twaite shad of 1.021 has been applied resulting in an increase in mean SZC impingement from 2,693 (Table 1) to 2,753 twaite shad per annum (Appendix A.5).

In conclusion, the screening for diurnal bias due to a higher proportion of daytime samples than predominantly night-time bulk samples has been shown to be species-specific and bi-directional. Therefore, the sampling did not introduce a consistent bias. Given that there were 100 valid bulk samples, and that bulk samples have a greater weighting when present, species-specific diurnal bias is relatively small. Correction factors ranging from 1.021 to 1.135 have been applied to twaite shad, smelt, eel and river lamprey to account for the potential for diurnal bias to result in underestimates in impingement predictions.

2.1.3 Entrainment predictions

Entrainment primarily impacts the early life-history stages of fish including eggs, larvae and post-larvae and, for some species, juvenile stages. The occurrence of these early life-history stages is highly seasonal and is directly related to the timing of spawning events. Between May 2010 and May 2011 the Comprehensive Entrainment Monitoring Programme (CEMP) collected 3-4 samples per month at SZB resulting in a total of 40 samples (BEEMS Technical Report TR318 [APP-324]). The CEMP has been used to determine the predicted rates of entrainment at SZC. The numbers presented in Table 3 represent the number of equivalent adults summed from the egg, larval and juvenile stages of the key species entrained. The range represents the variability in natural egg mortality rates (full details are provided in BEEMS Technical Report TR318 [APP-324]). The uncertainty analysis in Revision 1 of this report applied the range of entrainment EAV numbers to reflect the differing survival rates. In the **Marine Ecology and Fisheries Environmental Statement** [APP-317], entrapment estimates were based on the upper entrainment EAV numbers. In trying to achieve the most precautionary entrainment estimates this report mirrors the Environmental Statement and only applies the upper entrainment estimates (Table 3).

Entrained fishes are typically in the early stages of the life cycle and therefore have very low EAV. Gobies are an exception, with all life stages susceptible to entrainment. The earlier the life stage and the lower the EAV, the smaller the impact of a given rate of entrapment mortality on the population. The vast majority of early life stages would not survive to maturity if they had not been entrained. For all taxa except gobies (*Pomatoschistus* spp.) , entrainment losses converted to EAV represent a small proportion of impingement losses (see Section 2.1.4).

2.1.3.1 Entrainment mortality

Entrainment mimic unit (EMU) studies provide estimates of the survival of entrained sea bass and Dover sole eggs as well as survival of a range of invertebrates (BEEMS Technical Report TR318 [APP-324]). EMU studies have also demonstrated high survival rates of entrained glass eels. Where specific entrainment survival data from EMU studies are available it has been applied (Table 3), in all other instances mortality of early life stages has been assumed to be 100%. This is considered precautionary as studies at other power plants has demonstrated variable but higher survival. Observed survival rates as low as 3-5% have been reported for anchovy, whereas 59-97% survival has been shown for striped bass (ecological and morphological analogue of sea bass). Survival rates of entrained larvae of the different percoid fish (bass, perch, blennies and gobies) at Calver Cliffs Nuclear Power Plant ranged from 37 to 98% including 88-98% in gobies (Mayhew *et al.,* 2000). Whilst entrainment effects are minor for most species, assuming 100% mortality ensures our analyses are conservative, particularly for gobies (*Pomatoschistus* spp.).

Table 3. Range in annual equivalent adults predicted to be entrained at SZC for the key species entrained during monitoring at SZB. Numbers of eggs, larvae and juveniles that are entrained annually have been converted to equivalent adult values (EAV) numbers. Numbers included predicted entrainment survival for life stages with data available from entrainment mimic unit (EMU) studies. Where no data is available, 100% mortality is assumed. Uncertainty analyses apply the upper figures shown in bold and include the 'entrainment gap' for selected species.

Common name	Entrainment survival (TR318)	Entrainment EAV numbers (TR318) or weight in kg of equivalent silver eels (SPP104)		'Entrainment Gap' annual EAV number
		Lower	Upper	
Sprat	0% precautionarily assumed.	45,790	**199,715**	304,982
Herring	0% precautionarily assumed.	2,399	**23,992**	15,910
Whiting	0% precautionarily assumed.	0	**0**	
European sea bass	40% survival of eggs based on EMU studies.	36	**36**	
Gobies (*Pomatoschistus* spp.)	0% precautionarily assumed.	1,155,406	**2,892,198**	589,200
Dover sole	20% survival of eggs based on EMU studies. 0% precautionarily assumed of other stages.	592	**631**	
European anchovy	0% precautionarily assumed.	2,869	**2,869**	
Dab	0% precautionarily assumed.	21,810	**21,810**	
European eel (glass eel converted to silver eel biomass)	80% survival of glass eels based on EMU studies.	5.6kg	**18.9kg**	

2.1.3.2 Entrainment Gap

Entrainment monitoring is achieved by means of pumping water from the forebay at SZB and collecting samples in an array of plankton nets (BEEMS Technical Report TR318 [APP-324]). Stakeholders have raised the concern that there is a potential 'entrainment gap' whereby a proportion of fish that are too small to be impinged on the 10mm drum screen mesh are too large to be effectively sampled by the pump sampler during entrainment monitoring. This is based on the view that the pump used to sample water from the forebay "*is an effective sampler for non-swimming life stages (e.g. eggs) and weakly swimming stages such as fish larvae"* but "*is an ineffective sampler for actively swimming juvenile fish and never catches larger fish which are strong swimmers*" [REP2-481h].

A stakeholder raised particular concerns over the potential to underestimate entrainment for gobies (*Pomatoschistus* spp.) and sprat. Concerns were also raised regarding other pelagic species and slender bodied species such as the glass eel stage of European eel, river lamprey and sand eel. The issue of slender bodied species was addressed in **SZC Co. Written Submissions Responding to Actions Arising from ISH7: Biodiversity and Ecology – Parts 1 and 2** (Section 1.16 pg. 21 of [REP6-002])[8].

This report specifically considers the potential for an entrainment gap in sprat, gobies and herring and attempts to quantify the abundance of any missing size fraction. These species have been selected as they spawn in waters adjacent to Sizewell and are the three most abundant species in entrainment monitoring sampling and contribute to the top 95% of individuals in the impingement record. In the case of sprat, all life stages including eggs, larvae, juvenile and adults are identified in ichthyoplankton surveys (BEEMS Technical Report TR315 [APP-319]), entrainment and impingement monitoring. Sand gobies lay eggs on benthic substrates, primarily bivalve shells, and are not subject to entrainment. However, larvae, juveniles

[8] A further response to the response to written submission of Together Against Sizewell C is provided at Deadline 10 (Doc. Ref.9.120).

and adults are recorded in large numbers in entrainment and impingement monitoring. Herring lay eggs on seabed substrates but larvae and juvenile stages are recorded in ichthyoplankton surveys, entrainment monitoring and impingement sampling. Given the high abundance of these species and vulnerability of such life stages for entrainment they represent the species for which entrapment is most likely to be underestimated due to any inefficiencies in entrainment monitoring.

To estimate the effects of any entrainment gap, a three-step process was followed for each species. First, the distribution of body sizes retained during impingement monitoring was determined. Second, the distribution of body sizes recorded during entrainment monitoring was determined. These steps provided an estimate of the minimum body size for which impingement monitoring retained all individuals and the maximum body size for which entrainment sampling would retain all individuals. The difference between minimum size of efficient impingement and the maximum efficient size of entrainment represents the ‘entrainment gap’.

The third step includes incorporation of literature values for growth and mortality for each species to back-calculate the expected numbers of fish in the entrainment gap size range. The expected numbers in the entrainment gap versus observed numbers for the respective size class in the impingement record provides an estimate of the potential for missing fish in the entrainment gap.

In BEEMS Technical Report TR318 [APP-324] annual entrainment of sprat larvae by Sizewell C was predicted as 44,638,462 individuals and accounted for 18.9% of the total number of larvae entrained from all species. Entrainment rates of juvenile sprat was predicted to be 19,419,776 (38.9% of the total number of entrained juveniles). The absolute number of juvenile sprat in the entrainment gap was estimated at 3,118,423, with an EAV of this early life-history stage of 0.0978, which is equivalent to a prediction of 304,982 adult losses. This represents a 6% increase in the total entrapment numbers previously predicted, taking total entrapment losses to 5,127,842 equivalent adult sprat per annum (Appendix B.2). This increase in entrapment predictions has been added to the uncertainty analysis.

Gobies of the genus *Pomatoschistus* spp. are the key taxa most susceptible to the entrainment gap due to their high abundance and the greatest proportion of their life history occurring in the size window of the entrainment gap. Entrainment monitoring data collected from SZB was used to predict entrainment of 153,250,186 larvae (64.7% of the total larvae) and 22,375,425 juvenile (44.8% of total) sand gobies per annum. The total number of gobies (*Pomatoschistus* spp.) in the entrainment gap is estimated to be 2,960,806, with a calculated EAV of 0.199 equating to 589,200 equivalent adults per annum (Appendix B.4). This represents approximately 17.5% more equivalent adult gobies than previously reported to be lost to entrapment mortality. The additional numbers in the entrainment gap have been added to the uncertainty analysis. It should be noted that entrainment losses of gobies are highly precautionary in that they assume 100% mortality. Survival rates of entrained goby larvae has been reported between 88-98% at the Calver Cliffs Nuclear Power Plant (Mayhew et al., 2000).

Annual entrainment of herring larvae by Sizewell C was predicted as 17,921,743 and accounted for 7.6% of the total number of larvae entrained. Entrainment rates of juvenile herring were predicted to be 87,346 (0.2% of the total number of entrained juveniles). The absolute number of juvenile herring in the entrainment gap is estimated to be 4,792,065 with an EAV of this early life-history stage of 0.00332. Thus, equivalent adult losses of 15,910 are predicted. This represents a minor 1% increase in the total EAV numbers of herring entrapped per annum (Appendix B.3.). The additional numbers in the entrainment gap have been added to the uncertainty analysis.

In conclusion, sprat, gobies (*Pomatoschistus* spp.) and herring represent the three key species with the highest probability of entrapment predictions being subject to inefficiencies in entrainment sampling within the size window associated with the ‘entrainment gap’. Quantification of the total entrainment for sprat and herring led to increases of 1% and 6% in EAV numbers, respectively. Goby had the greatest increase in EAV numbers of approximately 17.5%.

Note on prey availability: Some stakeholders have questioned the application of EAV factors for juvenile stages raising concerns that this may underestimate the importance of these early life history stages as prey for designated species such as tern. Estimation of juvenile losses expressed as equivalent adults is a necessary step to determine effects on the sustainability of fish populations, assessed herein. Concerns relating to prey availability have been assessed in relation to localised depletion in BEEMS Scientific Position paper SPP103 Rev 5 [REP6-016]. The local depletion assessment does not require the application of EAV and specifically addresses concerns relating to prey in the entrainment size fraction.

2.1.4 Equivalent adult value (EAV)

EAV factors are used to convert an annual rate of loss due to impingement of predominantly juvenile fish into an annual rate of loss of fish that are maturing and joining the spawning population. High natural mortality of the young age classes impinged means that most of the impinged fishes would not survive to contribute to the adult spawning population in the absence of the station. The Cefas EAV method involves a forward projection of annual impingement mortalities, accounting for natural mortality, to give an equivalent annual rate of loss of mature fish. It is a straightforward adjustment to reflect the likelihood of impinged fish reaching maturity and contributing to the spawning population.

EAV factors are multiplied by numbers of impinged fish to estimate the number of equivalent adults that are lost (the EAV number) or multiplied by numbers of impinged fish and the individual body weight of mature fish in the population to give an EAV biomass. EAV numbers and biomass are expressed as annual rates.

Estimates of annual EAV numbers or biomass as a proportion of spawning population size can be used to assess if the annual rate of impingement mortality poses a risk to the population sustainability relative to pre-defined thresholds (see Section 2.2). An advantage of the EAV method is that it is not so data-demanding as more complex methods of population assessment (e.g. stock assessment). This advantage allows it to be applied to many species to screen for risks, as done by Cefas when assessing the effects of impingement.

During correspondence with SZC Co. regarding the scope of sensitivity analyses (23/07/2021), the Environment Agency requested that impingement predictions be updated to include repeat spawning, applying an extension proposed by the Environment Agency and termed “Spawning Production Foregone” (SPF). The Environment Agency also requested the underlying parameters used in the EAV calculations be checked to ensure they are suitably precautionary and apply the latest information. Whilst Cefas has checked the underlying parameters used in the EAV calculations, the SPF extension has not been used as the output from this calculation is not an annual rate of loss from the spawning population. The concerns with the SPF method are briefly considered below and are detailed in a Technical Note issued at Deadline 6[9] (Appendix F of [REP6-024]). Responses to Natural England and the Environment Agency comments on SZC Technical Note on EAV and stock size (Appendix I of [REP8-119][10]) was subsequently released at Deadline 8.

2.1.4.1 Is the EAV approach precautionary

Cefas EAV factors are calculated for each species based on the length distributions of the fish in the impingement record. Age at length keys are used to determine the ages of impinged fishes. Species maturity at age and age specific natural mortality rates from the literature are applied to determine the number of fish that would have been expected to survive to first maturity, had they not been impinged.

Fishing mortality has not been included when calculating the EAV. This means EAV numbers for first time spawners are overestimated, as in most species fishing mortality is recorded before the age of maturity in directed fisheries and as bycatch. By assuming no fishing mortality before maturity, the EAV assessment overestimates the chance of survival to maturity and is therefore precautionary, particularly for species such as cod, whiting and sea bass.

In repeat spawning fish populations that are subject to sustainable rates of additional mortality, the year in which a year-class of fish matures and recruits to the spawning population is not the year in which the egg production of the year-class is greatest. This is because the fecundity of fishes increases rapidly with size, thus age. But, at the age of first maturity there will be more fish present in any given year-class than at any subsequent age, because mortality will occur after first maturity. This means that an estimate of the number of entrapped equivalent adults (EAV number) that would have reached the age at first maturity will be greater

[9] Deadline 6 Submission - 9.63 Comments at Deadline 6 on Submission from Earlier Submissions and Subsequent Written Submissions to ISH1-ISH6 - Appendices - Revision 1.0 [REP6-024]
[10] Deadline 8 Submission - 9.99 Comments on Earlier Deadlines and Subsequent Written Submissions to CAH1 and ISH8-ISH10 - Appendices Part 1 - Revision 1.0. Appendix I of [REP8-119]

than the number that would have reached any subsequent age. Thus, when an EAV number at the age of first maturity is expressed as a percentage of spawning population numbers it provides a conservative estimate of impact. This is because every fish contributing to the EAV number has a value of one, which is the same as the value of one that is given to older, larger and more fecund fish in the spawning population.

An EAV number may be compared to a population number (as is the case for twaite shad where losses are compared to an estimated population number from a given river system, Table 6).

In most cases, the EAV number is converted into an EAV biomass for comparison with the Spawning Stock Biomass (SSB), by multiplying the EAV number of first-time spawners by the mean adult fish weight from the spawning population. The individual weight at the age at first maturity will be lower than the individual weight of older and more fecund fish in the spawning population. Multiplying lost numbers at the age of first maturity by the mean individual biomass in the spawning population will upweight losses of spawners due to entrapment and their effective contribution to the spawning population biomass. This results in a precautionary rate of EAV biomass loss as a percentage of spawning population biomass for repeat spawning species.

For species where there are very low numbers recorded in impingement samples or there are insufficient biological data to determine an EAV, a precautionary EAV of 1 has been assumed. Notably, this assumption was made for Twaite shad, river lamprey and European eel. This assumes all fish of these species would contribute to the spawning population. EAV values and comments on the degree of precaution assumed are provided in Table 4.

Cefas has reviewed the EAV input parameters and is satisfied they account for the latest evidence on species maturation and age specific mortality. Cefas recognises that the Environment Agency concern may pertain to whiting, for which recent ICES evidence indicated a lower age at first maturation in the south-western stock. However, the south-western whiting stock is separate from the North Sea stock and the maturation data applied to calculate EAVs correctly applied the average values for the period of observed impingement. Maturation data are updated by the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK) every year, the data for the years 2009-2017 remain unchanged.

EAVs used in this sensitivity analysis are consistent with those applied in the Environmental Statement [APP-317] and detailed in BEEMS Technical Report TR406.v7 [AS-238] and are provided in Table 4**.**

Table 4. Equivalent Adult Values (EAV) for the key species at Sizewell.

Common name	EAV	Comment
Sprat	0.751	
Herring	0.715	
Whiting	0.356	
European sea bass	0.224	
Gobies (*Pomatoschistus* spp.)	1.000	
Dover sole	0.213	
European anchovy	0.974	
Dab	0.445	
Thin-lipped grey mullet	0.083	
Flounder	0.462	
Smelt (cucumber)	0.761	
European plaice	0.345	
Atlantic cod	0.359	
Thornback ray	0.193	
Twaite shad	1.000	The assessment is precautionary as 46% of impinged fish are between 1- and 3-year-olds. Fish of this age have low maturity rates with 65-90% of males and 95-100% of females from England, Wales and Ireland mature at 4 y.o. or older (Aprahamian *et al.,* 2003). Up to 76% of fish in the impingement record may not have reached maturation.
River lamprey	1.000	River lamprey metamorphose into adults at a length of 90-120mm and at around 130mm they migrate to the sea (Maitland, 2003). 14% in the impingement record are below 130mm, some as small as 6.5-9.5mm have been recorded which may be washouts from river systems. River lamprey are semelparous (breed once then die) therefore an EAV of 1 represents the theoretical maximum value.
European eel	1.000	No silver eels (adults) have been impinged at Sizewell. The yellow eel stages would continue to mature in coastal waters before migrating to the Sargasso Sea to spawn. Eel are semelparous therefore an EAV of 1 represents the theoretical maximum value.
Horse-mackerel	1.000	Negligible impingement.
Mackerel	1.000	Negligible impingement.
Tope	1.000	Low impingement numbers.
Sea Trout	1.000	Single impingement record.
Sea lamprey	1.000	Negligible impingement. Sea lamprey are semelparous therefore an EAV of 1 represents the theoretical maximum value.
Allis shad	1.000	Single impingement record.

2.1.4.2 Spawning Production Foregone (SPF)

The SPF extension attempts to build upon the EAV by adding the probability of repeat spawning whereby a fish may spawn more than once over a number of years. By adopting this approach, the assessment necessarily estimates a multiannual rate of impingement losses and not an annual one. Such an approach would necessarily give inflated estimates of annual loss and annual loss as a percentage of spawning population size. This is because it would involve projecting and summing the future numbers of mature fish over several years (a multi-annual rate of loss) rather than estimating it for a single year.

Critically, rates that compile losses of spawning fish over several years and report these as a percentage of spawning population size cannot be related to the thresholds for an annual rate of loss (such as annual rates of fishing mortality that are known to be sustainable), because the rates would be multiannual. Accounting for repeat spawning would, in effect, generate a crude estimate of accumulated numbers of missing fish over many years.

A second important issue with the application of the SPF extension is the need to deal with fishing mortality. The Cefas EAV approach is already precautionary in that is assumes no mortality of the juvenile stages. To extend this assumption to the adult stages introduces over-precaution. For example, sustainable fishing mortality reference values vary in well studied commercial fish species between 19% for sea bass to 36% for plaice above natural mortality for the stocks of relevance to Sizewell (**Table 10** in **BEEMS Technical Report TR406.v7** [AS-238]). In their **Relevant Representations** [RR-0744], the MMO raise this point regarding the appropriate application of EAV approaches, acknowledging that both methods are precautionary but that *"care needs to be taken to avoid an over-precautionary approach"*. In their review of the EAV approaches, the MMO conclude [RR-0744]:

> *"The MMO consider the core method* [Cefas EAV method in comparison with the EAV-SPF] *is the better in that the end-point age is more likely to be reflective of reality in the context of currently fished seas, and because the MMO consider the extension method, while very precautionary, has conceptual challenges for EAV>1*[11] *and problems for comparing to SSB. The MMO is comfortable that all due efforts have been made to secure data at an appropriate scale."*

Cefas is confident in the precautionary nature of EAV-based risk assessment and maintain that the SPF method proposed by the Environment Agency is not fit for purpose because it does not estimate an annual rate of loss from the spawning population and thus it is inappropriate to relate the results from such an analysis to thresholds that are defined based upon an annual rate of loss. Rather than being a precautionary measure accounting for repeat spawning, SPF introduces further uncertainty and cannot directly be compared with estimates of annual rates of mortality that are known to be sustained (e.g., annual rates of fishing mortality). The SPF is therefore not considered further in this report.

If annual rates of EAV biomass were to exceed pre-defined thresholds for population sustainability, further detailed analyses or understanding of the species biology may be undertaken. A powerful analytical tool available for data rich species is to run a full ICES stock assessment whereby annual impingement from the station can be added as a source of mortality of the stock over multiple years to determine if the long-term impact of the station could affect population trends. Such data demanding approaches are not available for many of the species assessed at Sizewell and are restricted to data rich, typically commercially exploited species.

To provide the highest level of confidence available in the assessment of no significant effects, Cefas has undertaken a full ICES stock assessment for sea bass based on precautionary assumptions which was provided at Deadline 8 (BEEMS Scientific Position Paper SPP118 [REP8-131]). The stock assessment results were consistent with the EAV risk-based approach. No discernible effects on the population trends and only very minor effects on absolute SSB were observed despite the application of highly precautionary loss estimates. That is, the size of the spawning population would still have increased and decreased at the same times and at almost identical rates whether or not SZC impingement was occurring. The stock assessment results are summarised in Section 3.2.1.1.

2.1.5 Uncertainty in the performance of the LVSE mitigation

The LVSE intakes are designed to minimise impingement[12] by:

1. Reducing vertical velocities, which fish are ill equipped to resist, by means of velocity caps on the intakes.

[11] For many of the conservation species, and those impinged in low numbers a precautionarily EAV of 1 has been applied (Table 3).
[12] Small life-history stages typically entrained are not predicted to benefit significantly from the head design due to reduce swimming capabilities.

2. Limiting the intercept area 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 parallel with the tidal flow.
3. Reducing intake velocities into the head to a target velocity of 0.3m/s over as much of the length of the intake surface as possible to maximise the possibility of most fish avoiding abstraction.

Statutory consultees have questioned the effectiveness of the LVSE in the absence of an AFD. In its response to **Examining Authority question Bio.1.245** [REP2-140] the MMO state that

> *"It is recognised that the LVSE design has been put forward by the Environment Agency as a mitigation measure for cooling water abstractions (in its good practice guidance), although this tends to be accompanied by Acoustic Fish Deterrent (AFD) systems (which are not currently proposed for SZC). While it is feasible that the LVSE design, on its own, will provide some benefit in terms of reductions in fish impingement, even if the benefit was zero, the MMO does not believe this would not materially change the conclusions of the overall fish entrapment assessment."*

In acknowledgement of the lack of certainty in the current assessment of the effectiveness of the LVSE heads, the sensitivity analyses in this report assume no benefit of the LVSE. Impingement per cumec is therefore assumed to be no different than the current SZB head which, unlike SZC, has a velocity cap but is not LVSE by design. A value of 1.0 has therefore been applied in the sensitivity assessment (Table 5).

2.1.6 Uncertainty in the performance of the FRR mitigation

The 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. The FRR system has been designed and, following intensive design scrutiny, has received regulatory approval for HPC.

The predicted values of FRR mortality applied in the impingement assessments were based on Environment Agency (2005) guidance for survival through FRR systems, modified for the SZC specific trash racks, band screens and drum screens. A description of the approach is provided in BEEMS Technical Report TR406.v7 [AS-238]. Table 5 shows the predicted FRR mortality for each of the key species.

