# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

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# Executive summary

This report aims to assess the potential impacts of the Sizewell C (SZC) planned New Nuclear Build on the water quality within the local marine environment and to provide information that will support the assessment and setting of a discharge consent by the statutory regulator (Environment Agency) under the Environmental Permitting Regulations.

For marine discharges, the standard approach for determining the potential impacts to water quality from industrial aqueous discharges is to apply the Environment Agency/Defra screening of contaminant contributions from surface drainage sources from the Environment Agency's H1 Environmental Risk Assessment (Defra and Environment Agency Guidance, 2016).

The H1 screening methodology is applied here to identify any proposed chemical discharges that represent a potential risk to the marine environment including those which are then subject to detailed modelling to fully evaluate the acceptability of the discharge.

To assess the significance of specific chemical discharges the H1 methodology uses as its reference existing Environmental Quality Standards (EQSs). Where no EQS is available for a given substance then available toxicity test data are used to generate a Predicted No Effect Concentration (PNEC) as a reference for short term acute exposure and longer-term chronic exposure. Where insufficient or no toxicity data can be sourced then the marine background concentration for a substance from monitoring conducted adjacent to the Sizewell site is used as a point of reference.

The Greater Sizewell Bay (GSB) is considered as the initial reference area for the study site extending to Walberswick in the north with the southerly extent at the apex of the Thorpeness headland in the south. The seaward boundary extends to the eastern flank of the Sizewell-Dunwich Bank and includes the proposed cooling water infrastructure on the east side on the bank. Sizewell C site discharges from the combined drainage outfall (CDO) which would include those from the construction site and potentially those during commissioning and from the Fish Recovery and Returns (FRRs) would also occur within the GSB. Sizewell B intakes and outfalls are also located within the Sizewell-Dunwich Bank and discharge into the receiving waters of the GSB. This assessment predates the in-combination assessment including the temporary desalination plant which is considered in BEEMS TR552.

## Construction discharges prior to the availability of the combined drainage outfall

Prior to establishment of the CDO and sewage treatment plant, wastewater would be tankered off site for appropriate disposal. Groundwater discharge volumes during tunnelling have been incorporated into the assessment based on those identified for Hinkley Point C (HPC).

## Construction discharge assessment

Temporary and variable discharges to marine water will form part of the surface drainage strategy during the construction phase. The main expected contaminants in these discharges are suspended solids, hydrocarbons, Biochemical Oxygen Demand (BOD), some metals from groundwater sources and ammonia. Sediment and hydrocarbons in site drainage water will be managed with appropriate technology and good site management so that these discharges from the construction site are unlikely to affect the water quality status.

The groundwater metals contamination across the construction site varies so monitoring data are used to derive the 95$^{th}$ percentile concentration and these values are used in the initial screening assessment. Groundwater discharge volumes vary and are highest in the first 28 days so screening is conducted for this period. After the first 28 days of the construction schedule various overlapping processes lead to a combination of wastewater sources and different substance concentrations and therefore several time points

(Cases) during the schedule that are deemed worst case for different substance inputs are screened using H1 methodology.

**For first 28 days of the construction period - groundwater dewatering**

Chromium, copper, zinc and iron in the groundwater exceed EQS or equivalent values and fail the initial Test 1 of screening. Taking account of subsequent dilution upon discharge (Test 5) chromium fails and zinc is present at high background concentrations and so, as a precaution, was also considered in more detailed assessment.

Both chromium and zinc were modelled using the CORMIX US EPA supported mixing zone model and the validated Sizewell GETM model. CORMIX is used to predict the rate of chemical plume dilution and plume geometry from the Combined Discharge Outfall (CDO). The GETM model is a 3D hydrodynamic model with an inbuilt passive tracer to represent zinc and chromium. As a worst case, it was assumed that there was no loss of dissolved metals due to sediment absorption or biological uptake. Using these assumptions, concentrations were scaled, as the modelled concentration was simply a function of dilution.
Both zinc and chromium were modelled for the first 28 days of maximum groundwater discharge.

CORMIX shows that for zinc the outfall plume would no longer be detectable above background concentrations within 3m. For chromium the outfall plume would fall below the EQS within 25m. GETM was also used in support of modelling this discharge and slightly under-predicts the initial dilution and shows a 40-fold dilution in the first 25m, meaning the plume extends slightly further. The mean surface area in exceedance of the EQS for Chromium, predicted by GETM, is 5.49ha and for zinc, the total surface area for which the influence of the discharge plume would be detectable above background is 0.11ha

For both chromium and zinc the discharge concentrations predicted above EQS are localised and represent a negligible influence on water quality.

**From 28 days onwards in the construction period**

Once sewage treatment is available on site to treat sanitary waste from the workforce the treated effluent will contribute to a discharge via the CDO of ammoniacal nitrogen and nutrients as well as Biochemical Oxygen Demand (BOD), faecal indicator organisms Escherichia coli and Intestinal Enterococci and suspended solids.

Ammoniacal nitrogen inputs to the construction discharge are from groundwater and treated sewage effluent. The percentage of un-ionised form of ammonia is important to assess as this has a relatively high toxicity and as such has a derived annual average EQS of 21µgl$^{-1}$ $NH_3$-N. Various water quality parameters influence the proportion of ammonia that is un-ionised in seawater and so must be considered in any assessment i.e. higher pH, temperature elevation and reduced salinity all increase the relative proportion of un-ionised ammonia. The percentage of ammonia in the un-ionised form in the construction discharge was calculated for worst case discharge scenarios during the construction period and under mean and most extreme site values. The CORMIX model was used to determine the maximum distance required to achieve un-ionised ammonia concentrations below the EQS which was 6.3m.

Nitrogen in site discharges can contribute to nutrient enrichment in the marine environment therefore the input loadings during construction were assessed together with phosphorus loadings using a phytoplankton box model. The effect of chlorination at Sizewell B (SZB) and the proposed Sizewell C (SZC) on phytoplankton that pass through the power station was simulated with an emphasis on the spring bloom and summertime production using the phytoplankton box model. The combined loadings of nitrogen and phosphorus as described (section 5.4 and 5.5) from the construction and cold commissioning inputs together with relevant inputs from SZB resulting from the use of conditioning chemicals and the discharge of treated sewage were assessed. For much of the year light availability limits phytoplankton growth and the addition of relatively small quantities of nutrients has no effect. In the summer, nitrate is a limiting nutrient (when light is

not limiting) and is consumed rapidly. However, the exchange with the wider environment is much greater than the maximum proposed discharges, during construction and commissioning combined, so that no change in phytoplankton growth beyond natural variability would be observed. A model run over an annual cycle predicts a less than 0.13% difference in annual gross production of carbon and this level of change could not be discriminated above natural background variation and overall the effect observed would be a modest reduction in phytoplankton growth due to entrainment effects.

The background Biochemical Oxygen Demand (BOD) near to the Sizewell B cooling water discharge based on monitoring has a mean value of 2mg $l^{-1}$. Dissolved oxygen levels at the site are 'high' with a mean DO concentration of 6.27mg$l^{-1}$ when adjusted to an equivalent salinity of 35. The waters off Sizewell are well mixed vertically. Draw down of oxygen will only occur if the rate of consumption due to BOD is greater than that which is exchanged across GSB together with the oxygen transfer across the water surface. Indicative values of maximum oxygen demand of 40.6kg were calculated and this amount of oxygen would be transferred across 1.2ha in a day. Therefore, DO is likely to remain at high status. The discharges of BOD during construction are therefore considered to be of negligible significance for dissolved oxygen modification.

Under bathing water regulations discharges containing faecal bacteria must be treated to ensure that the concentration of key indicator organisms will meet a designated standard for coastal and transitional waters for which Good status for Transitional and Coastal waters requires that the colony forming unit (cfu) counts for intestinal enterococci are ≤200 cfu/100ml and for *Escherichia coli* are ≤500 cfu/100ml. The predicted numbers of *Escherichia coli* and intestinal enterococci in sewage effluent was calculated following different stages of sewage treatment and used in a modelling assessment taking account of dilution. CORMIX estimates show that the concentration of Intestinal Enterococci is likely to exceed the bathing water standard (200 cfu/100ml) only within 66m of the discharge for the maximum 30$ls^{-1}$ case for secondary treatment. With UV treatment, even at the higher discharge volume, exceedance would be limited to within less than 1 metre of the discharge.

As the microbiological modelling assessment indicates a relatively small distance over which indicator organism numbers would exceed the good bathing water standard and the nearest designated bathing waters are approximately 10km distant, there is a negligible risk to bathing water quality

**Tunnelling Discharge Assessment**

The offshore cooling water infrastructure consists of two subterranean intake tunnels and one outfall tunnel. Tunnels would be excavated by tunnel boring machines (TBMs) from land. In some TBM soil conditioning applications several different surfactant chemicals may be required. The use and discharge of two surfactant chemicals that are planned for use with the HPC tunnelling operation and that present higher risk quotients in terms of chemical properties are modelled for Sizewell (anti- clogging agent BASF Rheosoil 143 and the soil conditioning additive CLB F5 M). Both chemicals exceed their respective EQS and preliminary dilution assessment so were modelled using GETM. A third chemical bentonite a clay mineral may be employed in a slurry tunnelling method. Available data indicate that bentonite has very low toxicity and in its widespread use offshore in drilling processes it is classed as posing little or no effect. However, a modelling assessment was conducted to determine the 95$^{th}$ percentile and mean plume area to determine extent of any potential influence on water quality based on the limited effects dataset. For the soil conditioning chemical Rheosoil 143 there is no exceedance at the seabed and only very limited areas of exceedance at the surface of mean 1.01ha (5.83 as a 95$^{th}$ percentile). For CLB F5 M there was no exceedance at the seabed and the area at the surface exceeding the EQS was relatively small at 3.14ha for a mean assessment (25ha as a 95$^{th}$ percentile). A tunnelling discharge of bentonite at a concentration of 8.8mg$l^{-1}$ was modelled using GETM and the 95$^{th}$ percentile concentration of 10µg$l^{-1}$ was restricted to sea surface areas of <11ha (mean 1.35ha) with no influence on the bed. Limited data on survival of organisms exposed to bentonite suspensions indicate that the small areas affected, and the low discharge concentrations are likely to have negligible effects on water quality.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

### Commissioning Discharge Assessment

When the cooling water system is commissioned a range of tests will be conducted and conditioning of the entire plant will be undertaken with demineralised water and various chemical additives.

No operational cooling system will be available for the disposal and dilution of commissioning phase effluents during the cold flush testing stage during the phased development of the SZC site. Therefore, the only available discharge route for this wastewater stream will be through the CDO. During commissioning the input loading of phosphorus and nitrogen was assessed in combination with the construction discharge. In addition, the potential toxicity of three chemicals was also assessed during commissioning: hydrazine, ethanolamine (a water treatment chemical) and un-ionised ammonia. All three chemicals exceed their respective EQS in predicted commissioning discharges, but ethanolamine passed the initial dilution assessment (Test 5) and so hydrazine and un-ionised ammonia discharges were modelled using CORMIX and GETM. Using CORMIX the un-ionised ammonia discharge did not exceed its EQS after 25 metres and it was not possible to discriminate any areas of exceedance using GETM.

To assess the spatial extent of the hydrazine plume from a cold commissioning a upper bounding discharge concentration of 15 $\mu gl^{-1}$ was modelled (discharge volume maximum 1500$m^3$ over a 5 hour period). The results were compared with relevant PNEC values (chronic and acute), the mean and 95$^{th}$ percentile of the hydrazine concentration was extracted from a 31-day model run. For hydrazine the chronic PNEC value is 0.4 $ngl^{-1}$ for long term discharges (mean of the concentration values) and the acute PNEC value is 4$ngl^{-1}$ for shorter term discharges (represented by the 95$^{th}$ percentile). In addition to these two precautionary PNEC values considered in this report, the area exceeding 200$ngl^{-1}$ as a 95$^{th}$ percentile, as set by the Canadian Federal Water Quality Guidelines for hydrazine was evaluated.

The 95$^{th}$ percentile results show that the plume at the surface is shorter and thinner than the mean plume. The plume at the seabed shows a similar elongated narrow plume. The area exceeding the derived acute and chronic PNECs is less at the bed than the surface. The exceedance at the surface 12.9 and 30.5ha exceed the acute and chronic PNEC respectively. At the bed the exceedance is 2.92ha for both PNECs.

At the surface the exceedance for the 200$ngl^{-1}$ Canadian standard is 0.34ha, which represents three model grid cells (25 x 25 m) around and including the hydrazine discharge from the CDO.

As the hydrazine chemical plume lies completely inside the Outer Thames Estuary SPA and inside the Suffolk Coastal Waters, the areas of exceedance for the chronic and acute PNECs are the same as for the whole plume and area of intersection with specific sensitivities are further considered. To investigate the potential interaction of the hydrazine discharge concentration with relevant environmental sensitivities the results of both simulations are compared against three criteria: The likelihood that hydrazine could enter the Minsmere Sluice; levels of hydrazine at the seabed over the Coralline Crag and the area of intersection of the acute hydrazine plume with Little Tern foraging areas.

The hydrazine plume is transported northward towards Minsmere during the falling tide, meaning that the sluice water supply that is periodically used to add additional saltwater to the Minsmere salt marshes is unlikely to be exposed to hydrazine. The likelihood of any hydrazine exposure in the sluice water would also be made considerably less likely due to rapid degradation of hydrazine with a half-life of ca.,30 minutes. Eels at different life stages may move into or out of the Minsmere salt marshes via the sluice. The peak hydrazine concentrations predicted at the sluice are around 800,000 times below chronic toxicity data available for fish. Also, the concentration peaks occur just before the sluice opens and are therefore diminishing when any Eels could move via the sluice. The wider area concentration plume of hydrazine and concentration peaks in proximity to the sluice are therefore considered of sufficiently low concentration (and in the latter case duration), to not represent a significant barrier to Eel movement.

In terms of the coralline crag, the peak hydrazine concentration at the seabed over the crag does not exceed the acute PNEC and only exceeds the chronic PNEC for 15 minutes a day. In the Greater Sizewell Bay, the

hydrazine plume never intersects foraging areas for two of the three SPA breeding colonies of birds. Whilst the plume intersection with 15µgl$^{-1}$ release concentration regularly exceeds 1% of the foraging range for the little Tern colony, the duration of the plume is short, with concentrations exceeding the acute PNEC for no longer than 4 hours.

During the latter stages of commissioning that is hot functional testing (HFT) the objective is to test the reactor and associated systems under realistic operating conditions therefore it would be expected that the assessment for operational discharges via the cooling water system would also apply to that during HFT.

Coastal power stations normally require a means of chlorine dosing for biofouling control in the cooling water (CW) systems. Based upon the known risk of biofouling at Sizewell, EDF Energy would need to chlorinate the SZC CW system to maintain control over biofouling of critical plant. Testing of this system will be undertaken during the commissioning phase, but it is assumed that this would only occur once the full cooling water system was in place and operational. This assessment is, therefore, covered under operation.

## Operational Discharge Assessment

Potential discharges to the marine environment have been assessed for the operational phase of the planned SZC. For large cooling water discharges that are discharged to estuaries or coastal waters a specific screening assessment recommended by Defra and Environment Agency, (Clearing the Waters for All, 2016) is applied.

The annual and daily load of each of the chemicals used during operation is used to derive a predicted concentration in the cooling water discharge and this is compared to the relevant quality standard or other acceptable alternative reference for the substance. In the first phase of screening for operational chemicals chlorine and hydrazine fail screening and are assessed using more detailed modelling. As chlorination of seawater produces chlorination byproducts and bromoform was found to be the most dominant of those detected in laboratory simulations using Sizewell seawater it was also modelled in the cooling water discharge.

For the daily and annual discharge assessments of the cooling water inputs during operation several other substances including metals exceed the EQS screening criteria. However, in many cases these are screened out of further assessment as they are considered to have negligible likely effects as the actual discharge concentrations are below method detection limits, the concentrations are several orders of magnitude below their EQS (or PNEC or site background values) and/or the substances have low bioconcentration potential and are readily degradable. For phosphate and dissolved inorganic nitrogen further assessment of the contribution to nutrient status was made. Un-ionised ammonia passed initial screening for 24 hour and annual assessments but as temperature may influence the relative amount of un-ionised ammonia in the operational discharge a modelling assessment considering the influence of temperature elevation was also conducted.

As sewage effluent also contributes to the cooling water discharge during operation the influence of the biochemical oxygen demand and the numbers of intestinal enterococci and *Escherichia coli* likely to be present after treatment relative to the bathing water standard were also assessed.

During the operational phase biofouling of essential parts of the cooling water system results in the discharge of chlorine produced oxidants (or Total Residual Oxidants, TRO) at a predicted concentration of 150µgl$^{-1}$ at the outfall heads. To provide protection to the marine environment chlorine has an EQS of 10µgl$^{-1}$ TRO set as a maximum allowable concentration and expressed as a 95$^{th}$ percentile. The predicted TRO concentration in the cooling water discharge, based on an empirical demand/decay formulation derived from experiments with Sizewell seawater was modelled using the GETM Sizewell model. Two scenarios were considered: chlorination of SZB plus SZC operating in combination, and chlorination of SZB only. For each model run a month-long simulation was analysed and the mean and 95$^{th}$ percentile of the TRO concentrations were extracted. The total area of the plume that exceeds a concentration threshold of 10µgl$^{-1}$

was at a maximum for SZB and SZC operating in combination covering an area of 726ha at the surface and 167ha at the bed. For SZC alone 338ha of the surface and only ca., 2ha at the seabed are affected at a 95$^{th}$ percentile TRO of 10µgl$^{-1}$.

A Fish Recovery and Return system (FRR) is planned to provide a safe return of the more robust organisms directly into the marine environment. The possibility of residual chlorination of this system was initially evaluated but chlorination will be avoided by engineering design and so no further assessment of residual oxidants or chlorination by-products (CBP's) via this discharge route are relevant.

Chlorination of seawater may result in the formation of chlorination by-products. Laboratory studies of chlorinated Sizewell seawater showed that the major CBP that was detected was bromoform, so this was modelled for the cooling water discharge plume. Since bromoform is a product of chlorination, the same scenarios as for TRO were considered: chlorination of SZB plus SZC operating in combination and chlorination of SZB only. For each model run a month-long simulation was analysed and the 95$^{th}$ percentile of the bromoform concentrations was extracted. There is no published EQS for bromoform and so a calculated PNEC of 5µgl$^{-1}$ as a 95$^{th}$ percentile was used.

A maximum of 358ha at the surface and 130ha at the seabed is affected by a bromoform concentration in the discharge plume from SZB and SZC in combination that exceeds the 5µgl$^{-1}$ PNEC. For SZC alone a total area of 52ha at the surface and <1ha at the seabed exceeds the PNEC.

Hydrazine is an oxygen scavenger that is used in power plants to inhibit corrosion in steam generation circuits. Cooling water discharges exceed the acute and chronic quality standard (PNEC) values for both 24 hour and annual loadings. The worst-case daily discharges have been assessed in relation to an annual hydrazine discharge of 24kg per annum into the cooling water flow. Two discharge scenarios were studied for SZC: the first one considering a hydrazine discharge of 69ngl$^{-1}$ in daily pulses of 2.32h, and the second one of 34.5ngl$^{-1}$ of hydrazine discharged in daily pulses of 4.63h. The amount of mass that is released in each of these scenarios is the same. For each model run 28 days were analysed (two tidal cycles) and the mean and 95$^{th}$ percentile concentration for hydrazine were extracted. For hydrazine there is a chronic PNEC value of 0.4 ng l$^{-1}$ for long term discharges (mean of the concentration values) and an acute PNEC value of 4ng l$^{-1}$ for shorter term discharges (represented by the 95$^{th}$ percentile).

The total area exceeding the chronic PNEC at the seabed (0.4ng l$^{-1}$ as an average) is less than 1ha if hydrazine is released in the short or longer pulse scenarios. At the surface the area of exceedance of the chronic PNEC is very similar for short or longer pulses (ca., 157 and 158ha, respectively).

The acute PNEC (4ngl$^{-1}$ as the 95$^{th}$ percentile) is only exceeded at the seabed if hydrazine is released in short pulses and then for only for 0.22ha. At the surface, the area of exceedance for both scenarios is ca.,14ha if hydrazine is released in 2.3h pulses and ca.,17ha if hydrazine is released in 4.6h pulses.

Modelling that takes account of the site background un-ionised ammonia and the calculated additional input of un-ionised ammonia in the discharge was conducted. Average and worst-case combinations with respect to the percentage of un-ionised ammonia were simulated and show that no areas exceed the EQS of 21µg l$^{-1}$ $NH_3$-N as an annual mean. The 24-hour discharge figure for un-ionised ammonia is just over a third of the EQS at 7.92µg l$^{-1}$ but the site background concentration is low (maximum 5.2µgl$^{-1}$).

For annual discharges the screening assessment passed initial assessments but to provide more detailed assessment of the thermal influence on proportion of un-ionised ammonia the mean ammonia discharge at the outfall was added to regional background mean and 95$^{th}$ percentile values to predict the un-ionised ammonia level. All cases (including worst cases) for un-ionised ammonia show that all modelled areas are considerably below the EQS of 21µgl$^{-1}$ as an annual mean.

Assessment of un-ionised ammonia during operation indicates that daily and annual discharges would have negligible effect on water quality.

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During the operational phase, maximum daily loading for nitrogen reaches approximately 2% of the daily exchange for Sizewell Bay, but the average daily value is low at 0.2% of the daily exchange (again indistinguishable from background level).

For operational loadings phosphorus also passed the screening assessment but had one of the higher values in the screening test based on 24-hour loadings and would represent ca., 5% of the load present in the daily water exchange for the Greater Sizewell Bay. A more representative average daily value is very low at 0.03%. There is no equivalent EQS value for phosphorus and it is not normally the limiting nutrient in marine waters, and the discharge concentration is also below background concentrations for offshore waters based on mean winter nutrient concentrations in Atlantic seawater.

The effect of SZB and the proposed SZC during operation on phytoplankton that pass through the power station has been simulated using a phytoplankton box model. The observed cycle of plankton production has been simulated with emphasis on the spring bloom and summertime production. During operation the power stations discharge nutrients in the form of phosphate and nitrates resulting from the use of conditioning chemicals and the discharge of treated sewage. The influence of power station chlorination upon phytoplankton survival is also incorporated into the model.

Based on the DIN and phosphorus loading during operation the phytoplankton growth box model run over an annual cycle showed an insignificant increase in carbon levels (phytoplankton biomass) of 0.11%.

BOD loadings assessed during operation take account of maximum staff numbers on site during an outage and based upon HPC this is estimated as 1900 staff. The waters off Sizewell are well mixed vertically and reduction of oxygen concentration will only occur if the rate of consumption due to BOD is greater than the oxygen transfer across the water surface. The maximum BOD loading is 3.8kg which is equivalent to an oxygen requirement of 1.26kg which would be present in a volume of $183m^3$ which is very small relative to the daily exchange for GSB. An equivalent supply of oxygen to offset this demand would also be transferred across just over $1000m^2$ in a day. Therefore, DO is likely to remain at high status. The discharge of BOD during operation is therefore considered to be of negligible significance for dissolved oxygen modification.

Assessment of the sewage treatment level provided by secondary treatment and assuming dilution in the flow from a single operational EPR the estimated numbers of E.coli and intestinal enterococci in the discharge will meet the bathing water standard for Good status at the point of discharge.

The total biomass of moribund biota that potentially may be discharged from the FRR has been estimated. The additional loading of nutrients phosphorus and nitrogen added to the waters off Sizewell by the decaying biomass are considered low enough so as not to change an assessment of negligible influence on phytoplankton growth when considered in addition to the operational input of these nutrients.

Consideration is also made of the un-ionised ammonia contribution from decaying biomass. Calculated loadings for biomass produced from the FRR during April to September estimate that $NH_3$-N concentration could be at or above the EQS ($NH_3$-N, 21 $\mu gl^{-1}$) when accounting for natural background (and inputs from SZC operation) over an area of 1.2ha around the FRR. At maximum summer temperatures the area affected would increase to 3.8ha and for maximum loadings during March an area of 6.7ha would exceed the EQS.

The influence of biomass decay on the BOD was also assessed and daily re aeration over an area of ca., 14ha would be enough to meet this additional demand when considered with that of the operational discharge and this takes no account of water exchange for the Greater Sizewell Bay. For March when the highest discharge of moribund fish from the FRR is predicted the oxygen demand would increase to 0.6% of that available from daily exchange and would be equivalent to reaeration over 45.2 ha. Therefore, as waters off Sizewell are well mixed vertically facilitating reaeration at the surface, background dissolved oxygen levels are high and the water exchange rate of the GSB is enough to limit the extent and duration of any oxygen reduction, the input loading of BOD from biomass discharged from the FRR is predicted to have a negligible effect on dissolved oxygen concentration which is not significant.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

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## Conclusions

This report assesses the construction, commissioning and operation of two UKEPR units for the proposed SZC development.

A H1 type screening assessment together with more detailed modelling as required of the discharges during the construction, commissioning and operation periods has been completed. The assessment shows that resultant environmental concentrations of discharge chemicals during the construction period are likely to have a relatively localised and negligible influence on marine water quality. The influence of nitrogen and phosphorus inputs during construction combined with those during commissioning is considered to be insignificant for nutrient status of the Greater Sizewell Bay

Modelling of representative maximum scenarios for chemical release during tunnelling operations at Sizewell (based on information derived from HPC) show small areas in which conditioning chemicals may exceed the respective EQS but the impact on water quality is considered negligible.

During the commissioning phase the cold flush discharges from EPR unit 1 and 2 would be made in a low volume flow via the construction drainage system to the marine environment. Inputs of nitrogen and phosphorus have been combined with construction inputs and used as source terms for a phytoplankton box model. Combined nutrient inputs were shown to have negligible effects on phytoplankton growth. A hydrazine discharge during commissioning results in relatively small areas in exceedance of the derived PNEC values and for relevant ecological receptors the potential for effects appears negligible but these are further discussed in the Marine Ecology Chapter 21 of the Environmental Statement. Only a very small area at the surface of 0.34ha exceeded the more recently derived Canadian marine standard for hydrazine.

During operation the larger volume discharges for example of chlorinated cooling water have more potential for larger scale influences on the water quality of the Greater Sizewell Bay. The areas of exceedance of relevant quality standards or equivalents for chlorination TROs, bromoform and hydrazine although unlikely to affect long term water quality objectives for marine waters will need to be considered for individual receptors where areas of exceedance intersect.

Assessment of un-ionised ammonia during operation indicates that daily and annual discharges would have negligible effect on water quality.

Nutrient inputs during operation were assessed using a phytoplankton box model and were shown to have negligible influence on water quality and this included an in-combination assessment with the potential loading from dead biomass discharged from the FRR. Biochemical oxygen demand of the operational discharge was also shown to be negligible again in combination with BOD arising from dead biomass from the FRR.

Microbiological input from sewage discharge during operation is indicated to be compliant with bathing water standards at the point of discharge based on secondary treatment and within system dilution.

## Changes to this report

Edition 2. 3/5/2019

Included more detailed assessment and modelling of construction discharges based on the Hinkley Point C schedule but referencing expected groundwater contributions for the Sizewell C development. The screening assessment for the construction discharges was the same as for Edition 1 but Defra and Environment Agency, (Clearing the Waters for All, 2016) for large cooling water discharges was applied.

Edition 3. 15/8/2019

source data for several chemical inputs were updated by EDF Energy (based on information from HPC) and is incorporated into the assessments (Table 30). This applies to the phosphate and nitrogen loadings during operation based on higher numbers of staff present during an outage. The PNEC values have been updated for the demineralised water treatment (Sequestering) chemicals and this is reflected in the tabled values (Tables 5, 32 and 33). The nutrient inputs during construction and operation are assessed using a phytoplankton box model. Tunnelling chemicals are included, and their potential discharge assessed. Commissioning chemicals are also included and assessed. An additional assessment of the influence of potential biomass from dead organisms discharged by the Fish Recovery and Return is also included.

Edition 4. 29/10/2019

Some minor edits were made to the document and some additional supporting text related to the hydrazine habitats assessments was also added before this version was uploaded to the Aecom site.

Edition 5. 05/03/2020

Hydrazine cold commissioning discharge level has been revised to better reflect expected discharge level for permitting. An additional load assessment for trace metal contamination (cadmium and mercury) of raw materials used for water treatment has been added. Corrections have been made to some of the loading values for operational chemicals as more information has become available none of the changes has had significant implications for predicted impacts. This final version was uploaded to the Aecom site.

Edition 5 (Version 3). 18/10/2021

The supporting ecotoxicity data for PNEC derivation in Appendix B was updated at request of the Environment Agency to include additional justification of the derived hydrazine PNECs to bring this assessment in line with the approach adopted at Hinkley Point.

# 1 Background

## 1.1 Background to the site

NNB Generation Company (Sizewell C) Ltd is planning to build a new nuclear power station at Sizewell, Suffolk (SZC). The new power station will be located on the Suffolk coast, northeast of Ipswich and south of Lowestoft. The approximate National Grid reference for the centre of the proposed development is TM 4730 6410. The new power station will be built near and to the north of the Sizewell B station which will continue to operate after the commissioning of SZC. Sizewell A, which is located to the south of Sizewell B, ceased operation in 2006.

Any development at Sizewell that includes discharge to or operations in the adjacent marine environment must be considered in relation to its potential effect on coastal water quality.

Assessment is also made for Water Framework Directive (WFD) designations associated with the site which is in East Suffolk Zone (ESZ) of the Anglian River Basin District (RBD). Under WFD the Suffolk Waterbody is one of the main points of reference. The Suffolk Waterbody is designated as heavily modified based on coastal and flood protection and it is evaluated (2013 - 2016) as moderate status but must achieve good ecological potential by 2027.
(https://environment.data.gov.uk/catchmentplanning/WaterBody/GB650503520002).

Associated with the Suffolk Waterbody are the Walberswick marshes, the transitional waterbodies the Blyth and Alde and Ore and designated Bathing Waters at Lowestoft north and south of Claremont pier and at Southwold the Denes and Southwold.

The primary habitats designations associated with the site are the Outer Thames Estuary Special Area of Protection (SPA) and the Southern North Sea SAC.

## 1.2 Designation of zone of influence for modelling assessment

The Greater Sizewell Bay (GSB) is considered as the initial reference area for the study site. For the purposes of the Environmental Impact Assessment (EIA), the GSB extends to Walberswick in the north with the southerly extent bound by the geomorphic Coralline Crag formation at the apex of the Thorpeness headland in the south. The seaward boundary extends to the eastern flank of the Sizewell-Dunwich Bank and includes the proposed cooling water infrastructure on the eastern side of the Bank. The landward limit of the marine study area is delineated by Mean High Water Springs (MHWS). However, the GSB is not a closed system and water exchanges with the rest of the southern North Sea. The Zone of Influence (ZoI) for development impacts is, therefore, dependent on hydrodynamic processes.

For the EIA, the potential ZoI is dependent on several factors including; the position and duration of the discharge, the behaviour and persistence and/or degradation rates of the discharge components, bathymetry, and the state of the tidal cycle. Construction and operational discharges are predicted to occur from different point sources and may act cumulatively with discharges from Sizewell B, as is the case for thermal inputs. Therefore, the ZoI provides an initial reference point for considering the spatial and temporal area of impacts. Assessments will account for these factors and determine the absolute area of impact.

Sizewell B intakes and outfalls are located inshore of the Sizewell-Dunwich Bank (Figure 1) and discharge into the receiving waters of the GSB. Sizewell C site discharges from the combined drainage outfall (CDO) (which would include those from the construction site and potentially those during commissioning) and from

the Fish Recovery and Returns (FRRs) would also occur within the GSB and would be transported throughout the inner tidal excursion within the Sizewell-Dunwich Bank (Figure 1 and 2).

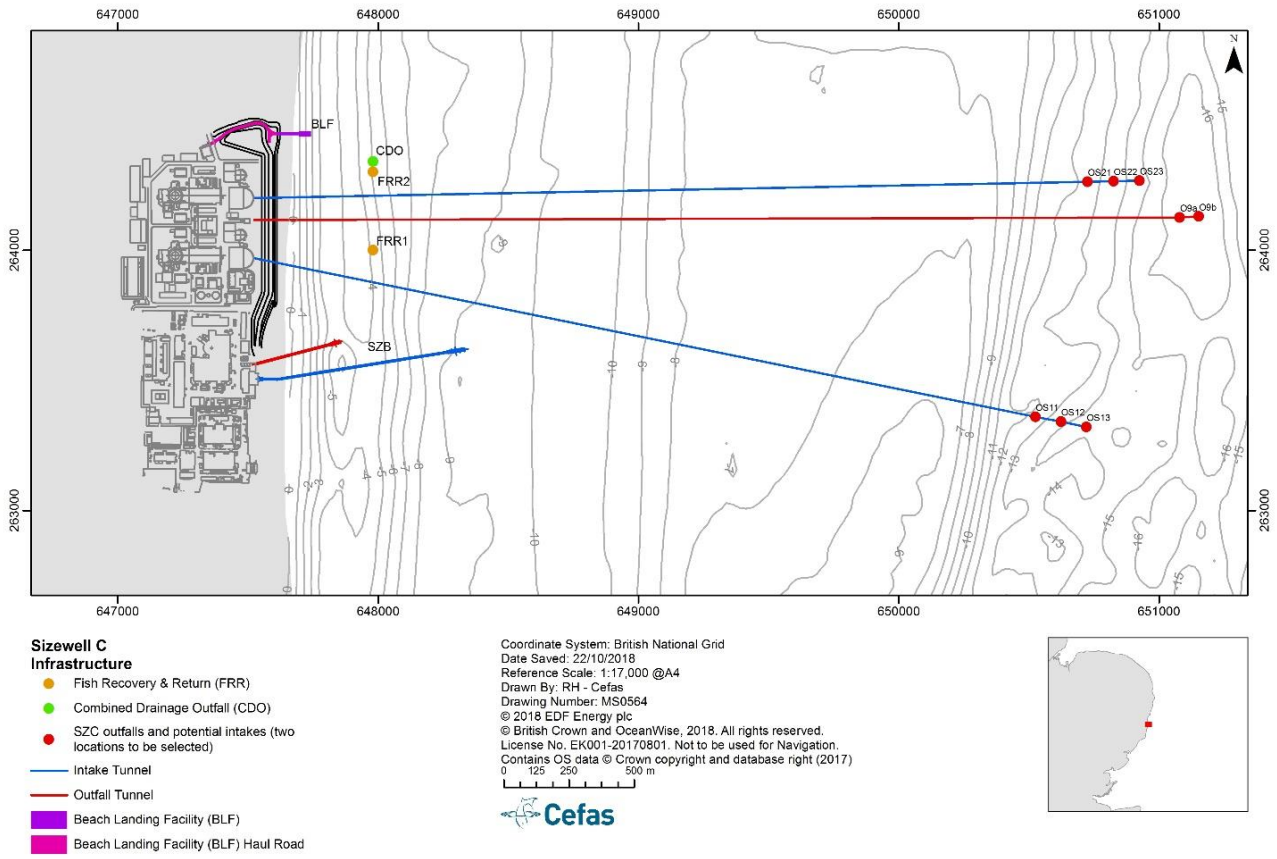


Figure 1: Schematic of development locations in the marine environment overlaid on bathymetry, blue indicates intake tunnels, red indicates outfall.

To determine the effects of entrainment on phytoplankton populations from Sizewell B and C, BEEMS Technical Report TR385 determined the approximate volume of water within the influence of the power station during a tidal cycle. Based upon a current meter (S2) deployed near the proposed Sizewell C intake locations, a progressive vector diagram (PVD) method indicated that the north – south excursion is approximately 15.9 km in each direction, and 1.4km east – west during spring tides. The trajectory of the tide flows both north and south, thus the tidal volume represents a body of water 31.8 km long and approximately 2.8km wide. The average depth was calculated at 12.5m giving a total volume of 1209.7 x $10^6$ $m^3$ (Table 1).

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Table 1 The volume of water associated with the Greater Sizewell Bay and the tidal excursion originally reported in BEEMS Technical Report TR385.

| Body of water defined in TR385 | Surface area (ha) | Average depth (m) | Volume (x$10^6$ $m^3$) |
|---|---|---|---|
| **GSB** | 4120 | 8.8 | 363.8 |
| **GSB + tidal excursion beyond the Sizewell-Dunwich Bank** | 9670 | 12.5 | 1209.7 |

The volumetric exchange rate has not been measured at Sizewell. Typical exchange rates in partially mixed tidal estuaries are 5% volume exchange on each tide (Dyer, 1979), thus 0.1 per day. In the Southern North Sea, an open sea area, the exchange is expected to be greater. Calibration of a Sizewell phytoplankton model found a volumetric exchange rate of 10% corresponded well to observation data and has subsequently been adopted (BEEMS Technical Report TR385). The exchange rate will vary in accordance to a spring-neap cycle. Furthermore, velocity observations show a net westward exchange as well as a net southward exchange, meaning a 10% exchange is estimated to be the minimum water exchange rate.

The tidal excursion is dependent on the stage within the spring-neap cycle but provides an estimate for the zone of influence. The method applied to determine the tidal excursion has a bearing on the calculation of the estimated area and volume. The following section details several methods applied to estimate the body of water potentially influenced by the power station.

For comparison, a harmonic analysis was conducted on the same S2 current meter (BEEMS Technical Report TR233) and provided similar results to the PVD method. The tidal ellipse indicates that the north – south excursion is approximately 17.2km, and 1.8km east – west during spring tides. The trajectory of the tide flows both north and south, thus the tidal volume represents a body of water 34.4 km long and approximately 3.6km wide.

Further analysis was undertaken to support the estimate of the tidal water volumes reported in BEEMS Technical Report TR385. To determine the Outer Tidal Excursion, a particle tracking study was considered but the trajectories exceeded the hydrodynamic model domain. Instead, without running a new model set-up, two alternative methods have been considered: a PVD and a harmonic analysis. The PVD method estimates the potential transport based upon measured velocity time-series (at a fixed location). The distance travelled between each time step of the record, is determined from using the U and V velocity components, and its trajectory plotted from the original starting point (i.e. the outfalls). The tidal excursion is then determined from an area encompassing the total trajectory path. For the harmonic analysis method, an idealised tidal curve was reconstructed, using the M2, S2 and N2 tidal constituents, to determine the major and minor axis of the tidal ellipse. This provides a maximum theoretical tidal excursion, excluding any meteorological forcing. The area and volume based upon the average depth, of the associated ZoIs are shown in Table 2.

To determine the volume of water that may be influenced by the CDO and FRR discharges, within the Sizewell-Dunwich Bank particle tracking associated with the FRR was completed (BEEMS Technical Report TR333). Particles were released from FRR Position 5 over a spring-flood tide and a neap-flood tide for May 2009. This is representative of the mean conditions for the area of Sizewell. The tidal excursion within the Sizewell-Dunwich Bank was then determined by defining an area encompassing every particle position at each time step of both runs combined. This indicates that the total tidal excursion is approximately 20.8km North-South and approximately 3.5km east-west.

Table 2 Approximate surface area and volume of the Zones of Influence based on the areas delineated in

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Figure 2.

| | Surface area (ha) | Average depth (m) | Volume (x$10^6$ $m^3$) |
|---|---|---|---|
| **GSB** | 4577.5 | 8.73 | 399.7 |
| **Inner Tidal Excursion** | 4323.2 | 8.49 | 367.0 |
| **Outer Tidal Excursion** | | | |
| PVD method | 7081.4 | 13.91 | 985.0 |
| Harmonics method | 10129.1 | 13.84 | 1401.9 |
| ***GSB + tidal excursion** | 9906.7 | 12.14 | 1202.9 |

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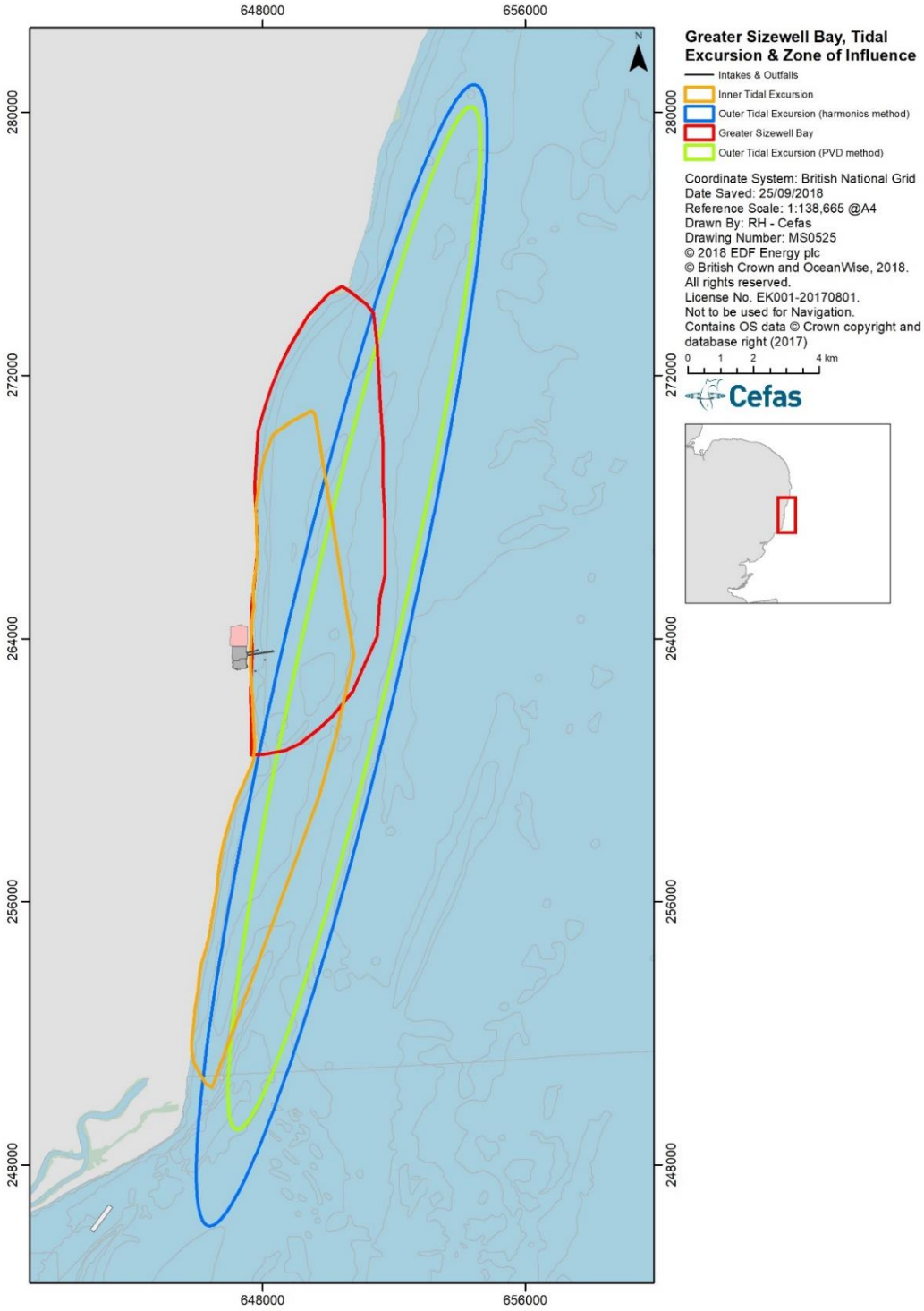


Figure 2 The area of the tidal excursion from the Sizewell C CDO/FRR and outfall during spring tides, the outer tidal ellipse and the Greater Sizewell Bay body of water.

## 1.3 Objectives

At each phase of the development (construction, commissioning, and operation) the potential and extent of any effects on water quality will be assessed. Assessment will take account of temporary and permanent discharges from the site and from the two proposed UKEPR units.

Sizewell C is a Nationally Significant Infrastructure Project (NSIP), therefore EDF Energy requires a Development Consent Order (DCO) to construct and operate the power station, and any associated development, under the Planning Act 2008. The application for development consent will comprise details of all development proposals and will be accompanied by an ES conforming to the Infrastructure Planning (Environmental Impact Assessment) Regulations 2009 (SI 2263) (as amended) (the EIA Regulations) and other relevant documents. In support of these requirements the main objective of this report is to assess the potential impacts on the water quality within the local marine environment and to provide information that will support the assessment and application for a Water Discharge Activity (WDA) environmental permit to the Environment Agency.

The WDA permit is determined based on the status of the receiving waterbody and upon existing pressures on water quality from other consented discharges. Once a permit is issued the operator must apply control measures to ensure compliance.

In December 2016, the Environment Agency released new guidance on how to assess the impact of any activity in transitional and coastal waters, “Clearing the Waters for All”. The process consists of three stages (screening, scoping and impact assessment). For the planned Sizewell C this report considers each of the three assessment stages for the discharges to the marine environment during construction, commissioning and operation.

In the screening stage those discharges and substances that are evaluated as having negligible likely effects are excluded from further scoping.

To assess the significance of specific chemical discharges the screening methodology applies existing Environmental Quality Standards (EQSs). Where no EQS is available approaches are described for derivation of an alternative reference value.

The focus of this report is the potential impact of activities upon water and sediment quality. Where relevant, more detailed chemical modelling of discharges is used to determine total areas of exceedance for those substances not screened out by preliminary assessment. This information is used to support the water quality assessment in BEEMS Technical Report TR306 (Water and Sediment Quality Synthesis). The same information but considering areas of overlap with the Water Framework waterbodies and Habitats are considered in BEEMS Technical Report TR483 or for individual biology receptors will be considered in the Marine Ecology section of the Environmental Statement.

# 2 Assessing potential concerns for marine water quality

## 2.1 Background

The water quality standards adopted for this assessment are those relevant to all expected discharges from the SZC site during all phases of the development. These mainly relate to Environmental Quality Standards (EQSs).

A detailed list of the currently available EQS values that have been assigned to water quality for both the freshwater and marine environments are described for other surface waters (Transitional and coastal waters, TraC Waters) for priority hazardous substances and other pollutants under Directive 2013/39/EU (implemented by the Water Framework Directive (Standards and Classification) Directions (England and Wales, 2015) which increased the list of chemicals to 45 and for substances classed as specific pollutants for which 29 substances are listed (Defra, 2014). Chemical status is recorded as 'good' or 'fail'. The chemical status classification for the water body is determined by the worst scoring chemical.

The water quality monitoring campaigns (defined periods of monitoring) for marine water quality are described in detail within BEEMS Technical Report TR189 and in BEEMS Technical Report TR314 (an update to TR189 that includes data on selected determinands from monitoring conducted in 2014/15). A Sizewell Water quality literature review TR131 provides historic information on background water quality for the Suffolk coastal waterbody. This document also provides details of all the relevant Screening EQS values for saltwater and the legislation and guidance documents from which they are derived.

## 2.2 Contaminants of concern for the combined drainage outfall (CDO)

Various chemical and physical standards for the protection of marine water may be affected by the discharges from SZC.

During construction and commissioning a CDO will be in place to collect and allow discharge of various wastewater streams to the marine environment. The discharge sources for contaminants of concern and flow rates used for the modelling at the CDO are:

1. Groundwater from the dewatering system which contains metals, ammoniacal nitrogen, dissolved inorganic nitrogen (DIN) and phosphate with a maximum flow rate of 124ls$^{-1}$ in the first 28 days and thereafter at 15ls$^{-1}$.
2. Treated sewage discharge which contains, ammoniacal nitrogen, dissolved inorganic nitrogen (DIN) and phosphate from permanent treatment units with a total flow rate of 13.3ls$^{-1}$.
3. Effluent from tunnel excavations potentially containing residual amounts of Tunnel Boring Machine (TBM) soil conditioning chemicals and variable quantities of groundwater containing metals, ammoniacal nitrogen and DIN.

Dewatering is required during the construction of SZC. In this process, groundwater is pumped from a network of deep boreholes. Atkins Ltd (Atkins) was commissioned by EDF Energy to measure and assess groundwater chemistry underlying the site (Atkins, 2016). This groundwater chemistry dataset (referred to as the 2014-2016 dataset see Appendix A) is used to derive the 95$^{th}$ percentile concentration for each of the substances of concern. These 95$^{th}$ percentiles are used to assess the potential for effects of discharged groundwater on the marine environment. The use of 95$^{th}$ percentiles provides a conservative assessment and is more robust that using maximum values for which there is lower confidence.

Sewage treatment will be available on site to treat sanitary waste from the workforce and treated effluent will contribute to ammoniacal nitrogen and nutrients as well as Biochemical Oxygen Demand (BOD), faecal coliforms, Escherichia coli, Intestinal Enterococci and suspended solids.

The main bulk of the tunnelling material (potentially with associated soil conditioning chemicals) is returned with the spoil to the muck bay. The tunnelling spoil will be re-used on-site in accordance with the site materials management plan. Sources of water from the tunnelling operations will include groundwater entrained within the tunnelling spoil, groundwater from the shaft dewatering, very minor seepages of groundwater into the tunnel, water used for cleaning equipment and dust suppression, surface run-off from the muck bay and groundwater seepage into the launch pits and Spray Concrete Lined (SCL) tunnels.

Construction work at the site is also likely to contribute concrete wash water to site discharges and these will be assessed accordingly.

During the early part of the commissioning phase, conditioning chemicals will also be discharged through the CDO. The chemicals present may include hydrazine (antioxidant), metals, and various organic and inorganic chemicals.

One of the issues when considering all three discharge streams (sewage, groundwater and tunnelling wastewater) is to consider the timescale of the likely discharges and potential maximum discharges and loads. This report considers when loads of a contaminant are at maximum levels or are likely to persist as discharges for a reasonable period. To determine realistic worst-case contributions that need to be assessed for specific contaminants from different discharge sources combined in the CDO, several ‘Cases’ are described for different phases during the construction period when input sources overlap and combine.

## 2.3 Evaluation of contaminants of concern during operation

During the latter phase of commissioning and during operation various process effluents e.g. treated sanitary wastes from welfare facilities for operational staff, waste chemicals from boiler cooling circuits would be combined with the cooling water and discharged from the single offshore discharge point. The discharge would include chlorine produced oxidants from chlorination of the cooling water, residual hydrazine, metals, and various organic and inorganic chemicals.

## 2.4 Key contaminants of concern during operation

Various chemical and physical standards derived for the protection of marine water quality may be affected by the physical and chemical nature of site discharges. These standards may be for absolute concentrations or temperatures (where discharges are added to background concentrations), or uplifts above ambient conditions. There is a temporal component to EQS exceedance which is typically maximum (100$^{th}$ percentile), 98$^{th}$ percentile, 95$^{th}$ percentile, or sometimes the mean.

EQS thresholds are based on toxicity data for the most sensitive species with a safety factor applied depending on the confidence in the data. Typically, data is derived from representative examples of algae, crustacea and fish and safety factors range for 10-fold for good data to 1,000 or more for data poor chemicals (in some cases an EQS may not be formally established and BEEMS has used an ‘applied EQS’ based on available data, see section 2.5 and Appendix B).

The main standards referred to in this report are shown in Table 3. Under the Water Framework Directive assessment of nutrient status of a waterbody (Water Framework Directive (Standards and Classification)

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Directions (England and Wales) 2015) there are four Waterbody 'Types' defined by annual mean concentration of suspended particulate matter (see Appendix C, Table 6).

The dissolved inorganic nitrogen (DIN) value referenced in Table 3 is based on the 99th percentile of the winter DIN values for 'Intermediate turbidity' waterbodies for classification of waterbodies as Good, Moderate, Poor, or Bad. The threshold value shown in Table 3 is derived based on the mean suspended particulate matter concentration at Sizewell. This would define Sizewell as of intermediate turbidity with associated threshold 99 percentile winter DIN values for coastal waters of 980µgl$^{-1}$ and 1470µgl$^{-1}$ for Good and Moderate respectively (Water Framework Directive Standards and Classification Directions 2015).

Table 3 Marine water quality standards applied in assessment of planned discharges during the SZC development – these represent Environmental Quality Standards (EQS) for other surface waters (TraC Waters) for priority hazardous substances and other pollutants (Directive 2013/39/EU); (Defra, 2014); microbiological standards from bathing water regulations (2013. No. 1675).

| **Determinands** | **WFD EQS Annual average values (µg l$^{-1}$)** | **WFD EQS Maximum Allowable Concentration (MAC) values (as 95 percentile) (µg l$^{-1}$)** |
|---|---|---|
| Cadmium and its compounds (dissolved) | 0.2 | 1.5 |
| Lead and its compounds (dissolved) | 1.3 | 14 |
| Mercury and its compounds (dissolved) | - | 0.07 |
| Nickel and its compounds (dissolved) | 8.6 | 34 |
| Chromium VI (dissolved) | 0.6 | 32 |
| Arsenic (dissolved) | 25 | Not applicable |
| Copper (dissolved) | 3.76 (2.677 x ((DOC/2) - 0.5)) µg/l dissolved, where dissolved organic carbon (DOC) > 1 mg l$^{-1}$ | Not applicable |
| Iron (dissolved) | 1000 | Not applicable |
| Zinc (dissolved) | 6.8 (plus ambient background 1.1 in salt water) | Not applicable |
| Boron (Total) | 7000 (pre Water Framework recommended standard)[1] | - |
| Chlorine | - | 10 |
| Un-ionised ammonia ($NH_3$)[2] | 21 | - |
| Winter dissolved inorganic nitrogen | | 980[3] |
| *Escherichia coli* | | ≤500 colony forming units/100ml[4] |
| Intestinal enterococci | | ≤200 colony forming units/100ml[4] |

[1]Mance et al, 1988; [2] Total ammonia values of 1100 (annual average) and 8000µg/l NH4-$^{N}$ are also recommended for habitats consideration (WQTAG086, 2005) [3]EQS for nitrogen is based on WFD 99 percentile standard for Good status for an intermediate turbidity waterbody. It should be noted that a more specific methodology for deriving 99$^{th}$ percentile values based on a relationship between SPM and DIN is recommended in draft Environment Agency guidance and for an annual average SPM of 55.2mgl-1 would

give a slightly lower value of 952µgl-1 as a 99th percentile but the screening here would only slightly change.; [4]This assessment is from bathing water regulations (2013. No. 1675) for coastal and transitional waters and represents Good standard

## 2.5 Applied EQS values

In the absence of EQS values for some toxic chemicals, the use of Predicted No Effect Concentration (PNEC) values is proposed. PNEC values have only been used where there is no existing EQS standard and where a relevant saltwater PNEC standard has been determined by independent authorities (as recommended in European Chemicals Bureau Technical Guidance, 2003 (TGD) and CIS, 2011).

PNEC values are determined such that they ensure the protection of all organisms in the receptor environment and they represent the predicted concentration of a given chemical where there should be no effects on the aquatic biota. The determination of PNEC values follows the Technical Guidance Document (European Chemicals Bureau Technical Guidance, 2003) and CIS, 2011 on risk assessment of new and existing chemicals following a review of the ecotoxicological literature. Under the guidelines from the Water Framework Directive (WFD) and the EQS Directives, WFD EQS values are, for the most part, also based on PNEC values.

A review of PNEC values for several discharge chemicals has already been undertaken by EDF Energy based on PNEC values determined by independent research organisations (e.g. INERIS). This document proposes acute and chronic PNEC values for hydrazine, morpholine and ethanolamine (details Appendix B). These derived PNEC values have been adopted in the present assessment.

Because of the inherent uncertainty in the derivation of marine PNEC values, they are not directly comparable with the H1 methodology which is based on comparison with annual average and maximum allowable concentration EQSs. Therefore, to assess the environmental significance of chemicals where a PNEC value has been adopted, the approach presented in the Technical Guidance Document of comparing the ratio between the PNEC value and Predicted Environmental Concentration (PEC) is used. If the PNEC:PEC ratio is less than or equal to 1, the discharges will have no environmental impact (the PNEC is a concentration of no effect for all organisms). A PNEC:PEC ratio > 1 indicates that a potential impact cannot be excluded.

Depending on the release pattern of a chemical and its environmental fate, chemical exposure may occur over long periods - or even continuously - in biota, in sediments, and even in the water column. In the water column, exposure may also occur intermittently for short periods e.g. coinciding with storm events or short periods of chemical use.

In order to cover both long- and short-term effects resulting from exposure, two water column EQSs will normally be required:

i. a long-term standard, expressed as an annual average (AA) concentration and normally based on chronic toxicity data
ii. a short-term standard, referred to as a maximum acceptable concentration (MAC) which is based on acute toxicity data.

AA data are usually based on taking the lowest chronic ecotoxicological value. The values derived for chronic PNEC are usually based on a No Observed Effect Concentration (NOEC) and are the chemical concentrations for which it is predicted that there will be no effect on aquatic biota or where this is not available an effect concentration for 50% of the test individuals (EC50). A safety factor is then applied by dividing with an assessment factor (1 to 10,000) depending on the quality, quantity, diversity, and specificity of the ecotoxicological data available following TGD guidance and CIS, 2011. The assessment factor therefore reflects the confidence that the lowest ecotoxicological datum represents the greatest number of taxa possible, the environment in which they live (freshwater or marine) and the type of discharge in terms of frequency (chronic, duration typically months to years or acute shorter term, hours to days). For exposures

resulting from shorter term (typically over 24 hours) exposures MAC values are derived from the lowest acute toxicity data and use 50% effect concentrations ($EC_{50}$) derived from studies of 24 - 96 hours duration.

As freshwater organisms are generally easier to obtain and test this has led to fewer marine toxicity datasets being available. This often leads to the development of marine PNEC values based on extrapolation from freshwater PNEC values or high assessment factors applied to marine ecotoxicological data (uncertainty regarding the sensitivity of other taxa). Because of the greater biological diversity in marine environments compared to freshwater environments and because of the general lack of marine ecotoxicological studies, it is often necessary to apply conservative assessment factors to the derived freshwater PNEC values to obtain a marine PNEC (unless there is justification that similar toxicity is seen in both environments).

For example, the derivation of PNEC values for hydrazine is based on the lowest valid ecotoxicological value: an $EC_{50}$ (50% effect on test species) value of 0.4µgl$^{-1}$ for the marine alga *Dunaliella tertiolecta*. To derive the chronic PNEC value an assessment factor of 1000 was applied because of the lack of studies available for other marine taxa. An assessment factor of 100 was applied to this $EC_{50}$ value to obtain the acute PNEC value.

For the assessment of the proposed SZC discharges of hydrazine, morpholine and ethanolamine, the chronic PNEC value has been applied to annual chemical loadings and the acute PNEC values to 24-hour discharges and these are shown in Table 4.

Table 4 Proposed PNEC values for chemical parameters based on EDF R&D review of ecotoxicity studies (EDF, 2008) (more detail is provided in Appendix B).

| Chemical Parameter | Acute Marine PNEC | Chronic Marine PNEC | Lowest ecotoxicological value used to derive PNEC values |
|---|---|---|---|
| Hydrazine | 0.004µgl$^{-1}$ | 0.0004µgl$^{-1}$ | $EC_{50}$ = 0.4µgl$^{-1}$on *Dunaliella tertiolecta* (marine unicellular alga) – chronic and acute PNEC |
| Ethanolamine | 160µgl$^{-1}$ | 160µgl$^{-1}$ | NOEC = 1.6mgl$^{-1}$on *Microcystis aeruginosa* (freshwater cyanobacteria) – chronic and acute PNEC |
| Morpholine | 28µgl$^{-1}$ | 17µgl$^{-1}$ | NOEC = 1.7mgl$^{-1}$on *Microcystis aeruginosa* (freshwater cyanobacteria) – chronic PNEC<br><br>$EC_{50}$ (96h) = 28mgl$^{-1}$on *Selenastrum capricornutum* (freshwater alga) – acute PNEC |

## 2.6 Application of data from ecotoxicity studies

For potentially toxic chemicals where there are no EQS or PNEC values, then data from ecotoxicity studies have been used for assessing the environmental significance of discharges from SZC. This approach has been adopted for the following discharges of by-products from sequestering agents used with the demineralisation plant. Sequestering agents may be used for functions such as the prevention of scale

formation by reacting with calcium salts present in water to prevent them reacting with other surfaces (see below).

### 2.6.1 Sequestering Agents

For the chemicals associated with the sequestering agents used in the demineralisation water plant (see Table 5), there are currently no saltwater EQS or EDF validated PNEC values available. Therefore, EDF validated ecotoxicity data (sourced from peer-reviewed publications and non-peer review literature such as industry reports) have been adopted for use in the H1 assessment. A precautionary approach has been adopted to determine the potential environmental significance of discharges of sequestering agent by-products, where the lowest concentration available in the ecotoxicity data is compared to the predicted concentration in the effluent following mixing of various process inputs. This approach has been used as the ecotoxicity data for most of the chemicals is only available for freshwater organisms. For annual discharges comparison has been made to chronic ecotoxicity concentrations and acute values used for 24-hour discharges.
A summary of the threshold values adopted for the H1 assessment is presented in Table 5. The footnotes indicate the reference sources for the dataset and application factors applied to these based on the CIS guidance, 2003. None of these substances have been analysed for in Sizewell seawater so there are no background concentration data to reference for the site.

Table 5 Derived PNEC values, and source and type of endpoint with application factor by which they were derived as used in the assessment of sequestering agents and by-products.

| Chemical | Chronic Concentration ($\mu gl^{-1}$) | Ecotoxicity Test Type | Test Species | Acute Concentration ($\mu gl^{-1}$) | Ecotoxicity Test Type | Test Species |
|---|---|---|---|---|---|---|
| ATMP | 74[a] | NOEC[1,2] (96h) | Freshwater alga (*Selenastrum capricornutum*) | 74[1] | NOEC[2,3] (96h) | Freshwater alga (*Selenastrum capricornutum*) |
| HEDP | 13[a] | NOEC(96h)[2, 2] | Freshwater alga (*Selenastrum capricornutum*) | 13[1] | $EC_{50}$ (96 hr)[3,4] | Freshwater alga (*Selenastrum capricornutum*) |
| Acetic Acid | 62.8[b] | NOEC (21 day)[2,5] | (*Daphnia magna*) | 301[d] | $LC_{50}$ (48 hr)[5,6] | Freshwater crustacean |
| Phosphoric Acid | 20[c] | $LC_{50}$ (72hr)[7] algae | Freshwater algae | 200[d] | $LC_{50}$ (72 hr)[7] algae | Freshwater algae |
| Sodium Polyacrylate | 11.2[8b] | NOEC (21 days) | Freshwater crustacean (*Daphnia magna*) | 180[8d] | $LC_{50}$ (96 hr)· | Freshwater algae (*Scenedesmus subspicatus*) |
| Acrylic Acid | 0.34[9b] | NOEC (72h) | Freshwater alga (*Selenastrum capricornutum*) | 1.7[9a] | $EC_{50}$ (96hr) | Freshwater alga (*Selenastrum capricornutum*) |

Table Notes:[a] application factor 100; [b] application factor 500; [c] application factor 10000; [d] application factor 1000; [1]this value is set at same level as the chronic value as a lower acute value would result otherwise [2]NOEC = No Observable Effect Concentration.[3]Jaworska et al (2002);[4]$EC_{50}$ = Effect concentration for 50% of the study individuals. [5]ECHA dossier Acetic acid; [6]$LC_{50}$ = Lethal concentration for 50% of the study individuals; [7]ECHA dossier phosphoric acid;[8]SDA, 1996 [9]Sverdup et al.,2001

## 2.7 Application of background mean concentrations

Several chemicals present within the expected marine discharges during the commissioning and operational phases of the site have no assigned saltwater EQS values that are at present accepted and are naturally present in marine waters (e.g. manganese, aluminium, lithium hydroxide, sulphate, sodium, chloride, suspended solids, phosphorus, chemical oxygen demand (COD) and biochemical oxygen demand (BOD)).

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Review of other screening assessments concerning marine discharges indicates that it is appropriate to use an ambient mean baseline concentration as a substitute benchmark value against which the significance of a chemical can be assessed. The mean baseline concentrations adopted as substitute benchmark water quality standards are based on the overall average values determined from the water quality monitoring undertaken during 2010 (BEEMS TR189) and in some cases from the supplementary studies during 2014/15 (BEEMS TR314).

# 3 Marine water quality baseline

The status of marine waters adjacent to Sizewell has been measured to determine whether the chemical composition of any planned discharges from the SZC development site will result in deterioration of marine water quality. The Suffolk waterbody which is adjacent to Sizewell is a heavily modified waterbody because of the presence of coastal protection and flood protection. As of 2015 the waterbody is considered of Moderate Ecological Potential because the dissolved inorganic nitrogen concentration; a supporting element is judged to be Moderate (Environment Agency, Catchment data explorer, 2019).

## 3.1 Data sources

Supporting data used in this assessment were derived from four main sources. Historic data relating to marine water quality were sourced from the scientific literature. Most of the data from this source focus on the quality of estuaries discharging into the Suffolk Waterbody. Water quality data were also obtained from the Environment Agency. Thirdly data on coastal sea surface temperatures were collated into time-series over 48 years based on data provided on behalf of Cefas, by councils, companies and other organisations. Finally, new data were obtained from a BEEMS monitoring programme focussed on current and planned cooling water discharge locations off Sizewell. The temperature and historic monitoring data are reported in detail in BEEMS Technical Report TR131. As the data extracted from the scientific literature do not specifically focus on the Suffolk waterbody or relevant ZoI for the site, most reliance in the following sections is placed on the other data sources.

## 3.2 Historic data

This section describes Environment Agency monitoring surveys for compliance and therefore the sites chosen, type of analysis and detection limits are set in this context. The data for dissolved metals covers the period 1989 to 2006 and include data for sites from off Felixstowe to just off the river Yare (Figure 3). However only four of the nine locations sampled in the original survey are within the Suffolk waterbody or ZoI and these are referred to below. Nutrients and inorganics data include samples collected between 1983 and the early part of 2014. The EQS are derived from Directive 2013/39/EU about priority substances, cadmium, lead, nickel, and mercury.

For the concentrations of metals in seawater from various sites within the Suffolk Waterbody only zinc exceeded the EQS off the Alde/Ore although high values were also measured in samples off Dunwich and off the mouth of the Orwell. There is no clear trend in concentrations measured and values below detection are interspersed with high values. For other determinands, for sample points outside the waterbody, cadmium exceeds its EQS value Off the River Deben, chromium VI at the Mouth of the Orwell, off Aldeburgh, off Dunwich and off the Yare. In the case of chromium VI there were only one or two measured values between 1991 -1994 and these led to EQS exceedance with subsequent values below detection up to 1999 (the last monitoring date). The lower revised EQS for cadmium, chromium VI and zinc relative to the high detection limits at the time of the original analysis means that it is not possible to determine the number of sites that might have breached the standard. Copper is also close to its EQS at the mouth of the Orwell but dissolved organic carbon values were not available and need to be taken account of in assessing compliance with the EQS.

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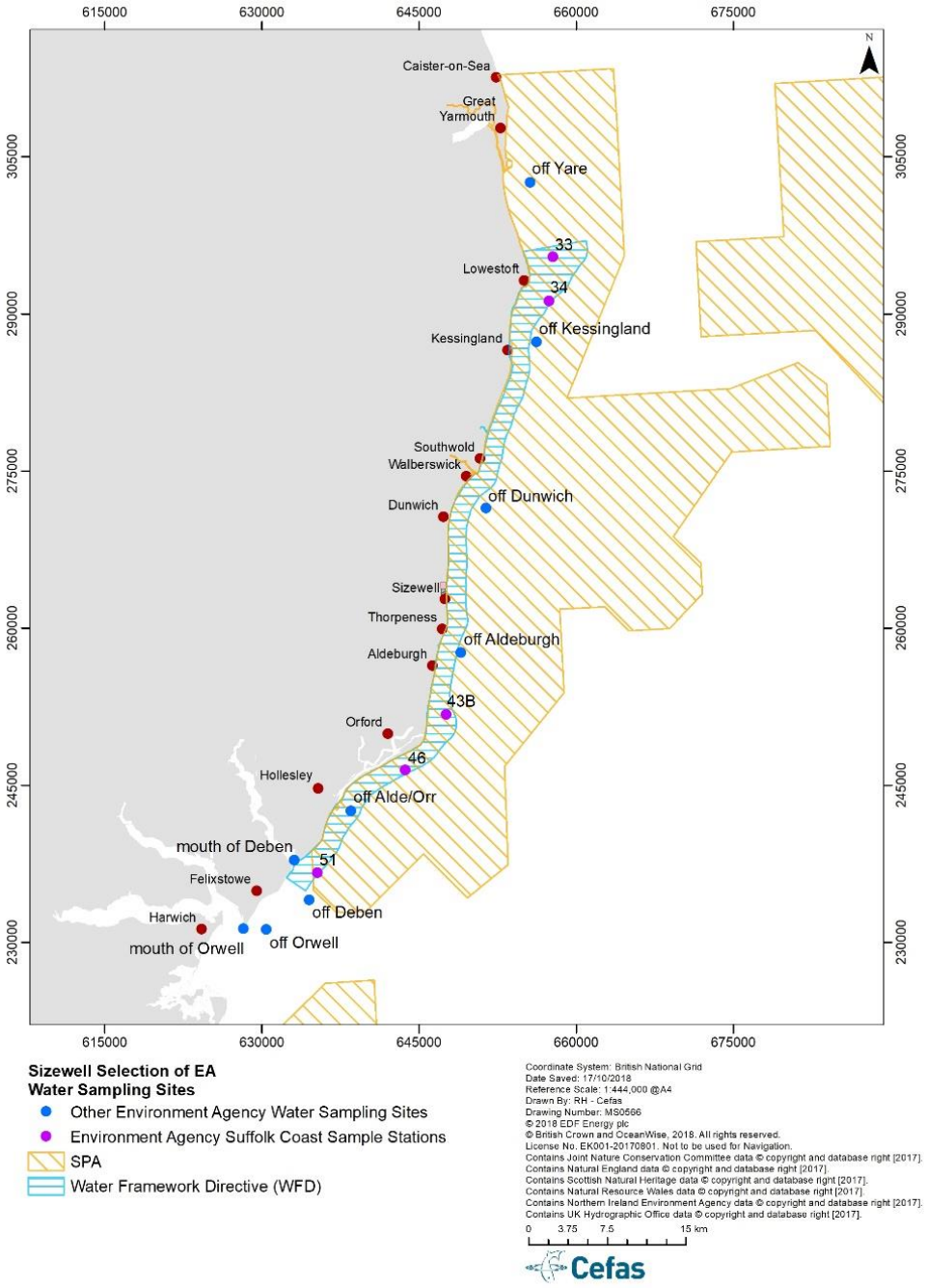


Figure 3 The Environment Agency (EA) sampling stations for which water quality data were available are shown in relation to Sizewell Power Station and major towns on the Suffolk coast. The numbered sample locations are the Suffolk Waterbody sampling points and the Suffolk Waterbody is delimited by the green hatched area near to shore. The brown hatched area extending further offshore shows the upper part of the Outer Thames Estuary SPA. Additional EA sampling points are shown as blue circles.

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## 3.3 BEEMS monitoring data

A marine water quality monitoring programme was established off the Suffolk coast near Sizewell B power station to assess the concentrations of many elements and compounds and their variation over a range of time scales. The programme ran from February 2010 to February 2011, and the programme's results are presented in BEEMS Technical Report TR189. A spatial survey was conducted at twelve sampling stations (Figure 4). The sampling was centred upon the existing cooling water outfall for Sizewell B, at station 5 (Figure 4). A tidal-cycle survey was carried out during which water samples were acquired at hourly intervals at station 5 over an ebb/flood tidal cycle during spring tide conditions. A seasonal survey was also carried out by acquiring water samples near slack water at stations 5 and 11 on 21 occasions throughout the programme.

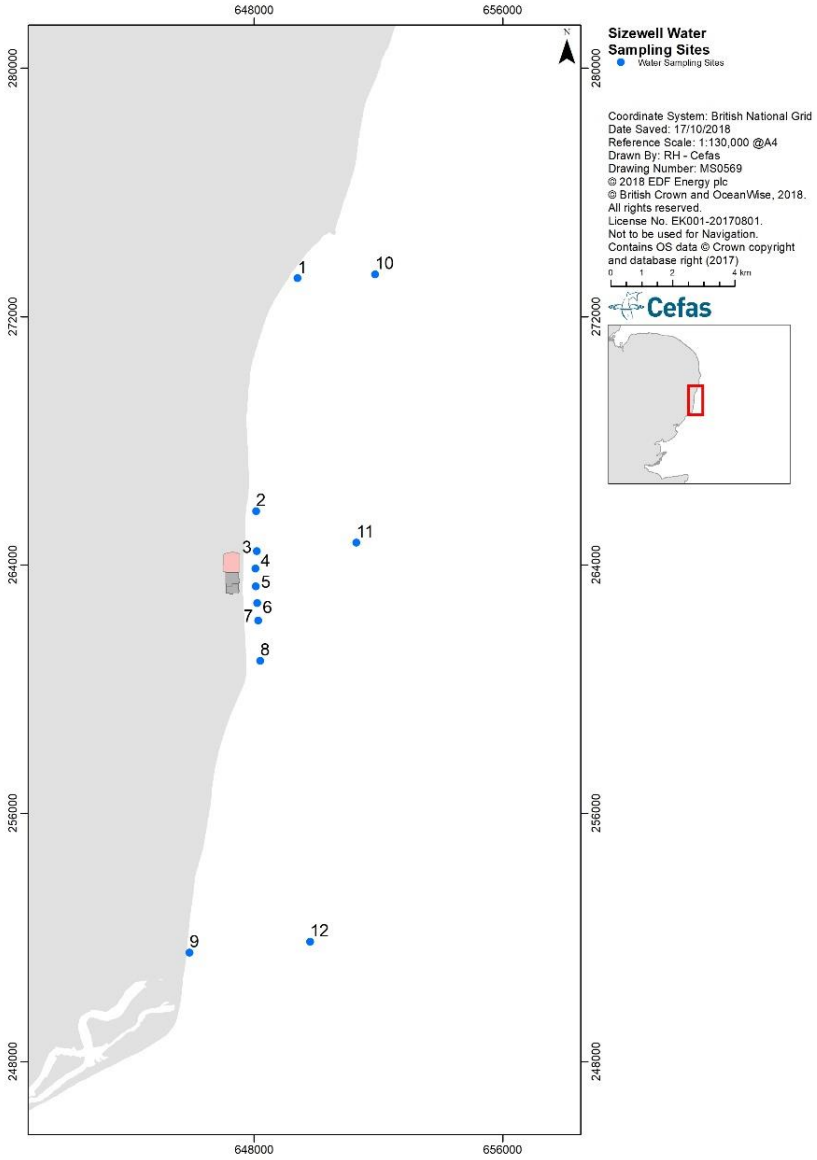


Figure 4 Location of the BEEMS sampling sites in the 2010 Sizewell monitoring survey

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Conductivity, temperature, and depth sensor (CTD) profiles showed that the waters sampled are well mixed for salinity. The temperature profiles indicate the presence of a thermally buoyant plume of water at the sea surface. Many of the chemical analyses give negative results, indicating that the analytes were either absent or present at concentrations below the limits of detection. Few differences between results from inshore of Sizewell Bank (stations 1 to 9) and offshore (stations 10 to 12) were noted, with the exceptions of suspended solids and turbidity. The higher measurements of suspended solids and turbidity inshore of Sizewell Bank are likely to be related to the shallower water depth and local sediment resuspension.

Concentrations of dissolved copper, arsenic, zinc, mercury, and cadmium exceed EQS levels on occasions. Some exceedance of the Environmental Quality Standard (EQS) concentrations for these metal and metalloid substances was detected at all stations except for stations 2 and 6. A small number of samples with concentrations more than their EQS was recorded for some polycyclic aromatic hydrocarbons (PAHs), biphenyl and bis (2-ethylhexyl) phthalate (DEHP), though most of the analyses for these compounds are negative. Exceedance of EQS concentrations for these organic compounds were detected at stations 1, 5, 9 and 12. These exceedances of organic EQSs were observed in samples acquired on three sampling dates: 7th and 8th April and the 19th May 2010.

Measured total residual oxidant (TRO) concentrations varied between 10 and 160$\mu gl^{-1}$. The EQS for TRO is 10$\mu gl^{-1}$ (0.01$mgl^{-1}$) but the limit of detection of the analytical method is of the order of 20$\mu gl^{-1}$ with the limit of quantification even higher therefore some caution must be applied to interpretation of measured results. The mean of all the TRO measurements (n = 725) was 40$\mu gl^{-1}$. Slight localised elevation of TRO was observed near the cooling water outfall and was below the level of detection within 2.4 km to the north and 500 m to the south. Elevated TRO was observed at the southern extremity of the survey area (at stations 9 and 12) but there was no spatial pattern to indicate that this elevation was connected to the power station outfall.

Hydrazine ($N_2 H_4$), an ammonia-derived compound and a strong reducing agent, is a chemical that is used in the secondary circuits of boilers and steam generators in power stations (including nuclear) because of its anti-oxidant properties and for this reason initial surveys included sampling and analysis for hydrazine. A wide range of hydrazine concentrations were apparently measured in the first 9 months of monitoring. Doubts about the validity of the ultraviolet-visible spectrophotometry results, based on the limits of quantification of the technique and potential interference, led to the use of an alternative analytical method. For the final three months of the programme a gas chromatography mass spectrometry (GC-MS) technique was used on water samples to measure hydrazine concentrations in addition to the spectrophotometric technique. The GC-MS technique was far more sensitive and indicated that hydrazine concentrations were generally below the limit of detection (0.01$\mu gl^{-1}$or 10$ngl^{-1}$). Prior hydrazine results are therefore considered unreliable. Three positive results were obtained from morpholine analyses conducted on water samples from stations 5 and 11. Morpholine is not used by Sizewell B power station therefore the origin and validity of these values is uncertain.

No concentrations of environmental concern were measured in the analyses carried out on sediment samples acquired at stations 5 and 11. All radionuclide concentrations measured in seawater samples were very low and were consistent with routine local radionuclide monitoring by the Environment Agency.

The results of this programme show that the concentrations of many elements and compounds are relatively uniform in the programme area. A small percentage of the samples acquired indicate that EQSs may occasionally be exceeded, but there is no indication that the Sizewell B power station causes these.

During 2014 and extending into the beginning of February 2015 additional water samples were collected monthly from up to four locations offshore of Sizewell B (Figure 5). The sample locations were the Sizewell B intake and outfall, the SZC planned combined intake/outfall and a BEEMS designated position just offshore of Dunwich labelled as SZ3 BEEMS reference station. The primary data referenced in this assessment are the measured nutrient concentrations which are discussed in more detail in the Sizewell supplementary monitoring data report, BEEMS Technical Report TR314.

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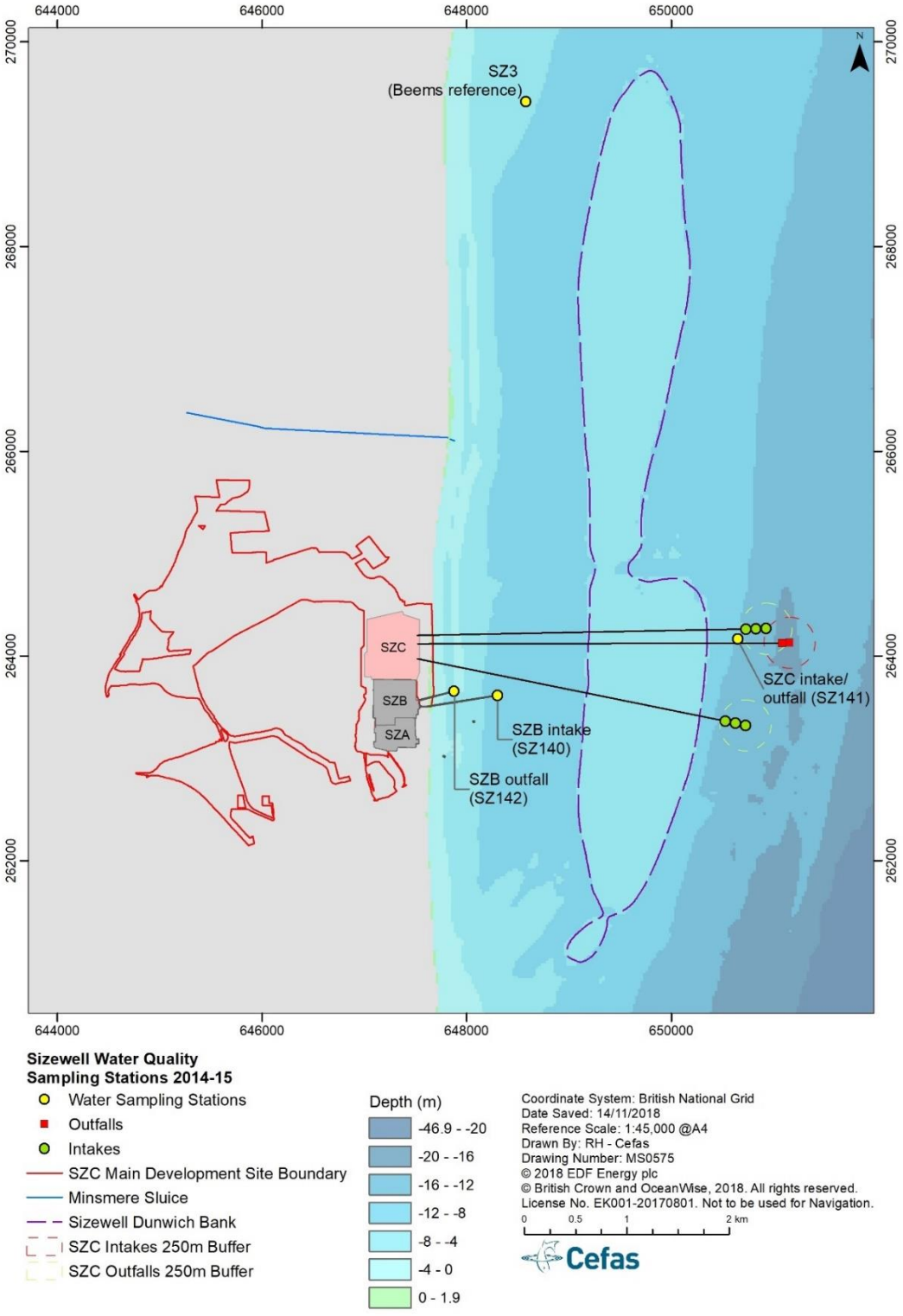


Figure 5 Map showing the marine water quality sampling locations during 2014/15 (BEEMS TR314) at Sizewell B intake and outfall, the SZC planned intake/outfall and a BEEMS reference station SZ3.

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# 4 Construction discharges

## 4.1 Background

The detailed information relating to expected chemical discharges during construction, commissioning, and operation of SZC was provided primarily in documents supplied by EDF and references with supporting information and approaches adopted for the Hinkley C development. It should be noted that some of the references quoted below are not specific to the Sizewell site and are based on the information provided in (publicly available) PCER documentation (PCER, 2009) and therefore are applicable to a generic UKEPR site. Regardless of information source appropriate adjustments are made where more specific details are known in this report to consider the proposal to construct two EPR units at SZC.

## 4.2 Drainage and wastewater

Wastes produced in the early phase of construction when no route for marine discharge is available will be tankered off site for appropriate disposal. Construction of the CDO and potentially the Fish Recovery and Return tunnels is likely to contribute much smaller quantities of groundwater and for a shorter period than those described and assessed in the following sections.

The CDO will be constructed by TBM and will be the primary discharge point for construction phase discharges of tertiary treated sewage, main site dewatering, TBM effluents and commissioning phase hydrazine releases (Figure 6). Discharges will be treated with a silt-buster or similar technology to minimise suspended solids being discharged into the receiving waters.

During construction of the CDO, the TBM is likely to be used to drag a pre-welded pipe into position. Pre-welding allows quality control prior to burial. The design of the CDO head has not yet been undertaken but is assumed to be the same as the FRR and comprise a concrete block with dimensions subject to final engineering design.

It is not planned for the CDO to function during the operational phase, however, the exact nature of all discharges that might occur during operation is not fully resolved. It is assumed that outages, every 12-18 months, will occur for each EPR separately and outage discharges, including hydrazine will be via the cooling water stream of the operational EPR. This is to say that both EPRs will not be offline simultaneously requiring outage discharges of hydrazine to be made via the CDO.

Construction phase drainage that may be discharged to the marine environment includes:

- Surface water drainage
- Effluent from the treatment of sewage plant and potable water by the on-site treatment works;
- Water pumped from both groundwater and excavations during construction dewatering activities.
- Wash water from cleaning concrete production equipment.
- Waste water from horizontal cooling water system tunnelling operations.

The main contaminants expected in the surface drainage from the construction area are suspended solids and hydrocarbons. Surface drainage water generation during the construction phase is likely to be highly variable over the course of the build period according to site activity and weather conditions.

The background concentration of metals measured in various groundwater sources at the SZC development site and the potential implications of their discharge are discussed below.

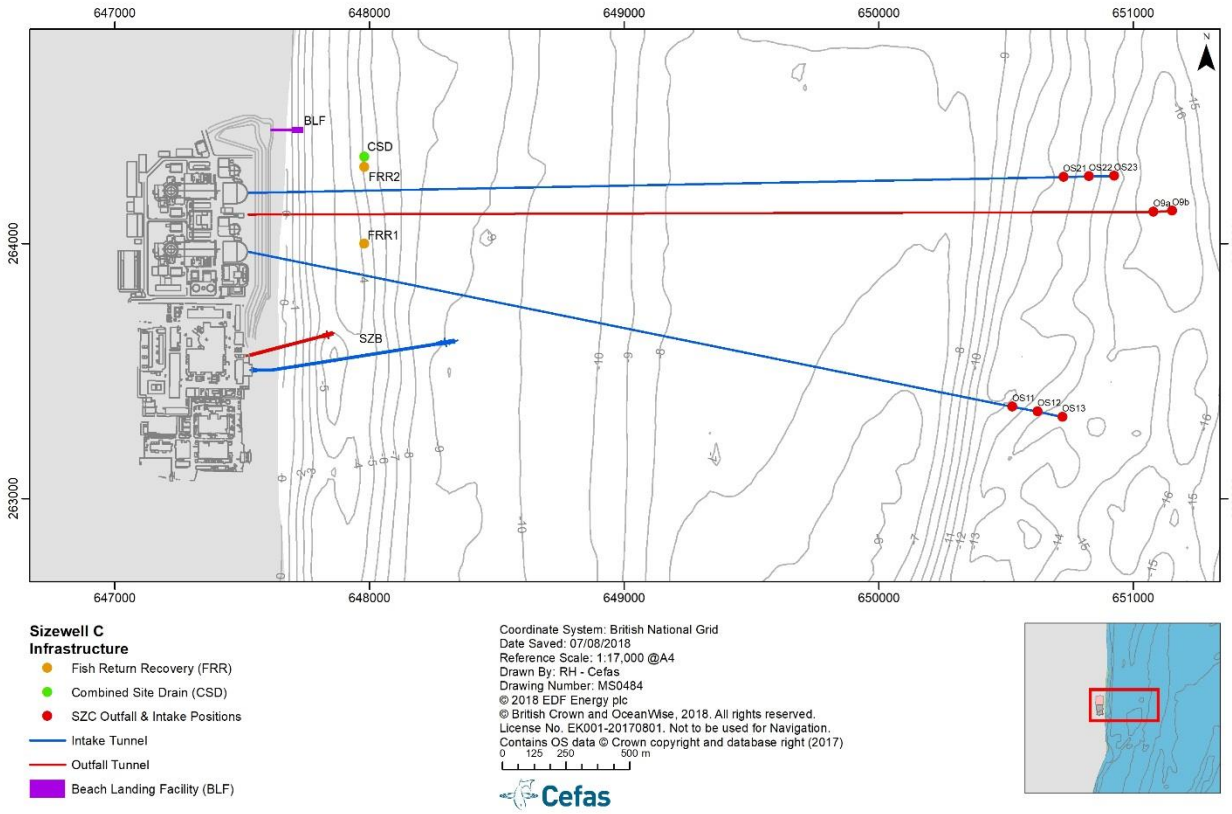


Figure 6 Schematic of development locations in the marine environment overlaid on bathymetry, blue indicates intake tunnels, red indicates outfall tunnels.

## 4.3 Construction discharge schedule

The CDO will be the primary discharge point for construction phase discharges of tertiary treated sewage, main site dewatering, TBM effluents and commissioning phase hydrazine releases.

As different site discharges may be present at the same time the timing, duration and magnitude of the likely discharges are important to determine. Table 6 shows an indicative set of construction discharge scenarios based on the example of phasing used for HPC but where known including specific information for SZC such as expected groundwater discharge rates and timing.

A cut-off wall will be constructed around the main construction site and over a 28-day period, groundwater will be lowered within this at an estimated discharge rate of $124ls^{-1}$ or $446m^3hr^{-1}$ (over this initial period the total waste water volume including the groundwater will be $0.155m^3sec^{-1}$). For the remainder of the construction period groundwater dewatering is estimated to occur at a rate of $15ls^{-1}$ or $54m^3hr^{-1}$.

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Package units for treatment of sewage and wastewater from welfare facilities would be established during the construction period with an estimated average discharge rate of 13.3ls$^{-1}$ and potential maximum of 30ls$^{-1}$ based on current plans at Hinkley Point.

Small amounts of concrete wash water are also likely to be discharged this is expected to contribute relatively small daily volumes up to 10m$^3$ a day (0.1ls$^{-1}$).

During tunnelling a combination of small quantities of groundwater and potentially residual concentrations of ground conditioning chemicals used in tunnelling may also be discharged.

A schedule of potential combined discharges during construction has been adapted based on that expected at Hinkley Point C and is shown in Table 6.

Table 6 Construction discharge scenarios during different phases (Case A-E) of construction at Sizewell C

| Date and activity change | Main site Groundwater (l s$^{-1}$) | Sewage (l s$^{-1}$) | Tunnelling wastes (and associated) discharges (l s$^{-1}$) | Case | Total Discharge (l s$^{-1}$) | Comments |
|---|---|---|---|---|---|---|
| WK 1 discharge available | | | | | | |
| | 124 | 0 | 0 | A | 124 | Worst Case Metals |
| WK 17 tunnelling start | | | | | | |
| | 15 | | 7 | B | 22 | |
| WK 26 permanent STP | | | SCL ramp up | | | |
| | 15 | 13.3 | 22 | C | 50.3 | |
| WK 49 | | | GW + soil conditioning 1 TBM | | | |
| | 15 | 13.3 | 26.7 | D | 55 | |
| WK 49 | | Occasional Max sewage | GW + soil conditioning 1 TBM | | | |
| | 15 | 30 | 26.7 | D1 | 71.7 | Worst Case Sewage |
| WK 81 | | | 2 TBMs | | | |
| | 15 | 13.3 | 6 | E | ~34 | Worst Case TBM |

- **Case A** is associated with the highest groundwater element and is the worst case for metals and will be screened at 124ls$^{-1}$
- **Case D** is the most usual case for sewage and faecal coliforms and includes groundwater from the main site and tunnelling. Additional contributions to N from hydrazine use during commissioning will also be considered with this scenario.
- **Case D1** is worst case for sewage of 30ls$^{-1}$ at 20,000µgl$^{-1}$combined with inputs from groundwater.
- **Case E** is the worst case for the TBM machines with the potential for 2 lots of ground conditioning chemicals to be discharged although recovery systems mean this is likely to be a negligible input.

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The following information is included to enable the plausible worst-case volume and contaminant concentrations to be considered for permitting. The schedule will inevitably change, but the summary of the worst-case conditions should provide conservative values representative of the likely changes. No seasonal dependence of the schedule has been considered therefore changes to the start or end times do not affect conclusions in the assessment: the assessment of impact is not dependent on the seasonality of the operations. The main seasonal factors affecting the discharge are wind variations and wave mixing. The modelling undertaken does not include wave mixing and so is conservative. Seasonal increases in wave height will increase mixing and reduce the areas of intersection (if any exist) between features and discharged waters above EQS concentrations.

Groundwater comprises the main dewatering flow (which after the initial dewatering phase remains constant at 15ls$^{-1}$ through the period considered) plus the contributions of groundwater resulting from the tunnelling and associated operations. Figure 7 shows that the groundwater discharge starts at 124ls$^{-1}$ from dewatering (Case A) which is the maximum groundwater contribution.