In Technical Brief: TB008 Fish Recovery and Return System Mortality Rates) the Environment Agency states

> "*The Technical Brief recommends a method to set a FRR mortality rate for each species and a range around the FRR mortality rate for each species. The range set accounts for the uncertainty in the underlying evidence used to set the FRR mortality rate, and in the efficiency of the bespoke FRR system proposed for Hinkley Point C (HPC)."*

Sizewell C will replicate the design of Hinkley Point C as much as possible. However, the reduced tidal range at Sizewell compared with Hinkley allows several design changes that are improvements over the HPC design:

a) The reduced tidal range means that the drum screens can be smaller – the diameter will be 4m less than at Hinkley Point C which means that the rotation time (and time that fish and biota will be in the bucket will be shorter than at Hinkley Point C);

b) Due to the reduced tidal range, and the elevations of buildings on the power station platform, the debris recovery building is at a suitable elevation to drain back to sea under gravity directly from its floor. At Hinkley Point C due to the large tidal range the material needs to be elevated to platform level by use of an Archimedes screw – which introduces an additional element of "fish handling" (i.e., manipulation) within the FRR. An Archimedes screw is not required at Sizewell.

c) The reduced tidal range and lack of the need for an Archimedes screw, allows each UKEPR unit to have its own, dedicated FRR tunnel to return fish to sea from the debris recovery building which is more direct and therefore reduces transit time for fish through the system.

Furthermore, at Hinkley Point C the trash rack would be fitted with 50mm spacing, whereas a 75mm spaced trash rack would be fitted at SZC. This increase in trash rack size reduces the impediment of the largest size fish (with the highest EAVs).

In summary, the FRR system at Sizewell C is predicted to have greater efficiency than that at Hinkley Point C. Therefore, for this sensitivity analysis, it is considered appropriately precautionary to apply the Environment Agency (TB008) Hinkley Point C FRR uncertainty ranges.

The Environment Agency best and worst-case values from TB008 have been used in the sensitivity analysis. Where worst-case ranges are lower than the FRR efficiency applied in BEEMS Technical Report TR406.v7 [AS-238], the higher values are used. The uncertainty analysis is completed twice, once with the FRR efficiency fixed to the values applied in BEEMS Technical Report TR406.v7 [AS-238] in the 'impingement assessment' and once with the Environment Agency TB008 range in the 'entrapment assessment'. The bootstrapping approach for the entrapment assessment draws from the FRR range assuming a uniform distribution (Table 5).

Table 5. Mitigation parameters applied in uncertainty analyses. Where the predicted FRR efficiency is greater than the Environment Agency worst case, the FRR efficiency value from BEEMS Technical Report TR406.v7 [AS-238] is applied as the worst-case. Sensitivity analyses apply the FRR efficiency (impingement assessment) and TB008 best and worst-case range (entrapment assessment).

Common name	LVSE benefit	FRR mortality (TR406.v7 [AS-238])	FRR mitigation range applied in uncertainty analysis based on Environment Agency HPC values (TB008)		
			TB008 predicted mortality	Realistic best case	Realistic worst case
Sprat	1.000	1.000	1.000	0.950	1.000
Herring	1.000	1.000	1.000	0.900	1.000
Whiting	1.000	0.551	0.552	0.410	1.000
European sea bass	1.000	0.551	0.608	0.300	0.950
Gobies (*Pomatoschistus* spp.)	1.000	0.206	0.200	NA	NA
Dover sole	1.000	0.206	0.200	0.050	0.206⁺
European anchovy	1.000	1.000	NA*	*0.900*	*1.000*
Dab	1.000	0.535	NA*	*0.206*	*0.535*
Thin-lipped grey mullet	1.000	0.551	NA	NA	NA
Flounder	1.000	0.231	0.200	0.050	0.231
Smelt (cucumber)	1.000	1.000	NA*	*0.900*	*1.000*
European plaice	1.000	0.206	0.200	0.020	0.206⁺
Atlantic cod	1.000	0.553	0.563	0.180	0.560
Thornback ray	1.000	0.206	0.545	*0.206*‡	0.550
Twaite shad	1.000	1.000	1.000	0.960	1.000
River lamprey	1.000	0.206	0.200	0.110	0.206⁺
European eel	1.000	0.206	0.200	0.110	0.206⁺
Horse-mackerel	1.000	1.000	NA*	*0.900*	*1.000*
Mackerel	1.000	1.000	NA*	*0.900*	*1.000*
Tope	1.000	0.206	NA	NA	NA
Sea Trout	1.000	1.000	1.000	NA	NA
Sea lamprey	1.000	0.206	0.407	NA	NA
Allis shad	1.000	1.000	1.000	NA	NA

* Where there is no FRR information of the species from the Environment Agency TB008 report a range has been applied for similar species groups, ranges are shown in italics. ⁺ Where the TB008 values are lower than those predicted in TR406 Rev 7, the TR406 values are applied. The lower value for best case FRR efficiency applies the TR406 Rev 7 predicted value rather than the Environment Agency TB008 reported value of 0.41, this is a result of the larger trash rack spacing between HPC and SZC.

2.1.7 Interannual variability in the population comparators

Rates of entrapment of fish at Sizewell are influenced by the abundance of fish of different life stages in the coastal waters. Recruitment drives the abundance of larvae and juvenile fish at Sizewell and the distribution of age classes which may be entrapped. Many juveniles inhabit inshore nursery areas. Older fish occur in the Greater Sizewell Bay during seasonal migrations. Most of the fishes impinged at SZB are juveniles. To assess the risks posed by the annual losses of these fish an EAV number or biomass, as described in Section 2.1.4, is compared to the relevant population comparator (Table 6). The stock unit comparators and justification for their application is described in greater detail in (BEEMS Scientific Position Paper SPP103 (Rev 5) [REP6-016]).

Entrapment predictions have been compared to the mean population comparator during the impingement monitoring period (2009-2017), whether it be SSB, landings or a population estimate. To account for the interannual variability in the population comparator, the sensitivity assessment accounts for the variance in the population comparator over the years of the impingement monitoring. The variability in the population comparator between 2009 and 2017 was considered from the mean value and standard error, using bootstrapping of 5,000 iterations assuming a normal distribution.

2.1.7.1 Twaite shad population comparator

There are no spawning populations of twaite shad on the UK east coast. The closest spawning river populations of shad occur in mainland Europe. The Shadow Habitats Regulations Assessment (HRA) Addendum [AS-178] scoped in European sites designated for twaite shad where the site is recorded as having breeding populations. The sites and distances from SZC are:

- Schelde - en Durmeëstuarium van de Nederlandse grens tot Gent SCI - (Scheldt) is located 197km from Sizewell C;
- Unterweser SCI – (Weser) 479km from Sizewell C;
- Weser bei Bremerhaven SCI - (Weser) 483km from Sizewell C;
- Nebenarme der Weser mit Strohauser Plate und Juliusplate SCI – (Weser) 475km from Sizewell C;
- Schleswig-Holsteinisches Elbästuar und angrenzende Flächen SCI – (Elbe) 509km from Sizewell C;
- Unterelbe SCI – (Elbe) 508km from Sizewell C.
- Mühlenberger Loch/Neßsand SCI – (Elbe) 563km from Sizewell C;
- Rapfenschutzgebiet Hamburger Stromelbe SCI – (Elbe) 565km from Sizewell C;
- Hamburger Unterelbe SCI – (Elbe) 582km from Sizewell C;
- Elbe zwischen Geesthacht und Hamburg SCI – (Elbe) 584km from Sizewell C;
- Marais du Cotentin et du Bessin - Baie des Veys SAC - 396km from Sizewell C;
- Tregor Goëlo SAC – 532km from Sizewell C.

Natural England in their Deadline 2 submission [REP2-153] advised that, consistent with a precautionary HRA approach, predicted losses of twaite shad from Sizewell C should be assigned against each breeding population given genetic information is not available to determine the source population.

Given the distance of the proposed development from the spawning rivers (hundreds of kilometres) and the fact the proposed development is in an open coastal environment, it is highly unlikely all fish impinged at Sizewell would come from any one riverine system. Fish monitoring programmes in German and Belgian estuaries are undertaken to determine trends in fish populations. However, to the best of our knowledge, absolute population estimates are not available for the designated sites. A summary of the known information on trends in European river systems and responses to Natural England and Environment Agency comments is provided in BEEMS Scientific Position Paper SPP103 Rev.5 [REP6-016].

In the absence of population estimates for European designated sites, Cefas estimated the twaite shad population of the Elbe and Scheldt located 500km and 200km from SZC, respectively, based on monitoring data provided by European organisations (BEEMS Scientific Position Paper SPP100 [AS-238]). The assumptions and limitations of attempting to estimate population estimates from the available monitoring data was raised in BEEMS Scientific Position Paper SPP100 [AS-238]. Natural England [REP2-153] and the Environment Agency [REP2-135] questioned the uncertainty in the methods applied to determine the population estimate, pointing to factors such as diurnal migration patterns, shoaling behaviours and assumptions of the distribution of fish across the estuary when scaling up estimates for the migratory period. Appendix C attempts to resolve these concerns and provide confidence intervals in the annual estimates. The revised data has also been used for the uncertainty analyses (Table 6). Whilst inherent uncertainty remains, these population estimates provide a best endeavours approach with the best available information from European rivers. It is important that the degree of uncertainty in the population estimate is considered alongside the precaution of apportioning all losses to a single river population and the very small scale of effects as described in Section 3.1.4. It would not be proportional to attempt to determine the population estimates for all twelve additional designated sites screened into the HRA when data is not available, and the predicted impacts are very low.

2.2 Statistical treatment

The uncertainty analyses were computed twice, the first run focusing on the 'uncertainty in impingement predictions' where the FRR mitigation values were fixed to those applied in **BEEMS Technical Report TR406.v7** [AS-238]. This first run only considered the uncertainty in the impingement predictions and population comparators and provided a direct comparison to the DCO assessments in the **Environmental Statement** with the latest results reported in **Appendix 7L** [REP2-110][13]. The second analysis 'uncertainty in entrapment predictions' included the full suite of parameters:

- Upper rate of entrainment.
- Rates of impingement.
- Effects of LVSE mitigation, where a worst-case of zero benefit has been applied.
- Effectiveness of the FRR systems (by applying a range of values proposed by the Environment Agency for the similar FRR design at HPC (TB008)).
- Baseline population comparator.

The resulting distributions of impingement and entrapment were summarised by taking the mean, median, 5th and 95th percentile of the 5000 calculated values. This analysis was carried out in the software R v4.0.2 (R Core Team, 2020) using the packages readxl (v1.3.1) for reading in the input file and dplyr (v1.0.0) for data handling. The detailed calculation steps are shown below.

The comparators to be used in the assessment of impact might be population numbers or biomass (e.g. SSB or catch /landings.

Respectively, scaled impinged numbers (N_{SZC}) or biomass at SZC (B_{SZC}) were calculated from impinged numbers at SZB (N_{SZB}) as:

$N_{SZC} = N_{SZB} \times 2.326 \times 1 \times EAV$,

where 2.326 is SZB to SZC scaling factor, 1 is the LVSE scaling factor, and EAV is Equivalent Adult Value factor. N_{SZC} is subsequently expressed as a proportion of population numbers.

For impinged biomass at SZC:

$B_{SZC} = N_{SZB} \times 2.326 \times 1 \times EAV \times W/1000$,

where 2.326 is SZB to SZC scaling factor, 1 is the LVSE scaling factor, EAV is Equivalent Adult Value factor, W is the mean weight of a mature fish in kg. B_{SZC} is subsequently expressed as a proportion of biomass in tonnes.

To include uncertainty in entrainment, iterations were drawn from a uniform distribution between the lower and upper values of the range of estimated entrained EAV numbers (Table 3), or weight in the case of eels. If the population comparator was expressed in weight (SSB or catch), entrainment numbers were multiplied by weight of a mature individual. Entrainment losses were added to the impingement losses described above to provide a total entrapment number or weight.

Uncertainty in entrapment accounted for the total uncertainty in impingement and entrainment. Uncertainty of the comparators was assessed from their mean values and standard errors assuming normal distributions. For each of the 5000 iterations, a separate value of the comparator was generated from the normal distribution. These values were checked to ensure they were greater or equal to the pre-FRR impingement plus entrainment, i.e., that impossible values of >100% entrapment were not simulated. For comparators without annual values (e.g., river lamprey), and for European anchovy that had highly variable landings, the mean was taken as a fixed value and applied to each iteration (Table 6.). In each case, the annual

[13] SZC Co. Responses to Examining Authority's Written Questions. Appendix 7L Detailed response to questions ExA Ref. Bio 1.242 and 1.243. [REP2-110].

entrapment rate as a percentage of the population comparator was then calculated. From the resulting 5000 estimates of % effect, the average (mean and median), lower (5th percentile), and upper (95th percentile) % effect values were calculated.

2.3 The threshold for effects

To have a negligible impact on the dynamics of a fish population, any predicted annual mortality rate must be considerably less than the rate of mortality that would prevent it from replacing itself on a year-to-year basis. Annual mortality rates of 10%-20% of SSB are typically considered sustainable in international fisheries management practice. Sustainable fishing mortality reference values, using precautionary approaches, vary in well studied commercial fish species between 19% for sea bass to 36% for plaice, above natural mortality for the stocks of relevance to Sizewell. The coefficient of variation of the SSB in species fished around Sizewell (e.g. herring, bass, whiting and others) is estimated by ICES to be 12-58% (Table 10 in BEEMS Technical Report TR406.v7 [AS-238]).

Given these relatively high rates of mortality are known to be sustainable for commercial species, a precautionary threshold of 1% annual mortality as a proportion of population size helps to gauge the risks posed by entrapment. It is important to note that this is not a threshold for changes in spawning population size attributed to entrapment. It is a threshold linked to annual rates of mortality which are deemed to be sustainable. The threshold is justified on the basis that it relates to losses of spawning fish that are an order of magnitude lower than those observed to be sustained by fished populations. Further consideration is given to species of conservation interest, such as the application of smaller population estimates e.g., single river systems in the case of twaite shad and river lamprey (Table 6).

For populations that are not targeted and caught by fisheries, a 1% threshold is even more precautionary and when annual mortality is a few percent of population size any observed variation or trend in spawning population size would be driven by factors other than SZC impingement. This is to say that the rates and timing of increases and decreases in spawning population size, with and without the additional effects of SZC entrapment, would be almost indistinguishable.

Table 6. Stock comparators and interannual variability between 2009-2017 used in uncertainty analyses. The stock areas are described in BEEMS Technical Report TR406.v7 [AS-238] and further justification is provided in BEEMS Scientific Position Paper SPP103 (Rev 5) [REP6-016].

Common name	Comparator	Biomass (t) or numbers									
		2009	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**Mean ‡**
Sprat	SSB (t)	184,795	185,165	164226	132,853	107,152	216,858	346,972	222,571	175,080	**192,852**
Herring	SSB (t)	2,043,590	2,164,870	2,583,390	2,746,510	2,517,680	2,450,220	2,275,330	2,684,890	2,331,180	**2,421,962**
Whiting	SSB (t)	130,622	154,317	142,719	147,948	139,669	132,966	141,379	148,121	156,088	**143,759**
European sea bass	SSB (t)	18,451	18,252	16,815	15,582	13,877	11,333	12,085	10,173	9,395	**13,996**
Gobies (*Pomatoschistus* spp.)	Population numbers	NA	NA	NA	NA	NA	NA	NA	NA	NA	**205,882,353**
Dover sole	SSB (t)	30,520	29,091	26,402	28,880	32,536	28,413	27,390	33,144	30,612	**29,665**
European anchovy	Landings (t)	1,045	1,205	633	842	207	1,042	8,954	1,041	13,039	**3,112**
Dab	Landings (t)	6,561	7,240	6,824	6,095	5,214	4,344	3,595	4,070	2,751	**5,188**
Thin-lipped grey mullet	Estimated SSB (t) based on landings	650.2	739.5	722.1	712.4	584.7	593.6	416.4	378.3	271.7	**563.2**
Flounder	Landings (t)	3,088	3,365	3,193	2,310	1,876	2,067	1,913	1,739	1,262	**2,313**
Cucumber smelt	Estimated SSB (t) (based on EA landings)	NA	NA	20.2 (3.2)	70.4 (11.3)	88.9 (14.2)	69.1 (11.1)	60.4 (9.7)	60.3 (9.6)	8.1 (1.3)	**53.9**
European plaice	SSB (t)	643,553	792,570	824,392	874,478	990,616	1,148,875	1,069,940	1,147,047	1,213,531	**967,222**
Atlantic cod	Landings (t)	16,460	16,333	12,178	11,004	8,591	10,552	10,302	8,539	6,156	**11,124**
Thornback ray	Landings (t)	532.0	490.8	624.8	661.9	752.7	744.0	663.9	717.6	905.0	**677.0**
Twaite shad[14]	Elbe: estimated adult numbers migrating upriver	11,942,180	1,360,327	1,073,925	40,151	96,188	246,775	7,626,827	7,911,800	4,250,363	**3,838,726**

[14] Following comments from the Environment Agency and Natural England estimates of the twaite shad run on the Elbe and Scheldt have been reanalysed see Appendix C.

Common name	Comparator	Biomass (t) or <u>numbers</u>									
		2009	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**Mean ‡**
	Scheldt: estimated adult numbers migrating upriver	No established population.			<u>53,772</u>	<u>7,212</u>	<u>24,543</u>	<u>23,718</u>	<u>160,951</u>	<u>NA</u>	**<u>54,039</u>**
River lamprey	Humber catchment population (t)	NA	NA	NA	NA	NA	NA	NA	NA	NA	**61.9**
European eel	Anglian RDB (t)	NA	NA	87.9	88.1	94.6	61.8	71.6	67.8	NA	**78.6**
Horse-mackerel	Landings (t)	44,533	24,046	27,619	21,023	18,628	13,370	9,354	12,186	13,344	**20,456**
Mackerel	Landings (t)	3,230,003	3,579,017	4,063,019	3,730,890	4,123,080	5,161,009	5,148,898	4,884,807	4,747,484	**4,296,467**
Tope	Landings (t)	649.9	564.4	511.5	466.1	483.3	462.4	500.8	453.7	460.2	**505.8**
Sea Trout	Population numbers	NA	NA	NA	NA	NA	NA	NA	NA	NA	**<u>39,795</u>**
Sea lamprey	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	**NA**
Allis shad	Estimated adult numbers migrating upriver	NA	NA	NA	NA	NA	NA	NA	NA	NA	**<u>27,397</u>**

‡ The reported mean is the average of the reported years. The uncertainty analysis determined the variability in the population comparator between 2009 and 2017 from the mean value and standard error, using bootstrapping of 5,000 iterations assuming a normal distribution.

3 Results & Discussion

3.1 Uncertainty in impingement predictions

The uncertainty analysis was run initially with impingement data and predicted FRR mitigation values as applied in BEEMS Technical Report TR406.v7 [AS-238] (Table 5). Permutations accounted for variation in the predictions of impingement and in the relevant population comparator over the impingement monitoring period. The results can therefore be compared to those in the DCO assessments in the **Marine Ecology and Fisheries Environmental Statement** (Volume 2, Chapter 22 [APP-317]) and with the updated results reported in Table 5 of Appendix 7L [REP2-110][15]. The simulation also allows the performance of the uncertainty analyses to be understood prior to the full entrapment run.

The absolute numbers of equivalent adult fish predicted to be impinged annually at SZC are presented in Table 7. The application of diurnal bias correction factors for the species of conservation interest results in proportional increases in predicted impingement. Absolute numbers of smelt increase by 1,872 equivalent adults per annum to a mean of 18,749. Twaite shad impingement increases by 56 equivalent adults per annum to 2,749. River lamprey and European eel increase by 73 and 66, equivalent adults to 609 and 573, respectively (Table 7). These estimates do not account for the uncertainty in mitigation or entrainment. Final entrapment mortality is considered further in Section 3.2.

Table 7 Predicted impingement numbers for each of the key species as SZC including predicted FRR rates. Conservation species of interest in bold have been treated with a correction factor to account for diurnal biases.

Common name	Impingement, FRR fixed (EAV numbers)			
	Lower 5%	Median	Mean	Upper 95%
Sprat	2,628,859	4,476,417	4,623,145	7,135,480
Herring	1,026,119	1,549,843	1,581,886	2,242,236
Whiting	226,569	290,393	293,089	368,138
European sea bass	41,271	75,847	79,079	126,179
Gobies (*Pomatoschistus* spp.)	47,960	94,406	99,598	169,061
Dover sole	6,821	9,141	9,257	12,143
European anchovy	49,763	123,067	144,491	313,388
Dab	19,639	29,303	30,571	46,417
Thin-lipped grey mullet	1,895	4,704	4,938	8,595
Flounder	2,713	3,382	3,423	4,275
Cucumber smelt	12,561	18,534	18,749	25,771
European plaice	1,069	1,530	1,562	2,166
Atlantic cod	1,425	3,159	3,279	5,603
Thornback ray	180	262	266	366
Twaite shad	1,529	2,642	2,749	4,392
	1,529	2,642	2,749	4,392
River lamprey	366	582	609	938
European eel	385	563	573	795
Horse-mackerel	581	1,364	1,560	3,232
Mackerel	26	217	277	719
Tope	0	10	11	35
Sea Trout	0	0	8	32
Sea lamprey	0	0	1	4
Allis shad	0	0	0	0

[15] SZC Co. Responses to Examining Authority's Written Questions. Appendix 7L Detailed response to questions ExA Ref. Bio 1.242 and 1.243.

For the key species at Sizewell, the mean impingement rates are below 1% of the relevant population comparators and corresponded to previous results. The exception is the effect of SZC impingement on twaite shad when the single River Scheldt population estimate is applied as the population comparator. Apportioning impingement losses to individual river populations follows the advice of Natural England and is explored in more detail in Section 3.1.4.

The only changes in the impingement uncertainty analysis since Revision 1 of this report, and as reported herein, are the application of a correction factor to account for potential diurnal bias in impingement rates due to bulk sample overflows, and modified population estimates for the Elbe and Scheldt twaite shad river populations in response to regulatory comments. In the case of smelt, after accounting for the potential for higher impingement risks at night, the rate of impingement as a percentage of the conservatively estimated Anglian population is predicted to be 0.60% (0.35% as a 5th percentile – 0.96% as a 95th percentile). Further discussion on smelt is provided in Section 3.1.3. River lamprey impingement increases to 0.08% (0.05% as a 5th percentile – 0.12% as a 95th percentile) of the Humber catchment population size. European eel impingement increases to 0.24% (0.16% as a 5th percentile – 0.34% as a 95th percentile) of the estimated numbers in the Anglian RDB.

The population level effects on the other species remain largely unchanged (within ± 0.5% relative change). These very small changes are reflective of the nature of the uncertainty analysis which is based on permutation testing. Sprat and herring are the most frequently impinged species accounting for 69% of total annual impingement numbers. Losses at the population level due to impingement at SZC, assuming no benefit from the LVSE heads or FRR mitigation, are predicted to be 0.03% for sprat and 0.01% for herring. In both cases upper 95th percentile losses are well below 0.05% of the population (Table 8).

Whiting is the third most impinged species at SZB and impingement losses from SZC after accounting for FRR mortality equate to 0.06% of the population as a mean and 0.07% as an upper 95th percentile (Table 8).

3.1.1 Sea bass impingement effects

The estimated annual loss of sea bass after accounting for FRR mortality is 0.87% of SSB with an upper 95th percentile estimate of 1.40% (Table 8). These estimates are considered to be precautionary because sea bass are not uniformly distributed within the Greater Sizewell Bay. Survey data has identified low catches in offshore otter trawl surveys, with 95% of sea bass caught inshore of the Sizewell-Dunwich Bank. This suggests that impingement predictions scaled-up from SZB may overestimate sea bass impingement at the offshore SZC intakes (see BEEMS Technical Report TR406.v7, Section 4.1.1.1. report pg. 30 of [AS-238]). Furthermore, the diurnal bias screening exercise (Section 2.1.2) indicated that bass may be more susceptible to impingement during daylight hours. Given the greater proportion of daylight samples used to calculate impingement predictions this may lead to an overestimate in sea bass impingement. On a precautionary basis, neither factor has been incorporated into the assessment.

To provide additional confidence in the assessment of no significant effects on viability of sea bass populations a full ICES stock assessment was completed (BEEMS Scientific Position Paper SPP118 [REP8-131]). The results of the stock assessment showed no discernible effects on population trends and only very minor effects on absolute SSB were observed despite the application of highly precautionary loss estimates. The stock assessment results are summarised in Section 3.2.1.1.

3.1.2 Cod impingement effects

The latest ICES advice points to growing evidence that cod in the North Sea may form two separate populations: the northern ‘Viking’ population, and the southern ‘Dogger’ population. Impingement at Sizewell would cause losses from the southern Dogger population. The SSB of the two populations remains unresolved, therefore the comparator applied is based on landings estimates from areas within the Dogger cod range (further details are provided in BEEMS Scientific Position Paper SPP103 (Rev 5) [REP6-016]). Losses of cod relative to conservative landings estimates are 0.08% as a mean and 0.14% as a 95th percentile. Such low losses relative to fisheries landings would have no significant impacts on the population of cod (Table 8).

3.1.3 Smelt impingement effects

Smelt in the coastal waters around Sizewell and in Suffolk are considered to belong to a population associated with the Norfolk Broads and the estuarine and brackish waters around Great Yarmouth and Lowestoft (Maitland, 2003b). 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 population and very large numbers have been observed in the River Elbe in Germany (BEEMS Scientific Position Paper SPP100).

No existing population estimate for smelt is available. For the purposes of assessing impacts on smelt, an 'Anglian' smelt population SSB has been estimated based on Environment Agency landings data from the Anglian Region. The Environment Agency manages a restrictive licensing of smelt fisheries. For the years with catch data the mean landings from the east coast Anglian Region between 2009-2017 were 8.63t. Based on the restricted landings an assumption is made that the landings represent the maximum sustainable harvesting rate for smelt in the region of approximately 16% (BEEMS Technical Report TR406.v7 [AS-238]). This implies that a conservative estimate of SSB is 53.9t. Losses of smelt from the proposed SZC station accounting for diurnal bias, but with no mitigation benefits assumed, represent 0.60% as a mean (0.35 as a 5th percentile and 0.96% as a 95th percentile of the SSB. Such losses will not have a significant effect on smelt population dynamics.

The Environment Agency have expressed concerns relating to the impingement of smelt and the potential for impact on the Alde & Ore Estuary with the justification that impingement rates may exceed the reproductive capacity of the fish in the Alde & Ore and the rate of immigration from other river systems. The Alde & Ore Estuary is located 25km south of SZC with the Suffolk Estuaries of the Orwell and Stour approximately 40km south. Approximately 30-40km north of SZC is the Bure & Waveney & Yare & Lothing complex for which previous tagging studies described riverine spawning migrations (Moore *et al.,* 2015). Greatest impingement of smelt occurs during summer feeding when smelt are in coastal waters and fish from a number of spawning rivers are likely to be impinged (see BEEMS Scientific Position Paper SPP103.v5, Section 3.6.2.1 and Figure 10 [REP6-016]). This is because the genetic homogeneity of smelt from at least the Ouse to the Thames indicates mixing of smelt from different watercourses and it is unlikely that the proposed development would affect smelt from a single watercourse.

Furthermore, factors other than entrapment at SZC are likely to have the overriding influence on the status of smelt in the Alde & Ore. Smelt spawn in suitable habitat in upper estuaries and in freshwater. Within the Alde & Ore there are barriers to upstream migration beyond the upper estuary. The tide gates at Snape Maltings are *"considered to be impassable for smelt and therefore likely to be hindering the reproductive capacity of the population due to restricted access to spawning habitat. Fish and eel pass feasibility assessments completed by the Environment Agency confirm that the structure is considered impassable for all fish species (Wood, Environment Agency 2016 pers. comm.)"* (extract from Natural England, 2018). Proposals being developed in consultation with the Environment Agency and described in the draft Fish Impingement and Entrainment Monitoring Plan (Doc Ref. 10.8) being submitted at Deadline 10, include installation of fish passes at Snape Maltings and Blythford Bridge as well as monitoring of smelt in the Alde and Blyth (secured by Schedule 11 of the Deed of Obligation (Doc. Ref. 8.17(H)). These measures have the potential to improve access to spawning habitat for smelt and benefit other diadromous species in the Alde & Ore and Blyth waterbodies.

3.1.4 Twaite shad impingement effects

In the case of twaite shad, two factors may potentially lead to changes in conclusions about the effects of impingement: the correction factor for diurnal bias and a revised estimate of the population comparator. At 1.021, the correction factor for diurnal bias has a minor influence on impingement predictions for twaite shad (Section 2.1.2.4).

There are no spawning populations of twaite shad on the UK east coast. The closest breeding populations of shad occur in rivers in mainland Europe.

Given the distance of the proposed development from the spawning rivers (hundreds of kilometres) and the fact that the SZC development is in an open coastal environment, it is highly unlikely that all fish impinged at

Sizewell would come from any given riverine system. However, such a scenario is considered for two European systems where population estimates have been made: the Elbe, approximately 500km from SZC, and the Scheldt, approximately 200km away. The approach to estimating the population size in these two systems is detailed in Appendix C.

The estimated mean impingement effect for twaite shad is highly skewed by the variance in the estimated population size. This statistical artefact is exemplified by the fact that the mean of the population effect for the Elbe (0.80%) is greater than the 95th percentile population effect (0.22%). In cases where there is large variance in the comparator, the median is a more reliable value and shows effects of 0.07% for the Elbe and approximately 5% for the Scheldt (Table 8). In both cases these increased values in comparison to Revision 1 of this report reflect the smaller population estimates adopted.

In the case of the Scheldt, the mean population size between 2012 and 2017 following recovery of a breeding population in 2012 was estimated at 54,039 with 95% confidence intervals between 20,412 to 130,203 fish (Appendix C). If all the twaite shad predicted to be impinged by SZC were from the Scheldt alone, the losses would account for nearly 5% of the estimated Scheldt population. Impingement monitoring at SZB has recorded twaite shad throughout the monitoring period (2009-2017), whereas recovery in the Scheldt occurred in 2012 with no spawning adults recorded in 2011. Therefore, it is not possible that all the twaite shad impinged at Sizewell originate from the Scheldt.

The number of twaite shad observed in the Scheldt Estuary varies greatly from year to year, both the number of migratory adults in the spring and the number of juveniles in summer and autumn. Adults are now found every year, however, recruitment of juveniles was observed in 2012, 2015, 2017, 2018, 2019 and 2020 (INBO, 2021). It is likely that this establishing population, is still dependent on the arrival of fish from elsewhere. Twaite shad exhibit site fidelity with > 90% of fish returning to natal rivers to spawn, the remaining fish (e.g. 3% reported in Davies *et al.,* 2020) may stray to other systems. The Elbe contributes to the greatest numbers of twaite shad in German estuaries and recovery of the Scheldt (and Weser) population is likely to be seeded from fish straying from surrounding systems such as the Elbe.

It is acknowledged that it is not possible to determine exactly where the twaite shad impinged at Sizewell have come from. However, genetic studies of North Sea twaite shad demonstrate mixing which is consistent with the assumption that the Weser and Scheldt population recoveries have been seeded from fish originating in the Elbe (see BEEMS Scientific Position Paper SPP103 (Rev 5) [REP6-016]). The predicted scale of losses from SZC are therefore considered to have negligible impacts on the breeding populations of shad in European rivers.

The results of the uncertainty analysis presented in Table 8 provide both the mean and the upper 95th percentile estimate for population impacts on the two river populations. In the *Appropriate assessment of the application to vary the water discharge activity permit for Hinkley Point C* (Environment Agency, 2020), the Environment Agency describe the application of upper (99th percentile) estimates to be overly precautionary when apportioning losses to a single river population. This was in recognition that there is a low probability of Hinkley Point C impinging twaite shad all originating from the same river. The rivers of concern for the Appropriate Assessment feed into the Severn Estuary. Given that the closest rivers with twaite shad are 200km from SZC it would seem consistent that the application of a 95th percentile is overly precautionary and in the case of the Elbe, notwithstanding residual uncertainties in the population estimate, the estimate of population effects should be taken as 0.22%.

Table 8. Uncertainty analysis for impingement with fixed predicted FRR rates for key fish species at SZC. Cells in green are below the initial 1% screening threshold. Cells in red indicate values in exceedance of the initial screening 1% threshold and are subject to further investigation

Common name	Impingement, FRR fixed (% of Comparator)				Comparator	Comment / change from SPP116.v1
	Lower 5%	Median	Mean	Upper 95%		
Sprat	0.014	0.025	0.026	0.041	SSB	Change in mean effect < ±0.5%
Herring	0.008	0.012	0.012	0.018	SSB	Change in mean effect < ±0.5%
Whiting	0.045	0.058	0.058	0.074	SSB	Change in mean effect < ±0.5%
European sea bass	0.447	0.836	0.871	1.397	SSB	Change in mean effect < ±0.5%
Gobies (*Pomatoschistus* spp.)	0.023	0.046	0.048	0.082	Population estimate	Change in mean effect < ±0.5%
Dover sole	0.005	0.007	0.007	0.009	SSB	Change in mean effect < ±0.5%
European anchovy	0.033	0.082	0.096	0.209	Landings	Change in mean effect < ±0.5%
Dab	0.015	0.023	0.024	0.038	Landings	Change in mean effect < ±0.5%
Thin-lipped grey mullet	0.178	0.436	0.461	0.822	Estimated SSB	Change in mean effect < ±0.5%
Flounder	0.009	0.012	0.012	0.016	Landings	Change in mean effect < ±0.5%
Cucumber smelt	0.351	0.570	0.602	0.964	Estimated SSB	+11.1% in mean effect due to diurnal effects see Section 3.1.3
European plaice	0.000	0.000	0.000	0.000	SSB	Change in mean effect < ±0.5%
Atlantic cod	0.033	0.074	0.078	0.136	Landings	Change in mean effect < ±0.5%
Thornback ray	0.084	0.124	0.126	0.177	Landings	Change in mean effect < ±0.5%
Twaite shad	0.033	0.071	0.796*	0.222	Elbe population estimate	-10.0% in mean effect due to diurnal effects & population comparator. See Section 3.1.4.
	2.110	4.933	9.624*	27.957	Scheldt population estimate	+14.0% in mean due to diurnal effects & population comparator. See Section 3.1.4
River lamprey	0.047	0.074	0.078	0.120	Humber population	+13.5% in mean effect due to diurnal effects see Section 3.1
European eel	0.158	0.236	0.241	0.340	RDB	+12.8% in mean effects due to diurnal effects see Section 3.1
Horse-mackerel	0.000	0.001	0.001	0.002	Landings	Change in mean effect < ±0.5%
Mackerel	0.000	0.000	0.000	0.000	SSB	Change in mean effect < ±0.5%
Tope	0.000	0.013	0.016	0.049	Landings	Change in mean effect < ±0.5%
Sea Trout	0.000	0.000	0.020	0.080	Catch numbers	Change in mean effect < ±0.5%
Sea lamprey	NA	NA	NA	NA	NA	NA
Allis shad‡	0.000	0.000	0.000	0.000	Population estimate	NA

* High mean values are a statistical artefact of extreme outputs generated due to the variance in the population estimate. In such a case the median is a more reliable comparator. In the case of the Scheldt, where population recovery only occurred in 2012, these estimates are not realistic worst-case as described in Section 3.1.4.

‡ A single allis shad was impinged on the 28th May 2009 in an invalid bulk sample, meaning impingement predictions are not available for the species. However, impact assessments continue to consider the species as present and acknowledge its occurrence in the impingement record.

3.2 Uncertainty in Entrapment Predictions: Full uncertainty analysis

The uncertainty in entrapment predictions presented in Table 10 provides a more comprehensive analysis of the effects of uncertainty in the input parameters. The full uncertainty analysis includes:

- Upper rate of entrainment.
- The potential entrainment gap for sprat, gobies (*Pomatoschistus* spp.) and herring.
- The full variation in the calculated distribution of predicted impingement rates.
- Application of a correction factor to account for potential diurnal bias in smelt, river lamprey, twaite shad and European eel.
- No benefit of the LVSE mitigation.
- The effectiveness of the FRR system (a range of values proposed by the Environment Agency for the similar, albeit more complicated[16], FRR design at HPC (TB008) is applied).
- Variation in the baseline population comparator.

The absolute numbers of equivalent adult fish predicted to be entrapped annually at SZC are presented in Table 9.

Table 9 Predicted entrapment numbers for each of the key species as SZC including ranges in FRR efficiency. Conservation species of interest in bold have been treated with a correction factor to account for diurnal biases, species underlined have been corrected for the potential 'entrainment gap'.

Common name	Entrapment, FRR range (EAV numbers)			
	Lower 5%	Median	Mean	Upper 95%
<u>Sprat</u>	3,070,074	4,858,726	5,011,192	7,451,279
<u>Herring</u>	1,013,128	1,512,085	1,543,899	2,174,881
Whiting	218,853	367,142	374,829	562,135
European sea bass	36,527	83,429	90,164	168,152
<u>Gobies (*Pomatoschistus* spp.)</u>	3,529,358	3,575,804	3,580,996	3,650,459
Dover sole	3,138	6,268	6,446	10,382
European anchovy	50,243	120,643	140,138	301,044
Dab	32,130	41,588	43,043	58,682
Thin-lipped grey mullet	1,895	4,704	4,938	8,595
Flounder	851	2,024	2,070	3,473
Cucumber smelt	11,910	17,565	17,819	24,622
European plaice	213	819	858	1,665
Atlantic cod	760	2,012	2,205	4,274
Thornback ray	251	472	489	788
Twaite shad	1,497	2,594	2,694	4,305
	1,500	2,584	2,693	4,304
River lamprey	254	443	468	765
European eel	317	480	496	721
Horse-mackerel	550	1,303	1,483	3,056
Mackerel	25	206	263	686

[16] The Hinkley Point C FRR system has an additional 'handling' element due to an Archimedes' screw which carries the fish to a sufficient elevation to drain back to see under gravity, larger drum screens resulting in longer retention times and a longer route of return to the sea.

Common name	Entrapment, FRR range (EAV numbers)			
	Lower 5%	Median	Mean	Upper 95%
Tope	0	10	11	35
Sea Trout	0	0	8	32
Sea lamprey	0	0	1	4
Allis shad	0	0	0	0

For the key species at Sizewell, the mean entrapment predictions are below 1% of the relevant population comparators for all species except gobies (*Pomatoschistus* spp.) and sea bass. The addition of the FRR uncertainty ranges (with upper estimates for survival below those predicted in the DCO assessments (BEEMS Technical Report TR406.v7 [AS-238])) caused small increases in predicted losses for a number of species including whiting, sea bass, and thornback ray. The upper uncertainty range for the FRR mitigation results in no benefit (100% mortality) for impinged whiting and only 5% reductions in mortality for sea bass. In these cases, the upper entrapment estimates reported in Table 10, represent a highly precautionary scenario with effectively no mitigation. Whiting is subject to negligible entrainment, but the uncertainty analyses incorporate the Environment Agency ranges in FRR effectiveness from 59% survival as a best case, to 0% survival as a worst-case (Table 5). As a result, the predicted losses of whiting increase from 0.06% of the population SSB as a mean (Table 8) to 0.08% of the SSB and 0.11% as an upper 95th percentile (Table 10). This precautionary uncertainty assessment demonstrates that the impact of the station is not significant at the population level.

The full uncertainty analysis included the addition of numbers in the entrainment gap and upper entrainment estimates (Table 3) for sprat, herring and gobies (*Pomatoschistus* spp.). Effects on gobies are considered in Section 3.2.1.2. In the case of sprat, the increase in the full uncertainty analysis was estimated to cause population level effects of 0.03% (0.02% as a 5th percentile – 0.04% as a 95th percentile), representing a minor increase in comparison to Revision 1 of this report. A similar situation is seen for herring where mean population level effects of 0.01% (0.008% as a 5th percentile – 0.02% as a 95th percentile) are now predicted (Table 10).

For many species including European eel and river lamprey, and for the epi-benthic species such as dab, flounder, plaice and sole, the range in FRR mortality proposed by the Environment Agency in TB008 indicates that the FRR may be more effective than assumed in the DCO assessments (BEEMS Technical Report TR406.v7 [AS-238]).

3.2.1.1 Sea bass population level effects

The predicted mitigation efficiency for sea bass is 0.551 indicating approximately 45% survival. The Environment Agency uncertainty data for FRR mitigation ranges from 70% survival to just 5% survival (Table 5). When the mitigation uncertainty is incorporated into the entrainment predictions the mean annual loss of sea bass is 0.99% of SSB with an upper 95th percentile estimate of 1.85% and a 5th percentile of 0.40% (Table 10). These figures do not account for the distribution of sea bass within the Greater Sizewell Bay and are likely to overestimate bass entrapment at SZC (see Section 3.1).

Whilst the EAV-based risk assessment indicates that the effects of the station do not pose a risk to the sustainability of the population, a full ICES stock assessment has been completed for this species incorporating losses from the station as an additional mortality term. Sea bass was selected for the stock assessment on the basis that it is the 4th most impinged species at Sizewell B and, along with gobies (*Pomatoschistus* spp.) , has the highest predicted annual rate of impingement as a proportion of spawning population size. Sea bass is a long-lived, repeat spawning species. As a commercially targeted species, sea bass is a data-rich species with information on the full life-history, migratory behaviour, population genetics and stock dynamics available. Well-established, internationally reviewed and accepted stock models are also available for assessing sea bass stock dynamics.

The full stock assessment is presented in BEEMS Scientific Position Paper SPP118 [REP8-131] with a summary provided herein. Annual impingement predictions for SZC under a range of precautionary scenarios were added as an extra source of mortality and included within the existing ICES sea bass stock assessment from 1985 to 2020 to demonstrate the long-term effects had SZC been operational throughout the assessment period. Mean and upper 95% confidence interval impingement estimates for SZC were incorporated into historic estimates of sea bass mortality to simulate a scenario with SZC operating for 35 years. The estimated sizes of the spawning populations of sea bass, with the simulated SZC impingement mortality was then compared to the core ICES assessment without SZC. Impingement predictions included an extreme worst-case scenario with the upper 95% confidence interval (U95) of annual unmitigated impingement rates assumed in every year for the 35-year assessment period. Assessments also considered the effects of the FRR system mitigation by assuming mean and U95 impingement predictions.

In all scenarios tested, including the extreme worst-case SZC scenario, impingement had no discernible effects on the population trends and only very minor effects on absolute SSB. That is, the size of the spawning population would still have increased and decreased at the same times and at almost identical rates whether or not SZC impingement was occurring. This is particularly evident during the periods of spawning biomass decline in the 1980's, and more recently during the 2010's. During this potentially sensitive period from 2010-2018 of low biomass (coinciding with CIMP) the population trends are barely discernible with or without the addition of SZC impingement mortality.

Commercial and recreational fisheries mortality dominate the mortality on sea bass with the addition of SZC impingement making negligible differences. This is to be expected as the vast majority of sea bass impinged at Sizewell are 0-3-year-old fish and below the minimum conservation reference size (MCRS) currently set at 42cm. Whereas fisheries mortality is more intensive and targeted at 4–15-year-old fish.

The application of the ICES stock assessments incorporating precautionary SZC impingement estimates for a duration of 35 years provides powerful evidence that there is no significant impact on population trends and impingement effects would not pose a risk to the viability of the population. The stock assessments confirm the results and conclusions drawn from the EAV-based risk assessment.

3.2.1.2 Gobies (Pomatoschistus spp.) population level effects

The term 'sand gobies' has been applied within DCO documents as a shorthand to describe a taxa comprising gobies of the genus *Pomatoschistus* spp. of which the sand goby (*P. minutus*) is the dominant species'. Together Against Sizewell C (TASC) in their Deadline 2 Submission - Written Representation - Ecological Impacts [REP2-481h] correctly point to the fact that in the southern North Sea the genus is represented by several species. The dominant species representing the *Pomatoschistus* spp. genus in the area relevant to Sizewell are sand goby *P. minutus*. For example, in research surveys carried out near Sizewell (ICES rectangle 33F1) from 1982 -2010, *P. minutus* represented over 95% of all captured Gobiidae of the different genera including unidentified confamiliars (70,635 out of 73,854 – Cefas data). As explained in BEEMS Technical Report TR318 [APP-324], 87% of all genera of gobies impinged at Sizewell B are *Pomatoschistus* spp., consequently this species group has been treated as a key taxa and assessed accordingly.

As an unexploited stock, data on population estimates for goby species in not available. Predicted entrapment losses of gobies of the genus *Pomatoschistus* spp. have been compared to a population estimate for *Pomatoschistus* spp. based on data from Cefas Young Fish Surveys (YFS). The approach is explained in BEEMS Technical Report TR406.v7 [AS-238]. As such, entrapment losses are compared to population estimates at the same taxonomic resolution.

Gobies (*Pomatoschistus* spp.) are the species most influenced by the addition of entrainment data in the overall entrapment assessment. This is because most gobies are entrained rather than impinged (Appendix B.4). The inclusion of the entrainment gap resulted in an increase of 17.5% in the total number of equivalent adult gobies estimated to be impinged by SZC. This additional mortality was factored into the assessment and the uncertainty analysis estimated mean annual entrapment of 1.74% of the population estimate with a 95th percentile of 1.77% and a 5th percentile of 1.71% (Table 10). The small range reflects the application of the upper entrainment estimate (rather than the range as applied in Revision 1 of this report) and the fixed FRR mortality rates (Table 5).

The entrapment losses for gobies are considered precautionary as the small impingement fraction is assigned a precautionary EAV of 1 and 100% mortality is assumed for the entrainment fraction. Survival rates of entrained goby larvae has been reported between 88-98% at the Calver Cliffs Nuclear Power Plant

(Mayhew *et al.,* 2000). As such the assumption of 100% mortality is likely to substantially overestimate entrainment losses.

Gobies (*Pomatoschistus* spp.) are a short lived, fast maturing, highly fecund species with high degrees of natural variability. They are ubiquitous in European coastal areas to at least a depth of 20m. The species produces pelagic larvae which are dispersed by tidal currents resulting in a lack of genetic diversity over the southern North Sea (BEEMS Scientific Position Paper SPP103 (Rev 5)). Because of the absence of a fishery, their short lifespan and early age of maturity, gobies will be able to sustain additional mortality rates greater than the precautionary 10% SSB threshold applied in the **Marine Ecology and Fisheries Environmental Statement** (Volume 2, Chapter 22 [APP-317]). Based on the principles of fisheries management, a sustainable harvesting rate for a short lived species with natural mortality (M) of 3.3 may be as much as 50% (Section 5.1.1 of BEEMS Technical Report TR406.v7 [AS-238]). Therefore, the precautionary assessment undertaken here where predicted losses are below 2% is not considered to have any significant effects on the viability of the population.

It is noteworthy that 10% thresholds for non-exploited species have previously been applied for major DCO projects. For example, the Thames Tideway Strategy Group comprising representatives from the Environment Agency, Port of London Authority, Thames Water and other stakeholders considered annual mortality rates of up to 10 % (due to hypoxia) to be sustainable for all species not subject to fishing mortality (further details are available in Section 5.1.4 of BEEMS Technical Report TR406.v7 [AS-238]). The predicted level of losses of gobies (*Pomatoschistus* spp.) is not regarded as significant at the population level.

3.2.1.3 European eel Anglian RDB level effects

Effects of SZC entrapment on European eel are predicted to be equivalent to a mean of 496 equivalent adults per annum (Table 9) representing 0.21% of the Anglian RDB biomass and 0.31% as a 95th percentile (Table 10). The increase in estimated eel losses in Revision 2 of this report reflect the application of a diurnal bias correction factor (Section 2.1.2.2) and the application of the highest entrainment estimate (Table 3). These levels of effects would not be significant on the Anglian RDB eel population.

It is acknowledged that whilst the impingement predictions are precautionary due to the application of the maximum EAV (Section 2.1.4), the Environment Agency maintain concerns relating to the uncertainty relating to entrainment of glass eels (Environment Agency Summary of Oral Case for ISH10: Biodiversity and Ecology [REP7-131]). Responses to these comments are submitted at Deadline 10 (Appendix B of Doc. Ref. 9.120).