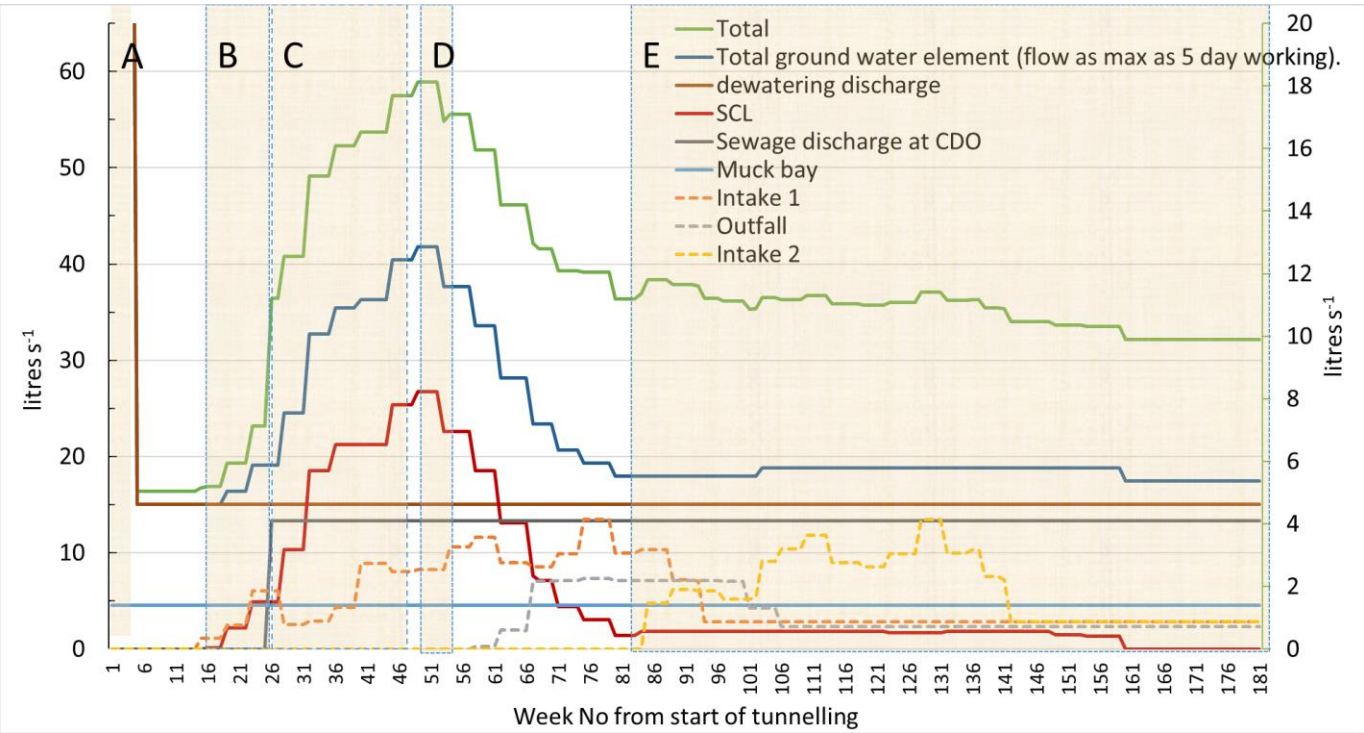


Figure 7 Likely flow volumes discharged at the CDO location from the start of tunnelling. Discharge volumes from 'Muck Bay' and TBM tunnelling for SZC intake 1, outfall and intake 2 are shown on the right hand axis. Timing is based on Hinkley C development and subject to change.

Groundwater reduces to 15ls$^{-1}$ after the first month and then, at around week 17, is added to by the discharge from the SCL (spray concrete-lined) works for approximately 50 weeks. Additional groundwater contributions during tunnelling increase groundwater input to around 42ls$^{-1}$ (Case D). Thereafter, the groundwater element of the discharge reverts to levels of around 15ls$^{-1}$ (Case E). Except for DIN and ammoniacal nitrogen the EVF calculation of groundwater derived substances uses only the groundwater volume with no assumption of additional dilution/contribution from the sewage discharge.

Figure 7 shows that the maximum discharges of flows that contain metals will occur during Case A. The maximum DIN input will be during Case D (between weeks 45 and 53 when the groundwater element reaches 42ls$^{-1}$). Case D is relatively transitory. Case D1, which includes an extreme case of sewage discharge, is also likely to be highly transitory. Once the SCL works are complete (Case E) the total

groundwater discharge falls to 15ls$^{-1}$. The waste from the TBM soil conditioning chemicals if present is likely to make the largest contribution during Case E. The total discharge volume during Case E is approximately 34ls$^{-1}$.

Wastewater will be generated if mud assisted drilling is adopted for construction of the horizontal cooling water tunnels. The initial estimated volume of wastewater generated during this process is based on that derived from the construction discharge schedule developed for Hinkley Point C

For assessment, maximum loads are to be addressed within modelling scenarios. The issues of concern being, maximum loads of; heavy metals, Un-ionised ammonia, Dissolved Inorganic Nitrogen (DIN), Biochemical oxygen demand, faecal coliforms from treated sewage effluent, metals and DIN from groundwater and any tunnel boring additives that are not recovered for reuse.

## 4.4 Total loads of cadmium and mercury

There are specific requirements for the minimisation of the annual loads of selected hazardous substances and cadmium and mercury are included and require assessment. Figure 8 shows the discharge rate for groundwater left axis and blue line. Groundwater discharge is very high (above left axis maximum shown) in the first 28 days (124ls$^{-1}$) during the main dewatering on site and then decreases rapidly to around 15ls$^{-1}$. From around week 16 to 76 groundwater varies due to overlapping contributions from tunnelling of intake 1, the outfall and intake 2. Over this whole period the cumulative load of cadmium and of mercury derived from the groundwater is shown by the brown and red lines and the scale on the right-hand axis. Over this 3.5 year period the cumulative load for cadmium is 0.45kg and for mercury is 0.05kg. Both these load figures meet the requirement to not exceed a significant annual load of 1kg for mercury or 5kg for cadmium. Trace contamination of raw materials used in demineralisation of water used during cold commissioning may contribute additional loadings of mercury and cadmium but based on maximum annual loadings during normal operation when the systems are in full use the additional annual loadings, cadmium 0.37kg and mercury 0.099kg (Table 29) would not result in exceedance of the significant loads.

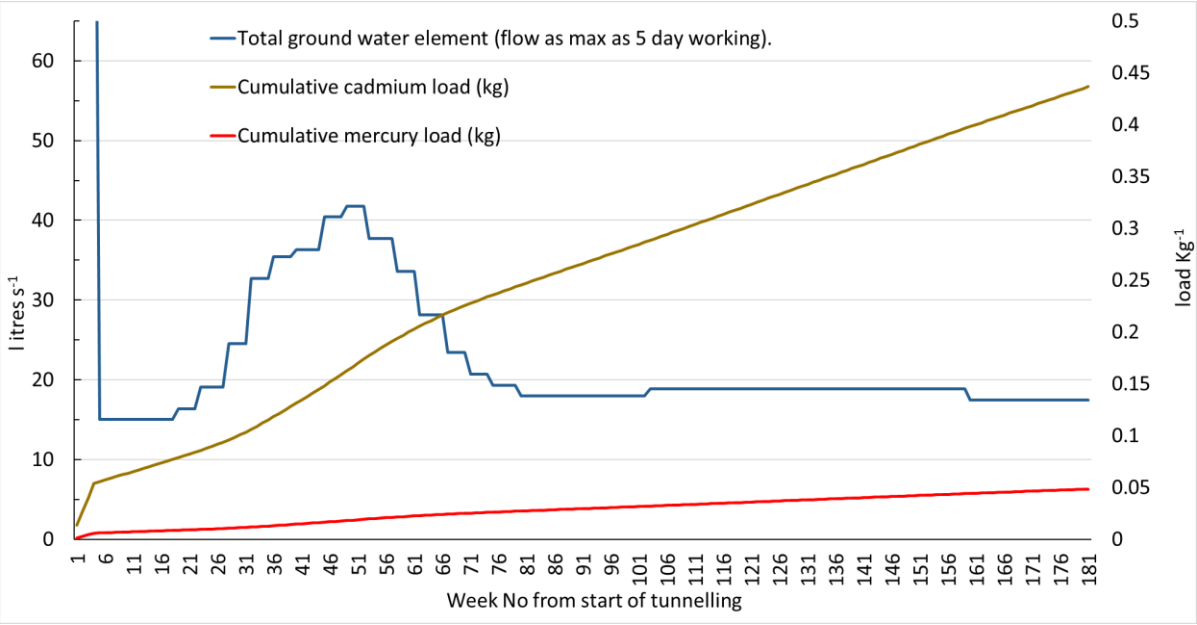


Figure 8. Just over 3.5 year timeline of groundwater discharge (ls$^{-1}$ left axis) and resulting cumulative metal load for Mercury and Cadmium (kg right axis).

### 4.5 Discharges screened out of assessment

Other temporary and more variable discharges to marine water may form part of the surface drainage strategy during the construction phase together with the range of expected discharges detailed above. The main expected contaminants in these discharges are suspended solids, Biochemical Oxygen Demand (BOD) and hydrocarbons. Assessment using the Environment Agency/Defra screening of contaminant contributions from surface drainage sources (Defra and Environment Agency Guidance, 2016) is not appropriate due to their highly variable nature over the construction period. Hydrocarbons can be removed from effluent prior to discharge by the incorporation of suitable oil separators within temporary drainage systems and any potential for chemical and oil spills during construction activities, whilst recognised, would be covered under the Government waste management guidelines. Therefore, no chemical release effects to the water and sediment quality of the local area are expected from these variable sources and they are therefore screened out of further assessment. Siltbuster or similar technology would be used to manage suspended sediments in drainage.

# 5 Screening assessment of construction discharge

### 5.1 Background

As part of a surface water risk assessment (Environment Agency and Department for Environment Food and Rural Affairs, 2016) the concentration of substances present in the discharge must be assessed against a list of specific pollutants and their Environmental Quality Standards (EQS). Initial screening tests (historically referred to as H1 tests) were conducted to determine if the concentrations of priority substances and specific pollutants in the discharge exceeded their respective EQS. For any substances that breach the EQS in the initial screening tests (Test 1) a further screening test is applied that takes account of initial dilution upon discharge (Test 5).

The EA Test 5 screening applies to the discharge from the CDO because the discharge is to the subtidal environment and beyond 50m from mean low water spring (MLWS) tidal level. Separate guidance is provided for assessment of large cooling water discharges that would occur during operation see section 9.

### 5.2 Handling of substance data

When calculating summary statistics for all substances, any values below the method detection limit were adjusted to a value equal to the detection limit. For metals, modelling tests use both total and dissolved concentrations to assess potential deterioration of surface water quality (Environment Agency, 2014). The total concentration of substances is used in the initial screen and in subsequent modelling to take account of uncertainty regarding the partitioning of substances into the dissolved phase as the groundwater mixes with the seawater. For several neutral hydrophobic chemicals and some metals, however, solubility would be expected to decrease under saline conditions (Turner, 2003). In this assessment only, dissolved substance data were available for the groundwater. The assessment includes the screening of the source terms against the saltwater EQS values presented in the Water Framework Directive (Standards and Classification) Directions (England and Wales) (WFD, 2015). EDF has reviewed the data from the boreholes that will form the longer-term network (those along the northern, western and eastern sides of the deep excavation) as well as wider data sets that are reflective of current arrangements, including temporary boreholes installed to enhance the efficacy of local dewatering. In each case, the 95th percentile for each of the substances of concern has been considered as this excludes anomalously high values while still providing a robust assessment. To enable a robust assessment of the potential impacts of the proposed discharge on the water environment and on the interest features to be made, the worst-case values have been selected from these

datasets. Summary statistics for the concentrations of substances measured in the site groundwater carried forward to the screening assessment are shown in Tables 7, 8 and 9.

The updated guidance for surface water pollution (Environment Agency, 2016) recommends the application of an initial test (Test 1) for discharges to Transitional and Coastal (TraC) waters in which the discharge concentration is compared to the relevant quality standard or equivalent for that substance. Where the discharge concentration exceeds the standard concentration, further assessment is required. When the discharge concentration is divided by the EQS in Test 1 any values of 0.5 and above are taken forward to the next stage of screening. As this construction discharge will be subtidal and is over 50 metres offshore, a further test ("Test 5") is recommended. Test 5 divides the concentration of a substance and volume discharged (the discharge specific Effective Volume Flux, EVF) by its EQS minus background concentration (the location specific Allowable Effective Volume Flux, AEVF). If the EVF is not greater than the AEVF, then the discharge is insignificant and is screened out. The AEVF references the discharge depth and this value can be up to a maximum of 3.5 metres. For Sizewell the discharge depth for construction relative to chart datum is greater than 3.5 metres therefore 3.5 this is the AEVF used for comparison as shown in Table 8 and 9.

The discharge concentrations in grey shading in Table 7 are those used in the EVF calculation. Theoretically, the mean values could be used in the EVF calculation with the annual average EQS, however, this assumes that the mean discharge is an annual average. As the discharge concentration is determined by the dewatering process it is not appropriate to assume a random process contributing to the discharge concentration, and the discharge is intended to occur over several years. There could, for instance, be many months when values above the mean are present in the chemical discharge. As a precautionary approach, the 95th percentile discharge concentrations have been used for calculating the EVF values.

As the suspended sediment concentration at a given location directly affects light penetration and the potential for increased phytoplankton growth, the reference concentration of dissolved inorganic nitrogen (DIN) for TraC waters for the Good/Moderate boundary also references the suspended sediment concentration. Several monitoring studies have measured the suspended sediment regime off Sizewell (BEEMS TR189). Suspended Particulate matter (SPM) data was also gathered from MODIS satellite database (Dolphin, Silva and Rees, 2011) for a project evaluating natural sediment variability in Regional Environmental Assessment areas in the North Sea and English Channel. Satellite data for suspended particulate matter showed average mean SPM value at Sizewell during April to August of 31$mgl^{-1}$(and average maximum 80$mgl^{-1}$) and during September to March 73$mgl^{-1}$(and average maximum 180$mgl^{-1}$). An annual mean SPM for these data was 55.3$mgl^{-1}$. With reference to the suspended sediment levels associated with WFD nitrogen standards (Appendix C) and based on the satellite data and previous monitoring surveys Sizewell is classed as of intermediate turbidity (Water Framework Directive Standards and Classification Directions, 2015).

The volume of water that would need to be disposed of during the initial dewatering phase is in the order of 300,000 $m^3$ based on the hydraulic properties of the materials within the cut-off wall around the main construction site. It is estimated that to lower groundwater within the cut-off wall to the design level will take 28 days at a rate of 124$ls^{-1}$. Following the initial lowering of water levels there will be some nominal ongoing discharge throughout the construction phase to deal with nuisance water (rainfall, seepage through the cut-off wall) but the volumes will be very small at estimated values of 15$ls^{-1}$. Groundwater samples were analysed during 2014-16 (ATKINS, 2016). A survey of exploratory boreholes across the site analysed for a suite of chemicals and polycyclic aromatic hydrocarbons, PCBs, various organic chemicals commonly present as contaminants in groundwater were below respective detection limits (details Appendix A). Ammonium, nitrite, and nitrate and eight metals analysed for were above detection and the results are summarised Table 7.

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## NOT PROTECTIVELY MARKED

Table 7 Metal concentration range measured in SZC construction site groundwater and relevant EQS values and marine background concentrations.

| Substance | Mean dissolved concentration µg $l^{-1}$ | 95% dissolved concentration µg $l^{-1}$ | Saltwater EQS AA µg $l^{-1}$ | Saltwater EQS MAC µg $l^{-1}$ | Marine Background concentration µg $l^{-1}$ |
|---|---|---|---|---|---|
| Arsenic | 3.55 | 11.5 | 25 | - | 1.07 |
| Cadmium | 0.10 | 0.18 | 0.2 | 1.5 | 0.05 |
| Chromium | 6.39 | 18.45 | 0.6 | 32 | 0.57 |
| Copper | 1.87 | 4.25 | 3.76 | - | 2.15 |
| Lead | 1.07 | 1.07[1] | 1.3 | 14 | - |
| Zinc | 7.34 | 17.5 | 6.8[2] | | 15.12 |
| Mercury | 0.013 | 0.023 | - | 0.07 | 0.02 |
| Iron | 395 | 1500 | 1000 | - | 50 |

1 The limited number of values above detection limits leads to a mean value higher than the 95 percentile which represents a value below detection limit therefore the higher mean value is used here 2: The EQS for zinc may be adjusted to take account of local background

Two assessments are made for groundwater substance inputs, one for the initial 28 days of construction period during which the groundwater contribution to site discharges is at a rate of 124$ls^{-1}$ (Table 8) and for the remainder of the construction period at 15$ls^{-1}$(Table 9).

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 8 Determination of whether after discharge and initial mixing the calculated contribution to dissolved metals from the site groundwater (124ls$^{-1}$, Case A) during the first 28 days of the construction period is likely to exceed the acceptable threshold above which significant impacts may occur

| Parameter value | 95 percentile Dissolved metal concentration µg l$^{-1}$ | TraC Water test 1 <100%EQS | Pass/Fail | TraC Water test 5 EVF<AEVF | Pass/Fail |
|---|---|---|---|---|---|
| Arsenic | 11.5 (n=82,151) [i] | **0.46** (11.5/25[ii]) | ≤1.0 (Pass) | 0.06 (11.5x0.124[iii])/(25-1.07[iv]) | 0.06<3.5[v] (Pass) |
| Cadmium | 0.18 (n=17,151) [i] | **0.9** (0.18/0.2[ii]) | ≤1.0 (Pass) | 0.149 (0.18x0.124[iii])/(0.2-0.05[iv]) | 0.149<3.5 [v] (Pass) |
| Chromium (mean) | 18.45 (n=111,152) [i] | **30.75** (18.45/0.6[ii]) | ≥1.0 (Fail) | **76.3** (18.45x0.124[iii])/(0.6-0.57[iv]) | 76.3>3.5 [v] **(Fail)** |
| Chromium (95 Percentile) | 18.45 (n=111,152) [i] | 0.58 (18.45/32[ii]) | ≤1.0 (Pass) | 0.07 (18.45x0.124[iii])/(0.6-0.57[iv]) | 0.07<3.5 [v] (Pass) |
| Copper | 4.25 (n=57,151) [i] | **1.13** (4.25/3.76[ii]) | ≥1.0 (Fail) | 0.33 (4.25x0.124[iii])/(3.76-2.15[iv]) | 0.33<3.5 [v] (Pass) |
| Zinc | 17.5 (n=134,151) [i] | **2.6** (17.5/6.8[ii]) | ≥1.0 (Fail) | N/A (17.5x0.124[iii])/(6.8-15.12[iv]) | N/A |
| Mercury (95 percentile) | 0.023 (n=31,151) [i] | **0.33** (0.023/0.02[ii]) | ≤1.0 (Pass) | 0.057 (0.023x0.124[iii])/(0.07-0.02[iv]) | 0.057<3.5 [v] (Pass) |
| Iron | +1500 (n=37,151) [i] | **1.5** (1500/1000[ii]) | ≥1.0 (Fail) | 0.196 (1500x0.124[iii])/(1000-50[iv]) | 0.196<3.5 [v] (Pass) |
| Lead | 1.07 (n=3,151) [i] | (0.82) (1.07/1.3[ii]) | ≤1.0 (Pass) | 0.44 (1.07x0.124[iii])/(1.3-1.0 [vi]) | 0.44<3.5 [v] (Pass) |
| Lead (95th percentile) | 0.58 (n=3,151) [i] | 0.04 (0.58/14[ii]) | ≤1.0 (Pass) | 0.005 (0.58x0.124[iii])/(1.3-1.0) | 0.005<3.5 [v] (Pass) |

[i]Number of values measured above detection and total number of values; [ii]Annual average EQS value (also includes 95 percentile for chromium and lead) [iii] total construction effluent discharge m$^3$/sec [iv] mean background concentration Sizewell TR314 2014/15 and Appendix E;[v] Allowable effective volume flux is taken as the maximum value of 3.5

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## NOT PROTECTIVELY MARKED

Table 9 Determination of whether after discharge and initial mixing the calculated contribution to dissolved metals from the site groundwater (uses maximum groundwater value, Case D) during the main construction period is likely to exceed the acceptable threshold above which significant impacts may occur

| Parameter value | Calculated dissolved metal discharge concentration µg $l^{-1}$ | TraC Water test 1 <100%EQS | Pass/Fail | TraC Water test 5 EVF<AEVF | Pass/Fail |
|---|---|---|---|---|---|
| Arsenic | 11.5 (n=82,151) [i] | **0.46** (11.5/25[ii]) | ≤1.0 (Pass) | 0.02 (11.5x0.042[iii])/(25-1.07[iv]) | 0.02<3.5[v] (Pass) |
| Cadmium | 0.18 (n=17,151) [i] | **0.9** (0.18/0.2[ii]) | ≤1.0 (Pass) | 0.05 (0.18x0.042[iii])/(0.2-0.05[iv]) | 0.05<3.5 [v] (Pass) |
| Chromium (mean) | 18.45 (n=111,152) [i] | **30.75** (18.45/0.6[ii]) | ≥1.0 (Fail) | **25.64** (18.45x0.042[iii])/(0.6-0.57[iv]) | **25.64>3.5 [v] (Fail)** |
| Chromium (95 Percentile) | 18.45 (n=111,152) [i] | 0.58 (18.45/32[ii]) | ≤1.0 (Pass) | 0.024 (18.45x0.042[iii])/(0.6-0.57[iv]) | 0.024<3.5 [v] (Pass) |
| Copper | 4.25 (n=57,151) [i] | **1.13** (4.25/3.76[ii]) | ≥1.0 (Fail) | 0.11 (4.25x0.042[iii])/(3.76-2.15[iv]) | 0.11<3.5 [v] (Pass) |
| Zinc | 17.5 (n=134,151) [i] | **2.6** (17.5/6.8[ii]) | ≥1.0 (Fail) | N/A | N/A |
| Mercury | 0.023 (n=31,151) [i] | **0.33** (0.023/0.02[ii]) | ≤1.0 (Pass) | 0.01 (0.023x0.042[iii])/(0.07-0.02[iv]) | 0.01<3.5 [v] (Pass) |
| Iron | 1500 (n=37,151) [i] | **1.5** (1500/1000[ii]) | ≥1.0 (Fail) | 0.07 (1500x0.042[iii])/(1000-50[iv]) | 0.07<3.5 [v] (Pass) |
| Lead | 0.58 (n=3,151) [i] | (0.45) (1.07/1.3[ii]) | ≤1.0 (Pass) | 0.15 (1.07x0.042[iii])/(1.3-1[vi]) | 0.15<3.5 [v] (Pass) |
| Lead (95th percentile) | 0.58 (n=3,151) [i] | 0.04 (0.58/14[ii]) | ≤1.0 (Pass) | 0.002 (0.58x0.042[iii])/(1.3-1[vi]) | 0.002<3.5 [v] (Pass) |

[i]Number of values measured above detection and total number of values; [ii]Annual average EQS value (also includes 95 percentile for chromium and lead) [iii] maximum groundwater construction effluent discharge (Case D) $m^3$/sec [iv] mean background concentration Sizewell TR314 2014/15 and Appendix E;[v] Allowable effective volume flux is taken as the maximum value of 3.5, [vi] lead background detection limit in BEEMS TR189

The Effective Volume Flux of the discharge (EVF) is defined as:

*EVF = (EFR x RC) / (EQS – BC)* $m^3 s^{-1}$

*Where:*

*EFR = the effluent discharge rate* ($m^3 s^{-1}$)
RC = release concentration of the priority substance of concern ($\mu g l^{-1}$)
EQS = EQS (AA) of the substance of concern ($\mu g l^{-1}$)
BC = mean background concentration at the discharge location ($\mu g l^{-1}$)

The metals contamination in boreholes can vary and the data provide dissolved concentrations only so the 95 percentile concentrations have been used to provide a more conservative assessment.

Dewatering discharges do not pass initial screening tests for chromium during the first 28 days of the construction period. Although zinc concentrations are relatively low at just over twice the EQS because the site background concentrations for zinc are above the EQS, Test 5 cannot be conducted. Both zinc and chromium are therefore taken forward for more detailed modelling.

## 5.3 Screening assessment of un-ionised ammonia resulting from construction discharge to the marine environment

Ammonia enters freshwater and marine water bodies from sewage effluent inputs, from industrial and agricultural activities and from the breakdown of organic matter. In the marine environment the toxicity of ionised ammonia ($NH_4$) should be considered. In waters, particularly at higher salinities, it has been shown that the ammonium ion can also permeate the gills, and so the concentration of total ammonia $NH_4$ can also be toxicologically significant. Total ammonia values of 1100 (annual average) and 8000µg/l $NH_4$-N (WQTAG086, 2005) are therefore set as guide values for habitats andthese are considered. In general, the un-ionised form of ammonia is more toxic than the ionised form. At higher pH values, un-ionised ammonia represents a greater proportion of the total ammonia concentration. Temperature increase also raises the relative proportion of un-ionised ammonia, but this effect is much less marked than for pH change, e.g. a temperature increase of 10°C (from 10 to 20°C) may double the proportion of un-ionised ammonia, but a pH change from a pH 7 to pH 8 produces an approximately tenfold increase (Eddy, 2005). A greater percentage of ammonia will also be in the un-ionised form when the salinity is lower.
The concentration of un-ionised ammonia can therefore be derived from knowledge of the total ammoniacal nitrogen concentration (i.e. $NH_4$ as N), the salinity, the pH and temperature using the EA calculator (Clegg and Whitfield, 1995). pH is the most important with an approximate doubling in un-ionised ammonia concentration between pH 7.5 and 8.

The EQS for un-ionised ammonia is $21 \mu g l^{-1}$ expressed as an annual average, however being consistent with the previous screening, this value is compared with the 95$^{th}$ percentile source contributions. The 95$^{th}$ percentile values used for the source terms were a groundwater ammonium concentration of $5557.2 \mu g l^{-1}$ as N and a treated sewage effluent maximum concentration of $20{,}000 \mu g l^{-1}$ as N. $20{,}000 \mu g l^{-1}$ as N represents the design standard of the sewage treatment plant.

Table 10 shows the un-ionised ammonia concentration in construction effluents based on initial physiochemical conditions and the hypothetical un-ionised concentration based on seawater conditions if the total ammonia ($NH_4$) concentration was undiluted. Cases A, D1, D and sewage only are considered. In each example case the un-ionised ammonia concentration in the source effluent under initial physicochemical conditions exceeds the EQS and so would fail Test 1 of the H1 assessment. Under seawater conditions

concentrations of un-ionised ammonia would exceed the EQS by a higher margin. Because the effluent discharges represent relatively small discharge volumes and the un-ionised ammonia concentrations are not many times above the EQS the test 5 dilution assessment results in all discharges passing the assessment.

At this stage, further assessment would normally not be required but further work has been conducted to assess the potential extent of the mixing zone that is predicted to be in exceedance of the EQS.

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Table 10. Starting concentrations before mixing with seawater for un-ionised ammonia concentrations for groundwater (Case A), treated sewage and combined discharge (D and D1) derived using the EA calculator as a source term before mixing (un-ionised ammonia levels are also shown for seawater conditions assuming no dilution of the source effluent).

| Discharge | Groundwater $ls^{-1}$ | Sewage $ls^{-1}$ | Ammoniacal nitrogen (N) ($\mu gl^{-1}$) | Salinity | Temp °C | pH | Un-ionised ammonia ($\mu gl^{-1}$) | Test 5 Result |
|---|---|---|---|---|---|---|---|---|
| Case A | 124 | - | 5,557 | 1 | 11.43 | 7.3 | 22.8 | 0.14 |
| Case D | 41.7(15+26.7)[1] | 13.3 | 9,049 | 1 | 11.43 | 7.3 | 37.2 | 0.10 |
| Case D1 | 41.7(15+26.7) | 30 | 11,600 | 1 | 11.43 | 7.3 | 47.6 | 0.16 |
| Sewage discharge only | - | 13.3 | 20,000 | 1 | 11.43 | 7.3 | 82.1 | 0.05 |
| Case A | 124 | - | 5,557 | 33.3 | 11.43 | 8.05 | 22.8 | 0.61 |
| Case D | 41.7(15+26.7)[1] | 13.3 | 9,049 | 33.3 | 11.43 | 8.05 | 37.2 | 0.44 |
| Case D1 | 41.7(15+26.7) | 30 | 11,600 | 33.3 | 11.43 | 8.05 | 47.6 | 0.73 |
| Sewage discharge only | - | 13.3 | 20,000 | 33.3 | 11.43 | 8.05 | 82.1 | 0.23 |

[1]groundwater from main site and from tunnelling

For some Cases small sources which would dilute the concentration, but which may not be present all the time have not been considered (e.g. in case D there could be 4 litres per second of additional water not containing DIN).

1) Case A total discharge is 124$ls^{-1}$ with a 95$^{th}$ percentile concentration of 5,557$\mu gl^{-1}$ ammoniacal Nitrogen as N.
2) Case D total discharge is 55$ls^{-1}$ with a 95$^{th}$ percentile concentration of 9,049$\mu gl^{-1}$ ammoniacal Nitrogen as N.
3) Case D1 total discharge is 71.7$ls^{-1}$ with a 95$^{th}$ percentile concentration of 11,600$\mu gl^{-1}$ ammoniacal Nitrogen as N.
4) Sewage only discharge is13.3$ls^{-1}$ at a planned maximum of 20,000$\mu gl^{-1}$ ammoniacal Nitrogen as N.

Mixing of the different sources contributing ammoniacal nitrogen and the ratio of un-ionised to ionised ammonia upon mixing with seawater is evaluated with dilution rates using CORMIX and these data are presented and discussed in section 6.

### 5.4 Dissolved inorganic nitrogen (DIN) contribution to nutrient status

Background winter DIN concentrations in Greater Sizewell Bay, are mean 25.5µmol (minimum 21, maximum 31) or, as N, 357$\mu gl^{-1}$ (minimum 0.30, maximum 0.43) (source: BEEMS Technical Report TR314).

The discharge of DIN at the CDO is made up of the following sources:

1. The total dewatering discharge (with a maximum flow during Case D of approximately 41.7ls$^{-1}$) with a mean concentration of 1021µgl$^{-1}$ as N;
2. The sewage treatment from the main plant construction with a flow of 1150m$^3$ day$^{-1}$ or 13.3ls$^{-1}$. With secondary treatment, this implies 5000µgl$^{-1}$ of ammoniacal nitrogen as N. This value is used conservatively as a mean but is most likely to represent the 95$^{th}$ percentile concentration.

In addition to these sources a further small volume of discharge ~3ls$^{-1}$ of discharge may be contributed from sources not containing DIN from the use of tunnelling chemicals. Combining the flow sources gives a maximum flow (during Case D) of 55ls$^{-1}$ with a concentration 1980µgl$^{-1}$(as N) if conservatively ignoring the dilution of 3ls$^{-1}$ which may not always be present.

### 5.4.1 Maximum concentration and flow

Considering additional contributions besides ammoniacal nitrogen to nitrogen in sewage the maximum concentration of DIN in the sewage discharge could be up to 23,000µgl$^{-1}$ of nitrogen as N (Table 11). The mean flow rate is 13.3ls$^{-1}$ but flow may peak intermittently up to 30ls$^{-1}$. It should be stressed that the 95$^{th}$ percentile concentration of the sewage treatment plant is still 5000µgl$^{-1}$. This value has been used as previously and is still a conservative estimate of the total loading discharged. Maximum discharge flow occurs during the first month at 124ls$^{-1}$ but consists only of groundwater contributions to DIN. It is possible that maximum discharge flow could occur during the Case D period. Using mean conditions for concentration and total maximum combined flow, regime D1$_{mean}$, becomes 71.7ls$^{-1}$ at 2,680µgl$^{-1}$ (as N). In a very unlikely case the maximum sewage flow (30ls$^{-1}$) and maximum concentrations for sewage (23000µgl$^{-1}$) and 95$^{th}$ percentile for ground water (5,636µgl$^{-1}$), would be 71.7ls$^{-1}$ at 12900µgl$^{-1}$ (as N) which is the D1 Case. The latter stages of the construction/commissioning period are E$_{mean}$ and E with flow rates of approximately 28.3ls$^{-1}$ (there would be further volume contributions from tunnelling wastewater, but these would not contribute DIN) and concentrations of 2,890µgl$^{-1}$ and 5,340µgl$^{-1}$ respectively.

The discharges during construction that may contain DIN are likely to be of variable duration and concentration. Table 11 illustrates some potential cases. For inorganic nitrogen the Water Framework Directive standard for Good status is based on the winter dissolved inorganic nitrogen 99$^{th}$ percentile for TraC waters of intermediate turbidity (suspended solids levels of 10 to <100mgl$^{-1}$) 99$^{th}$ percentile winter concentration that varies according to turbidity. For the H1 assessment a value of 980µgl$^{-1}$ as a 99$^{th}$ percentile for Good status has been adopted as a benchmark standard. In each case considered the background benchmark value exceeds the benchmark nitrogen value in the discharges (Table11).

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Table 11. DIN concentrations for groundwater (GW), treated sewage (STW) and combined discharge.

| Case | Groundwater flow $ls^{-1}$ | DIN concentration $\mu gl^{-1}$ | Sewage Flow $ls^{-1}$ | DIN concentration $\mu gl^{-1}$ | DIN Discharge concentration $\mu gl^{-1}$ | Test 5 Result |
|---|---|---|---|---|---|---|
| A | 124 | 5636 (95%) | 0 | 0 | 5636 | 1.26[1] |
| D1 mean | 41.7 | 1021 (mean) | 30 | 5000 | 2686 | 0.35 |
| D1 | 41.7 | 5636 (95%) | 30 | 23000 | 12901 | 1.67 |
| E mean | 15 | 1021 (mean) | 13.3 | 5000 | 2891 | 0.15 |
| E | 15 | 5636 (95%) | 13.3 | 5000 | 5337 | 0.27 |

[1] Test 5 is ($m^3$/sec x discharge concentration)/(EQS-background) for this example this is (0.124 x 5636)/(980-426) =1.26 It should be noted that a more specific methodology for deriving 99th percentile values based on a relationship between SPM and DIN is recommended in draft Environment Agency guidance unpublished and for an annual average SPM of 55.2$mgl^{-1}$ would give a slightly lower value of 952$\mu gl^{-1}$ as a 99$^{th}$ percentile but the screening here would only slightly change.

Applying Test 5 of the H1 assessment to the discharges of DIN from example cases during the construction period (Table 11) all values passed the assessment following initial dilution i.e. none of the values exceeds a value of 3.5. Although the DIN assessment indicates that the discharge concentration and volume are likely to have a limited extent of effect the natural background for DIN varies during the year and inputs during summer periods when nitrogen is more limited may have a greater effect on phytoplankton growth therefore further modelling assessment of the loadings is made to determine this.

The highest most continuous daily loadings will be contributed during Case $D1_{mean}$, which includes a maximum sewage discharge rate and highest groundwater discharge rate (except for the initial dewatering period in the first month of construction). The total flow rate during $D1_{mean}$ is $71.7ls^{-1}$and a concentration (represented by the 95$^{th}$ percentile for sewage) of $2680\mu gl^{-1}$ would lead to a discharge of $16.6kgd^{-1}$.

During commissioning, un-ionised ammonia is used (approximately $0.66kgd^{-1}$ average daily discharge) in the steam generator of the EPR and as this precedes construction/operation of the cooling water system the discharge will also occur through the CDO. Nitrogen input from commissioning is added to the groundwater and sewage loading derived for Case $D1_{mean}$ to provide a representative worst-case daily loading of $17.3kgd^{-1}$ DIN. This loading is therefore used for assessment of the potential impact on phytoplankton growth for the construction/commissioning period.

## 5.5 Phosphorus influence on nutrient status

Phosphorus load discharged during construction is contributed from groundwater, from treated sewage effluent and from use of phosphate during commissioning. A concentration 10mg $l^{-1}$ as P was derived for treated sewage from package units based on Natural England, 2016. For groundwater a 50$^{th}$ percentile value of 0.04mg $l^{-1}$ as TP was derived for Thames groundwater by Stuart and Lapworth, 2016 and is used here as a substitute prior to site data becoming available. For the cold commissioning input reference was made to HPC-EDECME-AU-000-RET-000063, 2017 and a maximum discharge of phosphate per day based on a period of hydraulic testing and preservation of closed cooling circuits, chilled water and electrically produced hot water systems). A value of 594kg $PO_4$ use over 85 days (covering various phases of EPR commissioning) was used as a reference to derive a daily value of 2.28kg as P. Adding the commissioning load to that of treated sewage (~26kg) and groundwater gives a total load of 28.2kg for assessment of combined nutrient inputs during construction and cold commissioning using a phytoplankton growth model.

## 5.6 BOD influence on dissolved oxygen

The Water Framework Directive applies to 1 nm from the coast (approx. 1850m) and from 2016 the Marine Strategy Framework Directive applies to the UK boundary. These standards use the same criteria for defining permissible dissolved oxygen (DO) concentrations, 4 – $5.7mgl^{-1}$ being good status and above $5.7mgl^{-1}$ is high status.
The background BOD near to the Sizewell B cooling water discharge based on monitoring done in 2010 (BEEMS Technical Report TR189) has a mean value of $2mgl^{-1}$. Any area not exceeding $1.5mgl^{-1}$ deviation from background is expected to generate less than $0.5mgl^{-1}$ impact on dissolved oxygen (OSPAR Comprehensive studies report, 1997). Based on Hinkley Point information, during construction of SZC the sewage treatment works is expected to achieve a maximum concentration of BOD of $40mgl^{-1}$ (i.e. over 5 days).
Based on the expected number of staff on site during the construction phase and waste water production of 100 litres/per head/per day a more typical sewage discharge of $13.3ls^{-1}$ (Case D) is expected through most of the construction phase but a maximum of $30ls^{-1}$ is also included as Case D1. Groundwater contribution is not yet confirmed so a value of $5mgl^{-1}$ BOD (representing Good status classification of surface waters of specific types) and this together with relevant groundwater flow rates is taken account of for Case A, D and D1 to allow assessment.

## 5.7 Coliforms, enterococci – bathing water standards and shellfish

This assessment is based on bathing water regulations (2013. No. 1675) for coastal and transitional waters for which Good status requires that at the bathing water monitoring points the colony forming unit (cfu) counts for intestinal enterococci are ≤200 cfu/100ml and for Escherichia coli are ≤500 cfu/100ml. The nearest designated bathing waters are Southwold the Denes (latitude 52.32º N, longitude 1.679º E) and Felixstowe North (latitude 51.96º N, longitude 1.355º E) and are approximately 10km and 35km distant, respectively. To ensure that there is no impact on compliance at these locations it is therefore necessary to confirm that treatment and dilution of the sewage effluents produced during the construction period meets the required standard. An assessment of effluent treatment levels was considered to derive initial loadings of sewage microorganisms and then the CORMIX model was used to assess dilution of the discharge and this is described in section 6.

## 5.8 Tunnelling wastewater and chemicals

The offshore cooling water infrastructure consists of two subterranean intake tunnels and one outfall tunnel. Tunnels would be excavated by tunnel boring machines (TBMs) from land. Spoil from the cutting face of the TBMs would be removed by a screw conveyor, then transported by conveyor belt to the landward muck bay for licenced disposal.

Groundwater would be generated from digging the galleries allowing access to the tunnels. During the transport of spoil material, groundwater and TBM chemicals can leach from the conveyor belts and fall to the tunnel floor. Wastewater on the tunnel floor would be discharged via the CDO. Discharges would be treated with a silt-buster or similar technology to minimise sediment inputs.

The waste from the TBM soil conditioning chemicals if present is likely to make the largest contribution during Case E as two tunnel boring machines would be in operation and two volumes of makeup water containing conditioning chemicals would be discharged. This assumption is based on the work conducted at HPC. The total discharge volume during Case E is approximately 34ls$^{-1}$ of which ~6ls$^{-1}$ is contributed by soil conditioning water and chemicals. It is uncertain whether similar chemical use to that planned for HPC will occur during tunnelling for Sizewell C but representative worst case use and discharge scenarios are modelled based on HPC to allow assessment of the potential influence of discharges upon water quality at Sizewell.

Various chemicals may be required during the tunnelling process:

- fuelling and lubrication of the TBM;
- sealing the tunnel walls against water/soil ingress, and;
- ground conditioning.

Fuel and lubricants would be subject to management protocols and oil/chemical spills will be contained by appropriate treatment and disposal. Sealants and greases are impervious to water and will remain associated with the tunnel walls or be removed with the spoil.

The underlying geology at Sizewell differs from Hinkley Point and a bentonite slurry tunnelling method is anticipated at Sizewell. Bentonite is a rock formed of highly colloidal and plastic clays composed mainly of montmorillonite, a clay mineral and is regularly used in construction and offshore drilling operations. Bentonite is included on the OSPAR list of PLONOR substances (pose little or no risk to the environment). These substances do not normally need to be strongly regulated as, from assessment of their intrinsic properties, the OSPAR Commission considers that they pose little or no risk to the environment. Although during operation of TBMs bentonite recovery systems are used (as bentonite is a valuable resource in the

tunnelling process) the potential release into the receiving waters is assessed. The World Health Organisation (WHO, 2005) reviewed information on environmental properties of bentonite and describe several short-term studies (24 hours) on marine fish, crustacea and molluscs for which no mortality was observed at an exposure concentration of 7500mgl$^{-1}$ sodium bentonite (Daugherty, 1951). A value of 19000mgl$^{-1}$ was also recorded as a 96hour $LC_{50}$ for the rainbow trout a freshwater fish species (Sprague and Logan, 1979).

A bentonite concentration in the 6l$^{-1}$ volume per second of tunnelling wastewater of 50mgl$^{-1}$ is estimated (EDF, 2010). The total volume of wastewater including groundwater generated during tunnelling is estimated as 34.3 l$^{-1}$ per second and the resulting bentonite concentration would therefore be 8.8mgl$^{-1}$. There is no EQS established for bentonite so the 95th percentile and mean plume area is derived to determine extent of any potential influence on water quality based on the limited effects dataset.

In some TBM soil conditioning applications several different surfactant chemicals may be required. The use and discharge of two surfactant chemicals the anti- clogging agent BASF Rheosoil 143 and the soil conditioning additive CLB F5 M that are planned for use with the HPC tunnelling operation and that present higher risk quotients in terms of chemical properties are modelled for Sizewell. This approach has been taken to provide a representative upper bounding assessment of potential effects of discharges from this process. The active substances in the TBM chemical products were identified from respective material safety datasheets. The substances identified are surfactants from chemical groups commonly found in household detergent products for which there are a range of toxicity studies available. Based upon common elements of their chemical composition, Predicted No Effect Concentrations (PNEC) have been established for representative surfactants and these are applied. PNEC values shown in Table 12 for each active substance are either taken directly from relevant risk assessment reports i.e. for 2-methyl-2-4 pentanediol (SIDS initial assessment report, 2001), or use the lowest PNEC from a substance group assessment i.e. PNEC values calculated for other alcohol ethoxylate sulphates are derived for representative carbon chain length substance or worst case if not known (Table 15 in HERA, 2004,) and for mono-C10-16-alkyl sodium sulphate (Table 13 HERA 2002). The Effective Flux Volume (EVF) is calculated for all the active substances and the discharge assessment is based on the component within the greatest EVF equating to the product that is present in the highest quantities and/or the lowest PNEC. In the case of the anti- clogging agent BASF Rheosoil 143, the active substance is sodium lauryl ether sulphate and for soil conditioning-additive, CLB F5 M the active substances with the lowest PNEC is from the mono-C10-16-alkyl sodium sulphate group (Table 12).

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Table 12 'H1' assessment of example ground conditioning chemicals and their active substances. The initial screening result for subtidal discharges in Transitional and Coastal waters (TraC)Test 5 is provided. Chemicals failing the TraC Test are assessed in greater detail.

| Conditioning Product | Estimated discharge concentration of active substance (mg/l) | Saltwater AA EQS ($\mu gl^{-1}$) [1] | Background concentration (µgl-1) | Effective volume flux (Case E) Total flow 34.3 l/s | TraC Water test 5 EVF < 3.0 (Pass/Fail) |
|---|---|---|---|---|---|
| BASF Rheosoil 143 | 23.13 | 40 | 0 | 19.89 | Fail |
| CLB F5 M Ethoxylated sulphates | 7.71 | 35 | 0 | 7.58 | Fail |
| CLB F5 M Mono- alkyl sodium sulphate | 7.71 | 4.5 | 0 | 58.94 | Fail |

[1] These EQS values were derived from HERA (2004) for BASF Rheosoil 143 (sodium lauryl ether sulphate) and CLB F5 M (Ethoxylated, sulphates). [2] A group of compounds known as alkyl sulphates (AS) are found in CLB F5 M. Toxicity of AS compounds increases with increasing alkyl chain length (C12-C18), whilst solubility is inversely related to chain length. As such, C14 has the lowest reported NOEC values of the AS group. PNEC values for each AS chain-length have been established by applying a factor of 10 to the lowest chronic NOEC; the PNEC for C14 is 4.5 µg/l (HERA, 2002). The C14 PNEC is over 4-fold lower than the next most toxic chain-length AS, however, it has been applied as the EQS value as a precautionary measure. The PNEC values are conservative and values derived from micro- and mesocosm studies have identified PNECs in the range of 7.5 – 224 µg/l, and 110 µg/l, respectively (HERA, 2002; and references therein).

The estimated discharge concentration for each of the two conditioning products (three component surfactants) screened using the H1 methodology exceeded their respective EQS values in Test 1 and taking account of initial dilution are also predicted to exceed the maximum EVF value of 3.5 in Test 5. As these TBM values exceed the discharge test they are assessed in more detail using CORMIX modelling and this assessment is provided in the following section.

# 6 Review and assessment of potentially significant construction discharges

## 6.1 Background

Potential discharges to the marine environment have been assessed for each phase of the planned SZC development these are during construction, commissioning and operation. The H1 annex D1 2014 guidance (Environment Agency, 2014) is not considered to be entirely appropriate for the highly variable discharge volumes that occur during construction or to the large volume discharges associated with cooling water but is used here to provide an initial screening approach to identify chemicals that require more detailed assessment.

The main expected contaminants in construction discharges are suspended solids, BOD and hydrocarbons and are associated with the preparatory works and the main building erection and the presence of construction staff on site. The level of suspended solids and hydrocarbons in site drainage will be monitored and controlled within acceptable limits.

Measurement of groundwater contamination showed that total petroleum hydrocarbons, polycyclic aromatic hydrocarbons, volatile organic compounds, polychlorinated biphenyls and several physical factors were below reasonable levels of detection (Appendix A).

## 6.2 Discharge assessment methodology

The release and mixing of metals in the construction discharge was modelled using CORMIX US EPA supported mixing zone model (CORMIX Version 10.0GT HYDRO1 Version 10.0.1.0 April 2017) and the validated Sizewell curvilinear GETM model. CORMIX is used to predict the rate of chemical plume dilution and plume geometry from the Combined Discharge Outfall (CDO). The GETM model is a 3D hydrodynamic model with an inbuilt passive tracer to represent zinc and chromium. As a worst case, it was assumed that there was no loss of dissolved metals due to sediment absorption or biological uptake. Using these assumptions, concentrations can be scaled, as the modelled concentration was simply a function of dilution. The GETM model setup, calibration and validation are described in British Energy Estuarine & Marine Studies (BEEMS) Technical Report TR229. The surface is forced with re-analysed data from a meteorological model (ERA40 interim from ECMWF). The boundary conditions were forced by the Danish Maritime Safety Administration (DaMSA) operational forecasting models, as described in BEEMS Technical Report TR229. The proposed discharge is a low volume of groundwater, treated sewage effluent and tunnelling waste with concentrations of some contaminants exceeding EQS levels. The location and basic properties of the proposed discharge are shown in Table 13.

Table 13 CDO discharge scenarios during different phases (Case A-E) of construction at Sizewell C

| Discharge Characteristics | Value |
|---|---|
| Location OSBG | 647980 E 264340 N |
| Charted water depth (surface to bed) at discharge location | At least 4.0 m |
| Discharge flow | Varies with Case. |
| Discharge salinity | 1 PSU |

### 6.2.1 Modelling buoyant plume

In this study, the GETM model domain used a discrete grid with dimensions of 25m by 25m (at its finest resolution) and 21 vertical layers in a sigma co-ordinate system in which the layer thickness changed with water depth. The discharge flow for Case A ($124ls^{-1}$) was small compared with the total volume in the model grid cell, so to avoid excessive initial dilution, the discharge was made into the model surface layer, which is consistent with the results of the near field CORMIX modelling of a buoyant plume (see Figure 18, Appendix D).