Whilst the risk of the station to glass eels is considered to remain very low, the commitment to entrainment monitoring within the Fish Impingement and Entrainment Monitoring Plan (Deadline 10 (Doc Ref. 10.8)) and enhancement measures has been agreed. As part of the ongoing consultation with the Environment Agency in relation to the Eels Regulations, SZC Co. has proposed to contribute to the installation of fish passes in relevant local rivers: at Snape Maltings on the River Alde and Blythford Bridge on the River Blyth. The provision for contributions to the Snape Maltings and Blythford Bridges schemes is secured in the Deed of Obligation (Doc. Ref. 8.17(H)). The schemes proposed will benefit not just eels but other diadromous fish that migrate between the sea and rivers, including smelt. A further contribution, also secured in the Deed of Obligation (Doc. Ref. 8.17(H)) to additional schemes may be made depending on monitoring results for smelt in relation to the Water Framework Directive (WFD).

Table 10. Full uncertainty analysis for entrapment of key fish species at SZC. Cells in green are below the initial 1% screening threshold. Cells in red indicate values in exceedance of the initial screening 1% threshold and are subject to further investigation.

Common name	Entrapment, FRR range (% of Comparator)				Comparator	Comment / change from SPP116.v1
	Lower 5%	Median	Mean	Upper 95%		
Sprat	0.016	0.027	0.028	0.043	SSB	+8.4% in mean effect due to treatment of entrainment data.
Herring	0.008	0.012	0.012	0.017	SSB	+1.9% in mean effect due to treatment of entrainment data.
Whiting	0.043	0.073	0.075	0.113	SSB	Change in mean effect < 0.5%
European sea bass	0.395	0.913	0.993	1.851	SSB	Change in mean effect < 1.0%
Gobies (*Pomatoschistus* spp.)	1.714	1.737	1.739	1.773	Population estimate	Changes driven due to treatment of entrainment data see Section 3.2.1.2.
Dover sole	0.002	0.005	0.005	0.008	SSB	Change in mean effect < 1.5%
European anchovy	0.033	0.080	0.093	0.201	Landings	Change in mean effect < 0.5%
Dab	0.024	0.033	0.034	0.048	Landings	Change in mean effect < 0.5%
Thin-lipped grey mullet	0.178	0.436	0.461	0.822	Estimated SSB	Change in mean effect < 0.5%
Flounder	0.003	0.007	0.007	0.013	Landings	Change in mean effect < 0.5%
Cucumber smelt	0.332	0.542	0.572	0.918	Estimated SSB	+11.2% in mean effect due to diurnal effects see Section 2.1.2.3.
European plaice	0.000	0.000	0.000	0.000	SSB	Change in mean effect < ±0.5%
Atlantic cod	0.018	0.047	0.052	0.104	Landings	Change in mean effect < ±0.5%
Thornback ray	0.117	0.223	0.232	0.376	Landings	Change in mean effect < 1.0%
Twaite shad	0.032	0.069	0.780*	0.218	Elbe population estimate	-10.0% in mean effect due to diurnal effects & population comparator see Section 3.1.4.
	2.075	4.828	9.425*	27.316	Scheldt population estimate	+13.9% in mean due to diurnal effects & population comparator see Section 3.1.4.
River lamprey	0.032	0.057	0.060	0.098	Humber population	+13.8% in mean effect due to diurnal effects
European eel	0.131	0.201	0.209	0.308	RDB	+16.7% in mean effects due to diurnal effects and treatment of entrainment data see Section 3.2.1.3.
Horse-mackerel	0.000	0.001	0.001	0.002	Landings	Change in mean effect < 0.5%
Mackerel	0.000	0.000	0.000	0.000	SSB	Change in mean effect < 0.5%
Tope	0.000	0.013	0.016	0.049	Landings	Change in mean effect < 0.5%
Sea Trout	0.000	0.000	0.020	0.080	Catch numbers	Change in mean effect < 0.5%
Sea lamprey	NA	NA	NA	NA	NA	NA
Allis shad ǂ	0.000	0.000	0.000	0.000	Population estimate	NA

* High mean values are a statistical artefact of extreme outputs generated due to the variance in the population estimate. In such a case the median is a more reliable comparator. In the case of the Scheldt, where population recovery only occurred in 2012 these estimates are not realistic worst-case estimates as described in Section 3.1.4.

ǂ A single allis shad was impinged on the 28th May 2009 in an invalid bulk sample, meaning impingement predictions are not available for the species. However, impact assessments continue to consider the species as present and acknowledge its occurrence in the impingement record

3.3 In built precaution in entrapment assessments

The uncertainty analysis has, where feasible, quantified the degree of uncertainty in the various input parameters underlying the population level assessment. The results of any monitoring programme or sampling campaign are bounded by the limitations and assumptions of sampling. Impingement and entrainment sampling is no different. However, the quality of the data used in the prediction of entrapment effects must be recognised.

Data used to predict the effects of SZC has been collected from an existing operational station where the intakes are less than 3km away. The volume of water sampled from impingement monitoring is substantially greater than would be achievable through fisheries surveys had an existing station not been operational. Whilst operational outages have occurred, when monitoring is not possible, impingement data has been collected for 8 years and consists of 205 monitoring visits. On 100 occasions a 24-hour impingement estimate has been achieved from daylight and overnight bulk samples. Whilst this may lead to potential diurnal bias, steps have been taken to account for underestimates in impingement predictions. For all species investigated, except for gobies (*Pomatoschistus* spp.), diurnal effects are minor. Seasonal sampling over multiple years allows seasonal and interannual variability to be accounted for and increases the probability of sampling rare species. The application of bootstrapping approaches allows variability in the data to be incorporated into annual predictions and the generation of associated confidence intervals. For these reasons the entrapment data set is a very powerful resource for predicting population level effects.

With the exception of sea bass and gobies (*Pomatoschistus* spp.), mean annual entrapment rates as a percentage of spawning population size are below 1%, as are the upper 95th percentile rates (Table 10). From this we infer very low risk for any of the species. Once the life-history of gobies and the distribution of sea bass in the Sizewell Bay relative to the proposed SZC intakes is accounted for, no significant effects on their population dynamics are predicted. In the case of sea bass the conclusions from the risk assessment are bolstered and confirmed by the application of a stock assessments that include SZC mortality (BEEMS Scientific Position Paper SPP118 [REP8-131]).

The uncertainty analysis has assumed no benefit from the LVSE head mitigation and considered a range of FRR effectiveness values produced by the Environment Agency for HPC (TB008). Revision 2 of this report has applied a correction factor to account for the potential diurnal bias introduced by a greater proportion of daylight samples in the CIMP and quantified the entrainment gap for three species most likely to be subject to underestimation in entrainment predictions.

Whilst it is possible that a degree of uncertainty remains for some species it is necessary to consider the magnitude of such uncertainties in relation to the magnitude of impacts, species by species, and given the already inbuilt precaution in the entrapment assessments.

The precautionary steps in the entrapment assessments include:

- **Fishing mortality has not been included when calculating the Equivalent Adult Value (EAV) factor.** This results in EAV numbers and EAV biomass being overestimated i.e., the juvenile fish entrapped would have less chance of surviving to contribute to the spawning population had fishing mortality during juvenile stages been considered (Section 2.1.4.1).
- **Precautionary EAV biomass.** The EAV biomass is calculated by multiplying the EAV number by the mean adult fish weight from the spawning population. The individual weight at the age at first maturity will be lower than the individual weight of older and more fecund fish in the spawning population. Multiplying lost numbers at the age of maturity by mean individual biomass in the spawning population will upweight apparent losses of spawners due to entrapment and their potential contribution to the spawning population biomass. This correctly results in a precautionary higher rate of annual EAV biomass loss as a percentage of spawning population biomass for repeat spawning species (Section 2.1.4.1).
- **For many species, an EAV of 1 has been assumed.** Notably these species include twaite shad, river lamprey and European eel. This assumes all fish impinged would have survived to contribute to the spawning population (Section 2.1.4.1).
- **No benefit of the LVSE head has been assumed** (Section 2.1.5)**.**

- **The FRR mortality is likely to be precautionary due to improved design features.** The uncertainty range in the FFR efficiency is based on Environment Agency values for HPC with a fine trash rack spacing and a greater tidal range. In addition, Sizewell has dedicated FRR tunnels for each EPR without the requirement for an Archimedes screw to raise the fish. Therefore, SZC FRR mortality rates would be expected to be lower than at HPC (Section 2.1.6).
- **Single river estimates:** Losses of conservation species such as twaite shad are considered precautionary as the losses are apportioned to single river systems hundreds of kilometers from Sizewell individually. The likelihood is the fish originate from a number of sources.
- **Only correcting for diurnal bias underestimates:** Where diurnal bias may lead to underestimates in impingement predictions, a correction factor has been applied to correct for the bias. However, when diurnal bias may have led to overestimates of impingement rates no such correction factor has been applied.
- **Entrainment mortality has precautionarily been assumed to be 100% for many species:** Gobies are a relatively robust species with low impingement mortality and entrainment studies at other power stations have identified relatively high entrainment survival rates (Section 2.1.3.1).

Considering the low level of predicted effects (Table 10), and the in-built precaution in the assessment, the conclusions in the **Environmental Statement** [APP-317] of no significant effects on population stability can confidently be determined.

4 Conclusions

This report describes the population level effects of entrapment and quantifies the sensitivity of the assessment to uncertainty in the operational performance of the proposed fish mitigation measures and uncertainties in sampling techniques. The sensitivity analysis also accounts for the natural fluctuations of fish stocks used as the comparator for losses.

Statistical bootstrapping approaches have been applied to entrapment predictions relative to the baseline population allowing estimates of the mean and 95th percentile effects to be established.

The three most commonly impinged species at Sizewell are sprat, herring and whiting, whilst gobies (*Pomatoschistus* spp.) are the most commonly entrained. The mean annual entrapment effect for sprat is predicted to be 0.03% of SSB (upper 95th percentile 0.04%). Herring entrapment is predicted to result in losses of 0.01% of SSB (upper 95th percentile 0.02%), and for whiting mean losses are 0.08% of SSB (upper 95th percentile 0.11%). Such losses are not significant at the population level.

Sea bass and gobies (*Pomatoschistus* spp.) are the only species where entrapment exceeds a 1% threshold.

The mean annual losses of sea bass due to SZC entrapment is predicted to be 0.99% of SSB with an upper 95th percentile estimate of 1.85%. These estimates are considered to be precautionary as they do not account for the greater distribution of sea bass within the Sizewell Dunwich Bank or the potential for a diurnal bias with greater numbers impinged during daylight hours. The effects on sea bass are not predicted to be significant at the population level. However, to provide the highest degree of confidence in the assessment a full ICES stock assessment was run. The results of the stock assessment confirmed this conclusion. No discernible effects on population trends and only very minor effects on absolute SSB despite the application of highly precautionary loss estimates.

The mean annual loss of gobies (*Pomatoschistus* spp.) is 1.74% with an upper 95th percentile estimate of 1.77%. Gobies are a short lived, fast maturing, highly fecund species with high degrees of natural variability. Because gobies are productive species with a short lifespan and early age of maturity, and because they are not fished, they will be able to sustain additional mortality rates greater than 10% of population size. The predicted level of losses of gobies (*Pomatoschistus* spp.) is not regarded as significant at the population level.

The uncertainty analysis has assumed no mitigation benefit from the LVSE intake head and considered a range of FRR effectiveness values produced by the Environment Agency for HPC (TB008). Correction factors have been applied to account for the potential diurnal bias introduced by a greater proportion of daylight samples and measures have been taken to quantify the entrainment gap for sprat and gobies (*Pomatoschistus* spp.) the species most likely to be subject to underestimation in entrainment predictions as well as herring. The application of correction factors has also been applied to account for diurnal biases in river lamprey, smelt, European eel and twaite shad. Whilst these measures to address uncertainty have resulted in increases in the relative population level effects in all cases where correction factors were applied no material changes in the assessment outcome were observed and the conclusion of no significant population level effects due to entrapment from SZC remains.

Where residual uncertainty remains, it is necessary to consider the magnitude of such uncertainties in relation to the predicted effects, species by species, accounting for the inbuilt precaution in the entrapment assessments.

This report provides further evidence that the proposed development of Sizewell C would not have significant effects on the population sustainability of the key species assessed. That is, the size of the spawning populations increase and decrease at the same times and at almost identical rates whether or not SZC is operating.

The results of the uncertainty analysis show that for all species the annual entrapment losses as a proportion of population size are below a 1% threshold that would pose a risk and therefore trigger further investigation for potential population level effects. The three most commonly impinged species at Sizewell are sprat, herring and whiting. The mean entrapment effect for sprat is <0.03% of the SSB (upper 95th percentile 0.04%), for herring entrapment is predicted to result in losses of 0.01% of SSB (upper 95th percentile 0.02%), and for whiting mean losses are 0.08% of SSB (upper 95th percentile 0.11%). Such losses are not significant at the population level.

Sea bass and gobies (*Pomatoschistus* spp.) are the only species that exceed the 1% threshold for annual entrapment losses as a proportion of population size. The mean annual losses of sea bass in the uncertainty analysis is 0.99% of SSB with an upper 95th percentile estimate of 1.85%. Sea bass are not uniformly distributed with low catch rates observed in surveys offshore and 95% of bass caught inshore of the Sizewell-Dunwich Bank suggesting that impingement predictions scaled-up from SZB may overestimate sea bass impingement at SZC. As such, the results are precautionary and no significant effects on population sustainability are predicted.

The uncertainty analysis also showed gobies (*Pomatoschistus* spp.) exceed the 1% threshold with a mean impingement rate of 1.74% and an upper 95th percentile effect of 1.77% of the population estimate. Because gobies are productive species with a short lifespan and early age of maturity, and because they are not fished, they will be able to sustain additional mortality rates greater than 10% of population size. The predicted level of losses of gobies (*Pomatoschistus* spp.) is not regarded as significant at the population level.

Overall, the results of the uncertainty analysis, and the in-built precaution in the assessment methodologies provide a high degree of confidence in the predictions of no significant effects at the population level.

5 References

Beare, D. J., Reid, D. G., Petitgas, P., 2002. Spatio-temporal patterns in herring (*Clupea harengus* L.) school abundance and size in the northwest North Sea: modelling space–time dependencies to allow examination of the impact of local school abundance on school size. – ICES J. Mar. Sci. 59, 469–479.

BEEMS Scientific Position Paper SPP103 (Rev 5). Consideration of potential effects on selected fish stocks at Sizewell. Cefas, Lowestoft. Submitted to ExA at Deadline 6 (Doc Ref: 6.14A).

BEEMS Scientific Position Paper SPP111. Sizewell C impingement predictions corrected for Sizewell B raising factors and cooling water flow rates. Cefas, Lowestoft.

BEEMS Scientific Position Paper SPP118. Sizewell C European sea bass, *Dicentrarchus labrax*, stock assessment. Cefas, Lowestoft. [REP8-131]

BEEMS Technical report TR339. Cefas comprehensive impingement monitoring programme 2014-2016. v.0.04. Cefas, Lowestoft.

BEEMS Technical Report TR406.v7. Sizewell C – Impingement predictions based upon specific cooling water system design. Cefas, Lowestoft.

Cardinale, M., Casini, M., Arrhenius, F., Håkansson, N. 2003. Diel spatial distribution and feeding activity of herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) in the Baltic Sea. Aquat. Liv. Res. 16, 283-292.

Dahlberg, M.D., 1979. A review of survival rates of fish eggs and larvae in relation to impact assessments. Mar. Fish. Rev. 41, 1–12.

Environment Agency (2005) Screening for intake and outfalls: a best practice guide. Science Report SC030231. Environment Agency, Bristol. 154 pp.

Environmental Agency 2020. Technical Brief: TB008 Fish Recovery and Return System Mortality Rates. Draft-04. 15 pp. Available at: https://ea.sharefile.com/share/view/s61339f123dad4ed794643b4b4f6932b9.

Froese, R. and D. Pauly. Editors. 2021. FishBase. World Wide Web electronic publication. www.fishbase.org, version (06/2021).

Gastauer, S., Fässler, S.M.M., Couperus, B., Keller, M.A. 2013. Target strength and vertical distribution of smelt (*Osmerus eperlanus*) in the Ijsselmeer based on stationary 200kHz echosounder recordings. Fish. Res. 148, 100-105.

Heath, M., Brander, K., Munk, P., Rankine, P., 1991. Vertical distributions of autumn spawned larval herring (*Clupea harengus* L.) in the North Sea. Continent. Shelf Res. 11, 1425-1452,

Horppila, J., Malinen, T., Nurminen, L. *et al*., 2020. A metalimnetic oxygen minimum indirectly contributing to the low biomass of cladocerans in Lake Hiidenvesi – a diurnal study on the refuge effect. Hydrobiologia 436, 81–90.

Huse, I., Korneliussen, R., 2000. Diel variation in acoustic density measurements of overwintering herring (*Clupea harengus* L.). ICES J. Mar. Sci. 57, 903–910

ICES. 2018. Report of the Working Group on southern horse mackerel, anchovy and sardine (WGHANSA), 26–30 June 2018, Lisbon, Portugal. ICES CM 2018/ACOM:17. 659 pp.

INBO 2021. Instituut voor Natuur- en Bosonderzoek. Monitoring van de visgemeenschap in het Zeeschelde-estuarium. Ankerkuilcampagnes 2020. Rapporten van het Instituut voor Natuur- en Bosonderzoek 2021 (1). Instituut voor Natuur- en Bosonderzoek, Brussel. doi.org/10.21436/inbor.29272200.

Lennox, R. J., Økland, F., Mitamura, H., Cooke, S. J., and Thorstad, E. B. European eel Anguilla anguilla compromise speed for safety in the early marine spawning migration. ICES J. Mar. Sci. 75, 1984–1991.

Maes, J., Pas, Z., Tallieu, A., Van Damme, P.A., Ollevier, F. 1999. Diel changes in the vertical distribution of juvenile fish in the Zeeschelde Estuary. J. Fish. Biol. 54, 1329–1333.

Mayhew, D.A., Jensen, L.D., Hanson, D.F., Muessig P.H. 2000. A comparative review of entrainment survival studies at power plants in estuarine environments. Environ. Sci.Policy 3 (1), 295 -301.

McCleave, J.D. Arnold, G.P. 1999. Movements of yellow-and silver-phase European eels (*Anguilla anguilla* L.) tracked in the western North Sea. ICES J. Mar. Sci. 56: 510-536.

Moore, A., 2016 Estuarine habitat requirements and distribution of diadromous fish. Defra Contract Report No. SF0264

Moore, A., Ives, M., Davison, P. and Privitera, L. (2016), A preliminary study on the movements of smelt, Osmerus eperlanus, in two East Anglian rivers. Fish Manag Ecol, 23, 169-171.

Munk, P., Kiørboe, T., Christensen, V., 1989. Vertical migrations of herring, *Clupea harengus*, larvae in relation to light and prey distribution. Environ. Biol. Fish. 26, 87–96.

Mustamäki, N., Jokinen, H., Scheinin, M., Bonsdorff, E., and Mattila, J. Seasonal shifts in the vertical distribution of fish in a shallow coastal area. ICES J. Mar. Sci., 73, 2278–2287.

Natural England. 2018. Marine Conservation Zones Natural England’s advice to Defra on Marine Conservation Zones to be considered for consultation in 2017. Annex 2: Advice on Tranche 3 MCZs with the species feature of conservation importance smelt (*Osmerus eperlanus*). Natural England Joint Publication JP026. June 2018.

Nilsson, F.L. A., Thygesen, U.H., Lindgren, B., Nielsen, B.F., Nielsen, J.R., Beyer, J.E. (2003) Vertical migration and dispersion of sprat (Sprattus sprattus) and herring (*Clupea harengus*) schools at dusk in the Baltic Sea. Aquat. Liv. Res. 16, 317-324.

O'Neill, R. 2017. The distribution of the European sea bass, *Dicentrarchus labrax*, in Irish waters. PhD Thesis, University College Cork.

Peck, M.A., Clemmese, C., Herrmann, J-P., Stacker, S., Temming, A. 2004. The feeding–growth relationship in post-larval Baltic sprat (*Sprattus sprattus* L.): Comparison of somatic, nucleic acid- and otolith-based growth rates. CM 2004 L:25. 21 pp.

Petitgas, P., Alheit, J., Peck, M.A., Raab, K., Irigoien, X., Huret, M., van der Kooij, J. Pohlmann, T., Wagner, C., Zarraonaindia, I., Dickey-Colas, M. 2012. Anchovy population expansion in the North Sea. Mar. Ecol. Prog. Ser. 444, 1–13

de Pontual, H., Lalire, M., Fablet, R., Laspougeas, C., Garren, F., Martin, S., Drogou, M., and Woillez, M. (2019) New insights into behavioural ecology of European seabass off the West Coast of France: implications at local and population scales. ICES J. Mar. Sci. 76, 501–515.

Quayle, V.A., Righton, D., Hetherington, S., and Pickett, G. 2009. Observations of the behaviour of European sea bass (*Dicentrarchus labrax*) in the North Sea. In Tagging and tracking of marine animals with electronic devices, pp. 103–119. Ed. by J. L. Nielsen, H. Arrizabalaga, N. Fragoso, A. Hobday, M. Lutcavage, and J. Sibert. Reviews: Methods and technologies in fish biology and fisheries 9, Springer.

R Core Team 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Righton, D., Westerberg, H., Feunteun, E. *et al*. 2016. Empirical observations of the spawning migration of European eels: The long and dangerous road to the Sargasso Sea. Sci. Adv. 2 (10), e1501694

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. [cited 23-07-2021]. Available from: https://www.marlin.ac.uk/species/detail/1204

Schurmann, H., Claireaux, G. and Chartois, H. (1989) Change in vertical distribution of sea bass (*Dicentrarchus labrax* L.) during a hypoxic episode. Hydrobiologia 371, 207–213.

Sizewell C Project. 2021. 6.14 Revision: 1.0 Applicable Regulation: Regulation 5(2)(a) PINS Reference Number: EN010012 Environmental Statement Addendum Volume 3: Environmental Statement Addendum Appendices Chapter 2 Main Development Site Appendix 2.17.A Marine Ecology and Fisheries

Stamp, T., 2021. The Ecology and Distribution of European Bass (*Dicentrarchus labrax*) in inshore and coastal waters of the U.K. PhD Thesis, University of Plymouth.

Teague, N., Clough, S.C. 2011. The survival of lamprey on travelling screens at potable water intakes WIT Transactions on State of the Art in Science and Engineering, 71: 101-109.

Turnpenny, A W H, 1988. The behavioural basis of fish exclusion from coastal power station cooling water intakes.Central Electricity Generating Board, Research Report No. TPRD/L/3270/R88, 28pp. + Figs. + Tables.

Turnpenny, A., Taylor, C. 2000. An assessment of the effect of the Sizewell power stations on fish populations. Hydroécologié Appliquée 12: 87-134.

Westerberg, H. 1979. Counter-current orientation in the migration of the European eel. Rapports et Proceès-verbaux des Réunions, Conseil international pour l'Exploration de la Mer 174, 134–143.

Westerberg, H., Lagenfelt, I., and Svedaäng, H. 2007. Silver eel migration behaviour in the Baltic. ICES J. Mar. Sci. 64, 1457–1462

Appendix A CIMP bulk overflow

A.1 Potential for diurnal bias in impingement estimates due to CIMP bulk overflows

The Environment Agency Written Representation on Sizewell C Development Consent Order at Deadline 2 [REP2-135] and in their Deadline 7 Submission - Comment on 9.67 Quantifying Uncertainty in Entrapment estimates for Sizewell C [REP7-132] (Revision 1 of this report), outlined concern pertaining to the overflow of the bulk samples during the Comprehensive Impingement Monitoring Programme (CIMP) at SZB.

As described in Section 2.1.2, impingement monitoring personnel cannot remain on the nuclear facility site outside normal working hours due to site security restrictions. Restricted site access at operational nuclear power stations means it is not possible to collect hourly samples outside normal working hours, nor monitor the collection of overnight bulk samples. Overnight bulk samples may overflow when the sample net becomes clogged. In summer months, overflow typically arises due to large numbers of ctenophores and jellyfish clogging the nets. Overflows may also result due to ingress of weed and/or mud, or in the winter months due to inundation with pelagic species, primarily sprat and herring, and demersal whiting. The causes of overflow are considered further in Section A.6 below. When a bulk sample overflows, the hourly samples collected that day are extrapolated to estimate the 24-hour impingement rates.

In summary, the Environment Agency concern is that when bulk samples overflow, extrapolation from hourly daytime samples may fail to account for diurnal patterns in species impingement and ultimately lead to underestimates of annual impingement rates. The specific concerns addressed herein include:

1. Uncertainty in SZB impingement rates when extrapolated may lead to potential underestimation of impingement rates at SZC, with particular concern for species of relevance to the DCO.
2. The loss of overnight sample data may underestimate impingement due to the frequent occurrence of overflowing, invalid overnight bulk samples,
3. The inclusion of valid overnight samples only may introduce bias, as when catches are high the sample is more likely to overflow and the data excluded. This leads to uncertainty in overnight impingement rates and cast doubt on the validity of extrapolating daytime hourly impingement rates to cover the overnight period.
4. Data from Sizewell A shows diurnal patterns of impingement, with peak catches during the night (Turnpenny 1988).
5. Eel and smelt display greater mobility at night and other species may be less able to see and avoid the intakes during the hours of darkness. This introduces uncertainty in the SZB impingement rates.
6. A corrective factor should be introduced to estimate impingement where there is no overnight sample data. This factor should also be applied to all Pisces data, regardless of whether the overnight sample is valid to account for the different proportion of valid and invalid overnight data between the two contractors and to address the uncertainty about which Pisces bulk samples did, or did not, overflow.

The **Environment Agency** [REP2-135] suggest that, by only including valid bulk samples, the analyses in **BEEMS Technical Report TR339** [AS-238] did not consider periods of maximum abundance when diurnal behaviour may be different. An attempt was made to address these concerns in Revision 1 of this report, however, in their Deadline 7 Submission [REP7-132] the Environment Agency raised further comments that were reiterated by the RSPB [REP7-154]. This report aims to further answer these concerns and apply a correction factor to address any cases where there were potential underestimations in fish impingement.

The purpose of this technical appendix is to address the concerns relating to the potential for diurnal bias in the sampling programme and to determine if reliance on daytime samples following incidences of bulk overflows introduces positive/negative bias for each of the key species.

A.2 Sampling intensity and overflowing overnight bulk samples

The CIMP programme undertaken at SZB, used to inform DCO impingement estimates, was undertaken for eight years from 2009-2013 and 2014-2017 and consisted of 205 samples. This provides a very powerful data set for determining impingement estimates at SZB and the proposed SZC station. Impingement monitoring at SZB was designed in a pseudo-random fashion to eliminate tidal biases whilst sampling the full year to capture seasonal patterns. To account for diurnal biases, samples consisted of six 1-hour samples during the day and an overnight bulk sample, thus providing a 24-hour impingement record.

Due to the aforementioned security and logistical issues associated with working on an operational nuclear facility, the collection of overnight bulk samples could not be monitored. On occasion this resulted in bulk samples overflowing due to net clogging and 24-h impingement estimates were extrapolated from hourly samples. This assumes that there are no differences in day and night impingement rates. If, through diurnal behaviour, species have different rates of impingement during the day or night this can lead to over-, or under-estimates in 24-hour impingement when bulk samples overflowed.

Of the 205 impingement samples collected seasonally over 8 years, there were a total of 100 valid bulk samples. As such, there are 105 occasions where the daytime hourly samples were extrapolated to establish a 24-hour estimate of impingement.

This annex aims to address whether excluding the overnight bulk samples and relying on the daytime samples introduces positive/negative bias in estimated 24-h impingement rates for each of the key species. Where bias is introduced that could lead to under-estimates in impingement rates a correction factor is determined to correct this bias. To screen for potential biases, all overnight sample data has been removed from the analysis. The hourly samples have been extrapolated to estimate 24-h impingement rates and the results are compared to previous results with 100 valid bulk samples included.

To note there were three occasions where insufficient, or no daytime samples were taken during a particular visit.

- 04.02.2009 – 24hr duration bulk sample
- 17.02.2009 – 24hr duration bulk sample
- 18.02.2009 – 24hr duration bulk sample

These samples have all been retained in the analysis as there should be no inherent bias in a 24-hour sample. However, the following samples have been excluded from the analysis, due to no or insufficient data in the daytime samples, which prohibit raising the data to 24 hours. Therefore, the updated data set without overnight bulk samples consists of 203 impingement samples, down from 205 (a 1% change in sample numbers).

- 04.06.2009 – 17.4hr duration bulk sample, plus two daytime samples only
- 23.11.2011 – 15.6 hr duration bulk sample, no daytime samples

There remained sufficient samples in each of these two quarters (Q2 2009 and Q4 2011) to calculate their quarterly rates as part of the impingement calculations.

A.3 Sizewell C impingement estimates with no overnight samples

By comparing the full data set with the situation where bulk samples are removed it is possible to see the relative effect the bulk samples have on estimates of impingement rate. An increase in impingement rate when bulk samples are removed, compared to the full data indicates that more individuals are caught in the daytime samples (or extrapolating from the hourly samples results in higher impingement rates than those recorded in a 24-h sample). In the case where impingement rates increase following the removal of all bulk samples it can be assumed that impingement is overestimated during incidents of bulk sample overflow. This remains an assumption as it implies that the probability of impingement during the incidence of bulk sample overflows is consistent with that during collection of valid samples. However, given there are 100 valid bulk samples throughout the dataset such an assumption is reasonable. In such instances no attempt has been made to correct impingement estimates to reduced rates. This means that there is a degree of precaution in the estimates of impingement rates for species more likely to be impinged during the day.

More importantly in this context, are species for which there is a decrease in impingement estimates when bulk samples are removed, as this indicates that incidences of overflowing samples could lead to under-estimates of 24-h impingement rates (again assuming that the diurnal probability of impingement during the incidence of bulk sample overflows is consistent with valid samples).

With the full data set (including valid bulk samples), eight species account for the top 95.1% of impingement numbers at SZC, these are sprat, herring, whiting, bass, gobies (*Pomatoschistus* spp.), Dover sole, anchovy, and dab. The same eight species account for 95.3% of the total impingement numbers when all bulk samples are removed. Changes in impingement estimates following the removal of overnight bulk samples are species specific (Table 11). For the eight most commonly impinged species, the removal of overnight bulk samples resulted in impingement estimates at SZB increasing (compared with estimates based on the full data set). The mean increase for the top eight species was 3%. These differences were species specific; sprat, herring, and sea bass increased between 6 and 10% with the removal of the bulk samples indicating that impingement rates for these species may have been overestimated by a small margin. Whiting, gobies (*Pomatoschistus* spp.), Dover sole and dab showed minor differences of less than 2% in either direction, whereas anchovy decreased by ~5% when the bulk samples were removed compared with the full data indicating a minor underestimate (Table 11).

In total, impingement for ten of the key species decreased, these were Dover sole, anchovy, dab, smelt, European eel, twaite shad, river lamprey, mackerel, tope and sea lamprey. Dover sole, anchovy, dab, and tope all decreased by <5% and would not significantly influence the results of the assessment which in all cases suggested impingement rates below 0.1% of the relative population comparator in Revision 1 of this report.

Sea lamprey and mackerel deceased by 100% and 48.6%, respectively. However only one sea lamprey was caught in the SZB CIMP in an overnight bulk sample, whereas mackerel are caught inconsistently and in low numbers, occurring in 9 out of 205 impingement samples. The estimated impingement numbers of these species are therefore too low to determine the potential for diurnal bias and impingement is negligible.

Four species have been identified as requiring further attention to account for potential underestimates in impingement rates, these are:

- smelt;
- river lamprey;
- European eel; and,
- twaite shad.

All four have been selected as they are species of conservation interest and, with the exception of twaite shad, impingement numbers decreased by >10% when bulk samples were removed compared to the full data analysis. Whilst twaite shad has a percentage decrease of just 2% it has been included on a precautionary basis, as it is a species of conservation interest (Table 11).

Table 11. Comparison of estimated annual Sizewell B impingement numbers at full operational capacity between estimates including valid bulk samples and with all bulk samples removed.

Common name	All valid data applied (SPP111.v2)			Corrected with no overnight bulk samples			Difference in mean (% change)
	Mean	Lower	Upper	Mean	Lower	Upper	
Sprat	2,646,189	1,364,820	4,478,852	2,906,236	1,409,466	5,124,269	9.8
Herring	951,056	563,376	1,441,666	1,008,291	534,621	1,688,739	6.0
Whiting	642,935	471,160	840,402	654,576	483,479	857,797	1.8
European seabass	275,802	127,651	478,914	304,116	147,374	517,977	10.3
Gobies (*Pomatoschistus* spp.)	207,900	88,386	394,005	210,890	94,790	399,431	1.4
Dover sole	90,766	62,984	125,047	90,604	63,555	122,835	-0.2
Anchovy	63,783	18,703	153,465	61,136	18,565	146,200	-4.1
Dab	55,245	32,813	92,227	54,582	33,398	87,103	-1.2
Thin-lipped grey mullet	46,269	14,356	89,305	48,143	17,487	91,984	4.1
Flounder	13,824	10,478	18,151	14,233	10,656	18,939	3.0
Plaice	9,441	6,078	14,071	9,578	6,693	13,089	1.4
Smelt	9,531	5,963	13,919	8,527	5,111	12,902	-10.5
Cod	7,097	2,458	13,247	7,747	2,616	14,452	9.2
Thornback ray	2,881	1,794	4,228	2,941	1,750	4,582	2.1
Eel	1,059	658	1,560	928	542	1436	-12.4
Twaite shad	1,158	576	2,017	1135	516	2174	-2.0
River lamprey	1,121	615	1,889	977	472	1772	-12.9
Horse mackerel	671	210	1,615	698	192	1784	4.1
Mackerel	119	6	394	61	0	168	-48.6
Tope	24	0	89	23	0	87	-3.3
Sea trout	3	0	20	14	0	82	292.7
Sea lamprey	2	0	11	0	0	0	-100.0
Allis shad	0	0	0	0	0	0	NA
Salmon							

Table 12. Comparison of annual estimates of unmitigated Sizewell C impingement numbers at full operational capacity between estimates including valid bulk samples and with all bulk samples removed.

Common name	**All valid data applied (SPP111.v2)**			**Corrected with no overnight bulk samples**			**Difference in mean (% change)**
	Mean	Lower	Upper	Mean	Lower	Upper	
Sprat	6,153,906	3,173,989	10,415,898	6,758,665	3,277,817	11,916,863	9.8
Herring	2,211,750	1,310,172	3,352,700	2,344,855	1,243,301	3,927,287	6.0
Whiting	1,495,192	1,095,717	1,954,416	1,522,265	1,124,367	1,994,870	1.8
European seabass	641,398	296,862	1,113,750	707,243	342,728	1,204,594	10.3
Gobies (*Pomatoschistus* spp.)	483,487	205,548	916,287	490,440	220,442	928,907	1.4
Dover sole	211,083	146,474	290,806	210,707	147,801	285,663	-0.2
Anchovy	148,332	43,495	356,894	142,177	43,174	339,999	-4.1
Dab	128,476	76,309	214,481	126,935	77,669	202,565	-1.2
Thin-lipped grey mullet	107,602	33,386	207,685	111,961	40,667	213,916	4.1
Flounder	32,149	24,367	42,211	33,099	24,781	44,045	3.0
Plaice	21,956	14,135	32,723	22,274	15,566	30,439	1.4
Smelt	22,165	13,867	32,370	19,830	11,886	30,004	-10.5
Cod	16,505	5,716	30,807	18,016	6,084	33,610	9.2
Thornback ray	6,700	4,172	9,833	6,838	4,070	10,656	2.1
Eel	2,463	1,530	3,628	2,158	1,260	3,341	-12.4
Twaite shad	2,693	1,340	4,691	2,639	1,199	5,056	-2.0
River lamprey	2,607	1,430	4,393	2,271	1,097	4,120	-12.9
Horse mackerel	1,560	488	3,755	1623	447	4148	4.1
Mackerel	277	14	915	142	0	391	-48.6
Tope	55	0	207	54	0	202	-3.3
Sea trout	8	0	48	32	0	190	292.7
Sea lamprey	4	0	26	0	0	0	-100.0
Allis shad	0	0	0	0	0	0	NA
Salmon							

A.4 Species of conservation interest

A.4.1 Smelt

Among the species of conservation interest, the largest impingement estimates are for cucumber smelt *Osmerus eperlanus*. For smelt, there was a 10.5% reduction in impingement estimates upon removal of the overnight bulk samples. This is in contrast to previous findings because Appendix F of BEEMS Technical Report TR339 [AS-238], reported no significant differences in mean smelt impingement rates between hourly and bulk samples when impingement rates were compared from 22 sample visits (with valid bulk samples), however, maximum impingement did occur during daylight hours. Furthermore, following the removal of 18 bulk samples, when potential overflows were identified and removed from the data set, the only key species where impingement rates changed more than 2.5% in either direction was cucumber smelt, with an 8.4% increase in predicted impingement suggesting higher numbers in the daytime estimates (BEEMS Scientific Position Paper SPP111.v2). On this evidence it was concluded that there was no indication that the impingement records underestimate smelt. Conversely, if smelt are more susceptible to impingement during daylight hours these impingement estimates may be precautionary and overestimate impingement. Two plausible hypotheses were proposed to explain the increase in smelt impingement following the removal of overflowing bulk samples:

a) Assuming smelt are equally susceptible to impingement throughout the day, the removal of the additional bulk samples could indicate that smelt were underestimated in the overflowing bulk samples previously used in the analyses. Removing these samples thereby increases impingement estimates. This explanation seems unlikely given the modest and bi-directional changes in other species.

b) Smelt are impinged in greater numbers during the day, thus the removal of bulk samples and extrapolation of raised hourly samples to determine 24-hour estimates, increases impingement estimates.

However, in contrast to previous findings, the removal of all overnight bulk sample data resulted in a decrease in smelt numbers. This suggests that whilst smelt impingement is variable, over the full data set impingement rates are higher in the overnight bulk samples (Figure 1; Table 11). Whist the removal of the overnight bulk samples decreases smelt impingement estimates, there is variability throughout the sampling period. The two discarded samples (04.06.2009 and 23.11.2011) in the daytime sample dataset both contained relatively large numbers of smelt with 145 and 45 fish, respectively. This contributes toward the reduction in numbers between the two datasets. When overnight data is included, there are 145 out of 205 samples (71%) where smelt were impinged, in comparison there are 130 samples containing smelt out of 203 samples (64%) when overnight data is excluded.

The mean impingement rates of smelt for both datasets are presented in Figure 1. The full CIMP data set and the truncated data with no bulk samples show similar patterns, with the full data having a slightly higher mean over the course of the sampling period. In June 2014 the single greatest difference between day and night samples occurred with higher numbers predicted from extrapolating day only samples. It is noteworthy that during the peak smelt abundance events including August of 2009, 2011 and 2012 there was no difference between the full data set and the truncated data with no overnight samples (Figure 2; Table 11).

Low numbers of larger fish are impinged at Sizewell in the period February – March (Q1), which coincides with the peak spawning run in freshwater in February - April, therefore these are likely to be mature adult fish heading past the station to spawn (BEEMS Scientific Position Paper SPP101 [AS-238]). During this period impingement rates are relatively consistent with and without overnight samples (Table 11). The greatest seasonal differences in smelt impingement estimates when the overnight bulk samples are removed occur in Q4, with a reduction in smelt numbers in all years, apart from 2017, for which there was no difference. Smelt impingement rates varies throughout the year, with more impinged in the warmer summer months, peaking in August, June and July and consisting primarily of 1+ y.o. juveniles (BEEMS Scientific Position Paper SPP101 [AS-238]). Impingement rates are low during the winter months which is characterised by the largest reductions in smelt impingement estimates when the overnight bulk samples are removed (Table 11).

The removal of overnight bulk samples and extrapolation of raised hourly samples to determine 24-hour estimates, decreases impingement estimates and potentially underestimates smelt impingement. The available evidence indicates that the current impingement data underestimates smelt when overnight bulk samples are excluded. The extent to which this is due to diurnal movements, or a result of encounter probability is unresolved.

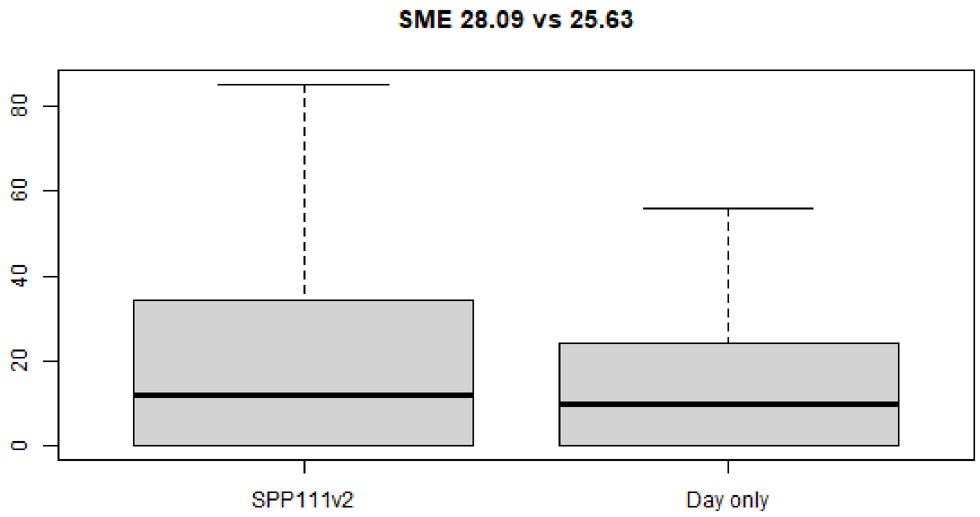

Figure 1 Smelt abundance raised to 24 hours and full capacity between the two datasets, SPP111v2 including overnight data and the amended analysis with overnight samples removed. The respective means are shown in the legend at the top of the graph.

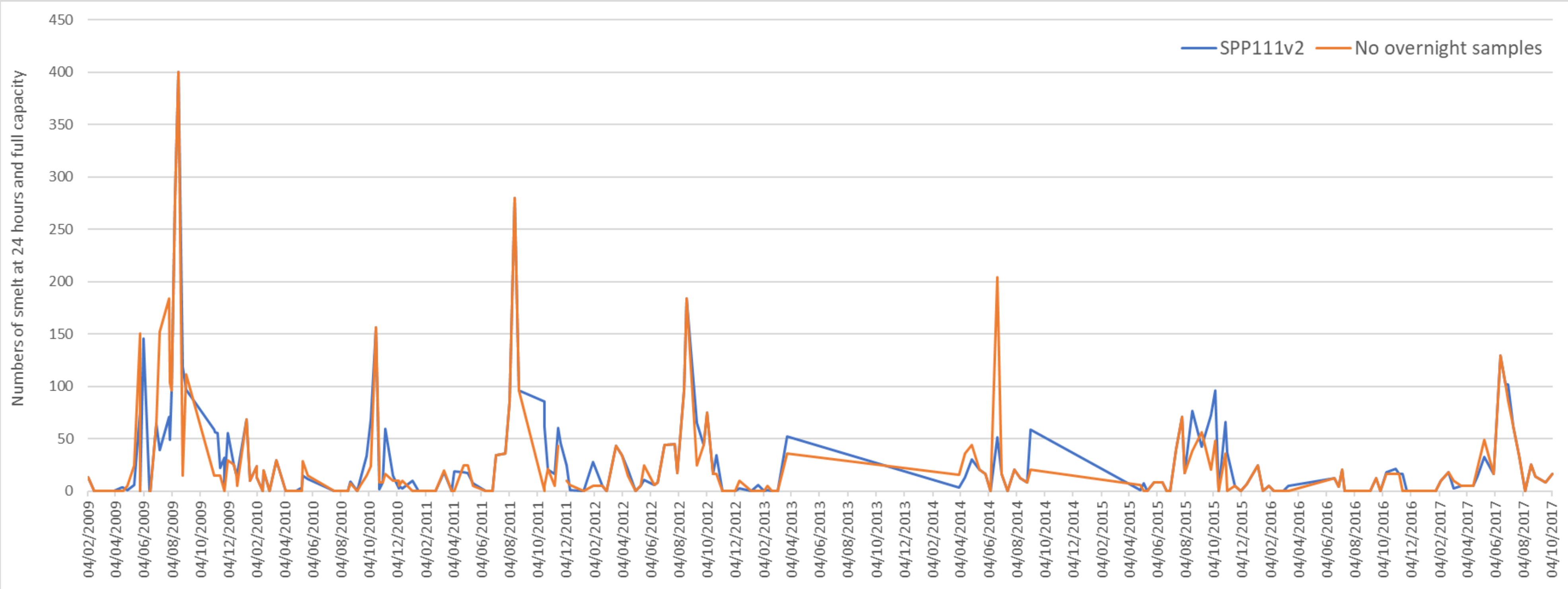

Figure 2 Smelt abundance raised to 24 hours and full capacity between the two datasets, blue line represents the full data (SPP111v2) including overnight bulk samples, the orange line shows the results with bulk samples removed and daytime samples raised.

Table 13 The seasonal difference in the mean number of smelt impinged per quarter with and without overnight samples. Quarters for which a reduction in impingement was recorded are highlighted in red.

Year	Quarter	Mean number excluding overnight samples	Mean number including overnight samples	Mean percentage difference	Quarterly mean (%)	Number of samples	
						Excluding overnight	Including overnight
2009	1	1.9	1.9	0.0	-12.2	7	7
2010	1	20.4	20.4	0.0		8	8
2011	1	2.2	3.1	-30.1		9	9
2012	1	10.6	15.2	-30.2		5	5
2013	1	5.8	8.4	-31.3		7	7
2016	1	4.2	4.9	-14.6		7	7
2017	1	6.0	5.0	20.8		7	7
2009	2	22.5	11.4	97.3	36.2	8	9
2010	2	7.2	4.9	47.3		6	6
2011	2	8.9	10.2	-12.6		6	6
2012	2	13.1	12.0	8.7		7	7
2014	2	48.0	19.1	150.7		7	7
2015	2	3.6	4.0	-10.1		6	6
2016	2	8.0	8.0	0.0		2	2
2017	2	53.8	49.7	8.1		6	6
2009	3	156.1	134.9	15.7	-13.2	9	9
2010	3	3.9	7.8	-49.7		7	7
2011	3	106.3	106.3	0.0		5	5
2012	3	65.1	70.8	-8.1		7	7
2014	3	12.7	19.1	-33.6		6	6
2015	3	34.6	45.6	-24.2		7	7
2016	3	4.6	4.6	0.0		7	7
2017	3	32.8	34.9	-6.0		7	7
2009	4	13.4	36.1	-62.8	-34.7	11	11
2010	4	26.6	32.3	-17.7		10	10
2011	4	12.9	38.4	-66.5		7	8
2012	4	16.6	18.2	-8.6		7	7
2015	4	13.6	30.0	-54.7		7	7
2016	4	6.9	10.2	-32.8		7	7
2017	4	16.0	16.0	0.0		1	1

A.4.2 European eel, twaite shad and river lamprey

European eel, twaite shad and river lamprey are all impinged in relatively low numbers and without large seasonal fluctuations in abundance. Therefore, they are more likely to be captured when the additional 18 hours of the bulk sample is included in the 24-hour estimates. As a result, these species are more susceptible to the effects of encounter probability, resulting in a greater proportion of individuals caught in the overnight bulk sample. Some species, eels in particular, have been shown to display greater activity levels at night compared to the day in certain habitats or at particular life stages.

As discussed in section 2.1.2.2 tracking of yellow and silver eels in the southern North Sea show that selective tidal stream transport was used day and night when it occurred. Other studies have shown midwater movements by night and low levels of movement on the seabed by day (Westerberg, 1979; Westerberg et al., 2007). It is feasible that a greater proportion of daylight samples may result in an underestimate of impingement of yellow eels at SZB. However, there is scarce literature evidence to support or refute the diurnal bias in impingement rates of yellow eels in coastal waters. An underestimate of impingement of yellow eels has been addressed with the application of the correction factor.