It should be noted that in a buoyant plume with a discharge in an offshore location, unless mixing occurs, there will be no impact on seabed features. Consideration of the tidal cycle is useful in understanding the likely modes of impact. When the flood tide is at its strongest (with flow to the south), the discharge plume will initially be buoyant, and will then be advected in a narrow surface streak and mixed down. As mixing occurs the concentration within the streak will rapidly drop. At high water, near slack tide, a pool of the discharged water will form at the surface which will be advected northwards as the ebb tide increases.

## 6.3 Modelling metals discharges

Several metals were present, and these were assessed using tests 1 and 5. Initial checks using 95 percentile discharge concentrations measured in groundwater samples (ATKINS, 2014/16 data) showed that chromium exceeded its annual EQS and failed test 5 which assessed exceedance of the EQS following initial dilution. The zinc 95 percentile concentration in the discharge exceeded the mean EQS for zinc. It was not possible however to evaluate the zinc discharge using the initial dilution test 5 as the background concentration data for zinc indicate that it exceeds the EQS. Chromium and zinc were therefore taken forward for modelling assessment.
The mean background concentration of zinc in the environment is $15.12\mu gl^{-1}$ whilst the EQS is $6.8\mu gl^{-1}$. Since the background levels are in exceedance of the EQS level, the EQS cannot be used as the threshold value for the CORMIX modelling. The detection limit for zinc in seawater samples (BEEMS TR314) is $0.4\mu gl^{-1}$. Therefore, the threshold value for Zinc is set at $15.12+0.4 = 15.52\mu gl^{-1}$, which represents the limit at which zinc would no longer be detected above the background concentration.

The mean background concentration of chromium in the environment is $0.57\mu gl^{-1}$ (BEEMS TR314) whilst the EQS is $0.6\mu gl^{-1}$.

Both zinc and chromium were modelled for Case A ($124ls^{-1}$) with a source concentration of $17.5\mu gl^{-1}$and $18.45\mu gl^{-1}$, respectively. CORMIX shows that for zinc the outfall plume would no longer be detectable within 3m. For chromium the outfall plume would fall below the EQS within 25m.

CORMIX output data suggest an initial dilution, for both zinc and chromium, was 47-fold at 25m from the discharge (i.e. the same size as a single grid cell in GETM). GETM slightly under-predicts the initial dilution with the discharge volume of $124ls^{-1}$ entering the model surface layer. The total volume in the upper grid cell is approximately $120m^3$. GETM shows a 40-fold dilution in the first 25m, meaning the plume extends slightly further. For chromium the plume concentrations are low, when evaluated against a chromium background of $0.58\mu gl^{-1}$ there was no exceedance at the bed but a mean surface area of 5.49ha exceeded the mean EQS $0.6\mu gl^{-1}$.There was no exceedance at the bed for zinc and the total surface area for which the influence of the discharge plume would be detectable above background is 0.11ha, or 2 grid cells (BEEMS TR488).

Both CORMIX and GETM are conservative estimates as they do not include additional mixing and dilution due to waves.

## 6.4 Modelling un-ionised ammonia discharges

Ammoniacal nitrogen exists in both ionised and un-ionised form in the combined groundwater and sewage discharges from the construction site with the ratio of each determined by pH, temperature and salinity. Un-ionised ammonia is generally considered more toxic and has an annual average EQS of 21µgl$^{-1}$.
When a primarily freshwater effluent (groundwater + treated sewage) is discharged and mixes with seawater the total ammonia concentration is diluted eventually to background levels, the effluent becomes more saline, pH increases and starting temperature eventually becomes the same as the background seawater. At each stage of dilution, the proportion of un-ionised ammonia is determined by the total ammonia concentration, the pH, temperature and salinity. By deriving the un-ionised ammonia concentration at successive stages of increasing effluent dilution the point at which the effluent is mixed enough to be below the EQS of 21µgl$^{-1}$ (considering natural background) can be determined.

Discharge modelling can then be applied to derive the distance at which this required level of mixing is achieved such that the proportion of un-ionised ammonia plus background is below the EQS.
The initial total ammonia ($NH_4$-N) concentration derived by combining the groundwater and sewage sources was presented in (Table 11) and this value is applied in the EA un-ionised ammonia calculator along with the starting conditions of groundwater + sewage, pH, temperature and salinity. The output from this assessment provides the initial un-ionised ammonia concentration in the groundwater and sewage mix.

The physicochemical parameters required for the effluent/seawater mixing plot are provided in the start and finish components of the mixing relationship:

a. freshwater, of average pH (7.3) and 95th percentile of ammoniacal nitrogen (Atkins, 2014, 2016 and permit), and an average temperature of 11.43ºC (BEEMS TR131 Edition 2).

b. seawater, with a mean temperature of 11.43ºC, 50$^{th}$ percentile salinity (33.3) (BEEMS TR189) and the 50$^{th}$ percentile seawater pH (8.05) (BEMS TR189). The mean ammoniacal nitrogen in the sea water background was 11.38µgl$^{-1}$ as N (BEEMS TR314).

A mixing figure (Figure 9) was used to take account of changing physicochemical conditions as the mainly freshwater discharge from the CDO mixes with seawater and becomes fully saline.

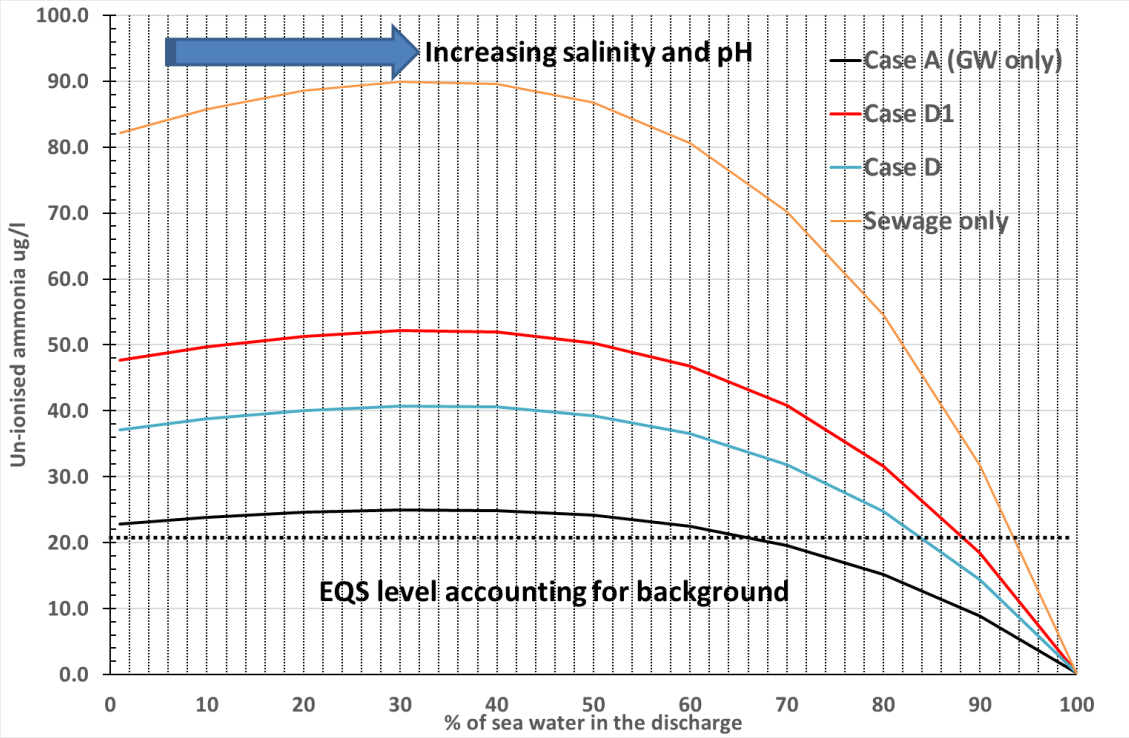


Figure 9. The change in proportion of un-ionised ammonia (µgl$^{-1}$) as the discharge, is mixed with seawater for sewage only, and cases Amax, D and D1

The associated change in the un-ionised ammonia concentration in the construction discharge relative to its annual average EQS was also assessed against the level of mixing. The calculations shown in Figure 9 are independent of the volume of the discharge, this graph therefore must be considered in combination with the estimated dilution rates derived from the CORMIX modelling. Case A, Case D1 and Sewage only discharges have been modelled with CORMIX. As Case D is a lower flowrate and source input, its impact will be lower, and was not modelled.

It is evident from Figure 9 that there is exceedance of the EQS (21µgl$^{-1}$) when less than 68% mixing has occurred for Case A, 84%mixing for Cases D, 88% for D1 and 94% for the sewage only case. In relation to Case A, it can be seen from Figure 19 (Appendix D) that a dilution factor of 2.13, (68% mixing) occurs after 3.67m for a discharge of 124ls$^{-1}$. Figure 20 (Appendix D) is relevant to case D1, showing that a dilution factor of 7.33 (88% mixing) occurs after approximately 3.89m. The sewage only case (Figure 21, Appendix D), which is unlikely to occur, would be compliant with a dilution factor of 15.67 (94% mixing). This dilution is likely to have occurred within 6.3m of the discharge.
In the marine environment the toxicity of ionised ammonia ($NH_4$) should be considered. In waters, particularly at higher salinities, it has been shown that the ammonium ion can also permeate the gills, and so the concentration of total ammonia $NH_4$ can also be toxicologically significant. Total ammonia values of 1100 (annual average) and 8000µg/l $NH_4$-N are therefore set as guide values for habitats (WQTAG086, 2005). The total ammonium concentration at the point of mixing described above is at background 11.38µg/l $NH_4$-N and well below levels of concern at mixing distance.

## 6.5 Assessment of total contributions to dissolved inorganic nitrogen (DIN) and phosphorus using a phytoplankton box model

The effect of chlorination at Sizewell B (SZB) and the proposed Sizewell C (SZC) on phytoplankton that pass through the power station was simulated with an emphasis on the spring bloom and summertime production using a phytoplankton box model. The combined loadings of nitrogen and phosphorus as previously

described (section 5.4 and 5.5) from the construction and cold commissioning inputs together with relevant inputs from SZB resulting from the use of conditioning chemicals and the discharge of treated sewage were assessed. For much of the year light availability limits phytoplankton growth and the addition of relatively small quantities of nutrients has no effect. In the summer, nitrate is a limiting nutrient (when light is not limiting) and is consumed rapidly. However, the exchange with the wider environment is much greater than the maximum proposed discharges, during construction, so that no change in phytoplankton growth beyond natural variability would be observed.
A model run over an annual cycle predicts a 0.13% difference in annual gross production (BEEMS TR385) of carbon and this level of change would not be discriminated above natural background variation.

## 6.6 Assessment of phosphorus loading

A total daily maximum phosphorus load of 28.2kg (daily maximum for sewage and groundwater added to maximum input from commissioning) was used as an input parameter combined with the DIN loads described in section 5.6 to run a phytoplankton growth model (Combined Phytoplankton and Macroalgae (CPM) model) (BEEMS TR385). A model run over an annual cycle showed an insignificant increase in carbon levels (phytoplankton biomass) of 0.13%.

## 6.7 Biochemical oxygen demand influence on dissolved oxygen

The background BOD near to the Sizewell B cooling water discharge based on monitoring done in 2010 (BEEMS Technical Report TR189) has a mean value of 2mg $l^{-1}$. Dissolved oxygen levels at the site are 'high' with a mean DO concentration of 7.5mg$l^{-1}$ (BEEMS Technical Report TR303) adjusted to an equivalent salinity of 35 this represents 6.27mg$l^{-1}$ (Water Framework Directive Standards and Classification Directions, 2015). The waters off Sizewell are well mixed vertically. Reduction of oxygen concentration will only occur if the rate of consumption due to BOD is greater than that which can be replenished by the daily exchange for the Greater Sizewell Bay and the oxygen transfer across the water surface.

Using 13.3l$s^{-1}$ and BOD of 40mg$l^{-1}$ and taking account of groundwater contributions a daily BOD of 121kg was estimated for Case D1 scenario, 64kg for Case D and 53kg for Case A. Every 1.5mg$l^{-1}$ BOD is estimated to result in 0.5mg$l^{-1}$ oxygen use (OSPAR Comprehensive studies report, 1997). Therefore, oxygen required to meet these BOD loadings would be D1 40.6kg/day, D 21.3kg/day and Case A 17.7kg/day. Daily water exchange for GSB is ca., 36 million $m^3$ so in relative terms the demand is very small. Typical values of oxygen flux are 100mmol $m^{-2}d^{-1}$ (Hull, 2016) or 3.2g$m^{-2}d^{-1}$. At a mean salinity adjusted background oxygen concentration of 6.27mg$l^{-1}$ a total of 40.6kg of oxygen would be transferred across 1.2ha in a day. Therefore, DO is likely to remain at high status. The discharges of BOD during construction are therefore considered to be of negligible significance for dissolved oxygen modification.

## 6.8 Modelling assessment of coliforms and intestinal enterococci

Based on data in support HPC development (pers. Comm., EDF), estimates were provided for maximum levels of faecal indicator organisms for the raw sewage input to the treatment plant (240 x $10^6$ E.coli and 13.6 x $10^6$ intestinal enterococci, Table 14). Secondary treatment implies a 100 factor (2 log) reduction in faecal indicator microorganisms, coliforms, and enterococci. If UV treatment is also applied a 5.4 log reduction is assumed.

Following either sewage treatment at a secondary or tertiary (UV) level the distance from the CDO discharge point at which enough dilution occurs to be below relevant microbiological standard levels, has been estimated using CORMIX for Case D (30l$s^{-1}$) sewage discharge and Case D1 (72l$s^{-1}$) (Table 14).

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The discharge plume from the CDO is buoyant and will be on the surface (Figure 18, Appendix D), but it should be noted that the CORMIX modelling does not include mixing due to waves and that mixing rates are most likely a significant under estimate as surface wave mixing will increase the mixing rate. Following either sewage treatment at a secondary or tertiary (UV) level the distance from the discharge point, at which enough dilution occurs to be below relevant microbiological standard levels, has been estimated using CORMIX for Case D ($30ls^{-1}$) sewage discharge and Case D1 ($72ls^{-1}$). These are shown in Figure 22 and Figure 23, in Appendix D. The discharge plume is buoyant and will be on the surface (Figure 18 Appendix D). CORMIX estimates show that the concentration of intestinal Enterococci cells are likely to exceed the bathing water standard only within 66m of the discharge for the $30ls^{-1}$ case, without UV treatment. For the larger discharge volume ($72ls^{-1}$) the bathing water standards are exceeded for 460m. With UV treatment, even at the higher discharge volume, exceedance is limited to less than 1 metre of the discharge. Typically, the sewage discharge may not be discharged on its own, but as part of other discharges, these other discharges will add direct dilution which compensates for the inhibition of mixing. The discharge has been modelled using the total volume although the sewage component is only a percentage of this therefore the assessment is conservative. The discharge point is not in designated bathing waters. Treatment from the plant is sufficient to ensure that *E.coli* concentrations in discharged waters comply with bathing water standards within a maximum of 3.1km from the discharge point (without UV treatment) and <1m (with UV treatment) (Table 14). The nearest designated bathing waters are Southwold the Denes (latitude 52.32º N, longitude 1.679º E) and Felixstowe North (latitude 51.96º N, longitude 1.355º E) and are approximately 10km and 35km distant, respectively. This assessment is based on bathing water regulations (2013. No. 1675) for coastal and transitional waters for which Good status requires that the colony forming unit (cfu) counts for intestinal enterococci are ≤200 cfu/100ml and for *Escherichia coli* are ≤500 cfu/100ml.

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Table 14 Estimate of minimum distance from point of discharge at which microbiological standards for bathing waters are met following different levels of sewage treatment

| Species | Standard cells/ 100ml | Discharge concentration cells / 100ml | 2ndry treatment 2 log reduction | Dilution required to meet bathing water standard | Maximum potential distance from the discharge at which meets bathing water standard $30ls^{-1}$ | $72ls^{-1}$ | UV treatment reduction[1] | Dilution factor required for discharge to meet bathing water standard | Maximum distance from the discharge at which it meets bathing water standard |
|---|---|---|---|---|---|---|---|---|---|
| *E.coli* | 500 | 240,000,000 | 2400000 | 4800 | ~1.7 km | ~3.1 km | 955.5 | 1.9 | <1 m pass immediately on discharge, for both cases. |
| Entero -cocci | 200 | 13,600,000 | 136000 | 680 | ~66 m | ~460 m | 54.1 | 0.3 | <1 m pass immediately on discharge, for both cases. |

[1]a log 5.4 reduction is achieved by UV treatment for *E. Coli* and a log 4.4 reduction for enterococci, assuming background concentrations are zero.

### 6.9 Modelling use and potential discharge of chemicals during tunnelling

As with the groundwater metals, the release and mixing of TBM chemicals in the construction discharge was modelled by considering them as passive tracers (no decay rate). As such, a single model run was carried out with single tracer at a release rate of 34.3 $ls^{-1}$ with an initial concentration of 100 $\mu gl^{-1}$. The results were then scaled to the appropriate concentrations for each chemical, as the modelled concentration was simply a function of dilution. The discharge was modelled as a freshwater input with no thermal uplift. Table 15 shows the model parameters used for TBM chemical modelling.

Table 15 GETM TBM modelling run parameters.

| Run ID | Description | Intake location | Discharge location | Metals discharge at the outfall (µg/l) | Discharge flow and Delta T ($m^3/s$ @ °C) | Time period |
|---|---|---|---|---|---|---|
| Sewage_construction-100ug | Discharge from SZB, with TBM discharge from SZC CDO | IB | OB | 0 | 51.5 @ 11.0 | /5/2009-1/6/2009 |
| | | None | CDO | 100 | 0.034@0 | |

A tunnelling discharge of bentonite at a concentration of 8.8$mgl^{-1}$ was modelled using GETM and the results are shown in Table 16. The concentration of bentonite in suspension is orders of magnitude lower than baseline suspended sediments concentrations predicted during construction (BEEMS TR480), with 95th percentile concentrations of 10$\mu gl^{-1}$ restricted to sea surface areas of 10.8ha and at mean concentration

1.35ha at the surface. No areas of the bed were affected at this concentration. Limited data on survival of organisms exposed to bentonite suspensions indicate that the small areas affected, and the low discharge concentrations are likely to have negligible effects on water quality.

Table 16: Area of the plume at different concentration levels of bentonite, with an 8.8mgl$^{-1}$ release concentration.

| Release Concentration | µg/l | Mean surface (ha) | Mean seabed (ha) | 95th percentile surface (ha) | 95th percentile seabed (ha) |
|---|---|---|---|---|---|
| 8.8 mgl$^{-1}$ | 2 | 19.06 | 0.90 | 235.7 | 170.0 |
| | 4 | 5.16 | 0 | 36.8 | 1.01 |
| | 6 | 2.47 | 0 | 19.7 | 0 |
| | 8 | 1.91 | 0 | 13.6 | 0 |
| | 10 | 1.35 | 0 | 10.8 | 0 |

For the soil conditioning chemical discharges, the total Rheosoil plume areas at the EQS (40 µgl$^{-1}$ as a mean and 95th percentile) were calculated and are shown in Table 17. There is a small area of exceedance at the surface 1.01ha and no exceedance at the bed for a mean assessment. There was no exceedance of the EQS for CLB F5 M at the seabed and the area at the surface exceeding the EQS were small (Table 18) with 3.14ha for a mean assessment.

Table 17: Area of the plume at different concentration levels of Rheosoil, with a 23.13 mgl$^{-1}$ release concentration. Values in bold exceed the EQS concentration.

| Release Concentration | EQS | µg/l | Mean surface (ha) | Mean seabed (ha) | 95th percentile surface (ha) | 95th percentile seabed (ha) |
|---|---|---|---|---|---|---|
| 23.13 mgl$^{-1}$ | 40 µgl$^{-1}$ | 5 | 22.20 | 1.91 | 321.73 | 224.61 |
| | | 10 | 5.49 | 0 | 39.25 | 1.68 |
| | | 20 | 1.91 | 0 | 14.24 | 0 |
| | | 30 | 1.35 | 0 | 8.63 | 0 |
| | | **40** | **1.01** | **0** | **5.83** | **0** |

Table 18: Area of the plume at different concentration levels of CLB F5 M, with a 7.71 mgl$^{-1}$ release concentration. Valued in bold exceed the EQS concentration.

| Release Concentration | EQS | µg/l | Mean surface (ha) | Mean seabed (ha) | 95th percentile surface (ha) | 95th percentile seabed (ha) |
|---|---|---|---|---|---|---|
| 7.71 mgl$^{-1}$ | 4.5 µgl$^{-1}$ | 1 | 119.98 | 58.54 | 1605.04 | 1386.37 |
| | | 2 | 13.91 | 0 | 132.88 | 80.20 |
| | | 3 | 6.17 | 0 | 45.08 | 5.27 |
| | | 4 | 3.81 | 0 | 30.39 | 0 |
| | | **4.5** | **3.14** | **0** | **25.01** | **0** |

The most toxic of the active ingredients for BASF Rheosoil 143 Sodium lauryl ether sulphate was modelled for the tunnelling discharge and is an example of an alcohol ethoxysulphate. Although tunnelling would occur over several years only very small areas at the surface are predicted to exceed the EQS for Rheosoil and

this group of surfactants are shown to be readily degradable with no indication for the formation of persistent or markedly toxic metabolites (HERA, 2004). The most toxic active component of CLB F5 M, mono- alkyl sodium sulphate is an example of an alkyl sulphate and experimental and field data also indicate this group to be readily degradable (HERA, 2002).

# 7 Screening assessment of commissioning discharges

## 7.1 Background

When the cooling water system is commissioned a range of tests will be conducted and conditioning of the entire plant will be undertaken with demineralised water and various chemical additives. During cold commissioning this process will generate wastewater containing several chemicals that will be discharged through the CDO.

During the commissioning phase the range of expected wastewater sources are:

- Surface drainage from across the site (assuming these discharges are made to local marine waters as part of the drainage strategy);
- Drainage from on-site purification plants (assuming these discharges are made to the marine environment as part of the site drainage strategy);
- Effluent from the demineralisation plant;
- Chemicals used for the conditioning of a range of circuits within the EPR units;

## 7.2 Screening assessment of Cold Flush Testing (CFT) of SZC Unit 1

No operational cooling system will be available for the disposal and dilution of commissioning phase effluents during the cold flush testing for construction of the first EPR. Therefore, the only available discharge route for this wastewater stream will be through the CDO.

Testing of the primary and secondary circuits requires them to be filled and flushed several times each with demineralised water and treatment chemicals. The maximum daily discharge volume is $1500m^3d^{-1}$, equivalent to the contents of the two $750m^3$ tanks that serve this waste stream. NNB GenCo proposes to empty each tank once a day, although not at the same time. No operational cooling system will be available for the disposal and dilution of commissioning phase effluents during the cold flush testing (CFT) stage for the first unit to be constructed during the phased development of the SZC site. Therefore, the only available discharge route for this wastewater stream will be through the CDO. If there is overlap in the period when each EPR is being commissioned this would increase discharge duration and load, but discharge concentration may be similar.

Cold flush testing involves cleansing and flushing the various plant systems with demineralised water to remove surface deposits and residual debris from the installation. NNB GenCo's intention would be for CFT effluent to be discharged to the Sizewell Bay via the CDO serving the SZC construction site. The discharges resulting from CFT will be subject to a separate, later water discharge activity permit application.

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### 7.3 Screening assessment of Cold Flush Testing (CFT) of SZC Unit 1

One of the chemicals that could potentially be included in the commissioning tests is hydrazine. Hydrazine is an oxygen scavenger and is used in power plants to inhibit corrosion in steam generation circuits. There is evidence that hydrazine is harmful to aquatic organisms at low concentrations with the lowest acute six-day $EC_{50}$ of 0.4ngl$^{-1}$ for growth inhibition of a marine alga, *Dunaliella tertiolecta* (see Appendix B for PNEC evaluation for hydrazine). Hydrazine persistence in the marine environment is low to moderate dependent upon its concentration and the water quality. There is no established EQS for hydrazine and so a chronic PNEC (Predicted No-Effect Concentration) of 0.4ngl$^{-1}$ has been calculated for long term discharges (expressed as a mean concentration value) and an acute PNEC of 4 ngl$^{-1}$ for short term discharges (expressed as a 95$^{th}$ percentile value). More recent assessments used in support of Canadian Federal Water Quality Guidelines for hydrazine indicate concentrations below 0.2µgl$^{-1}$ have a low probability of adverse effects for marine life, whilst a freshwater threshold of 2.6µgl$^{-1}$ is applied based on a greater availability of data in the freshwater environment (Environment Canada, 2013). Based on derivation of the more recent Canadian guidelines the lower chronic and acute PNEC derivations (0.4 and 4ngl$^{-1}$) are considered as precautionary triggers for further ecological investigation.

Prior to the release of hydrazine from the holding tanks, hydrazine would be treated to reduce the discharge concentration. Various treatment options are under investigation and it is anticipated that a discharge concentration of 15µgl$^{-1}$ is achievable as a representative upper bounding concentration equivalent to a 95$^{th}$ percentile. As a discharge concentration of 15µgl$^{-1}$ exceeds the EQS and fails the Test 5 dilution test this discharge concentration is modelled using GETM.

During commissioning two other chemical discharges for which use is anticipated are the circuit conditioning chemicals ethanolamine (at 4000µgl$^{-1}$) and ammonia principally in the un-ionised form (at 12000µgl$^{-1}$) due to the higher pH (ca.,10) maintained in the circuits. The same commissioning discharge volume and rate of discharge as that used for hydrazine are assumed. The expected concentrations used during commissioning were assessed using H1 test 1 and 5 (Table 19). Only the un-ionised ammonia fails Test 5 and requires further modelling assessment.

Table 19: H1 Test 1 and 5 for discharges of ethanolamine and un-ionised ammonia during commissioning.

| **Substance** | **Estimated discharge concentration µgl$^{-1}$** | **Saltwater AA EQS µgl$^{-1}$** | **Background concentration µgl$^{-1}$** | Effective volume flux Total flow 83.3 l/s | TraC Water test 5 EVF < 3.0 (Pass/Fail) |
|---|---|---|---|---|---|
| Ethanolamine | 4000 | 160 | - | 2.08 | Pass |
| Un-ionised ammonia | 12000 | 21 | 0.2 | 47.6 | Fail |

### 7.4 Hot functional testing

Hot functional testing begins following completion of CFT and when all the required systems are available. It takes place before fuelling the reactor and only once the cooling water infrastructure is in place and operational. The objective of HFT is to test the reactor and associated systems under pressure, temperature, flow and chemical conditioning as close to normal operating conditions as practicable without putting nuclear fuel at risk. The effluent produced during HFT would be diluted within the cooling water system before being discharged via the outfall tunnel to the adjacent marine environment.

Due to the current stage of the project and the long lead time until commissioning takes place, detailed information on the nature of the discharges during HFT is limited, but it is assumed that HFT can be considered as running the systems under normal operating conditions. It would therefore be expected that the assessment for operational discharges would also apply to that during HFT.

## 7.5 Chlorination system testing

Testing of the chlorination system will be undertaken during the commissioning phase, but it is assumed that this would only occur once the full cooling water system was in place and operational.

# 8 Impact assessment of commissioning discharges

## 8.1 Background

Modelling of the discharges from the commissioning of the EPRs when the cooling water system is unavailable assumes a maximum discharge rate of 83.3$l^{-1}$ per second from a total holding volume of 1500$m^3$ (two tanks of 750$m^3$).

The modelling has been undertaken using the validated GETM model of Sizewell that was used for thermal plume and chemical plume studies previously described in BEEMS Technical Report TR302 and TR303. The model was chosen to support the chemical runs because it is better able to reproduce the natural variability due to meteorological and tidal conditions.

The commissioning discharge is from the CDO, the salinity of the discharge is modelled as freshwater with no thermal uplift and discharge location parameters are as described for the construction discharge (further detail is provided in BEEMS TR494).

For the commissioning release of hydrazine, a release concentration of 15$\mu gl^{-1}$ released in daily pulses of 5.0 h starting at 12:00. This discharge period is enough to empty the total volume of both treatment tanks 1,500 $m^3$. The simulation was carried out for a period of 1 month (1$^{st}$ of May 2009-31$^{st}$ of May 2009), to encompass a full spring neap cycle. This is the same hydrodynamic GETM results that were used to investigate the effect of chlorination of the cooling water system to deter biofouling, as reported in BEEMS Technical Report TR303. The month of May was chosen due to having the highest phytoplankton growth which drives the whole marine ecosystem. Due to the pulse-like discharge, the interpretation of the short-term results (daily) is biased to the moment of the tidal cycle when hydrazine has been released. In order to evaluate this effect, an additional simulation was carried out with the release pulses starting at 18:00.

## 8.2 Hydrazine commissioning assessment

The initial assessment to determine the potential for an effect on water quality considers total areas of exceedance with results shown in Table 20.

To investigate the potential interaction of the hydrazine discharge concentration with relevant environmental sensitivities the results for assessment against the acute and chronic PNEC are compared against three criteria:

(a) The likelihood that hydrazine could enter the Minsmere Sluice;

(b) Levels of hydrazine at the seabed over the Coralline Crag;

(c) The area of intersection of the acute hydrazine plume with Little Tern foraging areas.

The Minsmere sluice controls the seawater that can flow into various drainage channels including those used to periodically supply a saline input to the Minsmere salt marshes. The sluice only opens for half an hour at high tide. Due to the proximity of the Minsmere sluice to the SZC construction discharge location an investigation was undertaken to determine whether the hydrazine plume could intersect with the sluice at concentrations above the chronic or acute PNECs and, if so, when. This is relevant, because if high hydrazine concentrations occur at times when the sluice is open, then it could enter channels that are used

periodically used to introduce saline water into the RSPB Minsmere reserve. The potential to influence movement of Eels into or out of the saltmarshes is also considered.

The Coralline Crag is a geological formation of special ecological interest in the area of Aldeburgh and Orford (Suffolk). Coralline Crag is a rock formation formed of bryozoan and mollusc microfossil debris that protrudes from the seabed. *Sabellaria spinulosa* has been found on the Coralline Crag and work has been conducted to determine whether the *Sabellaria* has formed reefs as defined under Annex I of The Habitats Directive (European Commission Council Directive EEC/92/43 on the Conservation of Natural Habitats and of Wild Fauna and Flora) (BEEMS Technical Report TR473). *Sabellaria* reefs are listed as a marine habitat to be protected for their role in harbouring diversity and the proximity to the SZC construction discharge site makes it susceptible to being exposed to hydrazine discharges.

Within 20 km of SZC are several nationally and internationally designated protected areas for flora and fauna, one of which is the Minsmere to Walberwick Special Area of Protection (SPA). Minsmere SPA is designated for the protection of several breeding, wintering and passage bird populations of European importance, including little tern (*Sterna albifrons*). Little terns feed by fishing in the top few centimetres of water column (del Hoyo *et al*., 1996) and have a limited foraging range of 2.4km offshore and 3.9km north and south (Parsons *et al*., 2015). Little terns are therefore potentially vulnerable to the impact of hydrazine if the hydrazine plume acts to deter fish from entering the little tern foraging area.

To assess the spatial extent of the hydrazine plume and compare the resulting concentrations with the PNEC values (chronic and acute), the mean and 95$^{th}$ percentile of the hydrazine concentrations was extracted from the 31-day model run. For hydrazine the chronic PNEC value is 0.4ngl$^{-1}$ for long term discharges (mean of the concentration values) and the acute PNEC value is 4ngl$^{-1}$ for shorter term discharges (represented by the 95$^{th}$ percentile).

The 95$^{th}$ percentile results show that the plume at the surface is shorter and thinner than the mean plume. The plume at the seabed shows a similar elongated narrow plume (BEEMS TR494). Table 20 provides a summary of the area of the plume that exceeds both concentration thresholds. For completeness, not only the chronic and acute PNEC values were included, but also other values between 0.1 and 0.5ngl$^{-1}$ for the chronic concentrations and between 1 and 5ngl$^{-1}$ for the acute concentrations. In addition to the two PNEC values considered in this report, the area exceeding 200ngl$^{-1}$ as a 95$^{th}$ percentile, as set by the Canadian Federal Water Quality Guidelines for hydrazine, have been included in Table 20.

The area exceeding the derived acute and chronic PNECs is less at the bed than the surface. At the surface ~27 and ~54ha exceed the acute and chronic PNEC respectively. At the surface the exceedance for the 200ngl$^{-1}$ Canadian standard is 0.34ha, which represents three model grid cells (25 x 25 m) around and including the hydrazine discharge from the CDO.

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Table 20: Area of the plume at different concentration levels of hydrazine, with a 15 µgl$^{-1}$ release concentration. Valued in bold exceed the respective PNEC concentrations.

| Release Concentration | | ng/l | 95$^{th}$ percentile surface (ha) | 95$^{th}$ percentile seabed (ha) | Mean surface (ha) | Mean seabed (ha) |
|---|---|---|---|---|---|---|
| 5:00h release 15µgl$^{-1}$ **at 83.3l/sec** | Chronic PNEC | 0.1 | | | 93.19 | 22.32 |
| | | 0.2 | | | 53.60 | 11.33 |
| | | 0.3 | | | 39.47 | 5.83 |
| | | **0.4** | | | **30.50** | **2.92** |
| | | **0.5** | | | 25.57 | 1.79 |
| | Acute PNEC | 1 | 52.03 | 21.53 | | |
| | | 2 | 27.36 | 10.99 | | |
| | | 3 | 18.17 | 6.17 | | |
| | | **4** | **12.90** | **2.92** | | |
| | | **5** | 10.54 | 1.57 | | |
| | Canadian Standard | **200** | **0.34** | **0.00** | | |

As the hydrazine chemical plume lies completely inside the Outer Thames Estuary SPA and the Suffolk Coastal waterbody, the areas of exceedance for the chronic and acute PNECs are the same as for the whole plume.

The hydrazine concentration at the surface, over the 31 days was modelled at the node closest to the location of the Minsmere sluice for the 15µgl$^{-1}$ release concentration in pulses of 5.0 h starting at 12:00, and the other starting at 18:00. Table 21 provides a summary of the peak concentrations. At no time are the concentrations above the chronic PNEC present. Concentrations of 0.07ngl$^{-1}$ can be found at the location of the Minsmere Sluice at the surface when hydrazine is released at 12:00 with a release concentration of 15µgl$^{-1}$. In all the cases, the plume does not stay in the vicinity of the sluice from after one high tide to the next (approximately 12h later). Since the Minsmere sluice only opens for half an hour after high tide, this means that the hydrazine plume does not coincide in time with the sluice opening. The hydrazine plume is transported northward towards Minsmere during the falling tide, meaning that the sluice water supply that is periodically used to add additional saltwater to the Minsmere salt marshes is unlikely to be exposed to hydrazine. The likelihood of any hydrazine exposure in the sluice water would also be made considerably less likely due to rapid degradation of hydrazine with a half-life of ca., 30 minutes.

Table 21: Summary of peak hydrazine concentrations at Minsmere sluice, time of occurrence and duration of the plume.

| Hydrazine Simulation results | | | Peak concentration (ng/l) | Time after high water (hours) | Duration above Chronic PNEC (hours) | Duration above Acute PNEC (hours) |
|---|---|---|---|---|---|---|
| 15µgl$^{-1}$ | Release at 12h | Surface | 0.07 | 0.0 | 0.0 | 0.07 |
| | | Bottom | 0.07 | 0.0 | 0.0 | 0.07 |
| | Release at 18h | Surface | 0.12 | 0.0 | 0.0 | 0.12 |
| | | Bottom | 0.11 | 0.0 | 0.0 | 0.11 |

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The potential for hydrazine concentrations in proximity to the Minsmere sluice to deter passage of Eels (which are a prey item for several bird species) into or out from the sluice is also considered. The European eel is catadromous, breeding in a specific region of the North Atlantic (thought to be the Sargasso Sea) (Wheeler, 1969; Arnold, J. D. and McCleave, 2002; van Ginneken and Maes, 2005). After hatching the young larvae (known as leptocephali) are transported back to European coasts by ocean currents (Wheeler, 1969; van Ginneken and Maes, 2005). As they approach the continental shelf, the leptocephali become glass eels, then transition into elvers. In the North Sea, elvers ascend rivers in March and April after metamorphosis, although some may remain in estuaries or coastal waters (Tzeng et al., 1997). Little is known about the residence times of glass eels in the southern North Sea. The eels reach the coast and seek a salinity cue to transition from oceanic waters to coastal ones, so the time spent in the open North Sea is dependent on when they sense this cue. Once in freshwater, the eels spend many years growing and feeding and after reaching a specific size range begin migrating out to sea.

There is limited data on the toxicity of hydrazine to marine fish, however, freshwater examples indicate the most sensitive species have a 96h LC50 value of 610µg/l (Environment Canada, 2013). This acute toxicity threshold is orders of magnitude higher than the source concentration from the proposed CDO commissioning discharge. There is the potential concern for sublethal effects particularly in relation to migratory eels. In the UK glass eels enter river systems from the sea in March and April whilst yellow eels migrate from the rivers back to sea in September to December. Commissioning discharges could coincide with the period of eel migration, as such the concentration of hydrazine at the Minsmere sluice (the closest entry point to freshwater from the CDO) was investigated. There is a paucity of data on sublethal effects of hydrazine on fish. One study identified evidence of behavioural responses including an increase in aggressive behaviours in laboratory trials with freshwater bluegill (*Lepomis macrochirus*), which the authors attributed to the irritant effects of hydrazine (Fisher *et al.* 1980). However, behavioural responses occurred at concentrations of 0.1mg/l and above, still considerably higher than the instantaneous maximum concentration at the sluice (0.12ng/l). Furthermore, hydrazine plumes would only intersect the sluice during an ebbing tide when water levels would be falling. As such, glass eels entering the sluice at high water are unlikely to be exposed. The predicted peak concentrations of hydrazine in proximity to the sluice in any case are many times below levels shown to cause sublethal effects in fish so Eels moving to or from the saltmarshes in the vicinity of the sluice would also not be exposed to significant concentrations of hydrazine. The hydrazine discharge plume has a low likelihood of affecting Eel migration along the coast since at the point of discharge the maximum concentration is 15µg/l (many times below known sublethal levels) and also degrades rapidly. It is therefore considered unlikely that commissioning discharges of hydrazine would affect eel migration given the low peak concentration relative to known effects levels and limited potential for exposure.

A similar assessment was conducted for the Coralline Crag. The model results show that at a 15µgl$^{-1}$ release concentration the chronic PNEC is not exceeded at the seabed and only for 0.25 h at the surface. But *Sabellaria* (as a benthic feature) would not be exposed to acute concentrations. Table 22 provides a summary of the peak concentration of hydrazine and the duration of the plume above the chronic PNEC. The peak concentration of hydrazine at the seabed, with a 15 µgl$^{-1}$ release concentration, is (0.05ngl$^{-1}$) and is below the acute and chronic PNEC.

Table 22: Summary of peak hydrazine concentrations at the Coralline Crag, time of occurrence and duration of the plume.

| Hydrazine Simulation results | | | Peak concentration (ng/l) | Duration above Chronic PNEC (hours) | Duration above Acute PNEC (hours) |
|---|---|---|---|---|---|
| 15µg/l | Release at 12h | Surface | 0.06 | 0.0 | 0.0 |
| | | Bottom | 0.05 | 0.0 | 0.0 |
| | Release at 18h | Surface | 0.46 | 0.25 | 0.0 |
| | | Bottom | 0.04 | 0.0 | 0.0 |

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In the Greater Sizewell Bay, there are three breeding colonies of little terns at Dingle, Minsmere and Slaughden (BEEMS Technical Report TR431). Little Terns have a foraging range of 2.4km offshore and 3.9 km north and south (Parsons *et al*., 2015). Results show that only the Minsmere colony is potentially affected by the hydrazine plume. The hydrazine plume never intersects with the Dingle colony to the north and the Slaughden colony to the south. At a release concentration of 15µgl$^{-1}$, the instantaneous area of intersection between the hydrazine plume and the foraging area of the Minsmere colony represents a peak of ca., 2.6% of the colony foraging area for the 12:00 and 18:00 releases (this assessment is based on the more precautionary PNECs not those derived under the Canadian standards).

Table 23 provides a summary of the peak area intersection with Minsmere little tern colony and the duration of the plume. Whilst the plume intersection with 15µgl$^{-1}$ release concentration regularly exceeds 1% of the foraging area, the duration of the plume is short, with concentrations exceeding the acute PNEC for no longer than ca., 4 hours.

It is possible during EPR commissioning that one EPR is operational and discharging via the cooling water system whilst the second is commissioned and so this is given further consideration in the operation assessment.

Table 23: Summary of peak hydrazine intersections with Minsmere little tern colony and the duration of the plume greater than the acute PNEC 4ng/l.

| Release concentration | Hydrazine release | Mean area intersection (%) | Peak area intersection (%) | Duration above 1% threshold (hours) | Duration above 1% threshold (% of day) | Number of acute exceedances per month |
|---|---|---|---|---|---|---|
| 15µg/l | Release 12:00h | 0.12 | 2.56 | 0.25 – 3.5 | 1.0 – 14.6 | 12 |
| | Release 18:00h | 0.12 | 2.02 | 0.25 – 2.0 | 1.0 – 8.3 | 16 |

Un-ionised ammonia commissioning assessment
The discharge of un-ionised ammonia during the commissioning phase of the EPR construction was modelled using the validated GETM model of Sizewell. The maximum ammoniacal nitrogen (N) concentration discharged during commissioning the turbine hall equipment and steam generator is expressed as 12000µgl$^{-1}$ with a system ca., pH 10 within a commissioning discharge of 83.3 l$^{-1}$ sec$^{-1}$. At this pH a total ammonium concentration will partition as 17,806µgl$^{-1}$ $NH_4$-N and 12000µgl$^{-1}$ $NH_3$-N. This ammonia concentration and the physicochemical conditions of the EPR commissioning demineralised water provide the initial point for construction of a dilution plot as the effluent mixes with seawater. Successive stages of $NH_4$-N dilution and decreasing pH from 10 to a seawater background of 8.05, and increasing salinity to that of seawater (33.3) are used to derive equivalent un-ionised ammonia $NH_3$-N at each stage of dilution using the Environment Agency provided calculator (Clegg and Whitfield,1995). Based on this dilution 94.4 mixing of the commissioning discharge is required to achieve an $NH_3$-N concentration less than the EQS of 21µgl$^{-1}$. (also accounting for a $NH_3$-N background of 0.2µgl$^{-1}$).

For the GETM model, an initial concentration of 12,000µgl$^{-1}$ assumed as $NH_3$-N was discharged at 83.3 l$^{-1}$ sec$^{-1}$ from the CDO location with a freshwater salinity. To avoid excessive initial dilution, the discharge is released in the surface layer of the model, as was done with other assessed chemical discharges (BEEMS Technical Report TR303). A mixing level of 94.4% is enough together with the changing pH, and salinity that occurs as the wastewater mixes with seawater to reduce the un-ionised ammonia below its EQS and is equivalent to a 16.8-fold dilution.

The modelling results from GETM show there is no plume in exceedance of the EQS for the un-ionised ammonia. In the direct vicinity of the outfall (<5m) the un-ionised ammonia of the discharge will exceed the EQS. But this behaviour is smaller than the model grid cell size (25m). By the time the discharge has got to the boundary of the initial grid cell, mixing would have reduced the plume such that the EQS is not

exceeded. Comparisons against previous nearfield modelling using CORMIX suggest a 16.8-fold dilution is achieved within approximately 10m. Therefore, the GETM model was unlikely to produce a plume. The maximum concentration at the surface and seabed is 50µgl$^{-1}$ and 1.39µgl$^{-1}$ $NH_3$-N, respectively. This represents the minimum value of the grid cell of discharge, an area of 25 m x 25 m. While this can be considered as a potential underestimate of the concentrations at the immediate point of discharge (i.e. <5 m), however, it demonstrates that exceedance of EQS would be highly spatially restricted to the area close to the immediate point of discharge. As for the construction discharge assessment the ammonia concentration at the point of mixing described above is at background 11.38µg/l $NH_4$-N and well below levels of concern (WQTAG086, 2005) at mixing distance.

# 9 Assessment of the source input data for operational discharges

## 9.1 Background

Expected discharges to local marine waters from SZC during the operational phase may be broadly characterised as:

- Surface drainage from across the developed site;
- Sanitary wastewater from on-site purification plants;
- Effluent from demineralisation plant;
- Chemicals discharged during the operation of the units; and
- Discharges associated with chlorination.

The data for chemical discharges during the operational phase are mainly provided as maximum loading rates over annual and 24-hour periods for most chemicals within the waste water effluent. Source term calculations for nitrogen and hydrazine which are included in the chemical discharges to the marine environment during the operational phase are discussed in the following sections.

## 9.2 Ammoniacal nitrogen load derivation

For the operational phase, SZCs nitrogen discharges several sources and waste streams are considered. The un-ionised ammonia figures in Table 24 were calculated using the Environment Agency calculator (Clegg and Whitfield, 1995) which requires input data for temperature, salinity, pH and total ammonia and takes account of typical (annual average) and worst-case (24 hour) temperature uplift. All these source physicochemical data were specific to the Sizewell site. The data recorded during the 2010 monitoring survey at Sizewell (BEEMS TR189) were the reference source for the relevant physicochemical data used to derive un-ionised ammonia values for screening. For the annual assessment a 98$^{th}$ percentile temperature value (19.4$^{\circ}$C) a 50$^{th}$ percentile pH (8.02) and the 50$^{th}$ percentile salinity 33.3 were used to calculate un-ionised ammonia concentration. These values together with the typical uplift of 11.6$^{\circ}$C for the cooling water from SZC (BEEMS TR302) provided the input parameters for the Environment Agency calculator together with the total ammonia concentration to derive the maximum annual loading of un-ionised ammonia. In a worst-case scenario when 2 out of 4 pumps are under maintenance the flow of cooling water would be halved but the heat content of 2 full power reactors would remain approximately the same raising the excess temperature at the outfall from 11.6$^{\circ}$C to 23.2$^{\circ}$C (BEEMS TR303). Hence a value of 23.2$^{\circ}$C with the 98$^{th}$ percentile temperature (19.4$^{\circ}$C), 95$^{th}$ percentile pH (8.2) and 5$^{th}$ percentile salinity (31.7) was used to derive the maximum 24h loading for un-ionised ammonia. Very similar summary statistics for physicochemical parameters were derived from a more recent monitoring survey reported in TR314 but as the differences in the datasets were not large and modelling was developed around the earlier dataset and the scenarios provide a precautionary assessment it was not considered necessary to re-run this modelling using slightly updated values

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Table 24 Operational phase chemical discharges of nitrogen from sum of waste streams for 2 EPR units (based on EDECME120678 PREL A, 2011 and adapted using input data from TR131 ED 2, TR314 and TR303)

| **Substance** | Maximum annual loading (kg $yr^{-1}$) | Maximum 24-hour loading (kg $d^{-1}$) |
|---|---|---|
| Nitrogen (as N) (excluding hydrazine, morpholine and ethanolamine) | 10130[1] | 332 |
| Nitrogen (in terms of ammonia ions $NH_4$ excluding hydrazine, morpholine and ethanolamine) | 13009 | 77[1] |
| Nitrogen (in terms of Un-ionised ammonia $NH_3$ | 958[2] | 27 |

1 nitrogen is potentially contributed by hydrazine, ethanolamine and morpholine and consideration is given to this in section 11.6 2 figures are back calculated from the un-ionised ammonia concentration derived from the un-ionised ammonia calculator using the $NH_4$ concentration that results from the combined sanitary and conditioning inputs and site background physicochemical data (see Table 26)

## 9.3 Hydrazine load derivation

The main operational waste streams that potentially contribute to discharges of hydrazine are shown in Table 25. Waste streams B+C are fed from the primary circuit and so the hydrazine loads are not factored into daily and annual discharge calculations as they have no daily discharge and only apply during start up or shut down periods. The worst-case daily hydrazine discharge would be after wet lay-up of steam generators. The assumption is that this would be treated until the hydrazine concentration falls below a level that is acceptable for a batch discharge. Wet lay-up is not expected in a normal refuelling outage (i.e. for Sizewell B this was ~15 years after first operation).

The D stream hydrazine loads only which are derived from the secondary circuit daily are therefore used in the following calculations.

Table 25 Operational phase chemical discharges of hydrazine from sum of waste streams for 2 EPR units (based on EDECME120678 PREL A, 2011)

| | Waste stream discharge B+C | Waste stream discharge D (kg $yr^{-1}$) |
|---|---|---|
| Hydrazine (daily) (kg $d^{-1}$) | 1 | 3 |
| Hydrazine (annual) (kg $y^{-1}$) | 3 | 24.3 |

## 9.4 Sanitary waste discharges and calculation of un-ionised ammonia combined inputs

Information on sanitary waste discharges during the operational phase are based on plans used for Hinkley point. For estimation of loadings from the treatment works into the cooling water for the H1 Assessment the following are assumed:

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- Maximum number of operational staff present during 24 hours (under outage conditions) based on Hinkley Point C – 1900 personnel;
- Waste water production per person – 100 l $d^{-1}$; and
- Discharge concentrations – BOD 20 mg $l^{-1}$, Total Ammonia 20 mg $l^{-1}$and Total Suspended solids 30mg$l^{-1}$. Based on these criteria the calculated discharge loadings are presented in Table 26. Further details relating to the calculation of these loadings are presented in EDF, 2011.

To obtain a total discharge loading value for the 2 EPRs during operation the values for total ammonia and suspended solids have been combined with other sources for each of the screening assessments.

For the EA screening assessment total ammonia concentrations from operational inputs (sanitary plus other inputs i.e. circuit conditioning) and the existing site background values are combined. Both a maximum 24 hour loading, and an average annual loading are considered. For the maximum loading assessment extreme values for temperature, pH and salinity are used in the EA un-ionised ammonia calculator with the 24 hour loading and site background ammonia to derive the maximum un-ionised ammonia value. For the annual assessment the annual ammonia value for combined operational sources plus background for the site are used with average pH, salinity and temperature data in the EA calculator to derive the annual un-ionised ammonia concentration. The ammonia background concentration in the seawater is based on monitoring data from BEEMS Technical Report TR314. The physicochemical data for the site are derived from BEEMS report TR189 (see Table 26).

Table 26 Calculated discharge concentration of un-ionised ammonia (as N) for treated sanitary effluent and combined inputs

| Parameter | Derivation of value | 24-hour value | Annual value |
|---|---|---|---|
| BOD | Sanitary loading | 3.8 (kg $d^{-1}$) | 1387 (kg $yr^{-1}$) |
| Suspended solids | Sanitary loading | 5.3 (kg $d^{-1}$) | 1916 (kg $yr^{-1}$) |
| Total Ammonia | Sanitary loading | 3.8 (kg $d^{-1}$) | 1,387 (kg $yr^{-1}$) |
| Total Ammonia (Circuit conditioning) | Circuit conditioning loading | 77.1 (kg $d^{-1}$) | 13009 (kg $yr^{-1}$) |
| Maximum ammonia concentration in discharge $NH_4$-N | Based on a 66[1] (24 h) and 116 cumec flow | 10.49 µg $l^{-1}$ | 3.06 µg $l^{-1}$ |
| Temperature data used in calculator | Based on maximum site background 19.5+ either thermal uplift of 23.2 or 11.6 °C[1] | 42.6 | 31.1 |
| pH data used in calculator | Based on 95 percentile and 50$^{th}$ percentile | 8.23 | 8.05 |
| Salinity data used in calculator | Based on 95 percentile and 50$^{th}$ percentile | 31.7 | 33.27 |
| Site background ammonia $NH_4$-N | Based on 95$^{th}$ percentile and mean | 26.3 µg $l^{-1}$ | 11.38 µg $l^{-1}$ |
| Total ammonia in discharge including background $NH_4$-N | 95$^{th}$ percentile and mean background added to respective mean and 95$^{th}$ percentile discharge | 36.78 µg $l^{-1}$ | 14.44 µg $l^{-1}$ |
| Un-ionised ammonia concentration $NH_3$-N | Calculated with EA un-ionised calculator (Clegg and Whitfield, 1995) using combined discharge concentration plus background ammonia | 7.32 µg $l^{-1}$ | 0.96 µg $l^{-1}$ |

[1] see TR302 'worst case scenario when 2 out of 4 pumps were under maintenance the flow of cooling water would be halved

## 9.5 Sanitary waste discharges and microbiological parameters

Based on the HPC estimates are made of maximum discharge concentrations of inputs into the sewage treatment plant. Secondary treatment implies a 100 factor (2 log) reduction in coliforms and enterococci. If UV treatment is applied a 5.4 log reduction is assumed. The dilution factor required to reduce the coliforms to levels that would comply with bathing water standards has been derived.

## 9.6 Chlorinated discharges

Coastal power stations require a means of chlorine dosing for biofouling control. Based upon the known risk of biofouling at Sizewell, EDF Energy would need to chlorinate the SZC cooling water (CW) system to maintain control over biofouling of critical plant. At those sites where chlorination is required, EDF Energy's operational policy for its existing UK fleet (based upon experiments and operational experience) is to continuously dose during the growing season to achieve a minimum Total Residual Oxidant (TRO) dose of 200$\mu gl^{-1}$ in critical sections of the CW plant and at the inlet to the condensers (BEEMS Technical Report TR316).

The expected discharges from the chlorination process include:

- Residual oxidants in the form of free chlorine and chlorinated compounds. The range and proportions of chlorinated compounds are variable and relate to the presence of organic material and bromine or bromide concentrations in the sea water being treated.
- Trihalomethanes which are present as bromoform at Sizewell.

A precautionary discharge source term of 150$\mu gl^{-1}$ was derived for the planned cooling water discharge based on the expected decay of TRO between the inlet to the condensers and the point of discharge from the cooling water outlet (BEEMS TR316). In laboratory studies six specific chlorination byproducts (CBPs) were analysed for in chlorinated seawater from Sizewell these were: bromoform; dibromochloromethane (DBCM); bromodichloromethane (BDCM) and dibromoacetonitrile (DBAN); dibromoaceitic acid (DBAA);) and 2,4,6 tribromophenol. Of the six CBPs analysed, DBCM was measured at close to its detection limit and the other CBPs except for bromoform were below detection (BEEMS TR217). Following additions of chlorine to Sizewell seawater of 250 -500$\mu gl^{-1}$ bromoform was detected at concentrations of 5 – 29$\mu gl^{-1}$. Based on interpolation of the chlorine dose required to achieve a target value of 200$\mu gl^{-1}$ TRO the equivalent bromoform concentration that results was 190$\mu gl^{-1}$ and so this value is used as the representative discharge concentration for the planned Sizewell C (BEEMS TR303).

## 9.7 Demineralisation plant discharges

Current estimations of discharge loadings from the demineralisation plant are largely based on extrapolation of information from the Flamanville 3 site (combined desalination and demineralisation plant) and local sea water quality. The assessment has adopted the discharge loading values for a combined desalination and demineralisation plant as this provides bounding conditions of a worst-case discharge scenario. The expected effluents from a combined desalination and demineralised plant are presented in Table 27. The values presented are based on the production of water for two EPR units. These maximum discharge values assume the desalination units run continuously and that the demineralisation unit runs for several hours each day with a regeneration cycle occurring every 30 days. Sequestering agents are used in the desalination

plant to prevent mineral deposits forming on the reverse osmosis membranes. For the SZC demineralisation plant one of two sequestering agents will be used i.e. either ATMP or a sodium polymer sequestering agent.

a) ATMP based sequestering agent

Amino tri-methylene phosphonic acid (ATMP) is the active ingredient in the commercial ATMP based sequestering agent. The discharge loading values for constituent chemicals and by-products associated with use of an ATMP sequestering agent are presented in Table 27.

For assessing the sodium component of the ATMP the loading values have been added to those from other sources in Table 28.

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Table 27 Constituent chemicals and by-products for an ATMP sequestering agent

| **Constituent chemicals** | Proportion of commercial solution | 24 hour loading (kg $d^{-1}$) | Annual loading (kg $yr^{-1}$) |
|---|---|---|---|
| ATMP[1] | 100 | 45 | 9100 |
| Sodium | 100 | 45 | 9100 |

1 ATMP = Amino Trimethylene Phosphonic Acid CAS No: 6419-19-8

The commercially available product comprises 10% alky-phosphonic acid, which on use degrades into several potentially toxic by-products and 90% sodium polyacrylate, which is also potentially toxic. Details on the calculations of the loading values in discharges are presented in Table 28.

Table 28 Constituent chemicals and by-products for a sodium polymer sequestering agent

| **Constituent chemicals** | By-Products | Proportion of commercial solution | 24 hour loading (kg $d^{-1}$) | Annual loading (kg $yr^{-1}$) |
|---|---|---|---|---|
| Alkyl phosphonic acid (10%) | HEDP | 9.75 | 4.5 | 890 |
| | Acetic acid | 0.15 | 0.1 | 14 |
| | Phosphoric acid | 0.13 | 0.1 | 12 |
| Sodium polyacrylate (90%) | Sodium polyacrylate (polymer) | 88.2 | 40 | 8030 |
| | Acrylic acid (residual monomer) | 1.8 | 1 | 165 |
| TOTAL | | 100% | 45 | 9100 |

## 9.8 Trace metals in raw materials

Water treatment chemicals such as sodium hydroxide, hydrochloric acid and sulphuric acid contain traces of substances such as cadmium and mercury, which are priority substances listed by the Water Framework Directive. The potential impact of these trace contaminants is discussed. As part of the H1 assessment there are specific requirements for the minimisation of the annual loads of the priority hazardous substances cadmium and mercury. An H1 assessment of discharge concentrations of cadmium and mercury was carried out to demonstrate that the levels discharge during normal operations will be small with a negligible environmental impact. This was based on operational experience and feedback from EDF's French fleet of nuclear power stations. Table 29 contains the estimated annual and 24 hour loadings for cadmium and mercury. Both these annual (and the daily worst case if scaled over a year) load figures meet the requirement to not exceed a significant annual load of 1kg for mercury or 5kg for cadmium.

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Table 29 Estimated annual contributions of cadmium and mercury from the raw material chemical use in water treatment systems

| Trace metal | Discharge loading | |
|---|---|---|
| | Annual (kg/y) | 24-hr (kg/day) |
| Cadmium | 0.37 | 0.005 |
| Mercury | 0.099 | 0.0011 |

The loadings for the non-radioactive contaminants associated with radioactive discharges for two EPR units are detailed in Table 30. For the assessment of discharges, it has been assumed that all metals within the effluent are present 100% in the dissolved state and therefore biologically available. This provides a worst-case scenario in terms of the modelling assessment.

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Table 30 Operational phase chemical loadings for 2 EPR units based on EDF, 2014 and subsequent modifications incorporated in HPC-EDECME-XX-000-RET-000061)

| **Substance** | Circuit conditioning (kg $yr^{-1}$) | Sanitary waste discharge (kg $yr^{-1}$) | Producing demineralised water (kg $yr^{-1}$)[1] | Maximum annual loading (kg $yr^{-1}$) | Maximum 24-hour loading (kg $d^{-1}$) |
|---|---|---|---|---|---|
| Boric acid ($H_3BO_3$) | 14000 | - | - | 14000 | 5625 |
| Boron | 2448 | - | - | 2448 | 984 |
| Lithium hydroxide | 8.8 | - | - | 8.73 | 4.4 |
| Hydrazine | 24.3 | | | 24.3 | 3 |
| Morpholine | 1680 | - | - | 1674 | 92.3 |
| Ethanolamine | 920 | - | - | 919 | 24.75 |
| Nitrogen as N | 10130 | 1595 | - | 11725 | 332 |
| Un-ionised Ammonia ($NH_3$) | - | - | - | 958[2] | 27[2] |
| Phosphates ($PO_4^3$) | 800 | - | - | 790 | 352.5 |
| Detergents | | - | 624 | 624 | - |
| Suspended solids | 2800 | 2080 | 88000 | 92879 | 870 |
| BOD | - | 1387 | - | 1387 | 3.8 |
| COD | 5050 | - | - | 5050 | 330 |
| Aluminium | 5.26 | - | - | 5.26 | 1.1 |
| Copper | 0.42 | - | - | 0.42 | 0.08 |
| Chromium | 8.37 | - | - | 8.37 | 1.7 |
| Iron | 34.97 | | 46000 | 46035 | 257 |
| Manganese | 3.33 | | | 3.33 | 0.67 |
| Nickel | 0.44 | | | 0.44 | 0.09 |
| Lead | 0.3 | | | 0.3 | 0.07 |
| Zinc | 5.6 | | | 6.0 | 1.2 |
| Chloride | | | 87100[3] | 87100 | 450 |
| Sulphates | | | 98400[3] | 98400 | 2000 |
| Sodium | | | 52400[3] | 52400 | 855 |
| ATMP | | | 9100 | 9100 | 45 |
| HEDP | | | 890 | 890 | 4.5 |
| Acetic Acid | | | 14 | 14 | 0.1 |
| Phosphoric acid | | | 12 | 12 | 0.1 |

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## NOT PROTECTIVELY MARKED

| **Substance** | Circuit conditioning (kg $yr^{-1}$) | Sanitary waste discharge (kg $yr^{-1}$) | Producing demineralised water (kg $yr^{-1}$)[1] | Maximum annual loading (kg $yr^{-1}$) | Maximum 24-hour loading (kg $d^{-1}$) |
|---|---|---|---|---|---|
| Sodium polyacrylate | | | 8030 | 8030 | 40 |
| Acrylic acid | | | 165 | 165 | 1 |
| Chlorine (TRO) and bromoform[3] | | | | | (150$\mu g l^{-1}$), 190$\mu g l^{-1}$ |

[1] figures represent combined demineralisation and desalination and therefore bounding;[2]These figures are back calculated from the un-ionised ammonia concentration derived from the un-ionised ammonia calculator using the $NH_4$ concentration that results from the combined sanitary and conditioning inputs;[3]Based on the expected chlorine dose required to achieve a target concentration of 200$\mu g\ l^{-1}$ at the condensers of the power station and taking account of subsequent decay of TRO a precautionary discharge source term of 150$\mu g l^{-1}$ is proposed in TR316 and for bromoform 180$\mu g l^{-1}$ (BEEMS TR303).

# 10 Operational assessment and modeling methodology

## 10.1 Background

Potential discharges to the marine environment have been assessed for the operational phase of the planned SZC. For large cooling water discharges that are discharged to estuaries or coastal waters a specific screening assessment recommended by Defra and Environment Agency (Clearing the Waters for All, 2016) is applied.

## 10.2 Screening operational discharges

Substances likely to be discharged in the cooling water are assessed as follows:

(i) Average background concentration for substance multiplied by average cooling water flow (to determine background load)

(ii) Average load of substance in process stream added to above load

(iii) Divide step (ii) result by total of average cooling water discharge volume and average process stream volume combined

(iv) Compare result of above to the EQS AA

A second assessment makes a comparison to the relevant EQS MAC

(v) Maximum background concentration for substance multiplied by minimum cooling water flow (to determine background load)

(vi) Maximum load of substance in process stream added to above load

(vii) Divide step (vi) result by total of minimum cooling water discharge volume and average process stream volume combined

(viii) Compare result of above to the EQS MAC

The aim of the process is to identify components of discharges that may contribute to the deterioration of a waterbody and so prevent achievement of target standards such as status objectives under the Water Framework Directive.

The guidance applies to continuous discharges and variable process discharges to freshwater and coastal waters (“surface waters”).

Substances are assessed in two stages: screening and modelling. For the modelling assessments the Cooling Water discharge locations are shown in Table 31.

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Table 31 Preferred location and depth of SZC cooling water outfall heads.

| | Latitude WGS84 (degrees N) | Longitude WGS84 (degrees E) | Easting BNG (m) | Northing BNG (m) | Depth ODN (m) |
|---|---|---|---|---|---|
| O9a<br><br>Same location as O9 in TR301 | 52.21807 | 1.67435 | 651080 | 264125 | 16.9 |
| O9b | 52.21803 | 1.67544 | 651155 | 264125 | 16.8 |
| WGS84: World Geodetic system 1984, BNG: British National Grid, ODN: Ordnance Datum Newlyn | | | | | |

### 10.3 Assumptions and limitations

Several assumptions were made to conduct the calculations for EQS AA and EQS MAC assessment for large cooling water discharges:

1) The discharge loadings used are those shown in Table 30.

2) The maximum daily and annual loading values have been adopted to provide a worst-case scenario in terms of contaminant loadings in the discharge. The use of daily chemical loading values needs to be treated with caution as the H1 methodology is developed for the assessment of long-term discharges. These discharge values are compared to EQS values which are normally based on annual average concentrations.

3) For chemicals in the discharge that do not have an EQS Predicted No Effect Concentrations are derived if enough toxicity data are available. Comparisons are made to any acute toxicity values where ecotoxicological data are limited and where no toxicity data are available comparisons are made to site background levels for the relevant chemical (see section 2).

4) For substances subject to intermittent release which is considered appropriate for 24-hour discharge assessments a factor of 100 would normally be applied to the lowest $L(E)C_{50}$ of at least three short term tests for species from three taxonomic groups to derive a short term PNEC. (EU Technical Guidance, 2003).

5) For annual discharge assessments where two long term test NOECs are available the lowest has a factor of 500 applied to derive a chronic NOEC for marine data and where three are available a factor of 100 is appropriate (EU Technical Guidance, 2003).

6) The maximum annual loadings are assumed to be discharged at a constant rate over the course of a year and to be mixed in the cooling water flows prior to discharge to the environment. It is assumed within the presented H1 calculations that for average annual concentrations the cooling water discharge flow, into which all discharges are mixed, is $116 m^3 sec^{-1}$ as a worst case under normal operational flow.

7) For 24 hour discharges the assessment has been made for a discharge flow of $66 m^3 s^{-1}$ to provide a worst-case “incidental” dilution scenario. This discharge volume assumes that only a single cooling water pump is operating for each EPR unit during a low water period. However, it should be noted

that 24-hour discharges are unlikely to occur exclusively under low tide conditions and when only one cooling water pump is functioning normally (and is therefore particularly conservative).

8) For metals it is assumed that annual loading figures relate entirely to metals in the dissolved phase. As dissolved metals are in a biologically available form, this assumption allows for assessment of a worst-case potential impact scenario.

9) The chemical discharge values consider any initial dilution or degradation of chemicals within holding tanks.

10) Mean background concentrations are used in place of EQS values for those substances which have no EQS and for which there is no or insufficient toxicity data to derive a predicted no effect concentration. Mean background concentrations are based on the results for the monitoring programme conducted in 2010 as reported in TR189 and in Sizewell 2014/15 supplementary monitoring report TR314.

    Discharge loadings have been used for both desalination and demineralisation processes as these combined sources represent a worst-case scenario.

## 10.4 Screening results for Operational discharges

Table 32 shows the assessment for large cooling water discharges that are discharged to TraC waters for 24h operational discharges and Table 33 shows respective results for the annual operational discharges. Table 32discharge concentrations are compared to the Water Framework Directive annual average environmental quality standards (WFD AA-EQS = Annual Average EQS), and in Table 33 the WFD EQS MAC = Maximum Acceptable Concentration EQS where these are available or AA values if not. In some cases, alternative or surrogate values have had to be referenced i.e Pre-WFD EQS values have been adopted for assessment of boron; Coastal and Transitional Water WFD EQS for chromium is for chromium VI; and in some cases, toxicity data values are compared. Where no toxicity data are available background concentrations measured at the site are compared. For nitrogen reference is made to the winter dissolved inorganic nitrogen 99$^{th}$ percentile for TraC waters of intermediate turbidity (suspended solids levels of 10 to <100mgl$^{-1}$, Appendix C)[1]. Calculations for the maximum 24h loadings are based on a discharge volume of 66m$^3$sec$^{-1}$ under maintenance conditions with a single operational EPR. The maximum annual discharge of 116m$^3$sec$^{-1}$ is based on a single EPR unit having a minimal operational cooling water flow of 58m$^3$sec$^{-1}$ under low tide conditions (worst-case scenario within ‘standard operation’).

[1] It should be noted that a more specific methodology for deriving 99th percentile values based on a relationship between SPM and DIN is recommended in draft Environment Agency guidance and for an annual average SPM of 55.2mgl-1 would give a slightly lower value of 952µgl-1 as a 99th percentile. However the screening assessment here would only slightly change and modelling using a combined macro algal and phytoplankton model is used to provide a definitive assessment of annual inputs.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 32 Screening MAC for large cooling water discharges for the maximum 24 hour loadings predicted for operational phase chemical discharges – bold underlined values indicate failure of the relevant test.

| **Substance** | EQS or surrogate value $\mu gl^{-1}$ | Derivation of surrogate | Discharge + background ($\mu gl^{-1}$) | Max discharge /EQS <1 |
|---|---|---|---|---|
| Boron[1] | 7000 | Pre WFD EQS | 4656 | 0.67 |
| Lithium hydroxide | 65[2] | Mean background | 90.2[2] | **1.39**[3] |
| Hydrazine | 0.004 | Acute PNEC | 0.53[4,5] | **131.5** |
| Morpholine | 28 | Acute PNEC | 16.18 | 0.58 |
| Ethanolamine | 160 | Acute PNEC | 4.34[5] | 0.03 |
| Nitrogen as N | 980[6] | WFD 99th percentile | 484.3[7] | 0.49 |
| Un-ionised Ammonia ($NH_3$.N) | 21 | WFD AA-EQS | 7.34[8] | 0.35 |
| Phosphates ($PO_4$-P) | 33.5 | Mean background | 127 | **3.79** |
| Suspended solids | 74000[3] | Mean background | 154[5] | 0.002 |
| BOD | 2000 | Mean background | 0.67[5,9] | 0.0003 |
| COD | 239000 | Mean background | 57.87[5] | 0.00024 |
| Aluminium | 12 | Mean background | 20.19 | **1.68** |
| Copper | 3.76 | WFD AA-EQS | 4.76 | **1.27** |
| Cadmium | 1.5 | WFD MAC-EQS | 0.13 | 0.09 |
| Chromium | 32 | WFD MAC-EQS | 2.48 | 0.08 |
| Iron | 1000 | WFD AA-EQS | 302 | 0.3 |
| Manganese | 2 | Mean background | - | - |
| Mercury | 0.07 | WFD MAC-EQS | 0.02[10] | 0.29 |
| Nickel | 34 | WFD MAC-EQS | 1.17 | 0.03 |
| Lead | 14 | WFD MAC-EQS | 3.94 | 0.28 |
| Zinc | 6.8 | WFD AA-EQS | 46 | **6.77** |
| Chloride | 14128000 | Mean background | 78.9[5] | 0.00 |
| Sulphates | 2778000 | Mean background | 350.7[5] | 0.00 |
| Sodium | 10400000 | Mean background | 150[5] | 0.00 |
| ATMP | 74 | NOEC (96h fw[11] algae) | 7.89[5] | 0.11 |
| HEDP | 13 | $EC_{50}$ (96 h fw algae) | 0.79[5] | 0.06 |
| Acetic Acid | 301 | $LC_{50}$ 48h fw crust | 0.02[5] | 0.00006 |
| Phosphoric acid | 200 | $LC_{50}$ 72h fw algae | 0.02[5] | 0.0001 |
| Sodium polyacrylate | 180 | $LC_{50}$ 96h fw algae; | 7.01[5] | 0.04 |
| Acrylic acid | 1.7 | $EC_{50}$ (96 h fw algae) | 0.18[5] | 0.1 |

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## NOT PROTECTIVELY MARKED

| **Substance** | EQS or surrogate value $\mu gl^{-1}$ | Derivation of surrogate | Discharge + background ($\mu gl^{-1}$) | Max discharge /EQS <1 |
|---|---|---|---|---|
| Chlorine (TRO) bromoform | (10) 5 | MAC-EQS | (150), 190 | **(15)38** |

1 Variable dissociation products of Boric acid and other boron compounds in seawater so assessment focuses on equivalent boron concentration. 2 Expressed as lithium. 3. Figures in bold exceed the EQS or reference value. 4 This loading does not include hydrazine from stream B+C because this would not be discharged except during start up and shutdown when hydrazine from stream D would not be discharged. 5 Discharge only does not include background or no background either measured or detected 6 It should be noted that a more specific methodology for deriving 99th percentile values based on a relationship between SPM and DIN is recommended in draft Environment Agency guidance and for an annual average SPM of 55.2mgl-1 would give a slightly lower value of 952µgl-1 as a 99th percentile but the screening here would only slightly change. 7 This figure includes a calculated 4.4kg day from sanitary effluent derived by calculation from permitted 23mg/l N from STW discharge – stream G. 8 These figures are back calculated from the un-ionised ammonia concentration derived from the un-ionised ammonia calculator using the NH4 concentration that results from the combined sanitary and conditioning inputs [69] 9 The BOD value is derived from stream G based on a BOD5-atu concentration of 20 mg/l and the derived concentration due to the discharge (0.67µgl-1) is negligible relative to the site background (2mgl-1) and not significant in terms of impact on dissolved oxygen when oxygen flux for vertically well mixed water column at site is considered.10 The mean is used in place of the 95$^{th}$ percentile as values below detection result in lower 11 fw represents freshwater species toxicity test data which determines PNEC

Reference to Table 32 shows that for the 24 hour discharge assessment, hydrazine, chlorine produced residual oxidants (TRO) and bromoform concentrations in the discharge during the operational phase will exceed the acute PNEC and so will be taken forward for more detailed modelling.

Discharge concentrations for copper and zinc also exceed EQS assessment criteria but, in each case, actual discharge concentrations are at least 30 times below the relevant AA EQS and are below their respective detection limits for analysis. It is the high derived 95$^{th}$ percentile background loadings that are responsible for this exceedance therefore no measurable exceedance resulting from the discharge itself would be detectable and so further assessment will not be conducted.

Lithium hydroxide, phosphate and aluminium do not have EQS or PNEC values but instead reference site mean backgrounds and so the 95 percentile load calculations which use site background 95 percentile values will invariably result in an exceedance. In the case of aluminium, the actual discharge contributes a sixtieth of the background and for lithium hydroxide the equivalent lithium input from the discharge is almost 300 times below the background in neither case are these inputs considered of significance. The phosphate input is several times above background and as phosphate can contribute to nutrient status it will be given further consideration in section 11.

Concentrations of other substances for which the discharge 24 hour loading concentration are present in the operational discharge at >40% of their EQS or equivalent reference value are also considered here, and these are boron (boric acid), morpholine, DIN, and un-ionised ammonia.

The boron background concentration in Sizewell seawater as a 95$^{th}$ percentile (as used in the 24h discharge calculation) is around 4564$\mu gl^{-1}$ and as the estimated discharge concentration of boron represents around one twentieth of this value it is the background concentration that has the most influence on the scale of the cooling water discharge concentration relative to the EQS. As the elevation of boron above the seawater background is relatively small and any influence will be localised to the area around the immediate discharge. As an essential element for many marine algal species the low elevation of boron concentration expected in short term discharges is likely to have negligible effects.

Morpholine was 58% of its derived PNEC for 24 hour discharges but is a readily degradable chemical and has a low likelihood of bioconcentration (see Appendix B) this coupled with its low toxicity indicates it would have negligible effects on marine species under this discharge scenario.

Un-ionised ammonia was 35% of its derived PNEC. As temperature may influence the relative amount of un-ionised ammonia the operational discharge has been further assessed considering temperature elevation and this modelling is described in section 11

The 24 hour discharge concentration of dissolved inorganic nitrogen was 49% of the site 99$^{th}$ percentile winter standard for water bodies of intermediate turbidity. As the loading of DIN may influence algal growth this is further assessed using a combined marcoalgal and phytoplankton model.

For annual loadings in the operational cooling water discharge hydrazine, chlorine and bromoform again exceed relevant PNEC or EQS values in the screening assessment and so more detailed modelling will also consider this discharge scenario. Discharges during the operational phase would also just exceed or equal the annual average PNEC for lithium hydroxide, phosphates, aluminium, and zinc (Table 33).

Lithium hydroxide, phosphate and aluminium do not have EQS or PNEC values but instead reference site mean backgrounds and so the mean load calculations which use site background mean values will invariably result in an exceedance. In the case of aluminium and lithium hydroxide, the actual discharge concentrations are below the method detection limit and are several orders of magnitude below the site background so the discharge contributions would have negligible effects. The phosphate discharge concentration is also below the method detection limit and although the discharge concentration is very low the input can contribute to nutrient status so it will be given further consideration in section 11.

Zinc fails the annual loading discharge assessment. However, it is the high background loading that is responsible for this exceedance and the actual discharge concentration would be below detection therefore this input is considered to have negligible effects.

In screening copper and chromium were 57 and 95% of their respective annual average EQS values but for both the predicted discharge concentrations are below method detection limits and are several orders of magnitude below their respective EQS (i.e. site backgrounds are not included) therefore negligible likely effects are predicted.

As was the case for the 24 hour screening assessment elevation of boron above the seawater background is relatively small and so any influence will be localised to the area around the immediate discharge. As an essential element for many marine algal species the low elevation of boron concentration is likely to have negligible effects and therefore this is screened out of further assessment.

For the annual discharge screening assessment as DIN at 37% of its background reference can contribute to nutrient status it is given further consideration in section 11. Un-ionised ammonia concentration was low at 0.05% of its EQS but is also given further consideration in section 11 in relation to the influence of temperature elevation on the percentage of un-ionised ammonia.

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Table 33 Screening Test for large cooling water discharges for average annual loadings predicted for operational phase discharges for 2 EPR units at SZC – bold underlined values indicate failure of test.

| **Substance** | EQS/surrogate value µg $l^{-1}$ | Derivation of surrogate | Discharge concentration including background (µg $l^{-1}$) | Annual Discharge/EQS <1 |
|---|---|---|---|---|
| Boron[1] | 7000 | Pre WFD EQS | 4145.67 | 0.59 |
| Lithium hydroxide | 65[2] | Mean background | 65[2] | **1.00**[3] |
| Hydrazine | 0.0004 | Chronic PNEC | 0.01[4] | **16.6** |
| Morpholine | 17 | Chronic PNEC | 0.46[5] | 0.03 |
| Ethanolamine | 160 | Acute PNEC | 0.25[5] | 0.001 |
| Nitrogen as N | 980[6] | WFD 99th percentile | 360.12[7] | 0.37 |
| Un-ionised Ammonia ($NH_3$-N) | 21 | WFD AA-EQS | 0.96[8] | 0.05 |
| Phosphates | 33 | Mean background | 33.57 | **1.00** |
| Detergents | - | - | 0.17[5,9] | 0.2 |
| Suspended solids | 74000[3] | Mean background | 25.4[5] | 0.0003 |
| BOD | 2000 | Mean background | 0.38[5,10] | 0.0002 |
| COD | 239000 | Mean background | 1.38[5] | 0.00001 |
| Aluminium | 12 | Mean background | 12 | **1.00** |
| Cadmium | 0.2 | WFD AA-EQS | 0.05 | 0.25 |
| Copper | 3.76 | WFD AA-EQS | 2.15 | 0.57 |
| Chromium | 0.6 | WFD AA-EQS | 0.57 | 0.95 |
| Iron | 1000 | WFD AA-EQS | 132.58 | 0.13 |
| Manganese | 2 | Mean background | - | 0.00 |
| Mercury | 0.07 | WFD MAC-EQS | 0.02 | 0.29 |
| Nickel | 8.6 | WFD AA-EQS | 0.79 | 0.09 |
| Lead | 1.3 | WFD AA-EQS | 1.0 | **0.76** |
| Zinc | 6.8 | WFD AA-EQS | 14.7 | **2.16** |
| Chloride | 14128000 | Mean background | 23.81[5,6] | - |
| Sulphates | 2778000 | Mean background | 26.90[5] | - |
| Sodium | 10400000 | Mean background | 14.32[5] | - |
| ATMP | 74 | NOEC 96h fw[11] algae | 2.49[5] | 0.03 |
| HEDP | 13 | NOEC 96h algae | 0.24[5] | 0.02 |
| Acetic Acid | 62.8 | NOEC 21d fw crust | 0.004[5] | 0.0001 |
| Phosphoric acid | 20 | $LC_{50}$ 72h algae | 0.003[5] | 0.0002 |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

| **Substance** | EQS/surrogate value µg $l^{-1}$ | Derivation of surrogate | Discharge concentration including background (µg $l^{-1}$) | Annual Discharge/EQS <1 |
|---|---|---|---|---|
| Sodium polyacrylate | 11.2 | NOEC 72h fw crust | 2.20[5] | 0.20 |
| Acrylic acid | 0.34 | NOEC 72 h fw algae | 0.05[5] | **0.13** |

1 Variable dissociation products of Boric acid and other boron compounds in seawater so assessment focuses on equivalent boron concentration. 2 Expressed as lithium. 3. Figures in bold exceed the EQS or reference value. 4 This loading does not include hydrazine from stream B+C because this would not be discharged except during start up and shutdown when hydrazine from stream D would not be discharged. 5 Discharge only does not include background or no background either measured or detected. 6 It should be noted that a more specific methodology for deriving 99$^{th}$ percentile values based on a relationship between SPM and DIN is recommended in draft Environment Agency guidance and for an annual average SPM of 55.2mgl-1 would give a slightly lower value of 952µgl-1 as a 99$^{th}$ percentile but the screening here would only slightly change.7 This figure includes a calculated 1595kg/y from sanitary effluent derived by calculation from permitted 23mg/l N from STW discharge – stream G. 8 These figures are back calculated from the un-ionised ammonia concentration derived from the un-ionised ammonia calculator using the NH4 concentration that results from the combined sanitary and conditioning inputs 9 Detergents are assumed to be non-ionic for cleaning reverse osmosis membranes (Beyer et al, 2017) and the PNEC is derived from Belanger et al 2006 cited in Table 4.37 Hera, 2009 for most toxic alcohol ethoxylates with chain length C18 and added application factor of 10 is applied to this value as it is based on freshwater data 10 The BOD value is derived from stream G based on a BOD5-atu concentration of 20 mg/l and the derived concentration due to the discharge (0.38µgl-1) is negligible relative to the site background (2mgl-1) and not significant in terms of impact on dissolved oxygen when oxygen flux for vertically well mixed water column at site is considered 11 fw represents freshwater species toxicity test data which determines PNE

# 11 Review and assessment of potentially significant operational discharges

## 11.1 Background

During operation, discharges from site are expected to be primarily via the main seawater cooling system (CFT). Various potential chemical discharges may occur at different times or continuously dependent on different operational processes. The main approach for modelling these discharges is to apply the GETM Sizewell model with input data related to the specific chemical decay and kinetics. More detail on the modelling is provided in BEEM TR303. The outputs from each modelling run enable an assessment of the area influenced by the relevant discharge plume that exceeds or is below a threshold value specific to the chemical of interest. This report supports an assessment of the overall influence of any discharges from SZC on water and sediment quality so total areas over which a given threshold value is exceeded are considered. Other BEEMS reports consider areas of exceedance relative to other receptors and therefore express areas of exceedance with reference to the location or distribution of the receptor of interest.

## 11.2 Cooling water chlorination modelling

Chlorine is commonly applied to the seawater to prevent biofouling of cooling water systems of coastal power stations. In seawater which typically has a high bromide concentration (68mgl$^{-1}$) chlorination results in displacement of chlorine atoms by bromine atoms with the formation of hypobromous acid, hypobromite ion and bromamines. The primary biocidal effects of seawater chlorination therefore result from oxidants associated with the bromine chemistry and these are measured and expressed as the total residual oxidant concentration or TRO.

Based upon the known risk of biofouling at Sizewell, EDF Energy would need to chlorinate the SZC cooling water (CRF) system to maintain control over biofouling of critical plant. At those sites where chlorination is required, EDF Energy's operational policy for its existing UK fleet is to continuously dose during the growing season of the principal biofouling species (e.g. the mussel) to achieve a minimum TRO dose of 0.2mgl$^{-1}$ in critical sections of the CW plant and at the inlet to the condensers. Chlorination of the cooling water has the potential to directly affect any entrained organisms and indirectly to affect organisms in the discharge footprint. To provide enough protection to the marine environment chlorine has an EQS of 10µg l$^{-1}$ for seawater set as a maximum allowable concentration and expressed as a 95 percentile (UKTAG, 2013, Defra 2014).

At those sites where chlorination is required, EDF Energy's operational policy for its existing UK fleet (based upon experiments and operational experience) is to continuously dose during the growing season to achieve a minimum Total Residual Oxidant (TRO) dose of 0.2mgl$^{-1}$ in critical sections of the CW plant and at the inlet to the condensers. The TRO discharge concentration from the CW systems at outfall heads would be 0.15mgl$^{-1}$. The FRR system would have its own discharge point but would not be chlorinated – currently assumed at location FRR1 Discharge Easting 647980 and Northing 264000 and FRR2 discharge Easting 647980 and Northing 264300, 4m depth ODN, 475m tunnel length).

Sizewell B has a permit to discharge cooling water with a maximum TRO concentration of 0.3mgl$^{-1}$ all year round and this source term has been used for the modelling studies to assess in combination effects in BEEMS Technical Report TR303.

The total residual oxidants (TRO) resulting from the combination of chorine and organic material in the water are modelled in TR303 using an empirical demand/decay formulation derived from experiments with Sizewell seawater and coupled into the GETM Sizewell model.

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Two scenarios were considered: chlorination of SZB plus SZC operating in combination, and chlorination of SZB only. A discharge of 132$m^3s^{-1}$ has been modelled for TRO for SZC. For each model run a month-long simulation was analysed and the mean and 95$^{th}$ percentile of the TRO concentrations was extracted. Table 34. Figures 10 and 11 present the area of the plume that exceeds a concentration threshold of 10$\mu gl^{-1}$. For SZC only, there is an area of 2.13ha exceeding the EQS at the seabed and 337.65ha at the sea surface. Figure 10 shows that the SZC plume does not mix with the SZB plume.

Table 34 Total areas exceeding the TRO EQS (These values are based on 132$m^3s^{-1}$ discharge from SZC).

| Model | | TRO =10$\mu gl^{-1}$ as a 95$^{th}$ percentile | |
|---|---|---|---|
| | | surface | seabed |
| SZB+SZC | ha | 726.21 | 167.08 |
| SZB only | ha | 388.56 | 164.95 |
| SZC only | ha | 337.65 | 2.13 |

Table 35 presents the area of the TRO plume that exceeds the EQS concentration threshold. For completeness, not only the EQS value was included but also other values between 1 and 20$\mu gl^{-1}$. Analysis of the TRO modelling runs shows that the EQS will be exceeded in the mixing zone at the surface and seabed for SZC and SZB. An important observation from this modelling is the separation of the TRO plumes from SZB and SZC discharges with no interaction between them down to the level of 1$\mu gl^{-1}$ of TRO (Figure 12). This is important because it implies that, within reason, the chlorination regimes of the two developments can be managed independently.

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Table 35: Area of the plume at different levels of TRO concentration (from BEEMS Technical report TR303 Edition 4).

| Model run | µg $l^{-1}$ | 95th percentile surface (ha) | 95th percentile seabed (ha) | mean surface (ha) | mean seabed (ha) |
|---|---|---|---|---|---|
| TRO_2outf_May - Chlorination of SZB + SZC | 1 | 5450.62 | 3662.9 | 1704.96 | 579.31 |
| | 2 | 3302.04 | 1415.19 | 869.52 | 234.26 |
| | 4 | 1710.23 | 428.1 | 412.22 | 129.41 |
| | 6 | 1214.69 | 251.52 | 238.07 | 64.03 |
| | 8 | 928.17 | 200.28 | 157.89 | 27.13 |
| | **EQS 10** | **726.21** | **167.08** | 112.81 | 16.82 |
| | 15 | 436.55 | 101.93 | 64.82 | 8.63 |
| | 20 | 289.87 | 52.03 | 44.07 | 4.93 |
| TRO_2outf_MayTROB - Chlorination of SZB only | 1 | 1652.14 | 1136.86 | 756.49 | 363.32 |
| | 2 | 1206.05 | 559.79 | 460.55 | 226.40 |
| | 4 | 821.86 | 332.71 | 257.02 | 126.72 |
| | 6 | 617.99 | 244.23 | 168.21 | 63.02 |
| | 8 | 483.09 | 197.14 | 122.90 | 27.03 |
| | **EQS 10** | **388.56** | **164.95** | 94.98 | 16.59 |
| | 15 | 264.98 | 101.26 | 60.11 | 8.41 |
| | 20 | 192.32 | 51.69 | 42.50 | 5.15 |
| SZC only | 1 | 3798.48 | 2526.04 | 948.47 | 215.99 |
| | 2 | 2095.99 | 855.4 | 408.97 | 7.86 |
| | 4 | 888.37 | 95.39 | 155.2 | 2.69 |
| | 6 | 596.7 | 7.29 | 69.86 | 1.01 |
| | 8 | 445.08 | 3.14 | 34.99 | 0.1 |
| | **EQS 10** | **337.65** | **2.13** | 17.83 | 0.23 |

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## NOT PROTECTIVELY MARKED

| Model run | µg $l^{-1}$ | 95th percentile surface (ha) | 95th percentile seabed (ha) | mean surface (ha) | mean seabed (ha) |
|---|---|---|---|---|---|
| | 15 | 171.57 | 0.67 | 4.71 | 0.22 |
| | 20 | 97.55 | 0.34 | 1.57 | - |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

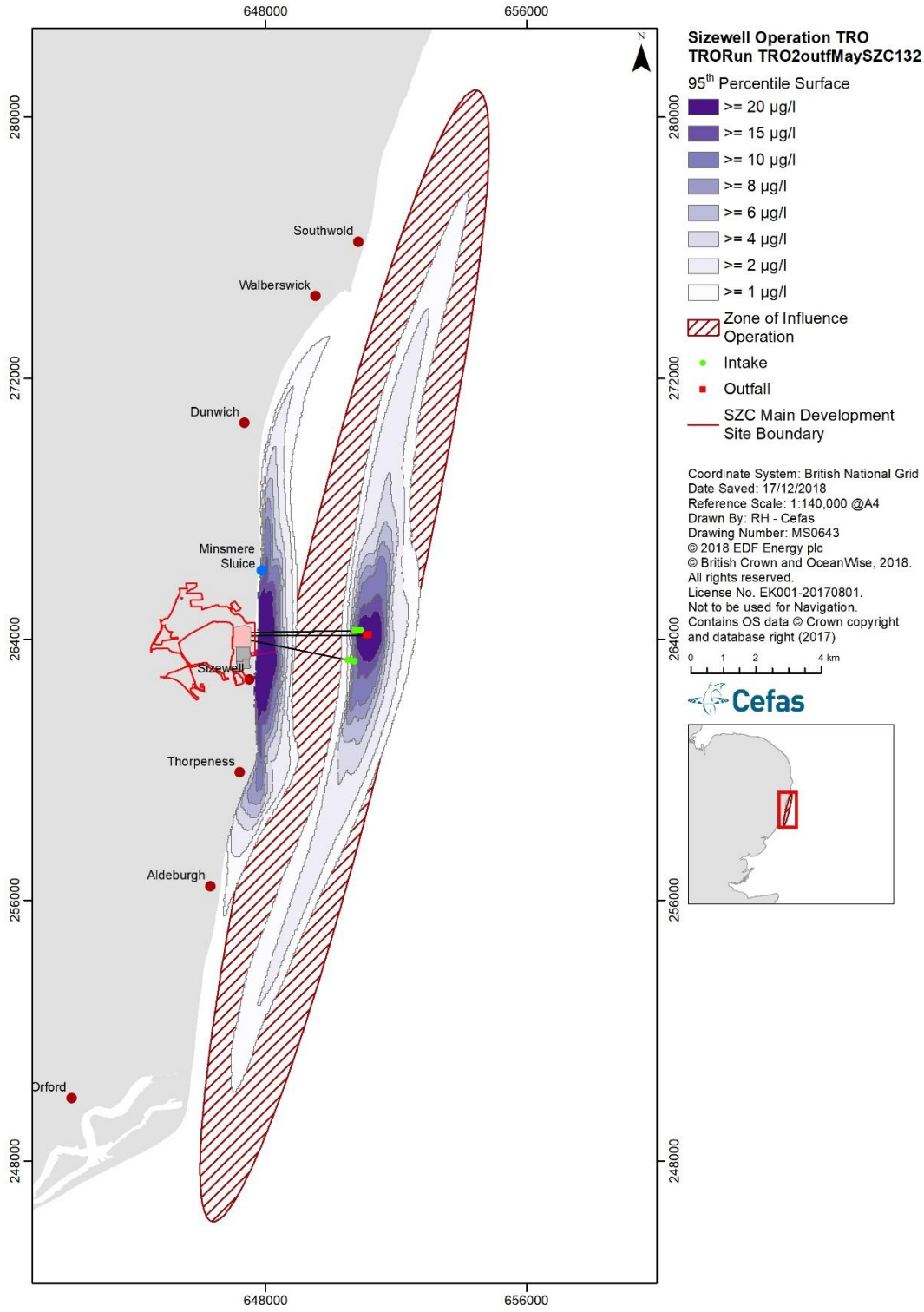


Figure 10: SZB + SZC modelling: 95th percentile of the TRO concentration at the surface (µgl$^{-1}$). The hatched area shows the outer tidal excursion.