As discussed in section 2.1.2.4 both twaite shad and river lamprey are impinged in relatively low numbers throughout the year. Therefore, impingement of these species will be susceptible to both the effects of potential diurnal biases and encounter probabilities. In either instance the reduction in the proportion of longer bulk samples (18 hours) may lead to underestimates. Impingement underestimates in both of these species has been addressed with the application of the correction factor.

A.5 Correction factors

To determine the impact of the missing overnight bulk samples on the four species of conservation interest, smelt, eel, twaite shad and river lamprey, a correction factor has been applied to adjust the SZB impingement estimates to account for these missing samples. The 24-hour values applying all the valid bulk samples were based on 205 sampling dates, 100 (48.78%) of these had both hourly samples during the day and a valid overnight bulk sample. As Figure 3 shows invalid samples are spread throughout the year, although there is seasonal variation in occurrence, with proportionally more invalid samples in June, July and August.

The correction factor is calculated from the mean impingement estimates when all bulk samples were removed (0% overnight) and when valid bulk samples were included (48.78% overnight), extended to estimate the effect for 100% of sample dates with overnight samples.

$$I_{100} = I + \frac{(I - I_0)}{P} * (100 - P)$$

Where I_{100} is the estimated impingement for 100% of dates with overnight samples, I_0 is the impingement for 0% with overnight samples and I is the impingement with the valid bulk sample data, which had P = 48.78% with overnight samples.

The correction factor applied in the uncertainty analysis is then $\frac{I_{100}}{I}$, the ratio of the estimated impingement for 100% of dates with overnight samples to the impingement estimates calculated with the valid bulk sample data (Table 14).The correction factor calculations assume that the ratio between day and night impingement rates is the same for all dates.

These correction factors have been applied to the four species of conservation interest to produce updated impingement estimates at SZB (Table 14). The correction factors have been incorporated into the uncertainty analyses for SZC.

Table 14 Correction factor to apply to SPP111 version 2 values within the uncertainty analysis of impingement estimates

Sizewell B impingement estimate	Mean impingement estimates with full data set (SPP111.v2)	Mean impingement estimates with no bulk samples	SPP111.v2		Impingement estimate with 100% overnight samples	Correction factor to 100% overnight
			% overnight samples	Change in Mean per %		
Smelt	9,531	8,527	48.78	20.573	10,584 (10,589)	1.111
Twaite shad	1,158	1,135	48.78	0.473	1,182 (1,182)	1.021
River lamprey	1,121	977	48.78	2.950	1,272 (1,272)	1.135
Eel	1,059	928	48.78	2.681	1,196 (1,197)	1.130

(Calculated estimate based on correction factor to 3dp.)

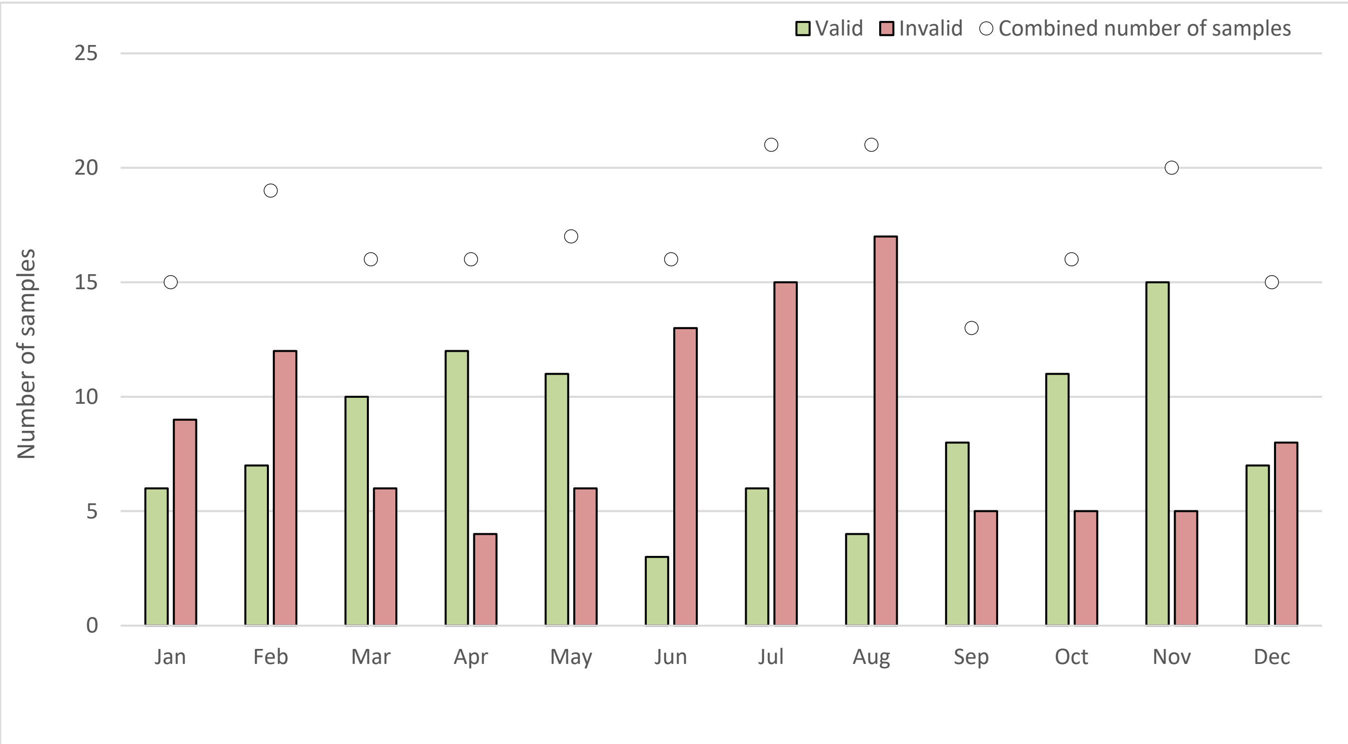

Figure 3 Bar chart showing seasonal changes in the numbers of valid and invalid overnight bulk samples, and the combined numbers of samples taken in each month

A.6 Factors contributing to overflowing bulk samples

To address the EA concern about why there is a higher proportion of overflowing bulk samples in the Pisces data series than in the Cefas data series the issue of overflowing overnight bulk samples has been examined in more detail.

Any bulk sample where the flow was redirected or may have overflowed has been treated as invalid and removed from subsequent analyses. This results in a total of 45 occasions when the bulk has been removed, equivalent to 36% of the 124 bulk samples during the Pisces data series. On four occasions a bulk sample collection was not deployed (3%). In the Cefas data series, the collection of 70 bulk samples was attempted between April 2014 and October 2017. On 49 occasions (70%) the bulk sample was deemed invalid and removed from the analyses. There were seven occasions (10%) when the bulk sample collection was not deployed for operational reasons.

The EA have suggested that a correction factor should be applied to all Pisces data, regardless of whether the overnight sample is valid, to account for the different proportions of valid and invalid overnight data in the Pisces and Cefas data series and to address the uncertainty about which Pisces bulk samples did, or did not, overflow. Whilst there are differences in the proportion of overflowing samples between the two different contractors (36% and 70% invalid overnight samples) there does not appear to be any patterns in the Pisces data series samples that support the suggestion that overflows were not recorded correctly. In fact, the proportion of overflowing overnight samples increased annually from 20% in 2009 to 65% in 2012, the latter percentage being more in line with that reported in the Cefas data series. In 2013 no invalid overnight samples were recorded, however only 3 months of data were collected that year.

A factor contributing to the difference in the proportion of bulk samples which have overflowed is the time of year that the bulk samples were deployed. Due to station outages in Q4 2014 and Q1 2015, 63% of Cefas samples were collected during Q2 and Q3. These periods, particularly Q3 when attempts to collect 25/70 of the bulk samples took place, are prone to the highest proportion of invalid bulk samples due to ctenophore and gelatinous zooplankton ingress. During periods of very high ctenophore biomass, fish impingement numbers are typically low (Figure 4). There are no annual or seasonal trends in ctenophore biomass over the full monitoring period, but spikes in ctenophore biomass were recorded in 2014 and 2015, during the Cefas data series. Quarter 3 in 2016, and to a lesser degree in 2017, corresponded with large spikes in jellyfish abundance (seasonal trends in jellyfish weight in Q3 between 2009-2017 were not significant; tau 0.42, $p = 0.14$, Theil-Sen slope = 4,873 [17]).

Annual sprat numbers have not shown a trend throughout the impingement monitoring period, but herring increased markedly between 2009-2017 (tau 0.86, $p < 0.01$, Theil-Sen slope = 115,542) and whiting also increased annually, although this was not significant at α 0.05 (tau 0.57, $p = 0.06$, Theil-Sen slope = 50,497). The increases in two of the three most abundant species may also have contributed to more samples overflowing in Q1 during the Cefas years.

It is not possible *a posteriori* to determine the extent to which the seasonal sampling strategy or changes in the abundance of species leading to overflows contributed to the differences in the proportion of invalid bulk samples in the Pisces and Cefas data series. However, a total of 100 valid bulk samples contribute to the impingement data series (21 in Cefas years, 79 in Pisces years) and any sample potentially subject to overflowing has been removed.

Further analyses examined the potential causes of bulk sample overflows. To assess which factors and species have the greatest influence the Cefas data series (2014 - 2017) was examined to determine whether more of a species or group were found per hour in daytime samples that which preceded the collection of overflowing bulk samples. Results were raised to weights in 24 hours by species and compared for the two groups using a two-sample *t* test and Mann–Whitney U test. If $p<0.05$ for either test the groups were deemed to be different.

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

The greatest weight differences in the Cefas data series were for jellyfish, ctenophores, mud and sprat (Figure 5). The mean weight of combined jellyfish in valid bulk samples was 99.57kg, compared to 1435.64kg in invalid overflowing samples, an increase of 1342%. The mean weight of ctenophores in valid bulk samples was 1154.50kg, compared to 2864.32kg, in invalid overflowing samples, an increase of 248%. Mud increased from 0.78 kg to 75.9 kg, or 9631% and sprat from 5.89 kg to 71.66 kg, or an 1117% increase. Whilst sprat and mud represent a large percentage increase, the absolute weights are an order of magnitude lower than jellyfish and two orders of magnitude lower than ctenophores. Other significant differences were detected, for example anchovy increased from 0.41 kg to 1.3 kg when there was an overflow. However, these small weights will not contribute to an overflow in isolation and likely reflective of the seasonal occurrence of anchovy in the summer months when jellyfish abundance is greatest. From these results it is concluded that the main causes of overflowing overnight samples are generally not ingresses of single fish species, with the exception of sprat. Rather, the causes are gelatinous zooplankton and mud, which block the mesh of the net and stop the water from adequately draining away. As a result, water levels rise and a proportion of the sample is lost, flowing over the edge of the collection basket. The weight of fish impinged during summer ctenophore and jellyfish blooms is small (Figure 5). Fish weights typically peak during winter and spring, whilst ctenophore and jellyfish biomass peak in summer. This suggests that overflows during the summer period will not miss large fish biomass events at night, but it is feasible they miss occasional species. This risk is reduced for species of conservation interest because overflowing bulk samples are visually inspected to identify such species.

The only fish species substantially contributing to overflowing bulk samples is sprat. However, the removal of bulk samples increases sprat abundance. Therefore, unless there was a different behaviour occurring when the overflowing bulk samples occurred, impingement estimates are precautionary.

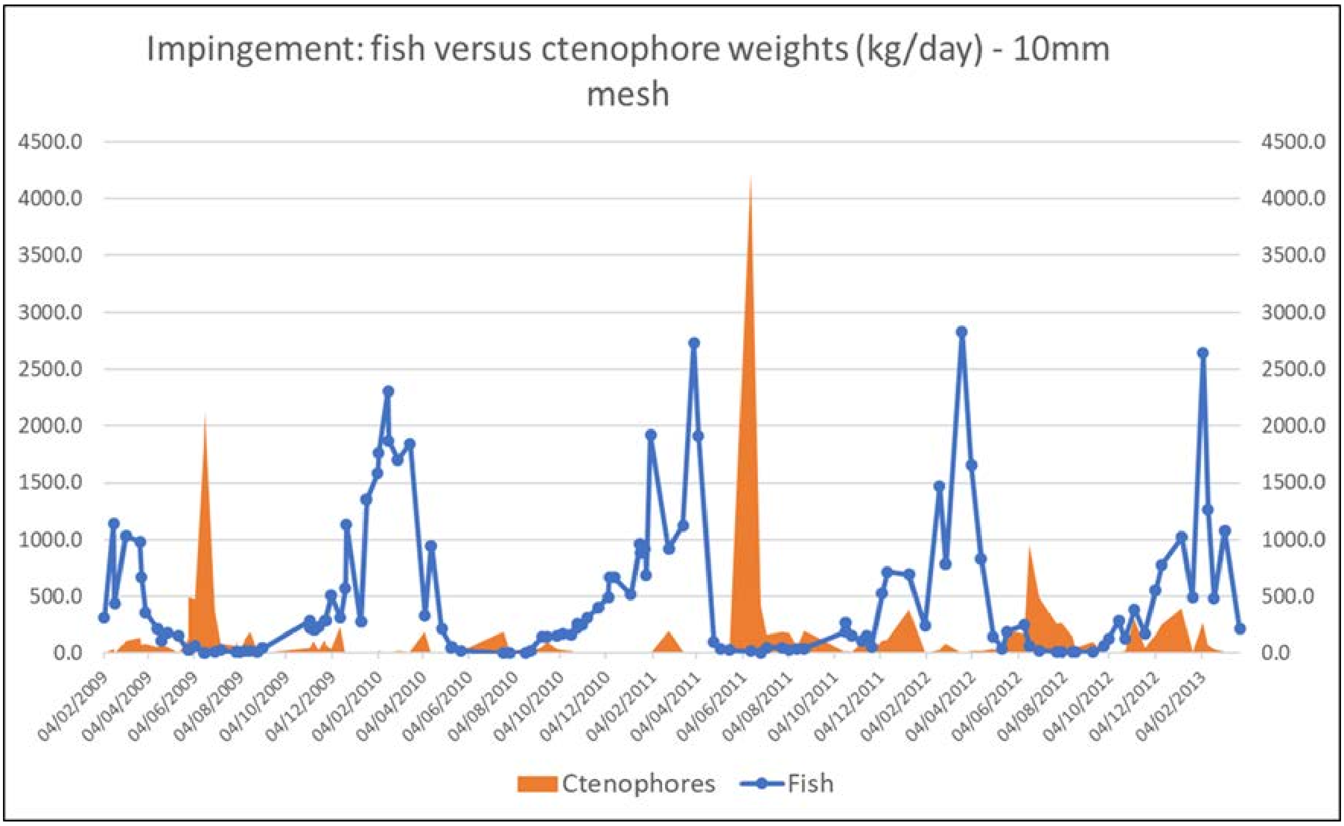

Figure 4 Comparison between estimated daytime impingement weights for ctenophores and fish from SZB full CIMP data set (BEEMS Technical Report TR406.v7).

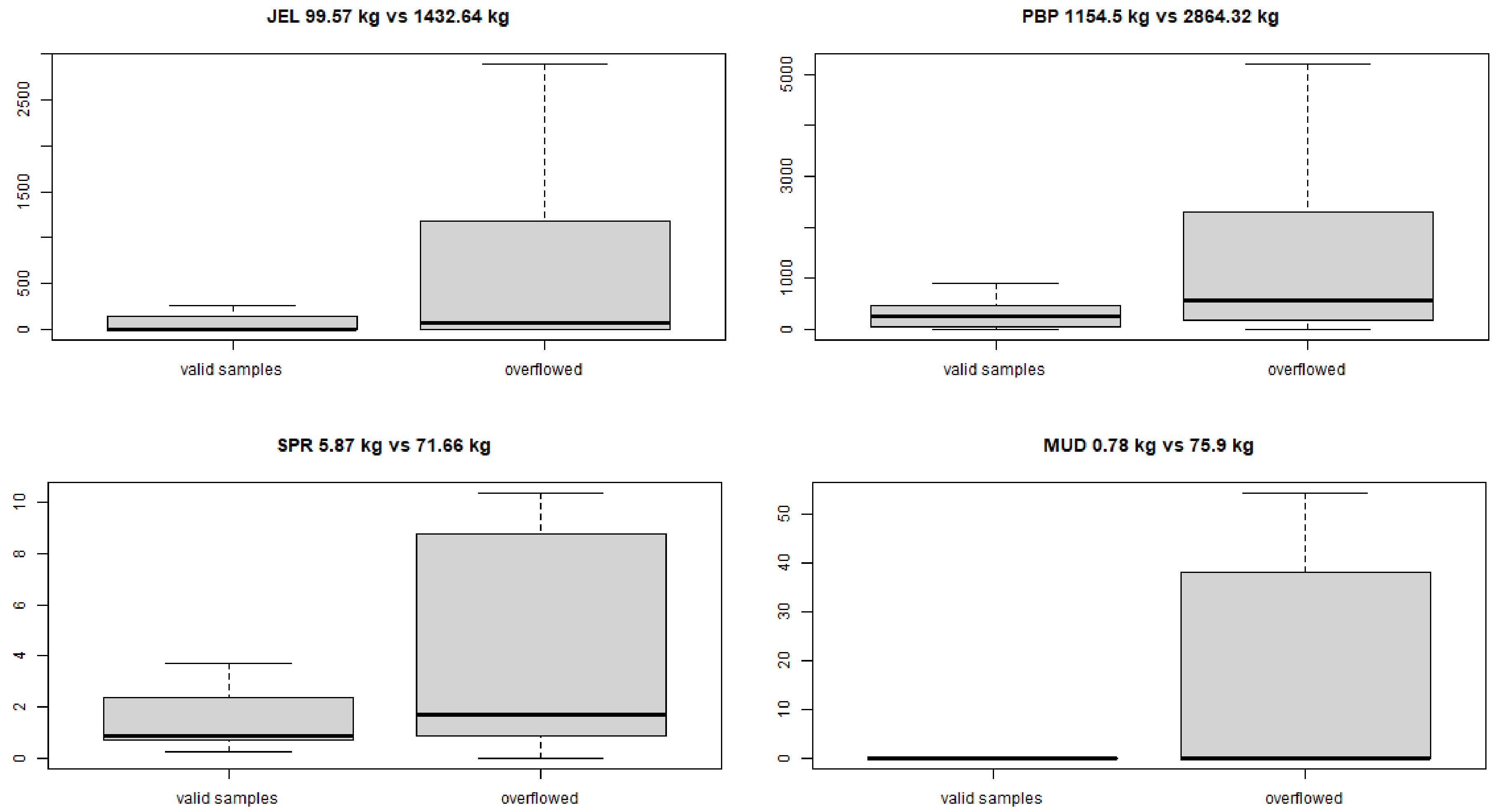

Figure 5 Weight of jellyfish (JEL), Ctenophores (PBP), mud (MUD) and sprat (SPR) in the 6h preceding the bulk sample, both when the overnight sample is valid and when it overflows. *Y*-axis scale is kg in the preceding daytime sampling period after raising to a 24h estimate. Bold horizontal lines indicate the median values of each group. Means for the valid samples group (left) versus that of the group containing only overflowed samples (right) are given in the bold title above each plot. Outliers are not shown but heavily weight the mean. Means can be considered a better representation of impingement estimates than the median in a zero-inflated series of data. By definition the median will be zero if a species is impinged on fewer than 50% of sampling visits within the data group.

Appendix B 'Entrainment Gap'

B.1 Estimating the uncertainty in the potential 'entrainment gap' between fish efficiently sampled by impingement and entrainment sampling.

In the written representation by Dr P. Henderson of Together Against Sizewell C (TASC) [REP2-481h] concerns were raised regarding the potential for an 'entrainment gap' whereby there are a proportion of fish that are too small to be impinged efficiently by the 10mm drum screen mesh but are large enough to be ineffectively sampled during entrainment monitoring. This is based on the view that the pump used to sample water from the forebay "*is an effective sampler for non-swimming life stages (e.g. eggs) and weakly swimming stages such as fish larvae"* but "*is an ineffective sampler for actively swimming juvenile fish and never catches larger fish which are strong swimmers*" [REP2-481h].

The primary species identified of concern are sprat and gobies (*Pomatoschistus* spp.). Both species spawn in waters adjacent to Sizewell. In the case of sprat all life stages including eggs, larvae, juvenile and adults have been identified in ichthyoplankton surveys (BEEMS Technical Report TR315 [APP-319]), entrainment (BEEMS Technical Report TR318 [APP-324]) and impingement (BEEMS Technical Report TR339 Rev. 3 of [AS-238]) monitoring. In the case of sand gobies, eggs are laid on benthic substrates primarily bivalve shells, and are not subject to entrainment, however, larvae, juveniles and adults have been recorded in large numbers in entrainment (BEEMS Technical Report TR318 [APP-324]) and impingement monitoring (BEEMS Technical Report TR339 Rev. 3 of [AS-238]).

Dr Henderson noted that the same applies to other clupeoid fish that are impinged at Sizewell including anchovy, pilchard and herring. He also noted that (page 15-16 [REP2-481h]): "*Juvenile smelt are recorded as impinged at Sizewell and appreciable mesh penetration of individuals less than 70 mm SL will occur. So they will be entrained*". Smelt ascent to upper estuaries and freshwaters in February to April to spawn. Most of the juvenile fish descend to the lower estuary by early autumn of their first year (Colclough and Coates, 2013) and by that time their lengths is ~ 6 cm TL (Scholle *et al.,* 2007). At this stage juvenile smelt have a body depth of approximately 10mm (Froese and Pauly, 2021), the size of the drum screen mesh. In the lowest part of Thames Estuary (Canvey Island) the smelt size is autumn is generally > 8 cm TL (Colclough and Coates, 2013). The size of juvenile smelt in the lower estuary is consistent with the size distribution of smelt impinged from marine habitats at Sizewell B. Length distribution data in Appendix E of BEEMS Technical Report TR339 Rev. 3 [AS-238] (pdf page 63) demonstrate very few smelt are impinged in the size range 6-7cm TL. It is highly unlikely that there is a significant gap in the smelt assessment as smelt small enough to be inefficiently sampled would be in the marine environment in very low abundance.

Anchovy larvae and juveniles have not been recorded in entrainment though occur in planktonic samples (BEEMS Technical Report TR315 [APP-319]) indicating low abundance in the water layers subject to water abstraction. Therefore, the problem of missed entrainment is not an appreciable concern. Juvenile sardines were also not recorded during entrainment monitoring, or in planktonic samples, whereas larger fish are impinged in negligible numbers.

In the case of herring, the potential for impacts of the station on the Blackwater herring population was raised. This potential for impacts has been considered in Section 6.6.5 of BEEMS Technical Report TR406 Rev. 7 (pdf page 150 of [AS-238]). The latest position on herring is presented in response to recent comments from Natural England in Section 2.3 of [REP6-016]. During ISH10, TASC also raised the potential for juvenile smelt to be missed from impingement sampling.