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

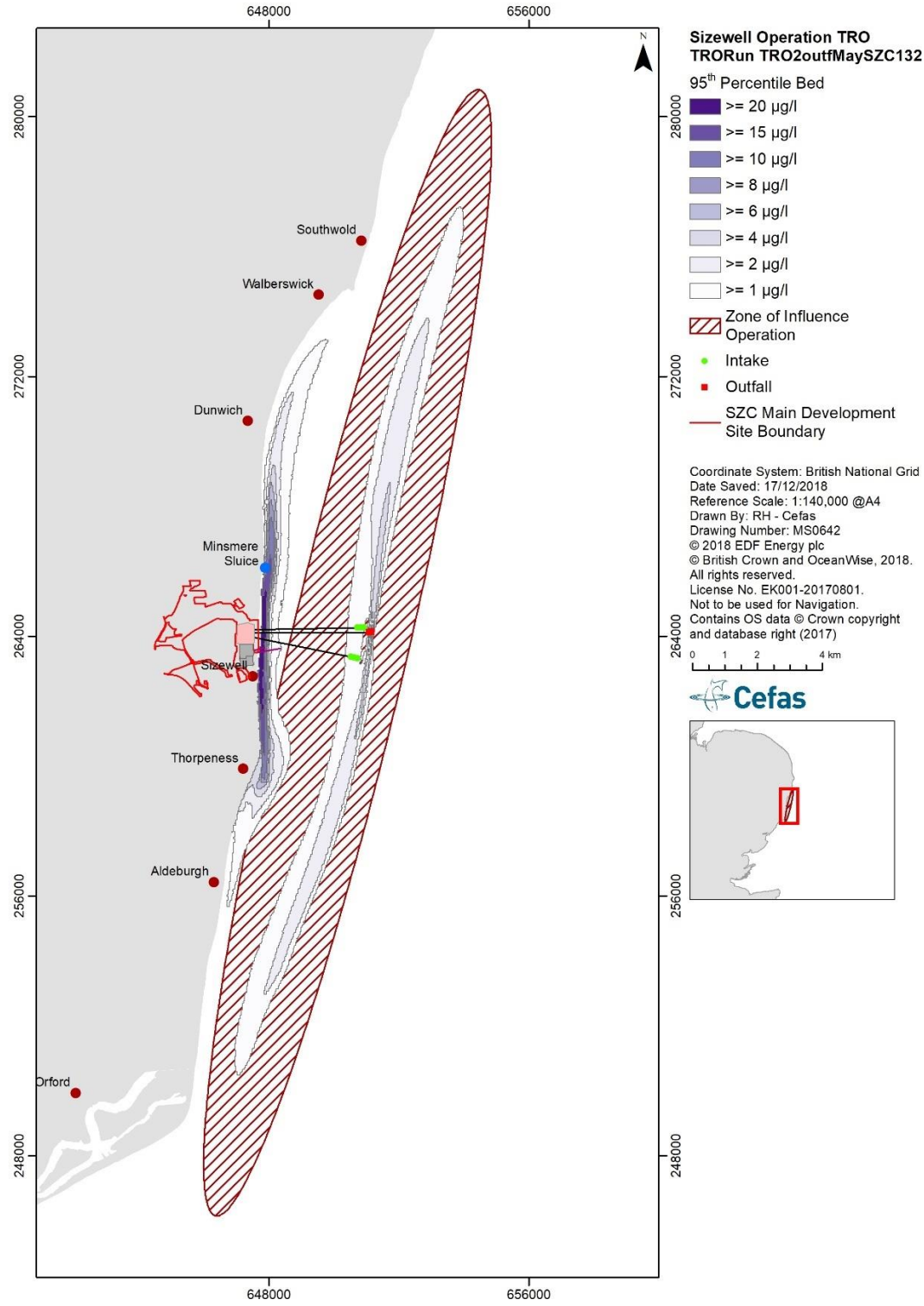


Figure 11 95th percentile of the TRO concentration at the seabed for chlorination from SZB and SZC (run TRO_2outf_May, from TR 303).

## 11.3 Cooling water chlorination by-products discharge modelling

Another consequence of the chlorination of the power station is the formation of chlorination by-products (CBP's) as a result of complex chemical reactions in seawater. Many products are formed, the number and type being dependent on the composition and physical parameters of the seawater. The dominant CBP's are, in order, bromoform, dibromochloromethane (DBCM), bromodichloromethane (BDCM), monobromaceitic acid, dibromoaceitic acid (DBAA), dibromoacetonitrile (DBAN) and 2,4,6 tribromophenol. Laboratory studies carried out with chlorinated Sizewell seawater only detected bromoform (BEEMS Technical Report TR217). Bromoform is lost through volatilization to the atmosphere, with the loss rate a function of the thermal stratification and values obtained from the literature (Mackay and Leinonen, 1975) and coupled into the GETM Sizewell model.

Since bromoform is a product of chlorination, the same scenarios as for TRO were considered: chlorination of SZB plus SZC operating in combination and chlorination of SZB only. For each model run a month-long simulation was analysed and the 95$^{th}$ percentile of the bromoform concentrations was extracted. There is no published EQS for bromoform and so a calculated PNEC of 5µgl$^{-1}$ as a 95%ile has been used (Taylor 2006). This value was predicted based on the results of a toxicological review and the application of Quantitative Structure Activity Relationships (the same figure was used In the HPC WDA permit application). Figure 12 and 13 and Table 36 show the area of the plume that exceeds the relevant concentration threshold.

Table 36: Total areas exceeding the Bromoform PNEC.

| Model | | PNEC = 5µgl$^{-1}$ as a 95$^{th}$ percentile | |
|---|---|---|---|
| | | Surface ha | Seabed ha |
| SZB+SZC | ha | 357.94 | 130.19 |
| SZB only | ha | 305.80 | 129.52 |
| SZC only | ha | 52.14 | 0.67 |

Like the TRO plume, the bromoform plume is a long, narrow feature parallel to the coast. Also, the SZB plume is always within the channel inshore of the Sizewell-Dunwich Bank and does not overlap with the SZC plume that is outside the Bank (Figure 12). Both plumes are strongly stratified with larger areas at the surface than at the seabed. The SZC plume is generally smaller and narrower than that due to SZB; the exception is at the 1µgl$^{-1}$ contour for the 95$^{th}$ percentile where the SZC plume has a longer extent but at higher concentrations the SZC plume is always smaller. This is due to the lower initial discharge concentration and greater water depth at the SZC outfall location (16m vs. 5m for SZB outfall).

The Bromoform plume areas that exceed the PNEC (5µgl$^{-1}$ as a 95$^{th}$ percentile) have been calculated and are shown in Table 36. For SZC only, the area exceeding the applied EQS at the seabed is 0.67ha and 52.14ha at the sea surface.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

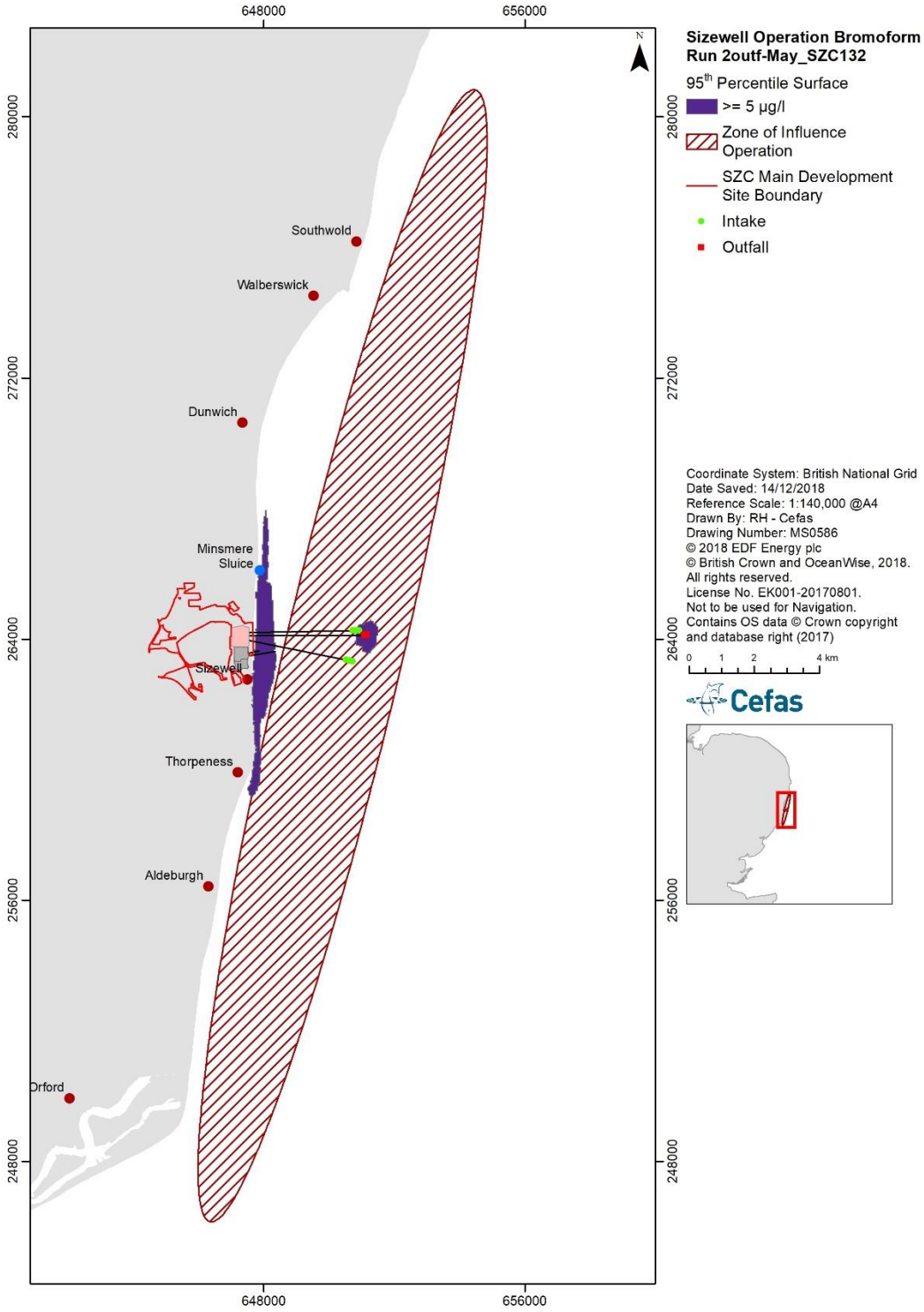


Figure 12: 95th percentile of the Bromoform concentration at the surface for chlorination from SZB and SZC (run Brom_2outf_May). Black line delineates the PNEC of 5µgl$^{-1}$. The hatched area shows the outer tidal excursion.

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

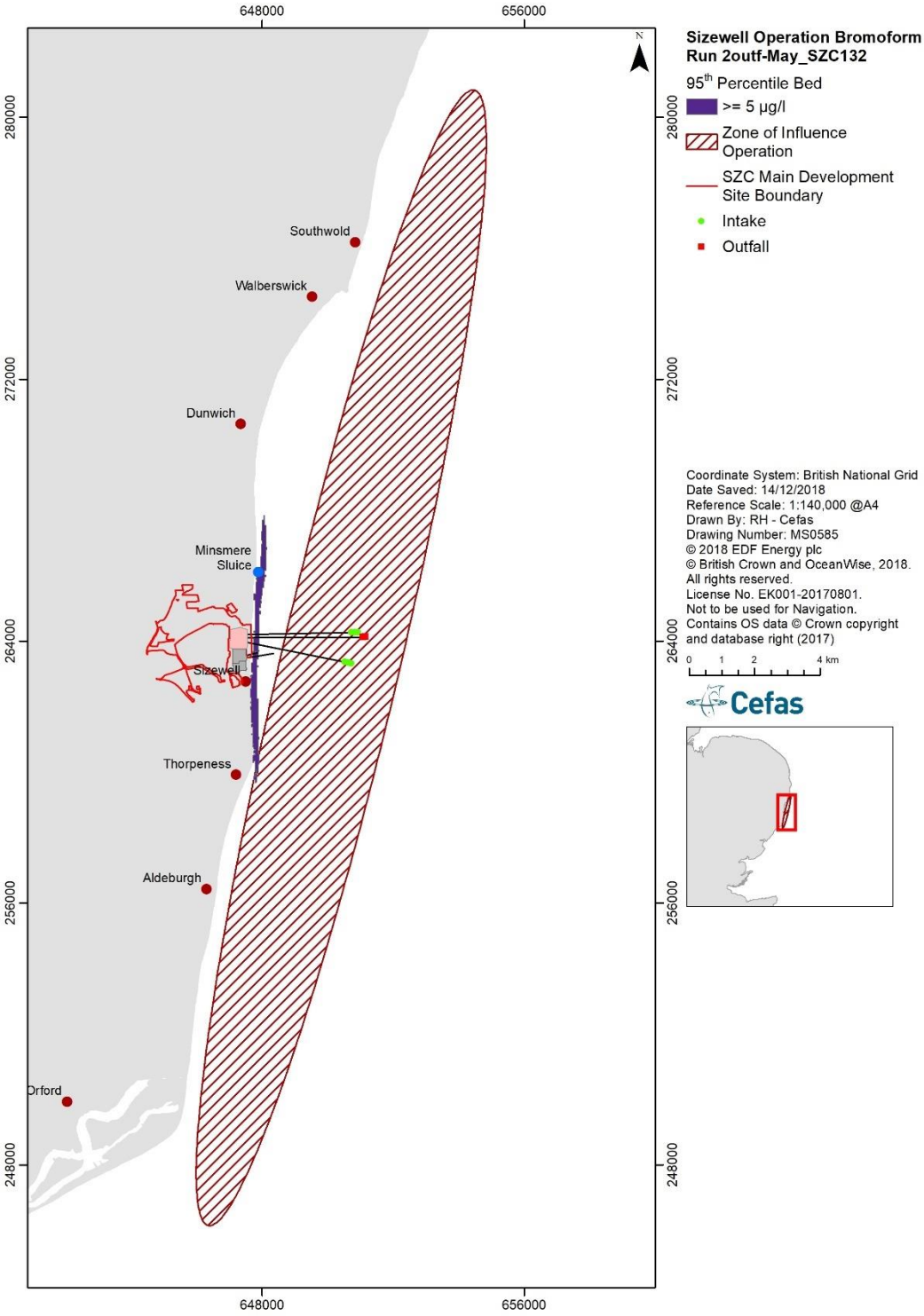


Figure 13. 95th percentile of the Bromoform concentration at the bottom for chlorination from SZB (run Brom_SZB_May-29).

## 11.4 Hydrazine assessment

There is evidence that hydrazine is harmful to aquatic organisms at low concentrations with the lowest acute six-day EC50 of 0.4µgl$^{-1}$ for growth inhibition of a marine alga, *Dunaliella tertiolecta*. Hydrazine persistence in the marine environment is low to moderate dependent upon its concentration and the water quality. There is no established EQS for hydrazine and so a chronic PNEC (Predicted No-Effect Concentration) of 0.4ngl$^{-1}$ has been calculated for long term discharges (calculated as the mean of the concentration values) and an acute PNEC of 4ngl$^{-1}$ for short term discharges (represented by the 95$^{th}$ percentile).

In this report the daily discharges from the Sizewell C secondary circuit have been modelled corresponding to an annual hydrazine discharge of 24kg per annum into the cooling water flow. For more detailed modelling it is assumed that the proposed Stream D annual discharge of hydrazine 24.3 kgy$^{-1}$ is discharged over 365 d i.e. no outages and a daily mean hydrazine discharge of 66.6 g into a 116 cumecs CW flow (concentration in treatment tank of 0.089 or 0.044mgl$^{-1}$ depending on whether one or two holding tanks are used and assumes worst case daily discharge volume). To understand the impact of different discharge rates from the treatment tanks and assuming no treatment, two discharge scenarios were studied for SZC: the first one considering a hydrazine discharge of 69ngl$^{-1}$ in daily pulses of 2.32h starting at 12pm, and the second one of 34.5ngl$^{-1}$ of hydrazine discharged in daily pulses of 4.63h duration starting at 12pm. The amount of mass that is released in each of these scenarios is the same. Due to the pulse-like discharge, the interpretation of the short-term results (daily) is biased to the moment of the tidal cycle when hydrazine has been released. In order to minimize this aliasing with the tidal signal, the simulation period has been fixed to 28 days (from the 1$^{st}$ of May to the 29$^{th}$ of May), which corresponds to two complete tidal cycles.

Hydrazine is modelled by using an empirical decay formulation derived in the laboratory and coupled into the GETM Sizewell model (BEEMS Technical Report TR145 and updated in TR352). The derivation of this decay constant has proved problematic in the past because of limitations in the stability and sensitivity of analytical methods for the measurement of hydrazine in seawater. The experiments described in TR145 used an analytical method with a limit of detection of approximately 10µgl$^{-1}$ and therefore had to perform decay experiments using initial hydrazine concentrations of 50 – 300µgl$^{-1}$ which are considerably greater than the estimated concentration of the daily discharges from Sizewell C. These experiments produced an estimated hydrazine half-life of 12-35 hrs which in agreement with previous reported work and was used in the modelling reported in Technical Report TR303. Previous work by Cefas and others has obtained indications that the half-life of hydrazine in seawater is concentration dependent however it has previously not been possible to confirm that the half-life continues to fall at concentrations of less than 100ngl$^{-1}$. More recent work has been conducted using a proven method developed by Cefas that has a Limit of Detection of 5ngl$^{-1}$. A more extensive set of studies has now shown that for concentrations of hydrazine between 30-3000 ng l$^{-1}$, the decay rate of hydrazine in Sizewell sea water follows first-order kinetics and has a half-life of 38 minutes. This work is reported in TR352.

In BEEMS Technical Report TR303 each hydrazine model run was for 28 days (two tidal cycles) and the mean and 95$^{th}$ percentile of the hydrazine concentrations was extracted. Table 37 presents the area of the plume that exceeds both concentration thresholds. For completeness, not only the chronic and acute PNEC values were included, but also other values between 0.1 and 0.5ngl$^{-1}$ for the chronic concentrations and between 1 and 5ngl$^{-1}$ for the acute concentrations.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 37: Total area of the plume at different levels of Hydrazine concentration.

| Model run | | ngl$^{-1}$ | 95$^{th}$ percentile surface (ha) | 95$^{th}$ percentile seabed (ha) | Mean surface (ha) | Mean seabed (ha) |
|---|---|---|---|---|---|---|
| Hydrazine_SZC_34ng _May – release of hydrazine in pulses of 4.63h a day starting at 12pm.<br><br>SZC only | Chronic<br><br>PNEC | 0.1 | | | 3914.09 | 3364.50 |
| | | 0.2 | | | 1269.19 | 795.85 |
| | | 0.3 | | | 389.46 | 1.46 |
| | | **0.4** | | | **156.88** | **0.34** |
| | | 0.5 | | | 66.16 | 0.11 |
| | Acute<br><br>PNEC | 1 | 446.42 | 15.14 | | |
| | | 2 | 132.54 | 0.78 | | |
| | | 3 | 54.72 | 0.00 | | |
| | | **4** | **17.38** | **0.00** | | |
| | | 5 | 1.23 | 0.00 | | |
| Hydrazine_SZC_69ng _May- release of hydrazine in pulses of 2.32h a day starting at 12pm.<br><br>SZC only | Chronic<br><br>PNEC | 0.1 | | | 4399.32 | 3788.72 |
| | | 0.2 | | | 1477.99 | 942.53 |
| | | 0.3 | | | 441.04 | 2.24 |
| | | **0.4** | | | **158.12** | **0.56** |
| | | 0.5 | | | 60.55 | 0.11 |
| | Acute<br><br>PNEC | 1 | 329.35 | 2.8 | | |
| | | 2 | 49.11 | 0.67 | | |
| | | 3 | 22.5 | 0.22 | | |
| | | **4** | **13.79** | **0.22** | | |
| | | 5 | 3.58 | 0.11 | | |

The hydrazine plume areas at the chronic PNEC (0.4ngl$^{-1}$ as an average) and the acute PNEC (4ngl$^{-1}$ as the 95$^{th}$ percentile have been calculated and are shown in Table 38.

Table 38: Absolute areas exceeding the Hydrazine PNEC.

| Model | **PNEC** | | Absolute area of exceedance | |
|---|---|---|---|---|
| | | | surface | seabed |
| Hydrazine_SZC_69ng_May mean | Chronic 0.4 ng $l^{-1}$ | ha | 158.11 | 0.56 |
| Hydrazine_SZC_34ng_May mean | Chronic 0.4 ng $l^{-1}$ | ha | 156.88 | 0.336 |
| Hydrazine_SZC_69ng_May 95$^{th}$ percentile | Acute 4 ng $l^{-1}$ | ha | 13.79 | 0.22 |
| Hydrazine_SZC_34ng_May 95$^{th}$ percentile | Acute 4 ng $l^{-1}$ | ha | 17.38 | 0.00 |

The chronic PNEC is exceeded at the surface and at the seabed, although for the seabed, an area of less than 1ha is affected for both discharge scenarios. The acute PNEC is exceeded at the surface (less than 18ha) and at the seabed, but only in the case of the 69ng$l^{-1}$ release for an area of 0.22ha.

Figure 16 and 15 show the mean predicted hydrazine concentration for a surface plume and at the seabed resulting from a daily hydrazine discharge of 69ng$l^{-1}$ from Sizewell C. Figures 16 and 17 show the 95$^{th}$ percentile hydrazine concentration at the surface and at the bed also for a daily discharge of 69ng$l^{-1}$ hydrazine.

### 11.4.1 Hydrazine discharge one EPR operational and one in commissioning

The timing of when the cooling water system is fully operational relative to the commissioning of the EPRs is not confirmed at this time. A worst-case assessment for discharge of hydrazine during commissioning with all discharges via the CDO is already made in section 7. Here the assumption is made that one operational EPR and one undergoing commissioning both discharge via the cooling water system. Adopting the discharge scenario with one secondary circuit wastewater storage tank (750$m^3$) discharging into the cooling water flow of 116$m^3sec^{-1}$ (this assumes one EPR operational with lowest volume water extracted under operational conditions) and a 50% hydrazine load discharge via one EPR results in a discharge concentration of 34.45ng$l^{-1}$ (69ng$l^{-1}$/2). Added to this discharge would be that from wastewater storage tank receiving hydrazine during commissioning and treated to a discharge concentration of 15µg$l^{-1}$ at a discharge rate of 83.3$l^{-1}sec^{-1}$ (adding an additional 10.77ng$l^{-1}$ to the cooling water discharge). The total discharge under this scenario would be 45.22ng$l^{-1}$. Discharges of hydrazine under this scenario are well represented by the scenarios that are already modelled for the operational discharges and so no further assessment is made here.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

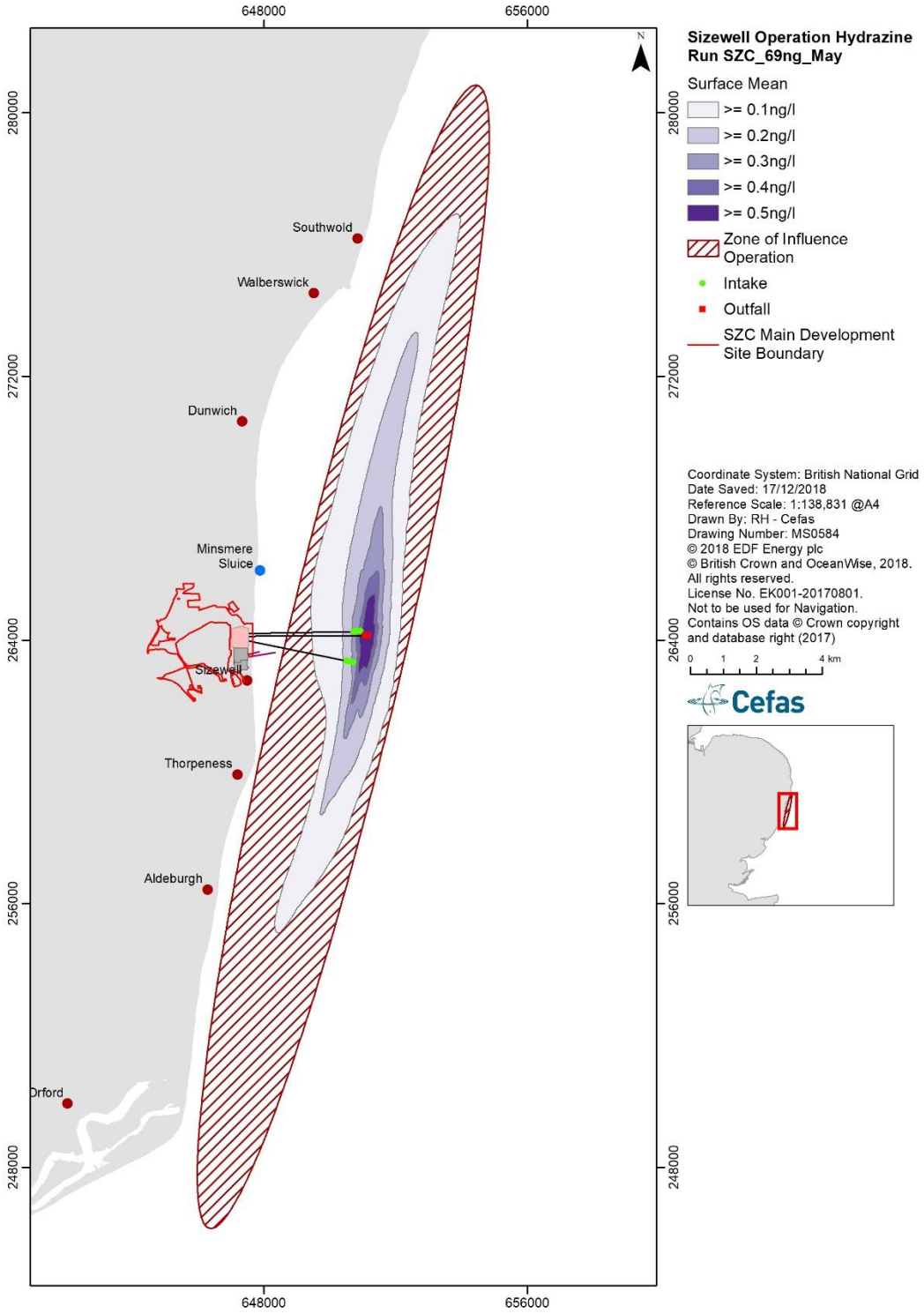


Figure 14: Mean hydrazine concentration at the surface after release of 69 ng $l^{-1}$ in pulses of 2.32h from SZC (run Hydrazine_SZC_69ng_May).

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

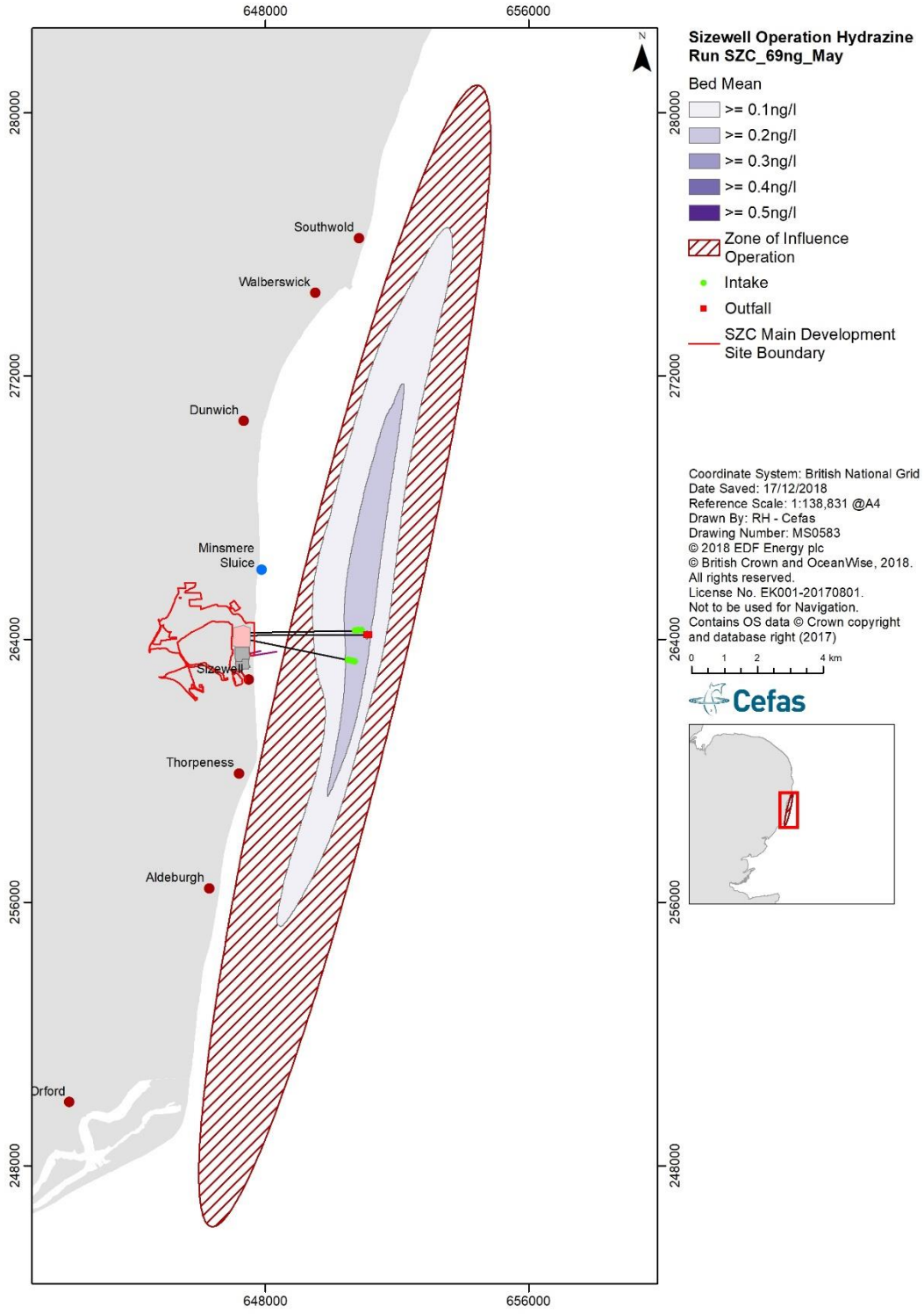


Figure 15: Mean hydrazine concentration at the seabed after release of 69 ng $l^{-1}$ in pulses of 2.32h from SZC (run Hydrazine_SZC_69ng_May).

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

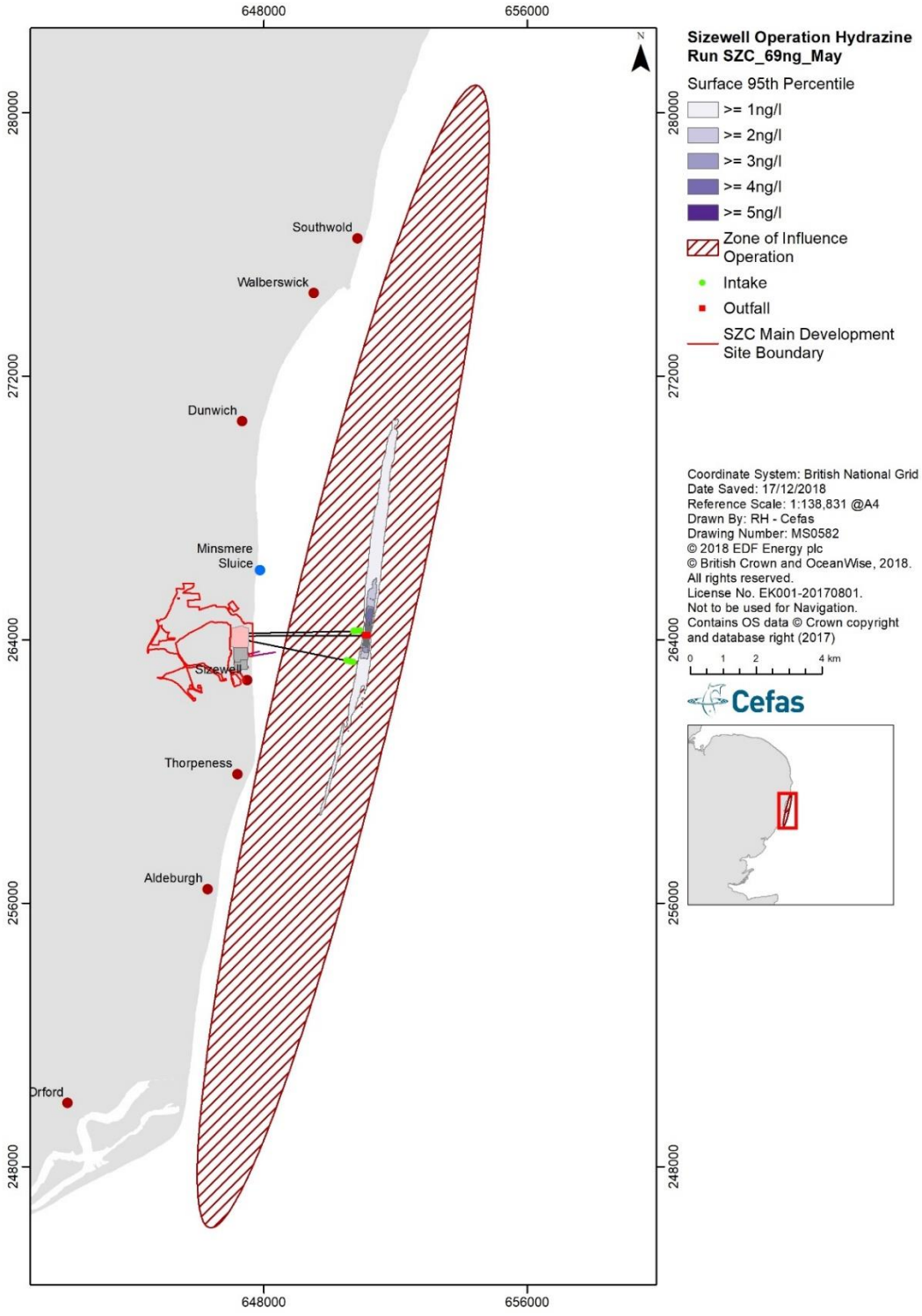


Figure 16: 95th percentile hydrazine concentration at the surface after release of 69ngl$^{-1}$ in pulses of 2.32h from SZC (run Hydrazine_SZC_69ng_May).

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

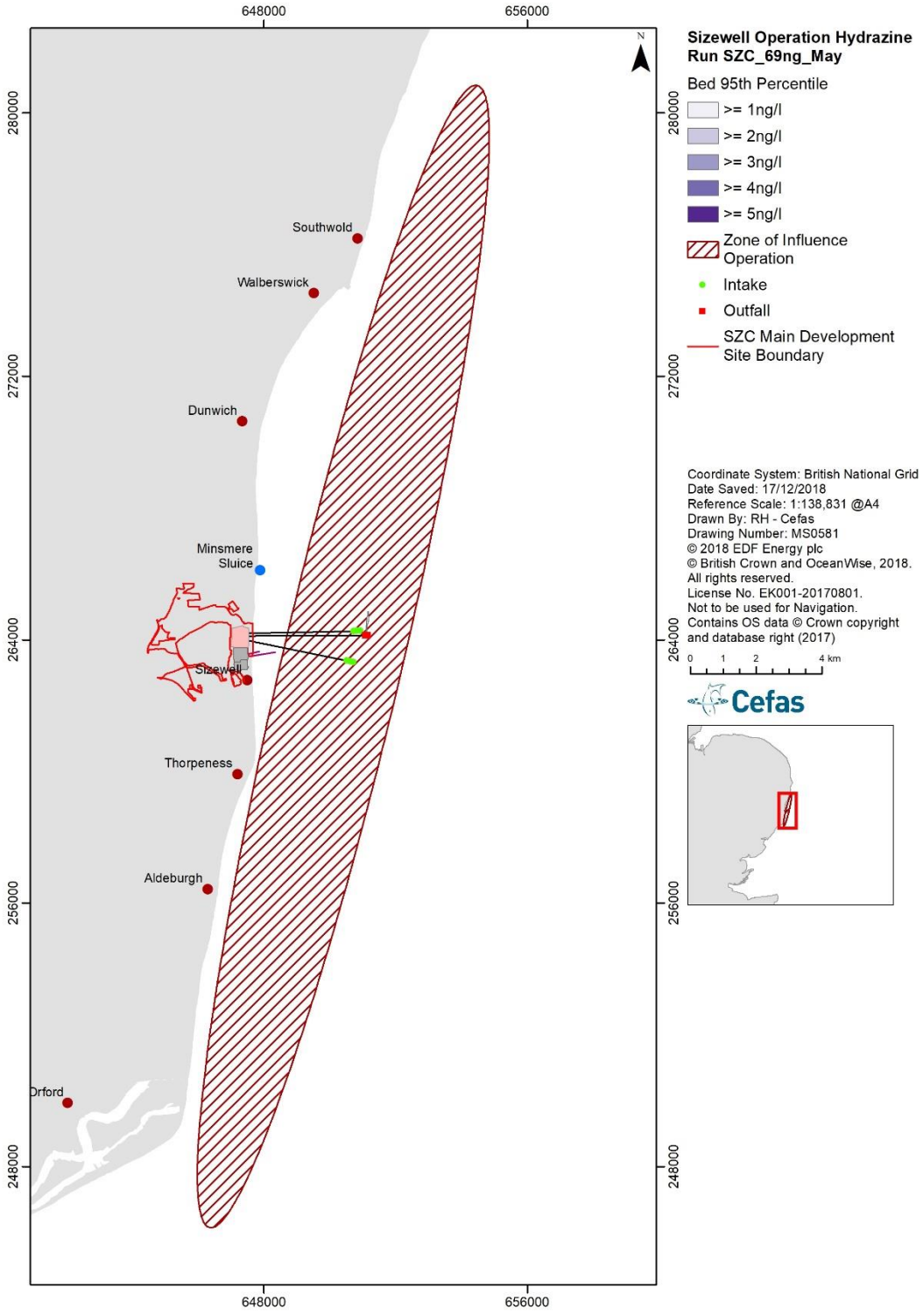


Figure 17: 95th percentile concentration at the seabed after release of 69 ng $l^{-1}$ in pulses of 2.32h from SZC (run Hydrazine_SZC_69ng_May).

## NOT PROTECTIVELY MARKED

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

### 11.5 Proportion un-ionised ammonia in discharge as influenced by temperature

In the operational phase SZC will discharge ammonia from plant conditioning chemicals and the on-site sewage treatment plant. The maximum annual discharge of nitrogen (as ammonia ions $NH_4$) from circuit conditioning for two EPRs is 13,009kg and the worst case sanitary loading during an outage is calculated to be 1,387kg giving a worst case ammonia discharge of 14,396kg which gives a calculated mean ammonia discharge concentration of 3.9µgl$^{-1}$ (3.06µgl$^{-1}$ $NH_4$-N) at the outfall assuming a worst case cooling water discharge rate of 116m$^3$s$^{-1}$ (This is the lowest volume of water abstracted under normal operating conditions and represent a worst-case scenario in terms of dilution of contaminants in the CW stream). As a conservative assumption this value has been added to the regional background mean and 95$^{th}$ percentile values to derive the un-ionised ammonia calculation.

Un-ionised ammonia concentrations have been calculated using the Environment Agency provided calculator (Clegg et al.,1995) using the GETM output for temperature elevation due to SZB+C and SZB alone and observed values for background temperature, salinity, pH and background ammonia levels. The regulatory approach for ammonia considers an annual average. The model runs replicate an annual cycle. Results have therefore been derived using an average temperature and average ammonia values. The derived un-ionised ammonia concentrations shown in the top five rows of Table 39 result in even the maximum un-ionised ammonia values being very low, 0.52µgl$^{-1}$ under the influence of the thermal input for SZB+SZC and 0.5µgl$^{-1}$ for that from SZB alone.

Results are also presented for more extreme conditions likely to result in highest un-ionised ammonia with 95$^{th}$ percentile background temperatures and mean ammonia, pH and salinity, and also 95 percentile values of pH and ammonia and the 5 percentile value of salinity with mean temperature and it is these that are also shown in Table 39.

A summary of the annual mean increases in un-ionised ammonia concentration predicted at the surface for Sizewell Bay Table 39 shows that in all cases (including worst cases) for un-ionised ammonia no areas in the model domain exceed the EQS of 21µgl$^{-1}$ as an annual mean.

Table 39: Summary of relative proportion of model domain areas associated annual mean increase of un-ionised ammonia concentration (EQS is 21µgl$^{-1}$ as an annual mean) at the surface

| **Un-ionised ammonia for mean temperature, mean ammonia, pH, salinity** | | |
|---|---|---|
| | Sizewell B + C | Sizewell B |
| 50 percentile | **0.25** | **0.25** |
| 95 percentile | 0.27 | 0.26 |
| 99 percentile | 0.29 | 0.27 |
| Maximum | 0.52 | 0.50 |
| **Un-ionised ammonia for 95 percentile temperature, mean ammonia, pH, salinity** | | |
| | Sizewell B + C | Sizewell B |
| 50 percentile | 0.8 | 0.46 |
| 95 percentile | 0.8 | 0.47 |
| 99 percentile | 0.9 | 0.52 |
| Maximum | 1.2 | 0.91 |
| **Un-ionised ammonia for mean temp, 95 percentile ph, ammonia , 5 percentile salinity** | | |
| 50 percentile | 0.8 | 0.81 |
| 95 percentile | 0.8 | 0.83 |
| 99 percentile | 0.9 | 0.88 |
| Maximum | 1.61 | 1.55 |

During operation the use of hydrazine, morpholine and/or ethanolamine have the potential to contribute to the ammonia input to the marine environment. Hydrazine breakdown during operation or subsequently

during holding and potentially treatment before discharge may result in nitrogen loss to the atmosphere and ethanolamine/morpholine may not readily breakdown to directly release ammonia however estimated maximum ammonia inputs from combined loadings of hydrazine, ethanolamine and morpholine could contribute a 4% increase on the annual loading. This additional potential loading is sufficiently small addition in terms of the assessment results Table 39 so as not to be of significance to background elevation.

## 11.6 DIN in operational discharges

During operation, the maximum number of people on site occurs when there are refuelling outages, during this time nitrate and phosphate loads are increased above background concentrations. The refuelling outages typically last four to six weeks but can occur at any time of year. During the winter period light is limiting and there is no effect resulting from the additional supply of nutrients. It is only in summer that the discharge needs to be considered. During operation the maximum 24-hour loading of nitrogen from all sources is 332kg and the maximum annual loading 11,725 kg per year (32.1kg $d^{-1}$). During the operational phase, maximum daily loading for nitrogen therefore reach approximately 2% of the daily exchange for Sizewell Bay, but the average daily value is low at 0.2% (again indistinguishable from background levels) (BEEMS TR385). The effect of SZB and the proposed SZC on phytoplankton that pass through the power station has been simulated using a phytoplankton box model. The observed cycle of plankton production has been simulated with emphasis on the spring bloom and summertime production. During operation the power stations discharge nutrients in the form of phosphate and nitrates resulting from the use of conditioning chemicals and the discharge of treated sewage. The influence of power station chlorination upon phytoplankton survival is also incorporated into the model and dominates the overall effects. For much of the year light availability limits phytoplankton growth and the addition of relatively small quantities of nutrients has no effect. In the summer, nitrate is a limiting nutrient (when light is not limiting) and is consumed rapidly. However, the exchange with the wider environment is much greater than the maximum proposed discharges, during operation so that no change in phytoplankton growth beyond natural variability would be observed. A Combined Phytoplankton and Macroalgae model Box model (BEEMS TR385) run over an annual cycle and incorporating nitrogen and phosphorus inputs showed an insignificant increase in carbon levels (phytoplankton biomass) of 0.11% within the Greater Sizewell Bay. Overall carbon levels decrease ca., 5% due to entrainment mortality and the added nutrients has a very minor influence on this.

During operation the use of hydrazine, morpholine and/or ethanolamine have the potential to contribute to the nitrogen input to the marine environment. Hydrazine breakdown during operation or subsequently during holding and potentially treatment before discharge may result in nitrogen loss to the atmosphere however estimated maximum nitrogen inputs from combined loadings of hydrazine, ethanolamine and morpholine could contribute 1.3kg/day. This additional potential loading is small relative to the 32kg from other sources and would be insignificant relative to the daily exchange and would not be expected to influence phytoplankton growth above that predicted for other operational inputs of nitrogen.

## 11.7 Phosphate discharge assessment

Phosphorus also passed the screening assessment but had one of the higher values in screening test 1 based on 24-hour loadings (352.5kg as $PO_4$). Converting this loading to $PO_4$-P gives a value of 115kg. The site background $PO_4$-P concentration is 33.5 $\mu g\ l^{-1}$. A predicted $PO_4$-P daily exchange in summer between Sizewell Bay and outer tidal excursion and the wider area is 2440kg (BEEMS TR385) therefore the planned daily $PO_4$-P loading from SZC would represent ~5% of this value but the average daily value is very low at 0.03%. There is no equivalent EQS value for phosphorus and it is not normally the limiting nutrient in marine waters, and the discharge concentration is also below background concentrations for offshore waters based on mean winter nutrient concentrations in Atlantic seawater (Foden *et al*., 2009). Although phosphorus is not normally the limiting nutrient for phytoplankton growth in marine systems a further modelled assessment of phytoplankton growth as influenced by operational inputs of phosphorus was conducted BEEMS TR385 to confirm whether there is likely to be any impact during the more critical summer period. When maximum daily inputs of phosphorus and nitrogen during the operational period were assessed using a phytoplankton box model a run conducted over an annual cycle showed an insignificant increase in carbon levels (phytoplankton biomass) of 0.11%.

## 11.8 Biochemical oxygen demand (BOD) assessment for operation

BOD loadings assessed during operation take account of maximum staff numbers on site during an outage based on HPC this is estimated as 1900 staff. The waters off Sizewell are well mixed vertically. Reduction of oxygen concentration will only occur if the rate of consumption due to BOD is greater than the resupply of oxygen through the estimated 10% daily exchange for GSB together with the oxygen transfer across the water surface.

The maximum daily BOD loading based on 1900 staff on site during an outage is 3.8kg. Every 1.5mgl$^{-1}$ BOD is estimated to result in 0.5mgl$^{-1}$ oxygen use (OSPAR Comprehensive studies report, 1997). Therefore 3.8kg BOD would be equivalent to (3.8/3) 1.26kg oxygen required. At a mean salinity adjusted background oxygen concentration of 6.27mgl$^{-1}$ a total of 1.26kg oxygen would be present in 183m$^3$. Daily water exchange for GSB is ca., 36 million m$^3$ so in relative terms the demand is very small. Typical values of oxygen flux are 100mmol m$^{-2}$d$^{-1}$ (Hull, 2016) or 3.2gm$^{-2}$d$^{-1}$. This amount of oxygen would also be equivalent to that transferred across just over 1000m$^2$ in a day. After mixing in the cooling water this loading is not expected to show measurable change in BOD background. Therefore, DO is likely to remain at high status. The discharge of BOD during operation is therefore considered to be of negligible significance for dissolved oxygen modification.

## 11.9 Assessment of coliforms, and intestinal enterococci

During operation the maximum number of staff on site is estimated at 1900 (with 100l$^{-1}$ per head per day effluent production) based on HPC and on numbers present during an outage. Mixing of the treated sewage effluent with the cooling water flow from one EPR (66m$^3$ s$^{-1}$) will achieve a dilution of ca.,33000. Assuming the same level of treatment is achieved during operation as for the construction period then application of secondary treatment alone will achieve compliance with the bathing water standards at the point of discharge.

## 11.10 Fish Recovery and Return modelling assessment

For the following assessments the monitoring data off Sizewell B provides the background parameters against which the inputs estimated from Sizewell C are considered and hence Sizewell B inputs are already factored in.

A Fish Recovery and Return system (FRR) is planned to provide a safe return of the more robust organisms from the drum screens directly into the marine environment. Both band and drum screens will be integrated within the Fish Recovery and Return (FRR) system. Intakes and tunnels will not be chlorinated. Chlorination first occurs after the drum and band screens but routing of the water sources that supply the FRR will mean that it is not chlorinated.

This section describes the impacts associated with the operation of the unmitigated FRR (alternative head designs are being evaluated and these would reduce impingement numbers, so the present assessment is very conservative). The FRR system is designed to minimise impacts on impinged fish and invertebrate populations. However, some species such as clupeids are highly sensitive to mechanical damage caused by impingement on the screens and incur high mortality rates. The return of dead and moribund biota retains biomass within the local food web but represents a source of organic carbon with the potential to enhance secondary production of carnivorous zooplankton and through the detrital pathways. In addition to organic loading, the potential for increases in nutrients, un-ionised ammonia concentration and reductions in dissolved oxygen are considered.

### 11.10.1 Calculation of moribund biomass discharge from the FRR and potential nutrient input and influence on un-ionised ammonia, and dissolved oxygen levels

The total biomass of moribund biota that potentially may be discharged from the FRR is estimated based on the level of abstraction (pump rates) for the planned Sizewell C intakes and the information on seasonal distribution of species and length weight distribution of the species impinged for the existing Sizewell B (BEEMS TR339 and TR381). The derived Sizewell C data indicate that the highest biomass discharged occurs during the months December to April. An average derived mean daily biomass for the year of 1065.5kg per day is predicted to be discharged from the FRRs. Between April to September a period more critical for potential nutrient enrichment the average daily biomass is much lower at 405.2kg per day.

#### *11.10.1.1 Nutrient inputs*

The recycling of nutrients from decaying fish biomass has been more frequently considered for freshwater systems e.g. decay of salmon carcasses in headwater streams. Several studies on salmonids indicate on a wet weight basis a phosphorus content of around 0.5% and nitrogen content of around 3.5% (Storebakken *et al*., 2000, Walker *et. al*., 2011 and Gende *et al*., 2004). The April to September period represents a time when sea temperatures and light levels at depth are increasing and phytoplankton growth is also increasing. At this time nutrients start to become less available and become a limiting factor for algal growth.
The potential decaying biomass between April to September has a mean value of 405.2kg per day during this more critical spring/summer period. Based on the percentage of nitrogen and phosphorus released per unit quantity of tissue with values derived from several studies, a maximum daily loading of ~14kg N and ~2kg P is indicated (see Appendix H).

Average daily nitrogen loading from operational inputs at SZC is 32kg which represents 0.2% of the daily exchange for Sizewell Bay. The additional inputs of N from decaying biomass represent an increase to a value of 0.3% of the daily exchange.

The daily average operational phosphorus loading is low at ca., 0.03% of the daily exchange for Sizewell Bay and the biomass input from the FRR represents a relatively high addition to this. Nevertheless, the P value only increases to ca., 0.1% of the daily exchange value for combined operational and FRR inputs.

Highest biomass discharge is predicted from January to March with lowest values during the spring summer period. However, nutrients derived from biomass during the winter period would not directly contribute to phytoplankton growth due to light limitation and lower temperatures. However, to provide a conservative assessment of potential nutrient inputs, values were derived based on the annual average biomass (1065.5kg). The predicted nitrogen and phosphorus loadings were 37.3kg per day N and 5.3kg per day P. These derived annual values for the FRR were combined with the predicted daily inputs during operation and used as source values in the Combined Phytoplankton and Macroalgae Model (BEEMS TR385). A model run over an annual cycle predicts a less than 0.29% difference in annual gross production of carbon and this level of change would not be discriminated above natural background variation. Overall carbon levels decrease ca., 5% due to entrainment mortality and the added nutrients have a small influence on this. The additional inputs of N and P from decaying biomass represent an increase to a value of 0.4% and 0.3% of the daily exchange, respectively,

This basic assessment is a worst case as it assumes that the fish are not consumed by other species and that the tissue nutrient content makes a direct contribution to nutrient levels when in fact it will take several days for the tissue to decay and to release nutrients. This assessment is conservative assuming rapid release of nutrients from the total biomass, therefore the nitrogen and phosphorus increase and potential contribution to phytoplankton growth is evaluated as negligible. The input loading of phosphorus and nitrogen from biomass discharged from the FRR is predicted to have a negligible effect on water quality separately and in combination with the operational input and which is not significant (more detail of the calculations is provided in Appendix H).

#### 11.10.1.2 Un-ionised ammonia inputs

Consideration is also made of the un-ionised ammonia contribution from decaying biomass. Studies on tissue of cod show ammonia contribution of 125mg $kg^{-1}$ $NH_4$-N (Timm and Jorgensen, 2002). This value is used as a proxy in the un-ionised ammonia calculator (Clegg and Whitfield, 1995), (along with relevant site background conditions for pH, temperature and salinity) to indicate the potential un-ionised ammonia contribution from decaying biomass at Sizewell. Based on the daily average biomass of fish discharged during the period April to September (and average pH, salinity and temperature) the estimated $NH_3$ loading could be at or above the EQS ($NH_3$-N, 21$\mu gl^{-1}$) over an area of 1.2ha around the FRR (including natural background and maximum predicted $NH_3$-N background from SZC operation with thermal elevation, 1.61$\mu gl^{-1}$, section 11.5). If the calculator input values are adjusted to consider 98$^{th}$ percentile temperature and 95$^{th}$ percentile pH which may occur during the summer period, the area of exceedance increases to 3.8 ha. Considering maximum predicted daily biomass from the FRR during March (3442kg) adjusted for an average March temperature (6.09°C, Cefas, 2013 and BEEMS TR131) an area of 6.7 ha would exceed the EQS.

#### 11.10.1.3 Influence on dissolved oxygen levels

The decaying fish biomass is also likely to contribute to the biological oxygen demand (BOD). An estimate of BOD loading of 3.5 g/g dry mass is derived based on BOD loadings from a study of particulate organic matter from fish cages (Stigbrandt et al., 2001). The source BOD value is used to derive an estimate of the BOD contribution from the daily average biomass (Based on annual mean biomass, 1065.5 kg). The estimate is 1342.5 kg BOD/day.

Any area that exceeds 1.5 $mgl^{-1}$ deviation in BOD from background is expected to generate less than 0.5$mgl^{-1}$ impact/reduction on dissolved oxygen (OSPAR Comprehensive studies report, 1997). Therefore, dividing the BOD loading by 1.5 and multiplying by 0.5 produces an estimate of the total oxygen reduction potential due to the BOD input which is 447.5 kg/day.

Based on a background concentration of 6.96 mg/l dissolved oxygen the calculated $O_2$ demand requirement (447.5 kg) is equivalent to oxygen available in 64,297 $m^3$. This volume represents 0.2% of the daily exchange for GSB.

Reaeration at the sea surface would also replenish oxygen levels. Typical values of oxygen flux are 100mmol $m^{-2}d^{-1}$ (Hull, 2016) or 3.2 g $m^{-2}d^{-1}$ therefore daily reaeration across 13.98 ha would be expected to compensate for the estimated daily oxygen consumption by decaying fish biomass.

During March when the highest daily biomass discharge would be predicted to occur via the FRR (3442 kg) oxygen demand would increase to 0.6% of that available from daily exchange and would be equivalent to reaeration over 45.2 ha.

This assessment assumes direct breakdown of all available biomass and no losses through predation. Reduction of oxygen concentration will only occur if the rate of oxygen use due to BOD is greater than daily exchange of the GSB and the oxygen transfer across the water surface.

Therefore as waters off Sizewell are well mixed vertically facilitating reaeration at the surface, the GSB has a relatively high exchange and the background dissolved oxygen levels are high there is unlikely to be a significant extent or duration of oxygen reduction associated with the input loading of BOD from biomass discharged from the FRR. BOD associated with moribund fish is therefore predicted to have a negligible effect on water quality which is not significant. There was no indication of low oxygen levels in monitoring surveys at Sizewell and average background BOD is 2$mgl^{-1}$ and this is assumed to include the influence from SZB.

# 12 Summary and Conclusion

## 12.1 Background

The aim of this report is to assess the potential impacts of the Sizewell planned New Nuclear Build on the water quality within the local marine environment and to provide information that will support the assessment and setting of a discharge consent by the statutory regulator (Environment Agency) under the Environmental Permitting Regulations.

For marine discharges, the standard approach for determining the potential impacts to water quality from industrial aqueous discharges is to apply the Environment Agency/Defra screening of contaminant contributions from surface drainage sources (Environment Agency's H1 Environmental Risk Assessment Defra and Environment Agency Guidance, 2016).

The H1 screening methodology is applied here to identify any proposed chemical discharges that represent a potential risk to the marine environment including those which are then subject to detailed modelling to fully evaluate the acceptability of the discharge.

To assess the significance of specific chemical discharges the H1 methodology uses as its reference existing Environmental Quality Standards (EQSs). Where no EQS is available for a given substance then available toxicity test data are used to generate a Predicted No Effect Concentration (PNEC) as a reference for short term acute exposure and longer-term chronic exposure. Where insufficient or no toxicity data can be sourced then the marine background concentration for a substance from monitoring conducted adjacent to the Sizewell site is used as a point of reference.

The Greater Sizewell Bay (GSB) is considered as the initial reference area for the study site extending to Walberswick in the north with the southerly extent at the apex of the Thorpeness headland in the south. The seaward boundary extends to the eastern flank of the Sizewell-Dunwich Bank and includes the proposed cooling water infrastructure on the east side on the bank. Sizewell C site discharges from the combined drainage outfall (CDO) which would include those from the construction site and potentially those during cold commissioning and from the Fish Recovery and Returns (FRRs) would also occur within the GSB.

## 12.2 Construction discharge assessment

Temporary and variable discharges to marine water will form part of the surface drainage strategy during the construction phase. The main expected contaminants in these discharges are suspended solids, hydrocarbons, Biochemical Oxygen Demand (BOD), some metals from groundwater sources and ammonia. Sediment in site drainage water will be managed with appropriate technology (e.g. Siltbuster) so that suspended sediment in discharges from the construction site are unlikely to affect the water quality status. Hydrocarbons can be removed from effluent prior to discharge by the incorporation of suitable oil separators within temporary drainage systems and any potential for chemical and oil spills during construction activities will be managed.

The groundwater metals contamination across the construction site varies so a dataset from 2014-2016 has been used to derive the 95$^{th}$ percentile concentration for each of the substances of concern and these are used for initial screening assessment. Groundwater discharge volumes are highest in the first 28 days so screening is conducted both for this period and for the highest volume groundwater discharge likely thereafter. For the first 28 days of construction and the following period chromium concentrations failed this initial screening and the elevated background concentrations of zinc derived from monitoring surveys of the site mean that the screening cannot be appropriately applied for zinc. So, both chromium and zinc were modelled using CORMIX US EPA supported mixing zone model and the validated Sizewell curvilinear GETM

model. CORMIX is used to predict the rate of chemical plume dilution and plume geometry from the Combined Discharge Outfall (CDO). The GETM model is a 3D hydrodynamic model with an inbuilt passive tracer to represent zinc and chromium. As a worst case, it was assumed that there was no loss of dissolved metals due to sediment absorption or biological uptake. Using these assumptions, concentrations can be scaled, as the modelled concentration was simply a function of dilution.
Both zinc and chromium were modelled for the first 28 day maximum groundwater discharge. CORMIX shows that for zinc the outfall plume would no longer be detectable within 3 m. For chromium the outfall plume would fall below the EQS within 25 m.

CORMIX output data suggest an initial dilution, for both zinc and chromium, was 47-fold at 25 m from the discharge (i.e. the same size as a single grid cell in GETM). GETM slightly under-predicts the initial dilution with the discharge volume of 124 l $s^{-1}$ entering the model surface layer. The total volume in the upper grid cell is approximately 120 $m^3$. GETM shows a 40-fold dilution in the first 25 m, meaning the plume extends slightly further. The mean surface area in exceedance of the EQS for Chromium, predicted by GETM, is 5.49 ha. For zinc, the total surface area is 0.11 ha, or 2 grid cells.

Sewage treatment will be available on site to treat sanitary waste from the workforce and treated effluent will contribute to ammoniacal nitrogen and nutrients as well as Biochemical Oxygen Demand (BOD), faecal coliform *Escherichia coli*, Intestinal Enterococci and suspended solids.

Ammoniacal nitrogen inputs to the construction discharge are from groundwater and treated sewage effluent. The percentage of ammoniacal nitrogen that takes the un-ionised form is important to assess as this has a relatively high toxicity and as such has a derived annual average EQS of 21µg$l^{-1}$ $NH_3$-N. Various water quality parameters influence the proportion of ammonia that is un-ionised in seawater and so must be considered in any assessment i.e. higher pH, temperature elevation and reduced salinity all increase the relative proportion of un-ionised ammonia.

The percentage of ammonia in the un-ionised form in the construction discharge was calculated for worst case discharge scenarios during the construction period and under mean and most extreme site values for physicochemical parameters that increase the percentage of the un-ionised ammonia. A mixing figure was used to take account of changing physicochemical conditions as the mainly freshwater discharge from the CDO mixes with seawater and becomes fully saline. The associated change in the un-ionised ammonia concentration in the construction discharge relative to its annual average EQS was also assessed against the level of mixing. The percentage mixing required to reduce the un-ionised ammonia concentration below the EQS was determined. The degree of mixing required in each case was considered in combination with the estimated dilution rates derived from the CORMIX modelling to determine the distance required to achieve un-ionised ammonia concentration below the EQS. The maximum distance to achieve a value below the EQS was 6.3 m which was for a case including treated sewage effluent only contributions.

To assess the contribution of DIN and phosphorus from the construction discharges (including additional inputs during commissioning) the predicted source inputs were combined and used in a phytoplankton box model to indicate the potential influence on phytoplankton growth. For much of the year light availability limits phytoplankton growth and the addition of relatively small quantities of nutrients has no effect. In the summer, nitrate is a limiting nutrient (when light is not limiting) and is consumed rapidly. However, the exchange with the wider environment is much greater than the maximum proposed discharges, during construction, so that no change in phytoplankton growth beyond natural variability would be observed.

A model run over an annual cycle predicts 0.13% percent difference in annual gross production (BEEMS TR385) of carbon and this level of change would not be discriminated above natural background variation.

The background Biochemical Oxygen Demand (BOD) near to the Sizewell B cooling water discharge based on monitoring has a mean value of 2 mg $l^{-1}$. Dissolved oxygen levels at the site are ‘high’ with a mean DO concentration of 7.5 mg $l^{-1}$ adjusted to an equivalent salinity of 35 this is equivalent to 6.27 mg$l^{-1}$. The waters off Sizewell are well mixed vertically. Draw down of oxygen will only occur if the rate of consumption due to BOD is greater than that which is exchanged across GSB and the oxygen transfer across the water surface.

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Typical values of oxygen flux are 100mmol $m^{-2}d^{-1}$ or 3.2 g $m^{-2}d^{-1}$. For a sewage discharge rate of 13.3 l $s^{-1}$ and BOD of 40mg$l^{-1}$ and taking account of groundwater contributions a daily BOD of 121kg was calculated which is equivalent to oxygen requirement of 40.6kg. This demand is very small relative to oxygen transferred as part of the daily exchange for GSB. Also, this amount of oxygen would be transferred across 1.2ha in a day.

Based on this assessment, the discharges of BOD during construction would be of negligible significance for dissolved oxygen modification.

Under bathing water regulations discharges containing faecal bacteria must be treated to ensure that the concentration of key indicator organisms will meet a designated standard for coastal and transitional waters for which Good status requires that the colony forming unit (cfu) counts for intestinal enterococci are ≤200 cfu/100 ml and for *Escherichia coli* are ≤500 cfu/100 ml.
The predicted numbers of *Escherichia coli* and intestinal enterococci in sewage effluent was calculated following different stages of sewage treatment with known reduction factors. The numbers after the final treatment stage were used in a modelling assessment taking account of dilution. CORMIX estimates show that the concentration of Intestinal Enterococci is likely to exceed the bathing water standard (200 cfu/100ml) only within 66m of the discharge for the 30l$s^{-1}$ case, without UV treatment (secondary treatment only). For the larger discharge volume (72l$s^{-1}$) the bathing water standards are exceeded for 460 m. With UV treatment, even at the higher discharge volume, exceedance is limited to within less than 1 metre of the discharge.

The offshore cooling water infrastructure consists of two subterranean intake tunnels and one outfall tunnel. Tunnels would be excavated by tunnel boring machines (TBMs) from land.
During the transport of spoil material, groundwater and TBM chemicals can leach from the conveyor belts and fall to the tunnel floor. Wastewater on the tunnel floor would be discharged via the CDO. Discharges would be treated with a silt-buster or similar technology to minimise sediment inputs but there would be residual concentrations of soil conditioning chemicals present.

The total discharge volume during Case E is approximately 34l$s^{-1}$ of which ca.,6l $s^{-1}$ is contributed by soil conditioning water and chemicals. It is uncertain whether similar chemical use to that planned for HPC will occur during tunnelling for Sizewell C but representative worst case use and discharge scenarios are modelled based on HPC to allow assessment of the potential influence of discharges upon water quality at Sizewell.

The underlying geology at Sizewell differs from Hinkley Point and a bentonite slurry tunnelling method is anticipated at Sizewell. Bentonite is on a list of substances that do not normally need to be strongly regulated as, from assessment of their intrinsic properties, the OSPAR Commission considers that they pose little or no risk to the environment. Although during operation of TBMs bentonite recovery systems are used (as bentonite is a valuable resource in the tunnelling process) the potential release into the receiving waters is assessed. The total volume of wastewater including groundwater generated during tunnelling is estimated as 34.3$l^{-1}$ per second and the resulting bentonite concentration would therefore be 8.8mg$l^{-1}$which was modelled using GETM. The concentration of bentonite in suspension is orders of magnitude lower than baseline suspended sediments concentrations predicted during construction, with 95$^{th}$ percentile concentrations of 10µg$l^{-1}$ restricted to sea surface areas of 10.8ha. Data on survival of organisms exposed to bentonite suspensions indicate that the limited areas affected, and the low discharge concentrations are likely to have negligible effects on water quality.

In some TBM soil conditioning applications several different surfactant chemicals may be required. The use and discharge of two surfactant chemicals the anti- clogging agent BASF Rheosoil 143 and the soil conditioning additive CLB F5 M that are planned for use with the HPC tunnelling operation and that present higher risk quotients in terms of chemical properties are modelled for Sizewell. As with the groundwater metals, the release and mixing of TBM chemicals in the construction discharge was modelled by considering them as passive tracers (no decay rate). As such, a single model run was carried out with single tracer at a

release rate of 34.3 l $s^{-1}$ with an initial concentration of 100µg$l^{-1}$. The results were then scaled to the appropriate concentrations for each chemical, as the modelled concentration was simply a function of dilution.
For the soil conditioning chemical discharges, the total Rheosoil plume areas at the EQS (40 µg$l^{-1}$ as a mean and 95$^{th}$ percentile) were calculated. There is no exceedance at the bed and only very limited areas of exceedance at the surface 1.01ha for a mean assessment. There was no exceedance of the EQS for CLB F5 M at the seabed and the area at the surface exceeding the EQS were relatively small with 3.14ha exceeding the EQS for a mean assessment.

## 12.3 Commissioning Discharge Assessment

When the cooling water system is commissioned a range of tests would be conducted and conditioning of the entire plant undertaken with demineralised water and various chemical additives. This process will generate wastewater containing several chemicals that will be discharged through the CDO.

Testing of the primary and secondary circuits requires them to be filled and flushed several times each. The maximum daily discharge volume is 1500$m^3d^{-1}$, equivalent to the contents of the two 750$m^3$ tanks that serve this waste stream. NNB GenCo proposes to empty each tank once a day, although not at the same time. No operational cooling system will be available for the disposal and dilution of commissioning phase effluents during the cold flush testing (CFT) stage for the first unit to be constructed during the phased development of the SZC site. Therefore, the only available discharge route for this wastewater stream will be through the CDO. If there is overlap in the period when each EPR is being commissioned this would increase discharge duration and load, but discharge concentration may be similar.

Four main chemicals are considered as part of the commissioning discharge. Phosphate loading and nitrogen from un-ionised ammonia addition were assessed in combination with the nitrogen and phosphorus inputs during construction using a phytoplankton growth model as described earlier. Un-ionised ammonia, ethanolamine (a water treatment chemical) and hydrazine were also screened for toxicity against their respective EQS or PNEC values in H1 Test 1. All failed the initial screening test and un-ionised ammonia and hydrazine failed the initial dilution assessment (Test 5) and so were modelled using CORMIX and GETM. Un-ionised ammonia did not exceed its EQs within 25 metres of the discharge.

Hydrazine, an oxygen scavenger used in power plants to inhibit corrosion in steam generation circuits was also modelled.

Prior to the release of hydrazine from the holding tanks, hydrazine would be treated to reduce the discharge concentration. Various treatment options are under investigation and it is anticipated that a discharge concentration of 15µg$l^{-1}$ is achievable as a representative upper bounding concentration equivalent to a 95$^{th}$ percentile. As a discharge concentration of 15µg$l^{-1}$ exceeds the EQS and fails the Test 5 dilution test this discharge concentration is modelled using GETM.

Modelling of the discharges from the commissioning of the EPRs when the cooling water system is unavailable assumes a maximum discharge rate of 83.3$l^{-1}$ per second from a total holding volume of 1500$m^3$. For the commissioning release of hydrazine, a release concentration of 15µg$l^{-1}$ released in daily pulses of 5.0 h starting at 12:00. This discharge period is enough to empty the total volume of both treatment tanks. To investigate the potential interaction of the hydrazine discharge concentration with relevant environmental sensitivities the results of both simulations are compared against three criteria: The likelihood that hydrazine enters the Minsmere Sluice and this also considers potential barriers to Eel movement; levels of hydrazine at the seabed over the Coralline Crag and the area of intersection of the acute hydrazine plume with Little Tern foraging areas.

To assess the spatial extent of the hydrazine plume and compare the resulting concentrations with the PNEC values (chronic and acute), the mean and 95$^{th}$ percentile of the hydrazine concentrations was extracted from the 31-day model run. For hydrazine the chronic PNEC value is 0.4ng$l^{-1}$ for long term

discharges (mean of the concentration values) and the acute PNEC value is 4ngl$^{-1}$ for shorter term discharges (represented by the 95$^{th}$ percentile).
The area of hydrazine concentration exceeding the derived acute and chronic PNECs is less at the bed than the surface. At the surface 12.9 and 30.5ha exceed the acute and chronic PNEC respectively. At the surface the exceedance for the 200ngl$^{-1}$ Canadian standard is 0.34 ha with no exceedance at the bed.

At no times are concentrations of hydrazine above the Chronic PNEC 0.4 ngl$^{-1}$ at the Minsmere Sluice. at the surface or bed when hydrazine is released at 12:00 with a release concentration of 15µgl$^{-1}$. In all the cases, the plume does not stay in the vicinity of the sluice from after one high tide to the next (approximately 12h later). Since the Minsmere sluice only opens for half an hour after high tide, this means that the hydrazine plume does not coincide in time with the sluice opening. The hydrazine plume is transported northward towards Minsmere during the falling tide, meaning that the sluice water supply that is periodically used to add additional saltwater to the Minsmere salt marshes is unlikely to be exposed to hydrazine. The likelihood of any hydrazine exposure in the sluice water would also be made considerably less likely due to rapid degradation of hydrazine with half-life of ca.,30 minutes. Information on fish sensitivity to hydrazine is limited but the information available suggests that sublethal effects may occur at concentrations ca. 800,000 times higher than peak instantaneous concentrations modelled at the sluice (0.12 ng/l. The fact that the sluice opens for a half hour after high tide also means that species moving through the sluice at this time are unlikely to encounter peak concentrations. The short residence time of the hydrazine plume in proximity to the sluice and the rapid degradation rate of hydrazine also limit the likelihood of exposure of species moving via the sluice.

A similar assessment was conducted for the Coralline Crag. The model results show that at a 15µgl$^{-1}$ release concentration the chronic PNEC is not exceeded at the seabed only at a concentration of 0.46ngl$^{-1}$ for a period of 0.25h at the surface. At the seabed, the acute PNEC is not exceeded with all release concentrations, meaning *Sabellaria* (as a benthic feature) would not be exposed to acute or chronic concentrations. The peak concentration of hydrazine at the seabed, with a 15µgl$^{-1}$ release concentration, is 0.46 ngl$^{-1}$, and is below the acute PNEC and marginally above the chronic PNEC.

In the Greater Sizewell Bay, there are three breeding colonies of little terns at Dingle, Minsmere and Slaughden and Little Terns have a foraging range of 2.4 km offshore and 3.9 km north and south. The hydrazine plume never intersects with the Dingle colony to the north and the Slaughden colony to the south. At a release concentration of 15µgl$^{-1}$, the instantaneous area of intersection between the hydrazine plume and the Minsmere colony regularly exceeds 1%, for both the 12:00 and 18:00 release. Whilst the plume intersection with 15µgl$^{-1}$ release concentration regularly exceeds 1% of the foraging range, the duration of the plume is short, with concentrations exceeding the acute PNEC for no longer than 4 hours.

During the latter stages of commissioning that is hot functional testing (HFT) the objective is to test the reactor and associated systems under pressure, temperature, flow and chemical conditioning as close to normal operating conditions as practicable. Due to the current stage of the project and the long lead time until commissioning takes place, detailed information on the nature of the discharges during HFT is limited, but it is assumed that HFT can be considered as running the systems under normal operating conditions. It would therefore be expected that the assessment for operational discharges would also apply to that during HFT.

Coastal power stations require a means of chlorine dosing for biofouling control. Based upon the known risk of biofouling at Sizewell, EDF Energy would need to chlorinate the SZC cooling water (CW) system to maintain control over biofouling of critical plant. At those sites where chlorination is required, EDF Energy's operational policy for its existing UK fleet (based upon experiments and operational experience) is to continuously dose during the growing season to achieve a minimum Total Residual Oxidant (TRO) dose of 0.2mgl$^{-1}$ in critical sections of the CW plant and at the inlet to the condensers.

Testing of this system will be undertaken during the commissioning phase, but it is assumed that this would only occur once the full cooling water system was in place and operational.

## 12.4 Operational Discharge Assessment

Potential discharges to the marine environment have been assessed for the operational phase of the planned SZC. For large cooling water discharges that are discharged to estuaries or coastal waters a specific screening assessment recommended by Defra and Environment Agency, (Clearing the Waters for All, 2016) is applied.

The annual and daily load of each of the chemicals used during operation enables derivation of a predicted concentration in the cooling water discharge and this is compared to the relevant quality standard or other acceptable alternative reference for the substance. In the first phase of screening for operational chemicals chlorine and hydrazine fail screening and are assessed using more detailed modelling. As chlorination of seawater produces chlorination byproducts and bromoform was found to be the most dominant of detected in laboratory simulations using Sizewell seawater it was also modelled in the cooling water discharge.

For the daily and annual discharge assessments of the cooling water inputs during operation several other substances including metals exceed the EQS screening criteria. However, in many cases these are screened out of further assessment as they are considered to have negligible likely effects as the actual discharge concentrations are below method detection limits, the concentrations are several orders of magnitude below their EQS (or PNEC or site background values) and/or the substances have low bioconcentration potential and are readily degradable.

The thermal elevation of the cooling water discharge can also influence the proportion of un-ionised ammonia present and as this represents a potential worst case for un-ionised ammonia this was modelled in support of the un-ionised ammonia assessment.

As sewage effluent also potentially contributes to the cooling water discharge during operation the influence of the biochemical oxygen demand and the numbers of intestinal enterococci and *Escherichia coli* likely to be present after treatment relative to the bathing water standard were also assessed.

During the operational phase the requirement to treat the cooling water to prevent biofouling of the condensers results in the discharge of chlorine produced oxidants (or Total Residual Oxidants, TRO) at a predicted concentration of 150$\mu gl^{-1}$ at the outfall heads. To provide protection to the marine environment chlorine has an EQS of 10$\mu gl^{-1}$ TRO set as a maximum allowable concentration and expressed as a 95$^{th}$ percentile. The predicted TRO concentration in the cooling water discharge, based on an empirical demand/decay formulation derived from experiments with Sizewell seawater has been modelled using the GETM Sizewell model. Two scenarios were considered: chlorination of SZB plus SZC operating in combination, and chlorination of SZB only. For each model run a month-long simulation was analysed and the mean and 95$^{th}$ percentile of the TRO concentration were extracted. The total area of the plume that exceeds a concentration threshold of 10$\mu gl^{-1}$ was at a maximum for SZB and SZC operating in combination covering an area of 726ha at the surface and 167ha at the bed. For SZC alone 338ha of the surface and only 2ha at the seabed are affected at a 95$^{th}$ percentile TRO of 10$\mu gl^{-1}$.

A Fish Recovery and Return system (FRR) is planned to provide a safe return of the more robust organisms directly into the marine environment. This system will not be chlorinated and therefore no discharge assessment is required.

A consequence of the chlorination of the cooling water system is the formation of chlorination by-products (CBP's) because of complex chemical reactions in seawater. In laboratory studies carried out with chlorinated Sizewell seawater the only CBP that was detected was bromoform. Bromoform is lost through volatilization to the atmosphere, with the loss rate a function of the thermal stratification and values obtained from the scientific literature. The volatilisation loss equation was coupled into the GETM Sizewell model. Since bromoform is a product of chlorination, the same scenarios as for TRO were considered: chlorination

of SZB plus SZC operating in combination and chlorination of SZB only. For each model run a month-long simulation was analysed and the 95th percentile of the bromoform concentrations was extracted. There is no published EQS for bromoform and so a calculated PNEC of 5µ $l^{-1}$ as a 95th percentile has been used. This value was based on the results of a toxicological review and the application of Quantitative Structure Activity Relationships. A maximum of 358ha at the surface and 130ha at the seabed is affected by a bromoform concentration in the discharge plume from SZB and SZC in combination that exceeds the 5µg$l^{-1}$ PNEC. For SZC alone a total area of 52ha at the surface and <1ha at the seabed exceeds the PNEC.

Hydrazine discharges exceed the acute and chronic quality standard (PNEC) values for both 24 hour and annual loadings. The worst-case daily discharges from the discharge tanks KER (Nuclear Island tanks), TER (additional capacity tanks) and SEK (Conventional Island tanks) have been assessed in relation to an annual hydrazine discharge of 24 kg per annum into the cooling water flow. It is assumed that the annual discharge of hydrazine would be discharged over 365 days i.e. no outages and a worst case daily mean hydrazine discharge of 66.6 g into a 116 $m^3 s^{-1}$ cooling water flow (concentration in treatment tank of 0.089 or 0.044 mg$l^{-1}$ depending on whether one or two holding tanks are used). Assuming no treatment, the daily discharge concentration in the CW flow would be 69ng$l^{-1}$ over 2.3h if one tank was used or 34ng $l^{-1}$ over a 4.6h period if two tanks were used. To understand the impact of different discharge rates from the treatment tanks two discharge scenarios were studied for SZC: the first one considering a hydrazine discharge of 69ng$l^{-1}$ in daily pulses of 2.32h starting at 12pm, and the second one of 34.5ng $l^{-1}$ of hydrazine discharged in daily pulses of 4.63h duration starting at 12pm. The amount of mass that is released in each of these scenarios is the same. For each model run 28 days were analysed (two tidal cycles) and the mean and 95th percentile of the hydrazine concentrations was extracted. For hydrazine there is a chronic PNEC value of 0.4ng$l^{-1}$ for long term discharges (mean of the concentration values) and an acute PNEC value of 4ng$l^{-1}$ for shorter term discharges (represented by the 95th percentile).

The total area exceeding the chronic PNEC at the seabed (0.4ng$l^{-1}$ as an average) is less than 1ha if hydrazine is released in the short or longer pulse scenarios. At the surface the area of exceedance of the chronic PNEC is very similar for short or longer pulses (ca.,158 and 157ha, respectively).

The acute PNEC (4ng$l^{-1}$ as the 95th percentile) is only exceeded at the seabed if hydrazine is released in short pulses and then for only for 0.22ha. At the surface, the area of exceedance for both scenarios is ca.,14ha if hydrazine is released in 2.3h pulses and ca.,17ha if hydrazine is released in 4.6h pulses.

Modelling that takes account of the site background un-ionised ammonia and the calculated additional input of un-ionised ammonia in the discharge was conducted. Sizewell temperature salinity and pH data were used to simulate average and worst-case combinations with respect to the percentage of un-ionised ammonia. For average and extreme combinations tested for un-ionised ammonia show that no areas exceed the EQS of 21µg $l^{-1}$ as an annual mean. The 24-hour discharge figure for un-ionised ammonia is just over a third of the EQs at 7.92µg $l^{-1}$ but the site background concentration is low (maximum 5.2µg$l^{-1}$). For annual discharges the screening assessment passed initial assessments but to provide more detailed assessment of the thermal influence on proportion of un-ionised ammonia the mean ammonia discharge at the outfall was added to regional background mean and 95th percentile values to derive the un-ionised ammonia calculation. All cases (including worst cases) for un-ionised ammonia show that no areas exceed the EQS of 21µg$l^{-1}$ as an annual mean.

During operation, the maximum number of people on site occurs when there are refuelling outages, during this time nitrate and phosphate loads are increased above background concentrations. The refuelling outages typically last four to six weeks but can occur at any time of year. During the winter period light is limiting and there is no effect resulting from the additional supply of nutrients. It is only in summer that the discharge needs to be considered.

During operation the maximum 24-hour loading of nitrogen from all sources is 332kg and the maximum annual loading 11,725 kg per year (32.1kg $d^{-1}$). During the operational phase, maximum daily loading for

nitrogen therefore reach approximately 2% of the daily exchange for Sizewell Bay, but the average daily value is low at 0.2% (again indistinguishable from background levels) (BEEMS TR385).

For operational loadings phosphorus also passed the screening assessment but had one of the higher values in the screening test based on 24-hour loadings (352.5 kg as $PO_4$). Converting this loading to $PO_4$-P gives a value of ca.,115kg. The predicted $PO_4$-P daily exchange in summer between Sizewell Bay and outer tidal excursion and the wider area is 2440 kg (BEEMS TR385) therefore the planned daily $PO_4$-P loading from SZC would represent ca.,5% of this value but the more representative average daily value is very low at 0.03%. There is no equivalent EQS value for phosphorus and it is not normally the limiting nutrient in marine waters, and the discharge concentration is also below background concentrations for offshore waters based on mean winter nutrient concentrations in Atlantic seawater.

The effect of SZB and the proposed SZC during operation on phytoplankton that pass through the power station has been simulated using a phytoplankton box model. The observed cycle of plankton production has been simulated with emphasis on the spring bloom and summertime production. During operation the power stations discharge nutrients in the form of phosphate and nitrates resulting from the use of conditioning chemicals and the discharge of treated sewage. The influence of power station chlorination upon phytoplankton survival is also incorporated into the model.

Based on the DIN and phosphorus loading during operation the phytoplankton growth Box model run over an annual cycle showed an insignificant increase in carbon levels (phytoplankton biomass) of 0.11%. However overall carbon levels decrease ca., 5% due to entrainment mortality and the added nutrients have a very minor influence on this.

BOD loadings assessed during operation take account of maximum staff numbers on site during an outage based on HPC this is estimated as 1900 staff. The waters off Sizewell are well mixed vertically. And reduction of oxygen concentration will only occur if the rate of consumption due to BOD is greater than the oxygen replenished by the daily exchange rate of GSB and the oxygen transfer across the water surface. The maximum BOD loading is 3.8 kg per day based on a maximum 1900 staff on site this is calculated to be equivalent to an oxygen requirement of 1.26 kg which would be present in a volume of 183$m^3$. This volume is extremely small relative to the exchange for the GSB. An equivalent supply of oxygen to offset this demand would also be transferred across just over 1000$m^2$ in a day. Therefore, DO is likely to remain at high status. The discharge of BOD during operation is therefore considered to be of negligible significance for dissolved oxygen modification.

Assessment of the sewage treatment level provided by secondary treatment and assuming dilution in the flow from a single operational EPR the estimated numbers of E. coli and intestinal enterococci in the discharge will meet the bathing water standard for Good status at the point of discharge.

For the period April to September nutrient limitation is more evident and phytoplankton growth is increasing, therefore the total biomass of moribund biota that potentially may be discharged from the FRR has been estimated. The additional loading of nutrients phosphorus and nitrogen added to the waters off Sizewell by the decaying biomass are considered low enough so as not to affect the assessment of negligible influence on phytoplankton growth when considered in addition to the operational input of these nutrients. For the combined nutrient data a model run over an annual cycle predicts a less than 0.29% difference in annual gross production (BEEMS TR385) of carbon and this level of change would not be discriminated above natural background variation. However, carbon levels decrease ca., 5% due to entrainment mortality and the added nutrients has a very minor positive influence on this.

Evaluation of the daily average un-ionised ammonia loading contributed by decaying biomass following discharge from the FRR estimates that it could be at or above the un-ionised ammonia annual average EQS of 21$\mu gl^{-1}$ $NH_3$-N, (taking account of natural background and input from SZC operation with thermal influence included) over an area of 1.2ha around the FRR. During the winter period the higher biomass loadings would increase the area above the EQS to 6.7ha.

The influence of biomass decay on the BOD was also assessed and daily re aeration over an area of 14ha would be enough to meet this additional demand when considered with that of the operational discharge and this takes no account of water exchange for the Greater Sizewell Bay. Therefore, biomass decay is expected to have a negligible influence on dissolved oxygen concentration.

## 12.5 Conclusions

This report assesses the construction, commissioning and operation of two UKEPR units for the proposed SZC development.

A H1 type screening assessment together with more detailed modelling as required of the discharges during the construction and operation periods has been completed. The assessment includes representative discharges of soil conditioning chemicals that may potentially be applied during TBM operation. The results of the assessments show that resultant environmental concentrations of discharge chemicals during the construction period are likely to have a relatively localised influence on marine water quality.

During the commissioning phase the cold flush discharges from EPR unit 1 and 2 would be made via the construction drainage system to the marine environment in a low volume flow. Several chemicals are likely to be discharged during commissioning, the most significant of which is hydrazine which has a high toxicity. Based on hydrazine wastewater treatment an indicative discharge concentration and rate was modelled. The potential area affected by concentrations in excess of the acute and chronic PNEC values for hydrazine were lower at the seabed with mean values exceeding the chronic PNEC of over 30.5ha at the surface and over 12.9ha at the surface exceeding the acute PNEC as a 95$^{th}$ percentile assessment. Three assessments were made also to determine potential influence of hydrazine on the Minsmere sluice, the Coralline Crag habitat and areas of offshore bird foraging for coastally based breeding colonies. The hydrazine plume is transported northward towards Minsmere during the falling tide, meaning that the sluice water supply that is periodically used to add additional saltwater to the Minsmere salt marshes is unlikely to be exposed to hydrazine. The likelihood of any hydrazine exposure in the sluice water would also be made considerably less likely due to rapid degradation of hydrazine with a half-life of ca.,30 minutes. The passage of eels into or out of the saltmarshes via the sluice is unlikely to be affected by the presence of hydrazine as hydrazine plumes would only intersect the sluice during an ebbing tide when water levels would be falling and the sluice would be closed. The predicted peak concentrations of hydrazine in proximity to the sluice in any case are ca. 800,000 times below levels shown to cause sublethal effects in fish so eels moving to or from the saltmarshes in the vicinity of the sluice would also not be exposed to significant concentrations of hydrazine.

In terms of the Coralline Crag the peak hydrazine concentration at the seabed over the crag does not exceed the acute PNEC and only exceeds the chronic PNEC for 15 minutes a day. In the Greater Sizewell Bay, the hydrazine plume never intersects foraging areas for two of the three SPA breeding colonies of birds. Whilst the plume intersection with 15µgl$^{-1}$ release concentration regularly exceeds 1% of the foraging range for the little Tern colony, the duration of the plume is short, with concentrations exceeding the acute PNEC for no longer than 4.5 hours.

During operation the larger volume discharges for example of chlorinated cooling water, chlorination byproducts (specifically bromoform) and hydrazine have more potential for larger scale influences on the water quality of the Greater Sizewell Bay. In terms of water quality, the influence of these discharges is relatively more limited but further assessment of areas of overlap with ecology receptors will be relevant. The FRR will potentially discharge dead organisms. An assessment of potential nutrient and oxygen demand from the decaying biota indicates that in combination with operational influence on the same parameters there is low likelihood of influence on phytoplankton growth or upon water quality.

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# 14 Appendix A Groundwater analysis 2014-2016

Table 40: Levels of detection for physical parameters, inorganic chemicals and metals measured in groundwater sampled from boreholes on the SZC New Build construction site

| Determinand | Accred. | SOP | Units | LOD |
|---|---|---|---|---|
| pH | U | 1010 | | |
| Electrical Conductivity | U | 1020 | µS/cm | 1 |
| Suspended Solids At 105C | U | 1030 | mg/l | 5 |
| Biochemical Oxygen Demand Low Level | N | 1090 | mg O2/l | 1 |
| Chemical Oxygen Demand Low Level | N | 1100 | mg O2/l | 1 |
| Alkalinity (Total) | U | 1220 | mg CaCO3/l | 10 |
| Chloride | U | 1220 | mg/l | 1 |
| Ammoniacal Nitrogen | U | 1220 | mg/l | 0.01 |
| Ammonium | U | 1220 | mg/l | 0.01 |
| Nitrite | U | 1220 | mg/l | 0.02 |
| Nitrate | U | 1220 | mg/l | 0.5 |
| Phosphate | U | 1220 | mg/l | 0.05 |
| Phosphorus (Total) | N | 1220 | mg/l | 0.02 |
| Sulphate | U | 1220 | mg/l | 1 |
| Total Oxidised Nitrogen | U | 1220 | mg/l | 0.2 |
| Cyanide (Free) Low-Level | N | 1300 | mg/l | 0.005 |
| Calcium | U | 1415 | mg/l | 5 |
| Potassium | U | 1415 | mg/l | 0.5 |
| Magnesium | U | 1415 | mg/l | 0.5 |
| Sodium | U | 1415 | mg/l | 0.5 |
| Arsenic (Dissolved) | U | 1450 | µg/l | 1 |
| Boron (Dissolved) | U | 1450 | µg/l | 20 |
| Cadmium (Dissolved) | U | 1450 | µg/l | 0.08 |
| Chromium (Dissolved) | U | 1450 | µg/l | 1 |
| Copper (Dissolved) | U | 1450 | µg/l | 1 |
| Nickel (Dissolved) | U | 1450 | µg/l | 1 |
| Lead (Dissolved) | U | 1450 | µg/l | 1 |
| Zinc (Dissolved) | U | 1450 | µg/l | 1 |
| Mercury Low Level | U | 1460 | µg/l | 0.01 |
| Iron (Dissolved) | N | 1470 | µg/l | 20 |
| Total Organic Carbon | N | 1610 | mg/l | 1 |

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Table 41: Levels of detection for total petroleum hydrocarbons and polycyclic aromatic hydrocarbons measured in groundwater sampled from boreholes on the SZC New Build construction site

| Determinand | Accred. | SOP | Units | LOD |
|---|---|---|---|---|
| Total TPH >C6-C40 | U | 1670 | µg/l | 10 |
| Naphthalene | N | 1700 | µg/l | 0.01 |
| Acenaphthylene | N | 1700 | µg/l | 0.01 |
| Acenaphthene | N | 1700 | µg/l | 0.01 |
| Fluorene | N | 1700 | µg/l | 0.01 |
| Phenanthrene | N | 1700 | µg/l | 0.01 |
| Anthracene | N | 1700 | µg/l | 0.01 |
| Fluoranthene | N | 1700 | µg/l | 0.01 |
| Pyrene | N | 1700 | µg/l | 0.01 |
| Benzo[a]anthracene | N | 1700 | µg/l | 0.01 |
| Chrysene | N | 1700 | µg/l | 0.01 |
| Benzo[b]fluoranthene | N | 1700 | µg/l | 0.01 |
| Benzo[k]fluoranthene | N | 1700 | µg/l | 0.01 |
| Benzo[a]pyrene | N | 1700 | µg/l | 0.01 |
| Indeno(1,2,3-c,d)Pyrene | N | 1700 | µg/l | 0.01 |
| Dibenz(a,h)Anthracene | N | 1700 | µg/l | 0.01 |
| Benzo[g,h,i]perylene | N | 1700 | µg/l | 0.01 |
| Total Of 16 PAH's | N | 1700 | µg/l | 0.2 |

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Table 42: Levels of detection for volatile organic compounds measured in groundwater sampled from boreholes on the SZC New Build construction site

| Determinand | Accred. | SOP | Units | LOD |
|---|---|---|---|---|
| Dichlorodifluoromethane | U | 1760 | µg/l | 1 |
| Chloromethane | U | 1760 | µg/l | 1 |
| Vinyl Chloride | N | 1760 | µg/l | 1 |
| Bromomethane | U | 1760 | µg/l | 5 |
| Chloroethane | U | 1760 | µg/l | 2 |
| Trichlorofluoromethane | U | 1760 | µg/l | 1 |
| 1,1-Dichloroethene | U | 1760 | µg/l | 1 |
| Trans 1,2-Dichloroethene | U | 1760 | µg/l | 1 |
| 1,1-Dichloroethane | U | 1760 | µg/l | 1 |
| cis 1,2-Dichloroethene | U | 1760 | µg/l | 1 |
| Bromochloromethane | U | 1760 | µg/l | 5 |
| Trichloromethane | U | 1760 | µg/l | 1 |
| 1,1,1-Trichloroethane | U | 1760 | µg/l | 1 |
| Tetrachloromethane | U | 1760 | µg/l | 1 |
| 1,1-Dichloropropene | U | 1760 | µg/l | 1 |
| Benzene | U | 1760 | µg/l | 1 |
| 1,2-Dichloroethane | U | 1760 | µg/l | 2 |
| Trichloroethene | N | 1760 | µg/l | 1 |
| 1,2-Dichloropropane | U | 1760 | µg/l | 1 |
| Dibromomethane | U | 1760 | µg/l | 10 |
| Bromodichloromethane | U | 1760 | µg/l | 5 |
| cis-1,3-Dichloropropene | N | 1760 | µg/l | 10 |
| Toluene | U | 1760 | µg/l | 1 |
| Trans-1,3-Dichloropropene | N | 1760 | µg/l | 10 |
| 1,1,2-Trichloroethane | U | 1760 | µg/l | 10 |
| Tetrachloroethene | U | 1760 | µg/l | 1 |
| 1,3-Dichloropropane | U | 1760 | µg/l | 2 |
| Dibromochloromethane | U | 1760 | µg/l | 10 |
| 1,2-Dibromoethane | U | 1760 | µg/l | 5 |
| Chlorobenzene | N | 1760 | µg/l | 1 |
| 1,1,1,2-Tetrachloroethane | U | 1760 | µg/l | 2 |
| Ethylbenzene | U | 1760 | µg/l | 1 |
| m & p-Xylene | U | 1760 | µg/l | 1 |
| o-Xylene | U | 1760 | µg/l | 1 |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 43: Levels of detection for volatile organic compounds measured in groundwater sampled from boreholes on the SZC New Build construction site

| Determinand | Accred. | SOP | Units | LOD |
|---|---|---|---|---|
| Styrene | U | 1760 | µg/l | 1 |
| Tribromomethane | U | 1760 | µg/l | 1 |
| Isopropylbenzene | U | 1760 | µg/l | 1 |
| Bromobenzene | U | 1760 | µg/l | 1 |
| 1,2,3-Trichloropropane | N | 1760 | µg/l | 50 |
| N-Propylbenzene | U | 1760 | µg/l | 1 |
| 2-Chlorotoluene | U | 1760 | µg/l | 1 |
| 1,3,5-Trimethylbenzene | U | 1760 | µg/l | 1 |
| 4-Chlorotoluene | U | 1760 | µg/l | 1 |
| Tert-Butylbenzene | U | 1760 | µg/l | 1 |
| 1,2,4-Trimethylbenzene | U | 1760 | µg/l | 1 |
| Sec-Butylbenzene | U | 1760 | µg/l | 1 |
| 1,3-Dichlorobenzene | N | 1760 | µg/l | 1 |
| 4-Isopropyltoluene | U | 1760 | µg/l | 1 |
| 1,4-Dichlorobenzene | U | 1760 | µg/l | 1 |
| N-Butylbenzene | U | 1760 | µg/l | 1 |
| 1,2-Dichlorobenzene | U | 1760 | µg/l | 1 |
| 1,2-Dibromo-3-Chloropropane | U | 1760 | µg/l | 50 |
| 1,2,4-Trichlorobenzene | U | 1760 | µg/l | 1 |
| Hexachlorobutadiene | U | 1760 | µg/l | 1 |
| 1,2,3-Trichlorobenzene | U | 1760 | µg/l | 2 |
| Methyl Tert-Butyl Ether | N | 1760 | µg/l | 1 |

Table 44: Levels of detection for Polychlorinated bipheyls measured in groundwater sampled from boreholes on the SZC New Build construction site

| Determinand | Accred. | SOP | Units | LOD |
|---|---|---|---|---|
| PCB 28 | N | 1815 | µg/l | 0.01 |
| PCB 52 | N | 1815 | µg/l | 0.01 |
| PCB 101 | N | 1815 | µg/l | 0.01 |
| PCB 118 | N | 1815 | µg/l | 0.01 |
| PCB 153 | N | 1815 | µg/l | 0.01 |
| PCB 138 | N | 1815 | µg/l | 0.01 |
| PCB 180 | N | 1815 | µg/l | 0.01 |
| Total PCBs (7 congeners) | N | 1815 | µg/l | 0.01 |
| Total Phenols | U | 1920 | mg/l | 0.03 |

# 15 Appendix B Supporting ecotoxicity data for PNEC derivation

The following sections provide details of the ecotoxicological data used to inform the risk of an impact from various chemical components of site discharges during different phases of the new build power station development. The reference source for the information on hydrazine, morpholine and ethanolamine is EDF 2008.