In relation to the concerns raised during ISH10 and in [REP2-481h] a summary with some technical detail is provided below:

- The design and deployment of the pump sampler followed the guidance set out by the BEEMS Expert Panel in the Science Advisory Report SAR005 and it is an efficient tool to sample eggs and larvae.
- For some species a fraction of the species life-history can be ineffectively sampled by both entrainment monitoring and impingement monitoring.
- To estimate the potential entrainment gap, SZC Co. set a maximum size for efficient sampling of entrainment at 30mm standard length (SL) and estimated the minimum size where all individuals of

a species would be impinged on the 10mm mesh at Sizewell B. This allowed the entrainment gap to be quantified as a size range (or size window) that was not fully selected by the sampling devices.
- The numbers of juvenile fish that would be expected to be present in this size range was estimated by back calculating the numbers of impinged fish by size class using existing data on growth and mortality.
- The difference between expected entrapment and observed impingement provides an estimate of the number of fish that would have been entrained but were not recorded (a detailed explanation is provided below).
- SZC Co. has provided an entrainment gap assessment for the two primary species of the Stakeholders' concern: sprat and gobies (*Pomatoschistus* spp.), as well as for herring. These species are selected as their larvae were the most abundant in ichthyoplankton surveys (BEEMS Technical Report TR315 [APP-319]) and entrainment monitoring (BEEMS Technical Report TR318 [APP-324]).
- For each of these three species, accounting for the entrainment gap resulted in an increased entrapment estimate. However, due to the small size of the fish in the entrainment gap the equivalent adult value (EAV) is small and the implications for population level effects are minimal.
- Applying precautionary approaches, the entrainment gap leads to underestimates of total sprat entrapment rates by 6.3%. That is the estimated number of equivalent adults lost due to the station increases by 304,962 equivalent adults per annum (6.3% of the estimated entrapment total).
- In the case of herring, accounting for the entrainment gap increases the total number of equivalent adults by 15,910, an increase of just 1.0%.
- For gobies (*Pomatoschistus* spp.), accounting for the entrainment gap increases the total number of equivalent adults by 589,200 or 17.5%.
- **The effects of the entrainment gap are incorporated into the uncertainty analysis used to assess population risks for sprat, herring and gobies (*Pomatoschistus* spp.).**

B.2 Sprat

Stakeholders stated that [REP2-481h]:
- *Unfortunately, sprat > 30 mm SL will be inefficiently caught by the entrainment pump sampler* (p.14)
- *On a 10 mm mesh as used at Sizewell B sprat need to be > 70 mm SL before they are always retained on the 10 mm filter screens* (p.14).

B.2.1 Entrainment

Metamorphosis of sprat larvae into juveniles occurs between 32 and 41mm TL, with some fish of 39mm TL already in the juvenile form (Lebour, 1921). Therefore, the minimum size of juvenile sprat missed by entrainment sampling is 39-41mm TL (32-33mm SL). Smaller specimens would be either larvae or larvae at the stage of metamorphosis ("half-juveniles") that were assigned as having been sufficiently small to be fully entrapped by the pump sampler'.

The Comprehensive Entrainment Monitoring Programme (CEMP) carried out between May 2010 and May 2011 provided an annual estimate of sprat entrainment at Sizewell B which has been used to predict entrainment at Sizewell C. In BEEMS Technical Report TR318 [APP-324] annual entrainment of sprat larvae by Sizewell C was predicted as 44,638,462 and accounted for 18.9% of the total number of larvae entrained of all species. Entrainment rates of juvenile sprat was predicted to be 19,419,776 (38.9% of the total number of entrained juveniles) (Tables B8 and B12 of BEEMS Technical Report TR318 [APP-324]).

It is assumed that all juvenile sprat recorded at entrainment sampling were mean 30mm TL, and at this size are effectively sampled by the pump as being < 30 mm SL. Consistent with the concerns raised by Stakeholders the entrainment gap for fish above the size of 30 mm SL (37 mm TL, size class 35-39 mm TL) was investigated.

B.2.2 Impingement

We do not agree with the position of TASC [REP2-481h] that sprat need to be > 70mm SL before they are always retained on the 10mm filter screens.

Sprat of size class 70-74mm SL corresponds to a body depth of 14.7 – 15.6mm as the fineness ratio for this species is 4.75 (Turnpenny, 1981). The Turnpenny (1981) equation for fish passage through a mesh is based on the assumption that, "*when aligned perpendicular to the screen, fish should not penetrate beyond the rear of the orbit of the eye, thereby ensuring that fish are stopped physically by the bony part of the head*" (Turnpenny, 1981). In the case of sprat, the body depth beyond the bony part of the head is too great to allow the fish to pass through a 10mm mesh based on the head dimensions (Figure 6). If the body depth of a fish appreciably exceeds 10mm, it is unlikely to be squeezed through the stainless steel mesh, but rather would be turned by the water flow to lie flat on the mesh surface and so be impinged.

Figure 6. An illustration of the body morphology of sprat showing why fish with a 10mm bony head are unlikely to pass through a 10mm mesh.

Although the suggestion that sprat need to be > 70mm SL to allow impingement is not supported by evidence, it is assumed on a precautionary basis that the minimum size suggestion by TASC stakeholders of 85-89mm TL (70-74 mm SL) should be used as the minimum size at which 100% fish are impinged. Therefore, this represents the starting point to back-calculate numbers of smaller fish between the maximum size of efficient entrainment (35-39mm TL) and the minimum size of 100% impingement (85-89mm TL). The numbers of sprat between these sizes will determine the impact of this entrainment gap.

To calculate the impact of this entrainment gap, it is assumed that growing fish decreases in numbers at a rate determined by the rate of natural mortality. Therefore, knowing numbers of a particular size class, the time that it takes to a fish to grow from one size class to another, and what proportion would die during this period, it is possible to calculate the abundance of the previous size class. To reconstruct the entrainment gap mortality rates, the equation of Gislason *et al*. (2010) was applied (as also used to estimate natural mortality in some equivalent adult value (EAV) calculations). Growth rates of larvae and early juveniles were taken from Alshuth (1989, Table 3) and averaged by size class. Growth rates of larger juveniles of 65-85mm TL were taken from the graphic representation as a mean 0.42mm/day (Baumann and Malzahn, 2007). All growth parameters are summarised in Table 15.

Table 15 Growth rates of juvenile sprat in the southern North Sea (Alshuth, 1989; Baumann, Malzahn, 2007).

Size class, mm	31-35	36-40	41-45	46-50	51-55	56-60	60-65	65-75	75-85
Mean growth rate, mm/day	0.393	0.402	0.375	0.313	0.285	0.276	0.275	0.42	0.42

Longevity of each size class (D, days) was estimated from the difference between its mid-point (mm) and the mid-point of the next size class (mm) divided by the growth rate (mm/d). For example, it would take 15.97 days for a juvenile sprat to grow from 47mm (size class 45-49mm) to 52mm (size class 50-54mm) at a rate of 0.313mm/day. Annual mortality from the Gislason formula (M) was then expressed as daily mortality (divided by 365.25) and the ratio between the unknown numbers of the precedent size class (N_1) and known numbers of the following size class (N_2) was estimated as:

$$\frac{N_1}{N_2} = \frac{1}{-\exp(DM)}$$

The resulting estimations of fish numbers that are entrapped annually but missed by entrainment and impingement sampling are presented in Table 16.

Table 16 Estimation of annual numbers of sprat entrained by SZC but inadequately recorded in pump samples. Numbers are scaled up to SZC based on SZB observed impingement.

Mid-point (TL, cm)	Length class (TL, cm)	SZC* impingement by length class	Natural mortality (annual)	Days of growing between stated size class and the next	Reconstructed numbers of expected entrapped juveniles	Entrainment Gap (difference between expected entrapment and impingement) based on assumption that all fish below 39 mm TL are recorded in pump samples
0.2	0.0-0.4	0		NA	NA	Size range efficiently sampled by entrainment pump sampler.
0.7	0.5-0.9	0		NA	NA	Size range efficiently sampled by entrainment pump sampler.
1.2	1.0-1.4	0		NA	NA	Size range efficiently sampled by entrainment pump sampler.
1.7	1.5-1.9	0		NA	NA	Size range efficiently sampled by entrainment pump sampler.
2.2	2.0-2.4	68		NA	NA	Size range efficiently sampled by entrainment pump sampler.
2.7	2.5-2.9	219		NA	NA	Size range efficiently sampled by entrainment pump sampler.
3.2	3.0-3.4	2,423		NA	NA	Size range efficiently sampled by entrainment pump sampler.
3.7	3.5-3.9	9,329		NA	NA	Size range efficiently sampled by entrainment pump sampler.
4.2	4.0-4.4	10,204	3.64	13.33	601,988	591,785
4.7	4.5-4.9	26,192	3.04	15.97	526,993	500,801
5.2	5.0-5.4	40,902	2.58	17.54	461,363	420,461
5.7	5.5-5.9	32,850	2.23	18.12	407,506	374,656
6.2	6.0-6.4	39,070	1.95	18.18	364,852	325,782
6.7	6.5-6.9	47,928	1.72	18.18	331,151	283,223
7.2	7.0-7.4	85,163	1.53	11.90	304,002	218,839
7.7	7.5-7.9	109,466	1.37	11.90	289,211	179,745
8.2	8.0-8.4	53,417	1.24	11.90	276,548	223,131
8.7	8.5-8.9	265,583				All fish impinged

* Scaled up from observed data at Sizewell B.

The absolute number of juvenile fish in the entrainment gap is 3,118,423. The EAV of this early life-history stage was calculated as 0.0978 based on the assumptions in (BEEMS Technical Report TR383). This results in total equivalent adult losses of 304,982. This represents a 6% increase on the numbers previously predicted, taking total entrapment losses to 5,127,842 equivalent adult sprat per annum (Table 17).

The numbers of sprat that are estimated to fall in the entrainment gap are included in the uncertainty analysis.

Table 17 Summary estimations of entrapped sprat numbers.

Sprat	Stage	Individuals	Adult equivalent	% of EAV mortality
Impingement numbers (Table 1)	Juveniles and adults.	6,153,906	4,623,145	90.2
Entrainment numbers TR318 entrained (Tables 11 and 12)	*Eggs*	*31,627,339*	*3,635*	
	Larvae	*44,638,462*	*171,029*	
	Juveniles	*19,419,776*	*25,052*	
Total entrainment (Table 3)	Mixed		199,715	3.9
Entrainment Gap	Mixed	3,118,423	304,982	6.0
Total losses			**5,127,842**	

B.3 Herring

The potential numbers in the entrainment gap for herring were calculated by applying the same approaches used for sprat.

Entrainment sampling at Sizewell B resulted in predicted annual entrainment of herring larvae by Sizewell C of 17,921,743 per year (7.6% of all larvae) equating to the loss of 23,992 equivalent adult fish. Entrainment of 87,346 juveniles (0.2% of all juveniles entrained) was estimated to result in negligible losses of adult fish as juvenile herring in the size range of 30mm have an EAV of 0.6×10^{-7} (Tables B8 and B12 of BEEMS Technical Report TR318 [APP-324]).

Metamorphosis of herring larvae occurs between 44-70mm TL (Blaxter, 1968) with the smallest fully metamorphised juveniles being 55mm TL (Lebour, 1921). As herring has the same fineness ratio as sprat, the considerations on the size range of sprat that may be susceptible to the entrainment gap apply for herring. We assume that entrainment sampling misses fish > 40mm TL (32mm SL) and the minimum size of fully impinged fish is the precautionarily assumed to be of the size class of 70 – 74mm SL, 85-89mm TL as suggested by TASC [REP2-481h].

Growth of metamorphic / juvenile fish between 45 and 85 mm TL is approximately linear (data from Arrhenius and Hansson, 1996) and the age of early juveniles in days (D) was estimated as:
$D=1.999 * TL – 0.5166$

Therefore, it takes ~10 days on average to grow between adjacent 5mm size classes. Observed growth rates of these juveniles are generally in the agreement with the v. Bertalanffy growth curve used in previous assessments of EAV at Sizewell.

The relative numbers of impinged herring of 75-120mm TL are low in relation to numbers in larger and smaller size classes. It is known that herring juveniles in estuaries are concentrated in the upper water layers and there are no changes in their vertical distribution between day and night (Maes *et al.,* 1999). It is possible that the low impingement rates of herring in the 75mm to 120mm size classes result from these juveniles having low persistence at the depth range where they might be subjected to entrapment. Consequently, the estimated numbers of fish in the range 40-84mm TL entrapped (impingement + entrainment) and back calculated from 85-89mm TL were lower than observed in the impingement data.

To account for this inconsistency, we assumed that the relative proportion of missed entrainment would be the same in herring and sprat. This is reasonable given their similar body shape. For example, if 72% (218,839 of 304,002, Table 16) of entrapped sprat of 70-74 mm TL were missed by the pump sampler and impingement monitoring, exactly the same proportion (161,452 of 206,681, Table 18) of similar-sized herring would be assumed to be missed.

Based on this assumption, the total number of juvenile fish within the entrainment gap was estimated as 4,792,065 in the range of 40-84mm TL (Table 18). Applying the same approach and using the same primary data to calculate EAV (BEEMS Technical Report TR383) the EAV for these juvenile herring 0.00332 resulted in an adult equivalent of 15,910. This represents a 1.0% increase in the adult equivalent numbers as previously predicted, taking total entrapment losses to 1,621,787 equivalent adult herring per annum (Table 19).

The numbers of herring estimated to fall in the entrainment gap are included in the uncertainty analysis.

Table 18 Estimations of annual numbers of herring entrained by SZC but missed from calculations based on the pump sampler.

Mid-point (TL, cm)	Length class (TL, cm)	SZC* impingement by length class	Proportion of juveniles missed at sampling	Reconstructed numbers of entrapped juveniles	Entrainment Gap (difference between expected entrapment and impingement) based on assumption that all fish below 39 mm TL are recorded in pump samples
0.2	0.0-0.4	0		NA	Size range efficiently sampled by entrainment pump sampler.
0.7	0.5-0.9	0		NA	Size range efficiently sampled by entrainment pump sampler.
1.2	1.0-1.4	0		NA	Size range efficiently sampled by entrainment pump sampler.
1.7	1.5-1.9	0		NA	Size range efficiently sampled by entrainment pump sampler.
2.2	2.0-2.4	63		NA	Size range efficiently sampled by entrainment pump sampler.
2.7	2.5-2.9	294		NA	Size range efficiently sampled by entrainment pump sampler.
3.2	3.0-3.4	4,115		NA	Size range efficiently sampled by entrainment pump sampler.
3.7	3.5-3.9	10,400		NA	Size range efficiently sampled by entrainment pump sampler.
4.2	4.0-4.4	32,788	0.98	1,967,193	1,934,404
4.7	4.5-4.9	39,700	0.95	838,480	798,781
5.2	5.0-5.4	60,580	0.91	743,913	683,333
5.7	5.5-5.9	32,982	0.92	442,124	409,142
6.2	6.0-6.4	38,802	0.89	401,160	362,357
6.7	6.5-6.9	40,410	0.86	319,615	279,206
7.2	7.0-7.4	45,229	0.72	206,681	161,452
7.7	7.5-7.9	6,253	0.62	22,773	16,520
8.2	8.0-8.4	28,369	0.81	175,239	146,870
8.7	8.5-8.9	22,433			All fish impinged

* Scaled up from observed data at Sizewell B.

Table 19 Summary estimations of entrapped herring.

Herring	Stage	Individuals	Adult equivalent	% of EAV total
Impingement numbers (Table 1)	Juveniles and adults	2,211,750	1,581,885	97.5
Entrainment numbers TR318 entrained (Tables 11 and 12)	Larvae	17,921,743	23,992	
	Juveniles	87,346	0	
Total entrainment (Table 3)	Total entrainment		23,992	1.5
Entrainment Gap	Juveniles	4,792,065	15,910	1.0
Total losses			1,621,787	

B.4 Gobies (*Pomatoschistus* spp.)

The sand goby *Pomatoschistus minutus* is the dominant species in the *Pomatoschistus* spp. genera at Sizewell. Life history parameters of *P. minutus* have been applied in back-calculations to determine numbers of fish in the entrainment gap for the sand goby group.

The CEMP carried out between May 2010 and May 2011 provided an annual estimate of entrainment of gobies (*Pomatoschistus* spp.) at Sizewell B from which Sizewell C entrainment was predicted as 153,250,186 larvae (64.7% of the total larvae) and 22,375,425 juveniles (44.8% of total) (Tables B8 and B12 of BEEMS Technical Report TR318 [APP-324]). All these juveniles are assumed to be of mean 30mm TL, which is conservative. The size range of these juvenile gobies is ~18-30 mm TL with smaller fish predominating; at smaller sizes mortality rates are higher and therefore the actual EAV applied is precautionary.

As TASC [REP2-481h] indicated for sprat where fish > 30mm SL are inefficiently sampled, we use the same threshold (30 mm SL) for gobies, and this size corresponds to 34mm TL. Therefore, we address gobies missed from sampling as being >35 mm TL.

Sand gobies have fineness ratio of 5.70 (Turnpenny, 1981) so 10mm body depth corresponds to 57mm SL and 67.5mm TL respectively. This size threshold of 100% impingement generally does not contradict visual representation of the sigmoid curve presented by TASC [REP2-481h], recreated in Figure 7, which predicts the retention rate at this length to be ~95-97%.

Sand gobies of 87mm SL (as suggested by TASC [REP2-481h]) would have a body depth of 15.3mm. This far exceeds the minimum size to be fully retained by 10 mm mesh. Therefore, we precautionarily assume the smallest size class subjected to 100% impingement by a 10mm mesh is 70-74mm TL (62-65mm SL) with a body depth of 10.9-11.4mm. This is based on the maximum body depth being 10% higher than 10mm mesh size.

Therefore, we precautionarily defined the entrainment gap as occurring between the size classes of 35-39mm and 70-74mm TL. Further, we complete a verification analysis extending the minimum size of effective entrainment sampling down to 20-24 mm TL (smallest juveniles) to find out whether backpropagated numbers of these fish would be incompatible with previous estimations of their numbers by the pump sampler.

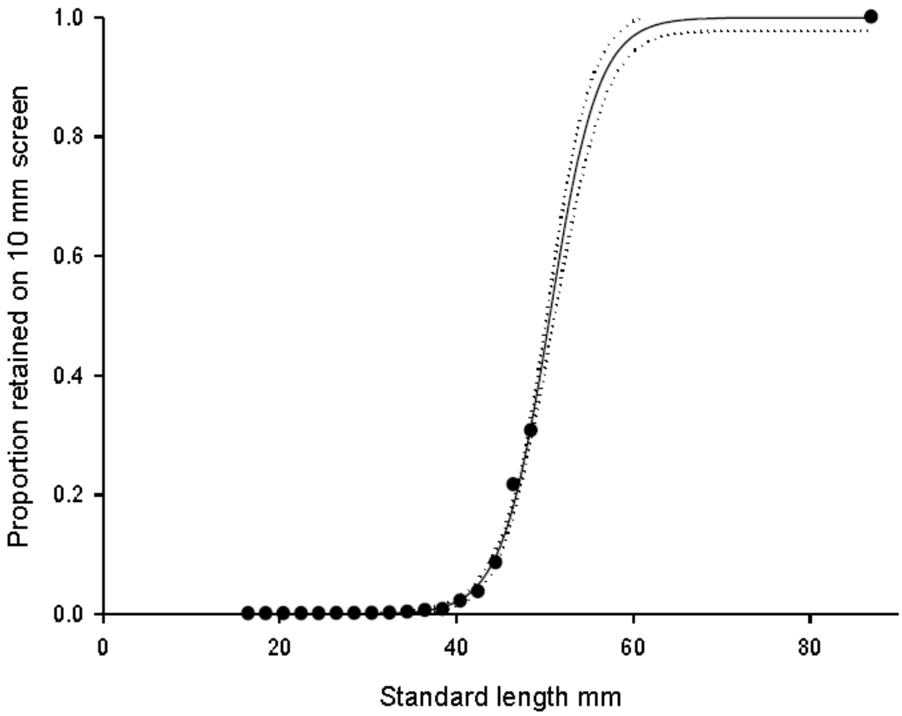

Figure 7. The proportion of sand goby *Pomatoschistus* spp. retained on a 10mm square mesh provided by TASC in REP2-481h.

To estimate size-specific mortality rates the equations of Gislason *et al.,* (2010) were applied. Growth rates of post-larval sand gobies were taken from FishBase for the southern North Sea (parameters of v.Bertalanffy equation are Linf=9.2 (cm), K=0.928, Froese and Pauly, 2021). This equation was used to calculate the mean age in days for the mid-point of each size class. The following step involved determining the longevity of each size class from the difference between its age and the age of the next size class (Table 20). The ratio between the unknown numbers of the precedent size class and known numbers of the following size class was estimated as for sprat and herring.

The previous prediction of gobies (*Pomatoschistus* spp.) being impinged by SZC was 483,487 with 152.8 million juveniles and larvae entrained (Table 20 Table 21). The entrainment gap approach resulted in an estimated total of 2,960,806 additional gobies (*Pomatoschistus* spp.) in the size range 35 to 70mm TL being entrained by SZC.

In [REP2-481h], TASC suggest that the missing entrainment fraction would result in total entrapment (entrainment + impingement) numbers for all life stages increasing from 153 million to 802 million per annum. The TASC estimate was based on the statement that (pages 12-13 of REP2-481h) "*Using the mesh penetration data, an impingement estimate of 100 individuals represents a total entrainment of 1.7 x 10^5 individuals in the size range 20-65 mm SL. For each sand goby impinged approximately 1700 pass through the system*". TASC did not provide calculations to support this value but from the Table on p.13 [REP2-481h] we consider that it applies for fish > 20mm SL (23mm TL). All gobies of this size and above should be considered as juveniles because the switch from larval to juvenile morphology and life style occurs in sand gobies at ~17-18 mm TL (Fonds, 1973; Riley, 2007). All gobies of this size range (20-34mm TL) should be efficiently sampled in entrainment monitoring and are already included in the assessment. This assumption might be verified if numbers of juveniles of 20-34 mm TL estimated by the CEMP were compared with back calculated numbers as estimated using our approach.

To compare our results with the TASC estimates, we recalculated missed goby entrainment based on the supposition of Dr Henderson and included all juveniles down to the minimum size of 20 mm TL, thus assuming the CEMP sampling did not capture these individuals. This resulted in additional 17,872,140 juveniles being entrained and not recorded, but this number is still far below the additional 648,740,400 fish referenced by TASC (REP2-481h). Among the ~17.9 million fish that were entrained and not recorded , some 14,911,333 are in the size 20-34 mm TL. This is similar to the numbers of juveniles of the same size recorded by pump sampler (22,375,425; BEEMS Technical Report TR318 - APP-324). Therefore, we assume that the CEMP effectively sampled all gobies bellow 35mm TL and this provides the suitable minimum size for the entrainment gap.

We conclude that the 802 million additional fish estimated by TASC is a considerable overestimate.

Table 20 Estimations of entrained fish missed by the pump sampler. Juveniles below 35 mm TL (not used in estimations as to avoid double counting) are bolded.

Mid-point (TL, cm)	Length class (TL, cm)	SZC impingement	Natural mortality (annual)	Age, days	Days of growing between this size class and the next	Reconstructed numbers of entrapped juveniles	Entrainment Gap (difference between expected entrapment and impingement) based on assumption that all fish below 39 mm TL are recorded in pump samples
2.2	**2.0-2.4**	**643**	**10.32**	**107.49**	**29.1**	**8,931,357**	**8,930,714**
2.7	**2.5-2.9**	**403**	**7.42**	**136.64**	**31.5**	**3,918,739**	**3,918,336**
3.2	**3.0-3.4**	**4334**	**5.65**	**168.12**	**34.4**	**2,066,617**	**2,062,284**
3.7	3.5-3.9	10,812	4.47	202.5	37.5	1,214,906	1,204,093
4.2	4.0-4.4	27,060	3.64	240.0	41.5	767,656	740,596
4.7	4.5-4.9	48,500	3.04	281.5	46.4	507,518	459,018
5.2	5.0-5.4	75,558	2.58	327.8	52.6	345,004	269,446
5.7	5.5-5.9	86,536	2.23	380.4	60.7	237,865	151,329
6.2	6.0-6.4	104,632	1.95	441.1	71.8	164,254	59,622
6.7	6.5-6.9	35,344	1.72	512.8	87.4	112,046	76,702
7.2	7.0-7.4	74265	1.53	600.2	87.4	74,265	All fish of this size impinged

The age of 50% maturity of *P. minutus* in the Northeast Atlantic is 1 year (Bouchereau and Guelorget, 1998). Impingement calculations precautionarily assumed all gobies (*Pomatoschistus* spp.) had an EAV on 1. To determine the EAV for the missing fraction we assume that the age of 100% maturity is 1.5 years as the maximum age is 2.7 years (Froese and Pauly, 2021) and that all gobies would be mature before the second year. The resultant EAV is 0.199 for fish in the entrainment gap. The total number of gobies (*Pomatoschistus* spp.) in the entrainment gap is estimated to be 2,960,806 individuals equating to 589,200 equivalent adults per annum (Table 21). This represents approximately 17.5% more sand gobies than previously estimated. However, actual losses would be much lower as in contrast to clupeoids and other vulnerable fish, survival of entrained Gobiidae is high even among larvae – 88-98% (Mayhew *et al*., 2000) and similar survival rates (84-89%) have been observed for juvenile gobies in impingement survival trials (Jacobs UK Ltd. 2016).

We also explored a possibility that the entrainment gap extends to fish as small as 20 mm TL, though we believe that this size category was fully sampled during CEMP protocol so does not belong to the missed entrainment. An EAV for the missed entrainment is 0.033 that would equal to losses of 589,781 equivalent adults and it is just ~600 fish more than losses estimated using the method detailed herein (Table 21).

Table 21 Estimations of annual numbers of gobies (*Pomatoschistus* spp.) entrained by SZC but missed from calculations based on the pump sampler.

Gobies (*Pomatoschistus* spp.)	Stage	Individuals	Adult equivalent	% of total
SPP111 impinged (Table 3); [REP2-110	Juveniles and adults	483,487	483,487	12.2
TR318 Tables B8 and B12 [APP-324]	Larvae	153,250,186		
	Juveniles	22,375,425		
Total entrainment (Table 12 APP-324)			2,892,198	72.9
Missed entertainment	Mixed	2,960,806	589,200	14.9
Total losses		**156,238,575**	**3,964,885**	
Missed entertainment calculated following the TASC approach in REP2-481h	Mixed	17,872,140	589,781	
Total losses following the TASC approach in REP2-481h		**171,149,909**	**3,965,466**	

References:

Alshuth, S. 1989. Growth larval and juvenile North Sea sprat, based on otolith growth increments. ICES C.M. 1989/H:15 12 pp.

Arrhenius, F. and Hansson, S. 1996. Growth and seasonal changes in energy content of young Baltic Sea herring (*Clupea harengus* L.). – ICES Journal of Marine Science, 53: 792–801.

Baumann H., Malzahn, A.M. 2007. New analyses of juvenile sprat growth and temporal origin in the German Bight (North Sea). GLOBEC International Newsletter. April 2007. 2 pp. Available at: https://epic.awi.de/id/eprint/16666/1/Bau2007b.pdf

Blaxter, J.H.S. 1968. Rearing herring larvae to metamorphosis and beyond. Journal of the Marine Biological Association of the United Kingdom 48, 17-28.

Bouchereau, J-L., Guelorget, O. 1998. Comparison of three Gobiidae (Teleostei) life history strategies over their geographical range. Oceanologica Acta 21, 503-517.

Colclough, S., Coates, S. 2013. A Review of the status of Smelt *Osmerus eperlanus* (L.) in England and Wales – 2013. Final Report. Colclough & Coates – SC2 Ltd. 60 pp.

Fonds, M. 1973. Sand gobies in the Dutch Wadden Sea (*Pomatoschistus*, Gobiidae, Pisces). Netherlands Journal of Sea Research 6: 417-478.

Froese, R. and D. Pauly. Editors. 2021. FishBase. World Wide Web electronic publication. www.fishbase.org, version (06/2021).

Gislason, H., Daan, N., Rice, J.C., Pope, J.G., 2010. Size, growth, temperature and the natural mortality of marine fish. Fish Fish. 11, 149–158.

Jacobs U.K. Limited 2016. The survivability of biota impinged on Pembroke Power Station cooling water screens. Document No JUKL/B1810700/R56. Southampton, 59 pp.

Lebour, M.V. 1921. The larval and post-larval stages of the pilchard, sprat and herring from Plymouth

District. Journal of the Marine Biological Association of the United Kingdom, 12, 427-457

Maes, J., Pas, Z., Tallieu, A., Van Damme, P.A., Ollevier, F. 1999. Diel changes in the vertical distribution of juvenile fish in the Zeeschelde Estuary. J. Fish. Biol. 54, 1329–1333.

Mayhew, D.A., Jensen, L.D., Hanson, D.F., Muessig, P.H. 2000. A comparative review of entrainment survival studies at power plants in estuarine environment. Environmental Science and Policy 3: 295-301.

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. [cited 10-09-2021]. Available from: https://www.marlin.ac.uk/species/detail/1204

Scholle, J., Schuchardt, B., Schultze, S., Venkenstedt, J. 2007. Situation of the smelt (Osmerus eperlanus) in the Ems estuary with regard to the aspects of spawning grounds and recruitment. BIOCONSULT Schuchardt & Scholle GbR. 101 pp.

Thorisson, K. 1994. Is metamorphosis a critical interval in the early life of marine fishes? Environmental Biology of Fishes 40,23-36.

Turnpenny, W.H. 1981. An analysis of mesh sizes required for screening fishes at water intakes. Estuaries 4, 363-368.

Appendix C Shad population estimates

C.1 Estimating uncertainty in the population estimates of shad in the Elbe and Sheldt

No spawning populations of twaite shad occur on the UK east coast. The closest breeding populations of twaite shad occur in mainland Europe. Natural England in their Deadline 2 submission [REP2-153] advised that consistent with a precautionary HRA approach, predicted losses of twaite shad from Sizewell C should be assigned against each breeding population given genetic information is not available to determine the source population. Fish monitoring programmes in German and Belgian estuaries are undertaken to determine trends in fish populations. However, to the best of our knowledge, absolute population estimates are not available for the designated sites. It would not be proportionate to attempt to determine the population estimates for all twelve European mainland designated sites screened into the HRA when data is not available, and the predicted impacts are very low.

Given the distance of the proposed development from the spawning rivers (hundreds of kilometres from Sizewell, Section 2.1.7.1) and the fact the development is in an open coastal environment, it is highly unlikely all fish impinged at Sizewell would come from any given river system. However, population estimates from two European systems: the Elbe, approximately 500km from SZC, and the Scheldt approximately 200km away where estimated by Cefas (BEEMS Scientific Position Paper SPP100 [AS-238]) and applied individually as comparators for population effects.

Natural England [REP2-153] and the Environment Agency [REP2-135] in their written representations at Deadline 2 questioned the uncertainty in the methods applied to determine the population estimates. Noting the inherent limitations associated with inferring population estimates from monitoring data, this section attempts to address regulatory concerns and provide confidence intervals to the mean population estimate.

C.1.1 Sampling uncertainties

In the Written Representation [REP2-135] Environmental Agency stated that assessments of shad in the estuaries of the Elbe and Scheldt are provided without addressing uncertainty of estimated numbers, and particularly mentions that:

"*There seem to be numerous assumptions being made (constant migration rate over the season, no difference in diurnal and nocturnal migration rate, constant migration rate over the width of the channel). Without confidence limits it is difficult to interpret the population estimates. Also, the inherent uncertainty might be why Magrath & Thiel presented results as individuals per 10^6 m^3 rather than population estimates*."

"No consideration is given to the shoaling behaviour of twaite shad. If a shoal is caught, the number of individuals per unit time may appear high but shad will not be evenly distributed in space and time".

This technical appendix attempts to address some of the uncertainties in the population estimate raised by the Environment Agency. In relation to the point raised regarding presented results as individuals per 10^6 m^3 rather than population estimates. We acknowledge and are aware of the limitations and difficulties in attempting to estimate a population estimate and understand why the cited authors chose to express results as a density when investigating population trends. However, the aim here is to provide a comparator value with respective uncertainty for the assessment of impacts of the station. Whilst density estimates are less prone to the necessary assumptions made herein, it is not possible to compare losses from the station against the density of fish in a European Estuary. For this reason, and in the absence of established population estimates, attempts have been made to estimate the population of twaite shad in the Scheldt and the Elbe to allow comparators for losses from the station.

The population estimates provided within this report are subject to the assumptions and limitations described in the following sections. Notwithstanding the remaining uncertainties it is important to note the scale of the predicted impact and the application of the best available information.

C.1.2 Migratory Period

Subadult shad forage in marine environment for several years before beginning their spawning migration into natal rivers to reproduce in fresh waters. This migration usually extends for three to four months in southern populations, and two to three months in northern stocks (Hillman, 2003; Aprahamian *et al.,* 2015). For example, the annual migration of the North Sea twaite shad stock into the lower Elbe Estuary occurs from April to June (Thiel *et al*., 1996). During this 2-3 month period fish enter the river in a series of waves (Aprahamian *et al*., 2015). The spawners move upstream and might be captured by passive fishing gears e.g., traps. Knowledge of intensity of migration (number of fish passing across the river section per day) and duration of the entire migration, permits an estimate of total numbers of spawners that ascended into fresh waters to reproduce to be made within established limitations.

The monitoring data in the Elbe and the Scheldt were taken during the twaite shad migratory period but does not encompass for the full period of migration. Estimations made herein, as well as those in BEEMS Scientific Position Paper SPP100 [AS-238] are based on extrapolation of these observed snapshots within the migration window scaled up to a migratory period assumed to be just 30 days. Taking into account that actual migratory activity lasts for at least two months, this approach should be considered as precautionary.

C.1.3 Density across the estuary

Migratory fish might move upstream at different depths (e.g. at the bottom or at the surface) and may be unevenly distributed across the river. This behavioural pattern for example is observed for Atlantic salmon that save energy to go against the current by preferring areas with high turbulence and low velocity (Lindberg, 2016). Twaite shad are also known to migrate in the lower half of the water column close to the river bed, where water velocity is lowest (Aprahamian *et al*., 2015).

As the anchor nets were extended all the way from the bottom to on or above the waterline (BEEMS Scientific Position Paper SPP100 [AS-238]), the patterns in vertical distribution of migrating shad did not impact our estimations. In relation to the assumption of homogenous density across the width of the channel, data used in the population estimate came from two Elbe stations, one of which (Kollmar) is situated in the mainstream and another one at Krausand/Gluckstadt relatively close to the shoreline. Assessments were done for both stations independently, and respective values were found to be very similar. The final value was taken as the average between two situations (BEEMS Scientific Position Paper SPP100 [AS-238]; Table 1). Whilst this represents only two points within the estuary width, the assumption of homogenous distribution may not be unreasonable, at least for the lower part of the Elbe Estuary. Moreover, the Elbe is a highly engineered water body. It is acknowledged that this is an assumption but there is no clear evidence to provide an alternative to this approach.

C.1.4 Schooling behaviour

Spawning shads are known to migrate upstream in large, loosely aggregated shoals (Gregory and Clabburn, 2003). The Environment Agency correctly noted that schooling behaviour might impact captured numbers so would result in high variability of observed catches as it was evidenced by personal observations; "*EA staff have participated in anchor netting on the Scheldt, catches vary hugely each day (with zero catch days not being uncommon)*" ([REP2-135] p.71).

Such an issue is a common phenomenon occurring throughout fisheries surveys and can be solved statistically through calculations of appropriate confidence intervals as completed below.

C.1.5 Anchor net catchability

No fishing gear is a 100% effective sampler. Therefore, fish abundance estimated from observed catches of a particular net surface would be always lower than in reality unless there is some concentrating factor in associated with the gear like doors of trawls. Such a concentrating device is absent in the framed anchor nets.

Efficiency of anchor nets in respect of catchability of adult shad is difficult to assess. There are three aspects to the selectivity and efficiency of this type of nets. First, like in all passive gear, their efficiency is directly related to the probability that a fish will encounter the net. This largely depends on fish density. The second aspect is the probability that once fish encounter the net, they enter it. This depends on water transparency and fish reaction to the mesh and the beams of the frame. The third aspect is the probability of fish that enter the net are retained. Tagging studies have demonstrated that in some species all fish large enough to be retained by the mesh escaped from trap nets within three days (Portt *et al.,* 2006). Catchability of anchor nets is well below 100%. For example, catch rates of stationary stow nets (a type of anchor net) compared to impingement by intake heads the Doel power plant in the Scheldt (Zeeschelde Estuary, Belgium), demonstrated that fish densities of all species combined and calculated from stow net catches were approximately 59.5% of those recorded in abstracted water (Maes *et al.,* 2001).

C.1.6 Accounting for uncertainty

To provide confidence intervals for the estimated abundance of shad in Elbe Estuary we used primary data as provided by Behörde für Umwelt und Energie – Hamburg“ (Daten 2000-2012) and „Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küstenschutz und Naturschutz – Betriebsstelle Stade“ (Daten 2013-2018) as described in BEEMS Scientific Position Paper SPP100 [AS-238]. The uncertainty was assessed for each individual year as well as for the entire period 2009-2017 to derive the probable range of values of the comparator.

The assessment of Scheldt population was based on the single station at Antwerp and used primary data provided as summaries in reports from the Instituut voor Natuur- en Bosonderzoek (INBO). Adults were recorded every year in the Scheldt. However, successful recruitment of juveniles was observed only in 2012, 2015, 2017, 2018, 2019 and 2020 (Breine *et al.,* 2021). Therefore, the recent recovery of the Scheldt and lack of annual recruitment suggests the population is not fully self-sustaining and is still supported by immigration from other rivers so does not represent a discrete stock. Confidence intervals in the “population” estimate for the Scheldt are provided only for interannual variability during the period 2012-2016.

C.1.7 Diurnal behaviour

The Environmental Agency [REP2-135] note that there might be differences between diurnal and nocturnal migration rates. Behavioural patterns during this estuarine to freshwater pre-spawning migration might be very different from that observed at Sizewell when impinged fish are foraging in marine environment and is far from maturity. A small dataset of Hillman (2003) demonstrated that Allis shad migrate both day and night, but rates of migrations might be higher in the daytime (Figure 8). A similar type of higher diurnal activity with fish shoals recorded mostly between 5 a.m. and 8 p.m. was suggested for twaite shad (Aprahamian *et al.,* 2003, 2015). However, it is not clear whether at night shad stop migrating altogether or rather disperse from loosely aggregated shoals and migrate individually as normally occurs in freshwater pelagic fish in rivers (Muška *et al.,* 2018). Whilst it is acknowledged that diurnal behaviours may differ for twaite shad in active migration in comparison to marine feeding, it is noted that diurnal bias in impingement predictions indicated minimal diurnal differences with slightly higher predicted impingement rates at night (Appendix A3).

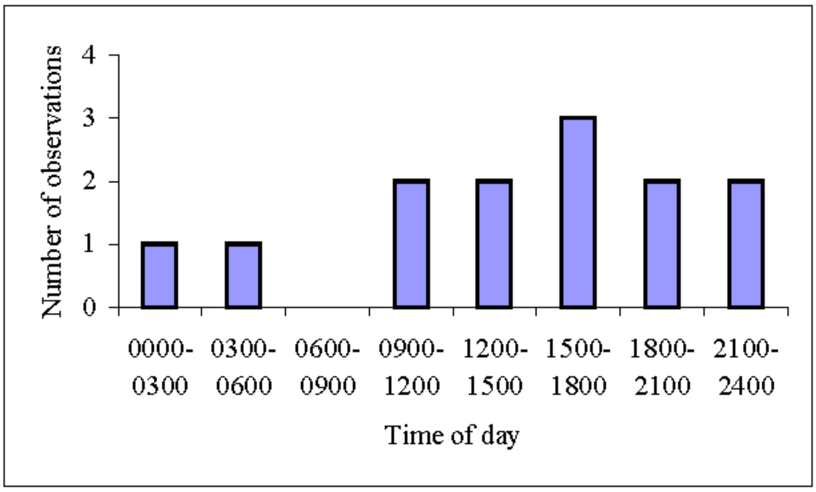

Figure 8. Diurnal timing of upstream Allis adult shad migration through Gunnislake fish pass (Fig. 37 from Hillman 2003)

To address possible bias in diurnal changes in migration rates, the light hours (day plus civil twilight) were assumed to be 15 hours in April, when the Scheldt Estuary was sampled, and 19 hours at the end of April - May, when the Elbe samples were taken (https://www.timeanddate.com/sun/uk/london?month=4&year=2021). Respective hours of darkness (night and dusk period of astronomical twilight excluding civilian twilight) were taken as 9 and 5 hours.

Conservatively, the rate of migration in darkness and the darker part of dusk was taken as 50% of that during the day and brighter part of dusk. As most observations on shad abundance were collected during daytime, we imply correction factors of 0.81 for Scheldt estimates (as equal to (1*15+0.5*9)/24) and 0.90 for Elbe estimates (as equal to (1*19+0.5*5)/24).

Available data per station (Elbe) or per sampling period (Scheldt) were bootstrapped with 10,000 iterations using R-package *boot* in the software R v. 4.1.0.

C.2 Scheldt

To estimate twaite shad in the Scheldt, the assessment was based on the assumptions that the migratory period lasts only 30 days and fish migrating across the entire width of the estuary but assumed 50% lower migration rates at night. The method also assumes 100% catch efficiency of twaite shad in anchor nets. The methodology was the same as applied in BEEMS Scientific Position Paper SPP100 [AS-238]. Following this approach, the shad catches per station, which duration was of 2-5 hours were scaled up to 24 hours and then to 30 days. Then these numbers were scaled to the full estuary width by multiplying by the estuary width at the sampling location and divided by the anchor net width (8m) as the net sampled the entire river depth from the bottom to the surface. The diurnal correction factor was applied *post-hoc.*

The estimated abundance of twaite shad in the Scheldt Estuary after applying the diurnal correction factor in 2012-2016 was estimated as **54,039** fish (Table 22). Lower and upper 95% confidence intervals from bootstrapping ranged from **20,412** to **130,203** adult fish.

The twaite shad numbers in 2018 and 2019 were also considered and estimated as 10,057 and 32,805 fish respectively. No spring survey was carried out in 2020. The summary estimation of shad abundance in the Scheldt Estuary (2012-2019) was calculated as mean of 44,722 with lower and upper limits between 20,402 and 105,965 adult fish respectively. The period from 2012-2016 is applied as the population comparator in the uncertainty analysis.

Table 22 Estimated numbers of twaite shad in the Scheldt Estuary (Antwerp) corrected for the diel bias.

Year	2012	2013	2014	2015	2016	2017	2018	2019	Mean 2012-2016
SPP100	66,385	8,904	30,300	29,281	198,705	NA			66,715
This study	53,772	7,212	24,543	23,718	160,951	NA	10,057	32,805	**54,039**

C.3 Elbe

The population estimate for twaite shad in the Elbe was estimated in a similar fashion to the Scheldt. Total abundance of shad in the Elbe Estuary for the period of 2009-2017 was estimated as **3,838,726** adult fish with lower and upper confidence intervals of **2,117,996** and **7,175,654** adult fish. Sampling of the different stations allowed bootstrapping and estimation of confidence intervals for each year. Uncertainty of annual stock size estimations are presented in Table 23.

Table 23 Estimated numbers of twaite shad in the Elbe Estuary corrected for diel bias (multiplied by 0.9) with annual 95% confidence intervals

Year	Mean	Low	High	SPP100
2007	1,812,897	416,429	3,080,161	2,023,858
2008	3,749,614	85,050	9,007,168	4,374,212
2009	11,942,180	954,273	28,774,945	12,946,294
2010	1,360,327	197,871	3,283,151	1,659,773
2011	1,073,925	368,586	1,853,950	385,132
2012	40,151	0	83,315	52,052
2013	96,188	11,716	168,944	117,650
2014	246,775	124,971	293,292	270,258
2015	7,626,827	1,748,594	13,486,500	8,475,065
2016	7,911,800	1,467,113	17,112,132	17,128,995
2017	4,250,363	2,504,371	5,787,394	5,081,855
Mean 2009-2017	**3,838,726**	**2,117,996**	**7,175,654**	5,124,119
Standard Deviation	4,357,458			6,377,150
Coefficient of Variance	1.14			1.24

The revised estimates of shad numbers in the Elbe Estuary are lower than those presented before in BEEMS Scientific Position Paper SPP100 [AS-238]. This is partially caused by the diel correction factor applied in new analysis and application of bootstrapping. Changes in the annual estimates differ notably in 2011 and 2016; in 2011 two additional sample points not previously included have been added to the data. Furthermore an error was identified in the analyses for 2016 in the original in BEEMS Scientific Position Paper SPP100 [AS-238] estimate leading to an overestimate in the annual population estimates, this has been corrected.

Twaite shad exhibit site fidelity with > 90% of fish returning to natal rivers to spawn. In the Severn Estuary 3% of twaite shad have been reported to stray from natal rivers (Davies *et al.,* 2020). If only 3% of fish from Elbe population stray to alternative river systems this would result in ~115,000 Elbe spawners supporting adjacent stocks. This number twice exceeds the estimated abundance of shad in the Scheldt Estuary. The Elbe contributes to the greatest numbers of twaite shad in German estuaries and it is likely that the recovery of the Scheldt (and Weser) population is seeded from fish straying from surrounding systems such as the Elbe.

References:

Aprahamian, M. W., Aprahamian, C. D., Baglinière, J. L, Sabatié, M. R. 2003. *Alosa alosa* and *Alosa fallax* spp. Literature Review and Bibliography. R&D Technical Report W1-014/TR. Environmental Agency, 374 pp.

Alexandrino, P.2001. Variation in the age of the first spawning of female twaite shad (*Alosa fallax fallax*) from the river Severn, England. Bulletin français de la pêche et de la pisciculture. 362/363: 941-951.

Aprahamian, M., Alexandrino, P., Antunes, C., Cobo, F., King, J. et al. 2015.Shads state of the art :3. Report of the ICES Workshop on Lampreys and Shads (WKLS) : 46-105. https://hal.inrae.fr/hal-02602614

Davies, P., Britton, R.J., Nunn, A.D., Dodd, J.R., Crundwell, C., Ve;terop, R., Ó'Maoiléidigh, N., O'Neill, R., Sheehan, E.V., Stamp, T., Bolland, J.D. 2020. Novel insights into the marine phase and river fidelity of anadromous twaite shad Alosa fallax in the UK and Ireland. Aquatic Conservation. Marine and Freshwater Ecosystems, 30: 1291-1298.

Gregory, J., Clabburn, P. 2003. Avoidance behaviour of *Alosa fallax fallax* to pulsed ultrasound and its potential as a technique for monitoring clupeid spawning migration in a shallow river. Aquatic Living resources 16: 313-316.

Hillman R. 2003. The Distribution, Biology and Ecology of Shad in South-West England. R&D Technical Report W1-047/TR. Environment Agency, Bristol 163 pp.

ICES. 2020. Benchmark Workshop for Demersal Species (WKDEM). ICES Scientific Reports. 2:31. 136 pp. http://doi.org/10.17895/ices.pub.5548

Lindberg, D.E. 2016. Flow preferences of upstream migrating Atlantic Salmon (*Salmo salar*). Doctoral Thesis. Swedish University of Agricultural Sciences, Umeå. 55 pp.

Muška, M., Tušer, M., Frouzová, J., Mrkvička, T., Ricard, T., Seďa, L., Morelli, F., Kubečka, J. 2018. Real-time distribution of pelagic fsh: combining hydroacoustics, GIS and spatial modelling at a fne spatial scale. Nature, Scietific Reports 8:5381 | DOI:10.1038/s41598-018-23762-z

Maes, J., Pas.J., Taillieu, A., Van Damme, P.A., Ollevier, S. 2001. Sampling of fish and crustaceans at the cooling water intake of an estuarine power plant: a comparison with stow net fishery. Archive of Fishery and Marine Research 49: 27-36.

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.

Portt, C.B., Coker, G.A., Ming, D.L., Randall, G.R. 2006. A review of fish sampling methods commonly used in Canadian freshwater habitats. Canadian Technical Report of Fisheries and Aquatic Sciences 2604. 51 pp.

Seaby, R.M.H., Henderson, P.A. 2007. Entertainment, impingement and thermal impacts at Indian Point power plant nuclear power station. Pisces Conservation Ltd. 57 pp.

Teague, N., Clough, S.C. 2011. The survival of lamprey on travelling screens at potable water intakes. WIT Transactions on State of the Art in Science and Engineering, 71: 101-110.

Thiel, R., Sepulveda, A., Oesmann, S. 1996. Occurrence and distribution of twaite shad Alosa fallax Lacépède) in the lower Elbe River, Germany. In (A.Kirhofer and D.Hefti, eds): Conservation of endangered fish in Europe. Birkhäuser Verlag, Basel. pp. 157-170.