## PNEC values for hydrazine,

The PNECs given below were obtained:

- based on bibliographic research into ecotoxicological data available in the literature in 2006,
- based on a critical review of these data and their categorisation by level of admissibility. The findings are categorised based on the living organisms studied, as well as their relevance, assessed after studying the original publication that presented them;
- depending on the various methods used, i.e. either the uncertainty factors method given in the EU Technical guidance,2003 or the species sensitivity distribution (SSD) or ACT-SSWD methods, for Acute to Chronic Transformation – Species sensitivity Weighted Distribution2,3.

This work by EDF R&D was submitted to INERIS for a second opinion in 2006. The values used are derived from this joint work. They are the same as those used by IRSN.

## Available ecotoxicity data for hydrazine

The bibliographic analysis was conducted by consulting the following databases: Aquire, Biological abstracts, Chemical abstracts, Dose-1998, ECDIN-1993, EINEC-1998, Environmental bibliography, HSDB-1997 (Hazardous Substances Data Bank), IRIS on-line (Integrated Risk Information System, U.S.EPA, http://www.epa.gov/iris/subst/0528.htm), IUCLID version 4.0-2001, Medline, OHM/TADS-1997, Pascal biomed, Pascal sciences et techniques, and Toxline. All of the ecotoxicological data regarding algae, vertebrates (fish, etc.) and invertebrates (crustaceans, etc.) have been listed.

A few of the lowest chronic and acute ecotoxicological values, selected as admissible after reading the publications, are given for each taxonomic group in the table below. A datum ultimately considered valid ($EC_{50}$ for *Dunaliella tertiolecta*) has been added. As a reminder, for the use of statistical methods, the sample of data used is larger than the one shown below.

Table 45: Acute and chronic toxicity data used for derivation of respective PNECs for hydrazine

| Species | Taxonomic group | Exposure times | $LC_{50}$-$EC_{50}$-$IC_{50}$ mg $l^{-1}$ | NOEC mg $l^{-1}$ | Authors |
|---|---|---|---|---|---|
| *Pseudokirchneriella subcapitata* | Algae (freshwater) | 72 hrs | 0.006 | 0.001 | Harrah, 1978 |
| *Dunaliella tertiolecta* | Algae (seawater) | 48 hrs | 0.0004 | | Dixon et al, 1979 |
| *Hyalella azteca - (amphipod)* | Crustacean (freshwater) | 48 hrs | 0.04 | | Fisher et al., 1980b - Anonymous, 1998 |
| *Daphnia pulex* | Crustacean (freshwater) | 48 hrs | 0.16 | | Velte, 1984 |
| *Daphnia pulex* | Crustacean (freshwater) | 96 hrs | 0.19 | | Velte, 1984 |
| Asillidae - (isopod) | Crustacean | 96 hrs | 1.3 | | Fisher et al., 1980b - |
| *Ictalurus punctatus* | Freshwater fish (vertebrate) | 96 hrs | 1 | | Fisher et al., 1980b - Anonymous, 1998 - IUCLID, 2001; - Dose, 1998; Richardson, 1992 |
| *Lepomis macrochirus* | Freshwater fish (vertebrate) | 96 hrs | 1.08 | | Fisher et al., 1978 ; Fisher et al., 1980a |
| *Notemigonus crysoleucas* | Freshwater fish (vertebrate) | 96 hrs | 1.12 | | Fisher et al., 1980b - |
| *Lepomis macrochirus* | Freshwater fish (vertebrate) | 72 hrs | 1.2 | | Hunt et al., 1981, cited by Velte, 1984 |
| Gasterosteus aculeatus | Freshwater fish (vertebrate) | 96 hrs | 3.4 | | Harrah, 1978; Klein and |
| Poecilia reticulata | Freshwater fish (vertebrate) | 96 hrs | 3.85 | | Slonim, 1977 |
| Pimephales promelas | 96 hrs | 4.5 | | Cowen et al., 1981 | 96 hrs |
| Pimephales promelas | 96 hrs | 5.98 | | Velte, 1984 | 96 hrs |
| *Ambystoma opacum* and *Ambystoma maculum(salamander)* | Amphibian (vertebrate) | 96 hrs | 2.12 | | Slonim, 1986 |
| *Ambystoma opacum* and *Ambystoma maculum(salamander)* | Amphibian (vertebrate) | 96 hrs | 4.11 | | Slonim, 1986 |

### PNEC obtained using uncertainty factors method

For this substance, INERIS currently recommends strictly applying the Technical Guidance Document, 2003 rules, and therefore the uncertainty factors method. Using this method, the chronic PNEC is obtained by applying an expansion factor of 1000 to the lowest $EC_{50}$ in *Pseudokirchneriella subcapitata* at 6µgl$^{-1}$(Harrah, 1978).

The value of chronic PNEC of hydrazine in fresh water used in the impact study is therefore 6/1000= 0.006 µg l$^{-1}$ or 6ngl$^{-1}$.

### PNEC obtained using uncertainty statistical method

Though the PNEC derived from statistical calculations is merely additional information for the analysis, it is presented anyway:

The ACT-SSWD statistical method makes it possible to calculate HC5_5%, which protects 95% of species, with a confidence interval of 95%. This HC5_5% value may be likened to a PNEC.

By using all the data about hydrazine for the three freshwater categories of algae, vertebrates (fish, etc.) and invertebrates (crustaceans, etc.), and weighting them all equally, a HC5_5% of 5.0 µg l$^{-1}$ is obtained.

The 3.7 µg l$^{-1}$ value obtained for acute HC5_5% (see next section) may seem contradictory, because it is on the same order of magnitude as the chronic value. This is explained by the presence of a value penalising algae in the data set used (EC10 72 hrs for 0.003 mg l$^{-1}$ of *Pseudokirchneriella subcapitata*). Owing to the lifespan of this sort of organism, the algae tests are chronic in nature. Strictly speaking, there is no acute test for algae. Therefore, the same data needs to be used for both acute and chronic PNEC, if this important taxon in the aquatic environment is to be taken into consideration.

As a precaution, it will therefore be assumed that the hydrazine PNEC values derived from statistical calculations are the same for both acute and chronic toxicity. The lower of the two values (which are on the same order of magnitude), i.e. 3.7 µg l$^{-1}$, is therefore used.

### Chronic seawater PNEC

As there are not enough ecotoxicological values available for hydrazine in marine species, there is no way to calculate a seawater PNEC based on them. The TGD makes it possible to use ecotoxicological data from freshwater species to assess the seawater PNEC of a substance. If so, an additional factor of 10 is applied when using the value of the freshwater PNEC in seawater, in order to take account the many uncertainties which govern the environmental impact of chemicals in marine environments (i.e. an expansion factor of 10,000, in our situation). However, owing to expert opinion (as hydrazine cannot bioaccumulate and is nonpersistent), a suggestion has been made to adopt expansion factors of 1000 and 100 instead for the chronic and acute seawater PNECs, while still holding $EC_{50}$ to be valid for 0.4 µg l$^{-1}$ of *Dunaliella tertiolecta*).

**The chronic PNEC of hydrazine in seawater used in the impact study is therefore equal to 0.4/1000 0.0004 µg l$^{-1}$, or 0.4 ng l$^{-1}$.**

### Acute freshwater PNEC

Using all of the acute data on hydrazine for the three freshwater categories of algae, invertebrates and vertebrates, and weighting each one equally, a HC5_5% (likened to a PNEC) of 3.7 µg l$^{-1}$ is obtained using the ACT-SSWD statistical method. Using the TGD's uncertainty factors method as currently recommended by INERIS, the chronic PNEC is obtained by applying an expansion factor of 100 to the lowest EC50 in 6 µg l$^{-1}$ of *Pseudokirchneriella subcapitata* (Harrah, 1978).

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**The value of acute PNEC of hydrazine in fresh water used in the impact study is therefore 6/100= 0.06 µg $l^{-1}$, or 60 ng $l^{-1}$.**

### Acute seawater PNEC

As stated above, an expansion factor of 100 is applied to EC50 for 0.4 µg $l^{-1}$ of *Dunaliella tertiolecta*.

**The value of acute PNEC of hydrazine in fresh water used in the impact study is therefore 0.4/100= 0.004 µg $l^{-1}$, or 4 ng $l^{-1}$.**

**GLOSSARY**

PNEC: Predicted No Effect Concentration
EC50: Half maximal effective concentration
IC50: Half maximal inhibitory concentration
LC50: Median lethal dose for 50% of organisms
HC: Hazard Concentration
NOEC: No Observed Effect Concentration

### Other hydrazine risk assessments

In 2010 the Canadian Ministers of environment and Health (Environment Canada, 2011) produced a draft risk assessment of hydrazine. The assessment refers to the series of studies by Scherfig et al., (1977) and Dixon *et al*., 1979 which indicated greater than 22% loss of the test substance in the static algal toxicity tests described using the marine unicellular algal species *Dunalliela tertiolecta*. Based on this the Canadian risk assessment considered only results of toxicity tests lasting up to, 96 hours in exposures without renewal of hydrazine. Tests on micro algae were also considered short-term. Information regarding hydrazine toxicity on freshwater organisms was gathered from sixteen toxicity papers ranging in date of publication from 1976 to 1984; no data published after that date were found. Only one of the sixteen papers, a study on the effects of hydrazine on two different species of algae (Scherfig *et al*., 1977), reported data from a chronic (long-term) test. Six of the papers reporting acute (short-term) data did so for fish, three for amphibians, two reported data on fish and invertebrates, and one for bacterium and protozoa. These papers in total yielded data on fourteen different species, six fish, three invertebrates, two amphibians, one algal species, one bacterium, and one protozoan. Robust study summaries were completed for all studies, and only those meeting the Canadian Council of Ministers for the Environment (CCME) reliability standards were included in the assessment. Most of the acute and short-term effect values found for marine organisms were obtained with exposures of early life stages of phaeophytes (brown seaweed) to hydrazine. The lowest effect concentration obtained with this group of algae, a 96-h Lowest Observed Effect Concentration (LOEC) of 2×10-3 mg/l, was selected as the Critical Toxicity Value (CTV) for marine environments.

Predicted No Effect Concentrations (PNEC) were derived for freshwater and marine organisms based on available empirical toxicity data. The CTV for freshwater organisms was based on the HC5 of the SSD for acute toxicity (0.026 mg/l). An assessment factor of 10 was applied to this value to estimate a long-term no-effects concentration from short-term acute endpoints, yielding a chronic PNEC of 0.0026 mg/l. For marine organisms, the CTV was the 96-h LOEC (0.002 mg/l) for the gametophyte of a Brown algae exposed to hydrazine. Similarly for freshwater organisms, an assessment factor of 10 was applied to this value to estimate a long-term no-effects concentration from short-term acute endpoints, yielding a chronic PNEC of 0.0002 mg/l.

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Toxicity studies reviewed on the European Chemicals Agency (ECHA) website (ECHA profile hydrazine) indicate three freshwater studies that evaluated for acute toxicity: for a fish species (*Lebistes reticulatus,* 96h $LC_{50}$ 0.61 mg/l) which was evaluated as reliable without restriction; for invertebrates (*Daphnia pulex*, 48h $EC_{50}$ 0.19 mg/l) evaluated reliable with restriction for algae; and for algae (*Desmodesmus subspicatus* formerly *Scenedesmus subspicatus*, 72h $EC_{50}$ 0.017mg/l and $EC_{10}$ of 0.008 mg/l) which was also evaluated as reliable without restriction. In addition to the acute data a chronic dataset for *Daphnia magna* considered reliable without restriction was also reported and produced a 21d NOEC of 0.01mg/l for reproduction.

Based on these ECHA data, a factor of 1000 would be applied to the lowest acute $EC_{50}$ to derive an acute PNEC for marine species (Table 25, Section 4.3 page 149 EU Technical guidance part II). A factor of 1000 is considered appropriate as there are three trophic level acute datasets and there are supporting studies for other marine groups, [Technical Guidance: Lowest short-term L(E)$C_{50}$ from freshwater or saltwater representatives of three taxonomic groups (algae, crustaceans and fish) of three trophic levels, + two additional marine taxonomic groups (e.g. echinoderms, molluscs) -see hydrazine datasets Table 46.

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Table 46: Supplementary acute and chronic toxicity data for hydrazine for several marine and freshwater species

| Species | Taxonomic group | Exposure times | $LC_{50}$-$EC_{50}$-$IC_{50}$ mg $l^{-1}$ | NOEC mg $l^{-1}$ | Authors |
|---|---|---|---|---|---|
| *Desmodesmus subspicatus* (previous name: *Scenedesmus subspicatus*) | Algae *(freshwater)* | 48 hrs | 0.017 (growth rate) | 0.006 (growth rate) | ECHA Registration Dossier (Currenta, 2009) |
| *Tisbe battaglai* | Crustacean (copepod) (marine) | 48 hrs | 0.006 | 0.027 | BEEMS TR175 |
| Daphnia magna | Crustacean (freshwater) | 21 days | 0.123 (EC50) | 0.01(EC10) | Daphnia magna |
| Hemigrapsus oregonensis | Crab | 96 hrs | 3.6 | | Hemigrapsus oregonensis |
| *Gasterosteus aculeatus* (three-spine stickleback) | Fish (marine) | 96 hrs | 5.4 (95% CI 4.5-6.6) | | Klein and Jenkins, 1978 a065562.pdf (dtic.mil) |
| *Gasterosteus aculeatus* (three-spine stickleback) | Fish (marine) | 336 hrs (14 day) | 1.07 (95% CI 0.78-1.47) | | Klein and Jenkins, 1978 a065562.pdf (dtic.mil) |
| *Mytilus edulis* | Mollusc (mussel) (marine) | 96 hrs | ≤5.7 No effect | | Klein and Jenkins, 1978 a065562.pdf (dtic.mil) |

Applying a factor of 1000 to the algal $EC_{50}$ results in a derived acute PNEC of 17 ng/l. For derivation of the chronic PNEC there is no appropriate marine data so as there are two chronic datasets (for freshwater/or marine species) the *Daphnia magna* chronic reproduction test result and the algal (*Desmodesmus subspicatus)* 48h NOEC of 0.006 mg/l for growth rate (which is also considered chronic data) an application factor of 500 is applied to produce a chronic PNEC of 12 ng/l [Based on EU Technical guidance marine: *Two long-term NOECs from freshwater or saltwater species representing two trophic levels (algae and/or crustaceans and/or fish) assessment factor 500*].

### Summary hydrazine risk assessment

Hydrazine decay in seawater has been demonstrated to be relatively rapid with a half-life of around 38 minutes in Sizewell seawater (BEEMS TR352). Hydrazine is not indicated to bioaccumulate based on its low bioconcentration factor (BCF) in studies with fish (Slonim and Gisclard, 1976) and its low partition coefficient (-2.07 reported in Environment Canada, 2011). However, although not persistent, hydrazine is relatively toxic. Some algal groups have been shown to be sensitive to hydrazine exposure and this may relate to the lack of a rigid cellulose cell wall (Dixon *et al.,* 1979). Algal test data for low-persistence substances is difficult to derive due to the required test media for running algal tests, meaning that a static test with non-renewal of the test solution is most frequently adopted. The inability to maintain test concentrations was the primary reason that some more recent assessments of hydrazine toxicity have rejected toxicity data for the marine species *Dunaliella tertiolecta.* It is apparent that certain groups of marine microalgal species may be more sensitive to hydrazine, but other taxonomic groups may be less so, based on the limited range of available data. Even within microalgal species, sensitivity may range over several orders of magnitude i.e. in Cefas studies *Micromonas pusilla* (which has similar cell wall characteristics to *Dunaliella tertiolecta*) had a 96 hour $EC_{50}$ of 1.27 µg $l^{-1}$ hydrazine. Two diatom species, by contrast had lower sensitivity: *Skeletonema sp.* had an $EC_{50}$ 37.61 µg $l^{-1}$ hydrazine based on the area under the growth curve (AUGC), and *Thalassiosira weissflogii* had a 72 hour $EC_{50}$ of 140.5 µg $l^{-1}$ hydrazine (AUGC). Applying the EU Technical guidance approach (Section 4.3 page 149, EU Technical guidance part II) to available robust toxicity data for hydrazine (the dataset described in Table 46) produced acute and chronic PNECs of 17 and 12ng/l respectively. However, it is recommended that the PNECs derived by the original review (EDF, 2008) of 4 ng/l (acute) and 0.4 ng/l (chronic) be retained to provide a precautionary screening assessment level to identify the need for further ecological evaluation for specific receptors.

### Available ecotoxicology data for morpholine

The PNECs given below were obtained:

- based on a bibliographic search of the ecotoxicological data available in the literature in 2001, the summary prepared by GRNC4, and three series of tests conducted by the CIT5 laboratory (at the request of EDF R&D), as the bibliographic analysis had demonstrated a lack of data;
- based on a critical review of this data and categorising them by level of admissibility. The findings are categorised based on the living organisms studied, as well as their relevance, assessed after studying the original publication that presented them;
- based on the uncertainty factors method described in the TGD6.

This work by EDF R&D was submitted to INERIS for a second opinion in 2006. The values used are derived from this joint work.

The sources of information used to gather the ecotoxicology values are the work of GRNC and the IUCLID database (2000).

As the bibliographic analysis had demonstrated a lack of data, three series of tests were conducted (at the request of EDF R&D) by the CIT laboratory (a 72-hour growth inhibition test on algae, a 21-day microcrustacean reproduction inhibition test, a 28-day offspring weight gain test on fish).

The test on *Microcystis aeruginosa* (Bringmann and Kühn, 1978) was added, because the value was ultimately considered valid after a second opinion was given.

A few of the lowest ecotoxicological values in fresh water, selected and considered admissible after reading the publication, are given for each organism in Tables 35 and 36 below.

Table 47: Chronic toxicity data used for derivation of respective PNECs for morpholine

| Organism | Species | Chronic toxicity mg $l^{-1}$ | Authors |
|---|---|---|---|
| Algae | *Selenastrum capricornutum* | 80 (NOEC 24-144hrs) | Adams et al., 1985 |
| | *Scenedesmus subspicatus* | 20 (NOEC 72hrs) | CIT,2003 |
| | *Microcystis aeruginosa* (cyanobacterium) | 1.7 (TT 8d considered to be NOEC) | Bringmann and Kühn, 1978 |
| Crustacean | *Daphnia magna* | 2.56 (NOEC 21d) | CIT,2003 |
| Fish | *Oncorhynchus mykiss* | >or= 100 (NOEC 28d) | CIT,2003 |

Table 48: Acute toxicity data used for derivation of respective PNECs for morpholine

| Organism | Species | Acute toxicity mg $l^{-1}$ | Authors |
|---|---|---|---|
| Algae | *Selenastrum capricornutum* | | Calamari et al., 1982 |
| Shellfish | *Daphnia magna* | 100(EC50 24hrs) | Bringmann and Kühn, 1978 |
| Fish | *Leuciscus idus* | 240 ($LD_{50}$ 48hrs) | Juhnke and Ludermann, 1978 |
| Fish | *Oncorhynchus mykiss* | >or= 100 (NOEC 28d) | Calamrai et al., 1982 |

## Chronic freshwater PNEC

The uncertainty factors method given in the TGD is applied, meaning that the PNEC is deduced from the listed and adopted NOEC data, and particularly the lowest one to which an uncertainty factor is applied based on the number of data points for each trophic level.

The data shows that there are three NOECs for three different trophic levels. In accordance with the TGD, a factor of 10 has been applied to the algae NOECs (the lowest of the three trophic levels), which is equal to 1.7mg$l^{-1}$ (see table above).

Thus, the chronic PNEC of morpholine in fresh water used in the impact study is 1.7/10 = 0.170mg $l^{-1}$, or 170µg$l^{-1}$.

## Chronic seawater PNEC

Regarding marine environment, only acute data for fish exist ($LC_{50}$ for 96hrs on *Chelon engeli* at 100mg$l^{-1}$ (McCain and Peck, 1976). In such cases, the TGD suggests applying an additional expansion factor of 10 to the chronic freshwater PNEC in order to set a chronic seawater PNEC.

Thus, the chronic PNEC of morpholine in seawater used in the impact study is 170/10 = **17µg $l^{-1}$**.

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## NOT PROTECTIVELY MARKED

### Acute freshwater PNEC

The uncertainty factors method described in the TGD is applied, meaning that the PNEC is deduced from the listed and adopted LD50 data, particularly the lowest one, 28mgl$^{-1}$ (EC50- 96hrs for Selenastrum capricornutum), to which an uncertainty factor of 100 is applied. Thus, the **acute PNEC of morpholine in fresh water used in the impact study is 28/100 = 0.280mgl$^{-1}$, or 280 µg l$^{-1}$.**

### Acute seawater PNEC

The TGD suggests applying an additional expansion factor of 10 to the acute freshwater PNEC in order to set an acute seawater PNEC, whenever there is too little seawater data. Thus, the chronic PNEC on morpholine in seawater used in the impact study is 280/10 = 28 µg l$^{-1}$.

### Additional information

Morpholine is readily biodegradable at 93 % after 28 days in a non-GLP Guideline study following protocol OECD 301 E (modified OECD screening test; 10 -d window kept; inoculum: effluent from municipal wwtp) [BASF AG 1990, Report No. 1901337].

According to a MITI study (corresponding to OECD 305C), morpholine does not significantly accumulate in aquatic organisms (BCF < 2.8) [CITI, 1992].

Calculated logKoc-values of 0.8666 and -0.61967 are available based on estimates from MCI and log Kow respectively (BASF SE, KOCWIN v2.00, 2010). The molecule is not expected to adsorb to suspended solids and sediment based upon the log Koc as calculated for the uncharged molecule.
https://echa.europa.eu/registration-dossier/-/registered-dossier/13364/5/3/2

**GLOSSARY**
PNEC: Predicted No Effect Concentration
EC50: Half maximal effective concentration
IC50: Half maximal inhibitory concentration
LC50 , LD0: Median lethal dose for 50% (0%) of organisms
HC: Hazard Concentration
NOEC: No Observed Effect Concentration
TT: Toxicity Threshold may be considered to be a NOEC

### Available ecotoxicology data for ethanolamine

The PNECs given below were obtained:

- based on bibliographic research into ecotoxicological data available in the literature in 2006,
- based on a critical review of this data and categorising them by level of admissibility. The findings are categorised based on the living organisms studied, as well as their
- using various methods, i.e. either the uncertainty factors method given in the TGD8, or the SSD or ACT-SSWD methods, for Acute to Chronic Transformation – Species sensitivity Weighted Distribution9, 10 The SSWD method is a variant of the SSD method suggested in the TGD, if the dataset allows.

This work by EDF R&D was submitted to INERIS for a second opinion in 2006. The values used are ultimately less than those derived from this initial shared work. This choice is conservative.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

The bibliographic analysis was conducted by consulting the following databases: Aquire, Biological abstracts, Chemical abstracts, Dose-1998, ECDIN-1993, EINEC-1998, Environmental bibliography, HSDB-1997 (Hazardous Substances Data Bank), IRIS on-line (Integrated Risk Information System, U.S.EPA, http://www.epa.gov/iris/subst/0528.htm), IUCLID version 4.0-2001, Medline, OHM/TADS-1997, Pascal biomed, Pascal sciences et techniques and Toxline. All of the ecotoxicological data available at the time of the search regarding algae, vertebrates, and invertebrates was listed. A few of the lowest chronic and acute ecotoxicological values, selected as admissible, are listed for each organism in the table below. As a reminder, for statistical methods after reading the publication, the data sample used is larger than this one.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 49: Chronic toxicity data used for derivation of respective PNECs for ethanolamine

| Organism | Species | Chronic toxicity mg $l^{-1}$ | Authors |
|---|---|---|---|
| Freshwater algae | *Desmodesmus subspicatus* | 4 (NOEC) | IUCLID, 2000 |
| Freshwater algae | *Microcystis aeruginosa* | *1.6 (TT 8 days)* | Bringmann and Kuhn, 1978 |
| Freshwater crustacean | *Daphnia magna* | 7.8 (NOEC 21day) | EDF R&D, 2006 (ref. HP77-2006-003970-FR) |
| Marine invertebrates | *Hydractinia echinata* | 128.28 (3hrs) | Chicu et al., 2000 – Personal communication, 2006 |
| Marine crustaceans | *Crangon crangon* | >100 (NOEC) | Portmann and Wilson, 1971, ECDIN, 1993 |
| Freshwater fish | *Salvelinus fontinalis* | 1.77 (NOEC 100d) | Groth et al, 1986- IUCLID 2000 |

Table 50: Acute toxicity data used for derivation of respective PNECs for ethanolamine

| Organism | Species | Chronic toxicity mg $l^{-1}$ | Authors |
|---|---|---|---|
| Freshwater algae | *Desmodesmus subspicatus* | 8.42 (72 hrs) | Eisentraeger et al., 2003 |
| Freshwater algae | *Desmodesmus subspicatus* | 15 (72 hrs) | IUCLID, 2000 |
| Marine algae | *Isochrysis galbana* | 80 (96 hrs) | Roseth et al, 1996 |
| Freshwater crustacean | *Daphnia magna* | 65 (48 hrs) | IUCLID, 2000 |
| Marine invertebrates | *Hydractinia echinata* | 128.28 (3hrs) | Chicu et al., 2000 – Personal communication, 2006 |
| Freshwater amphibians | *Xenopus laevis* | 220 (48 hrs) | De Zwart and Sloof, 1987 |
| Freshwater fish | *Lepomis macrochirus* | 329.16 (96hrs) | Wolverton et al,1970 |
| Freshwater fish | *Brachydanio rerio* | 3683.4 (96 hrs) | Groth et al, 1993 – IUCLID 2000 |
| Freshwater fish | *Carassius auratus* | 170 (96hrs) | IUCLID, 2000 |

### Chronic freshwater PNEC

The TGD's requirements are applied, and the uncertainty factors method is used. There are chronic NOECs for freshwater species at three different trophic levels. In accordance with the TGD, a factor of 10 was applied to the lowest of the chronic NOECs available, which is equal to 1.6 mg $l^{-1}$ (tested on Microcystis aeruginosa).

The chronic PNEC of ethanolamine in fresh water used in the impact study is therefore 1.6/10 = 0.16 mg $l^{-1}$, or 160 µg $l^{-1}$.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

### Chronic seawater PNEC

For the marine environment, little marine ecotoxicological data exists (2 acute values). In such cases, the TGD suggests applying an additional expansion factor 10 times the ecotoxicological value chosen in order to assess the freshwater PNEC. However, in the TGD, the application of higher uncertainty factors to deduce marine PNECs is mainly justified by the fact that the food webs in marine ecosystems are often more complex than that of freshwater ecosystems. Greater safety factors for the marine environment are therefore relevant for bioaccumulative and persistent substances. This is not true of ethanolamine. It is therefore suggested to adopt the same PNEC values for seawater as for freshwater.

Thus, the chronic PNEC of ethanolamine in seawater value used in the impact study is 160 µg $l^{-1}$.

### Acute freshwater PNEC

The uncertainty factors method described in the TGD is applied based on current INERIS recommendations, meaning that the PNEC is deduced from the listed and adopted EC50 data, and particularly the lowest value of 8.42 mg $l^{-1}$ (EC50-72hrs for *Desmodesmus subspicatus*), to which an uncertainty factor of 100 is applied.

As a result, an acute PNEC of ethanolamine in fresh water of 8.42/100=0.084 mg $l^{-1}$ is obtained, which is less than the chronic PNEC of ethanolamine in fresh water. It is therefore proposed to adopt the same value for both acute and chronic PNEC.

Thus, the acute PNEC of ethanolamine in fresh water value used in the impact study is 28/100 = 160 µg $l^{-1}$.

Though the PNEC derived from statistical calculations is merely additional information for the analysis, it is presented anyway: It is possible to use the SSWD technique developed jointly by EDF R&D and INERIS (see references above), with an acute dataset, in order to take into account the diversity of data available and to obtain a more realistic view of the various species' sensitivity to the substance being studied. Using all of the acute data on ethanolamine for the three freshwater categories of algae, invertebrates and vertebrates, and weighting each one equally, a HC5_5% (likened to a PNEC) of 2.8 mg $l^{-1}$ is obtained.

### Acute seawater PNEC

For the same reasons as given in the section above, the acute PNEC of ethanolamine in seawater used in the impact study is therefore 160 µg $l^{-1}$.

**GLOSSARY**
PNEC: Predicted No Effect Concentration
EC50: Half maximal effective concentration
IC50: Half maximal inhibitory concentration
LC50 , LD0: Median lethal dose for 50% (0%) of organisms
HC: Hazard Concentration
NOEC: No Observed Effect Concentration
TT: Toxicity Threshold may be considered to be a NOEC

# 16 Appendix C Extract from the Water Framework Directive (Standards and Classification) Directions (England and Wales) 2015

**Table 16**

| **Dissolved inorganic nitrogen standards for coastal water (salinity 32), or part of such water, (coastal waters categorised by type in accordance with paragraph 3 of Schedule 2)** | | | | |
|---|---|---|---|---|
| *Mean dissolved inorganic nitrogen concentration (micromoles per litre) during the period 1st November to 28th February* | | | | |
| | *Dissolved inorganic nitrogen concentration (micromoles per litre)* | | | |
| *Type* | *High* | *Good* | *Moderate* | *Poor* |
| | Mean for the period 1st Nov to 28th Feb | | | |
| Clear | 12(i) | 18(i) | 27 (i) | 40.5(i) |
| | | 99 percentile standard for the period 1st Nov – 28th Feb | | |
| Intermediate turbidity | 12 | 70 | 105 | 157.5 |
| Turbid | 12 | 180 | 270 | 405 |
| Very turbid | 12 | 270 | 405 | 607.5 |

(i) The standard refers to the concentration of dissolved inorganic nitrogen at a mean salinity of 32 for the period of 1st November to 28th February.

**Table 6**

| **Criteria for identifying types of transitional and coastal water to which the dissolved inorganic nitrogen standards for transitional and coastal water apply** | |
|---|---|
| *Type* | *Annual mean concentration of suspended particulate matter (mg/l)* |
| Very turbid | > 300 |
| Turbid | 100 - 300 |
| Intermediate turbidity | 10 < 100 |
| Clear | < 10 |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

**Table 17**

| **Dissolved inorganic nitrogen standards for transitional water (salinity 25), or part of such water, (transitional waters categorised by type in accordance with paragraph 3 of Schedule 2)** | | | | |
|---|---|---|---|---|
| *Mean dissolved inorganic nitrogen concentration (micromoles per litre) during the period 1st November to 28th February* | | | | |
| | *Dissolved inorganic nitrogen concentration (micromoles per litre)* | | | |
| *Type* | *High* | *Good* | *Moderate* | *Poor* |
| | Mean for the period 1st Nov to 28th Feb | | | |
| Clear | 20(i) | 30(i) | 45(i) | 67.5(i) |
| | | 99 percentile standard for the period 1st Nov to 28th Feb | | |
| Intermediate turbidity | 20 | 70 | 105 | 157.5 |
| Turbid | 20 | 180 | 270 | 405 |
| Very turbid | 20 | 270 | 405 | 607.5 |

(i) The standard refers to the concentration of dissolved inorganic nitrogen at a mean salinity of 25 for the period of 1st November 28th February.

# 17 Appendix D CORMIX modelling used in support of discharge assessment for the CDO

The results of the CORMIX simulations are presented as dilution curves, with dilution plotted against distance from the source along the centreline of the plume jet. CORMIX calculates the distance at which plume concentrations drops below a designated Water Quality Standard.

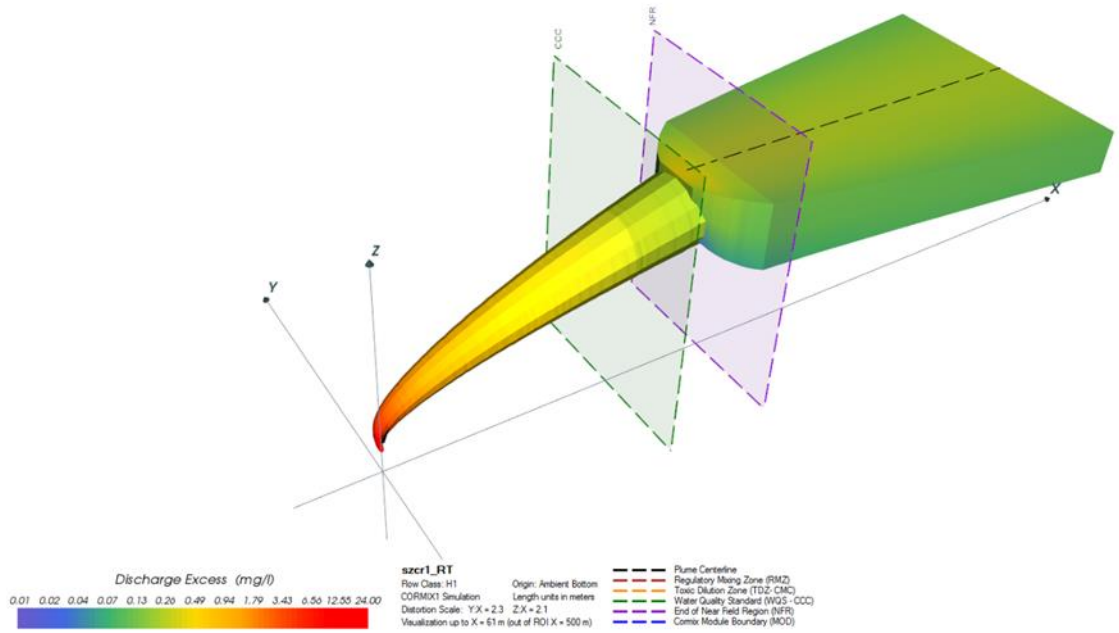


Figure 18: CORMIX output at rising mid tide, showing the buoyant nature of the plume 124 $ls^{-1}$ discharge.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

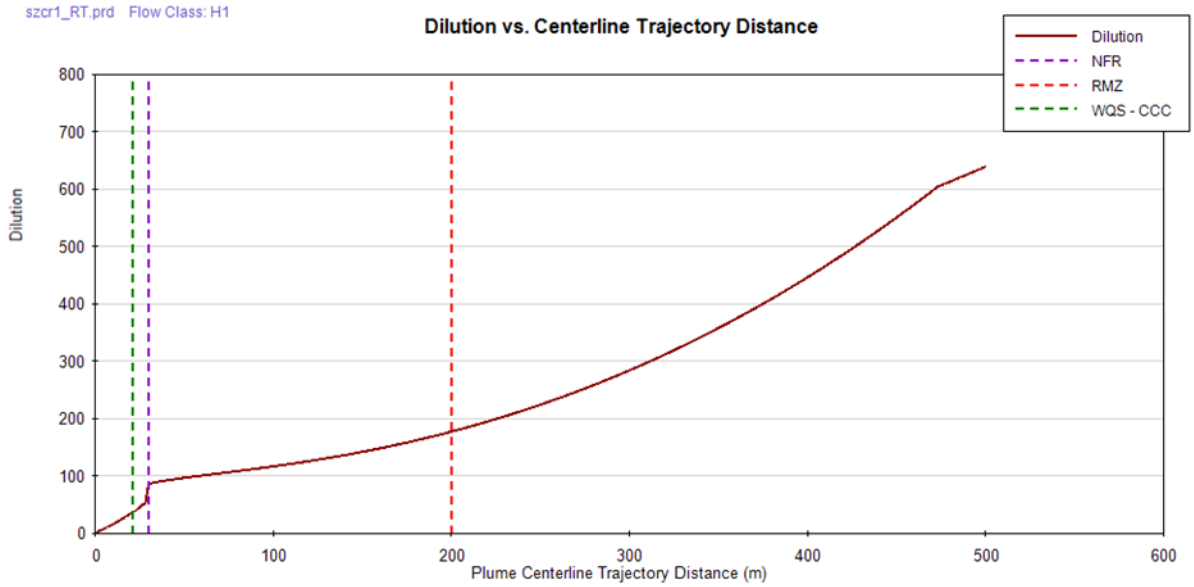


Figure 19: Dilution curve for a 124 l $s^{-1}$ discharge at the CDO. Relevant for Case A.

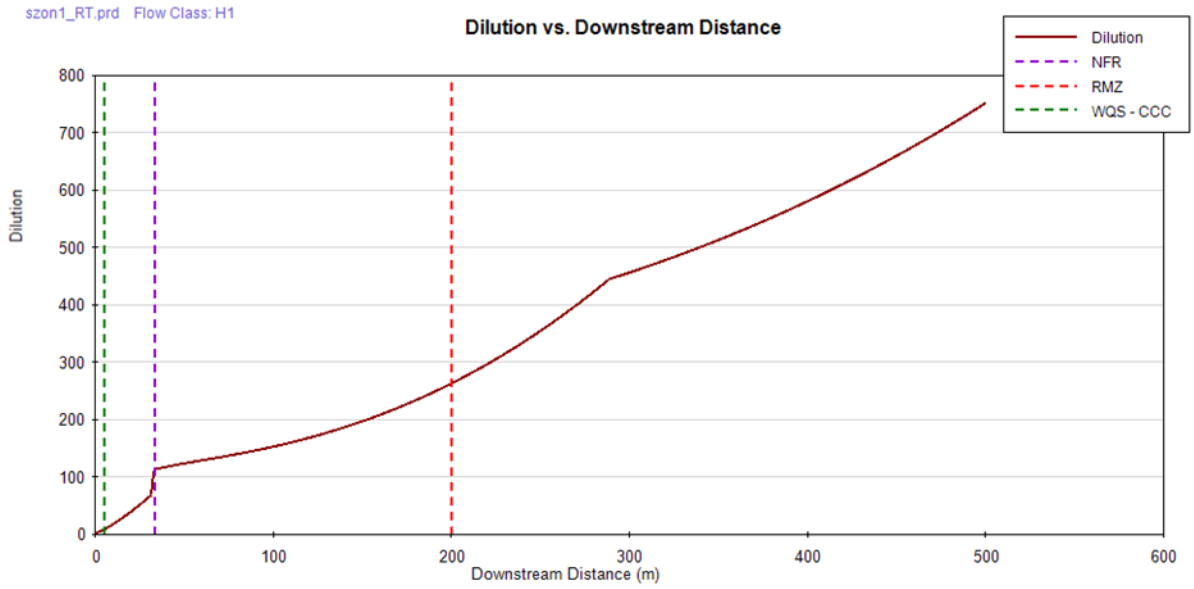


Figure 20: Dilution curve for a 72 l $s^{-1}$ discharge at the CDO. Relevant for Case D1.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

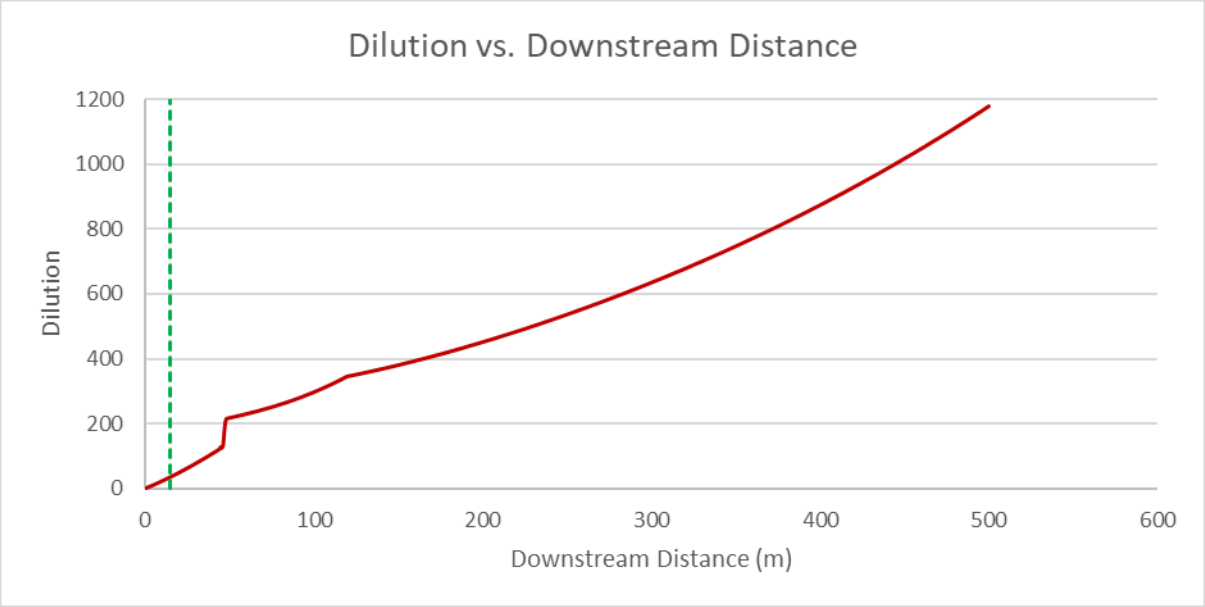


Figure 21: Dilution curve for a 30 l $s^{-1}$ discharge at the CDO. Relevant for sewage only.

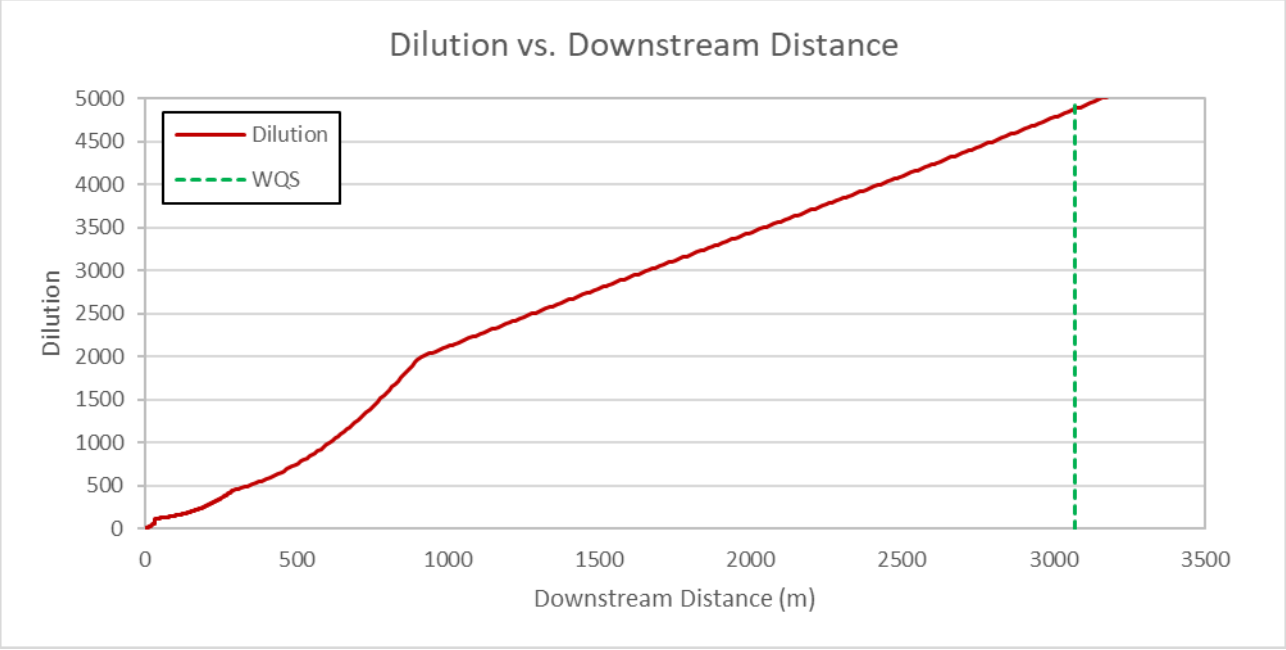


Figure 22: Dilution curve of E.coli for a 72 l $s^{-1}$ discharge at the CDO. Relevant for Case D1.

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

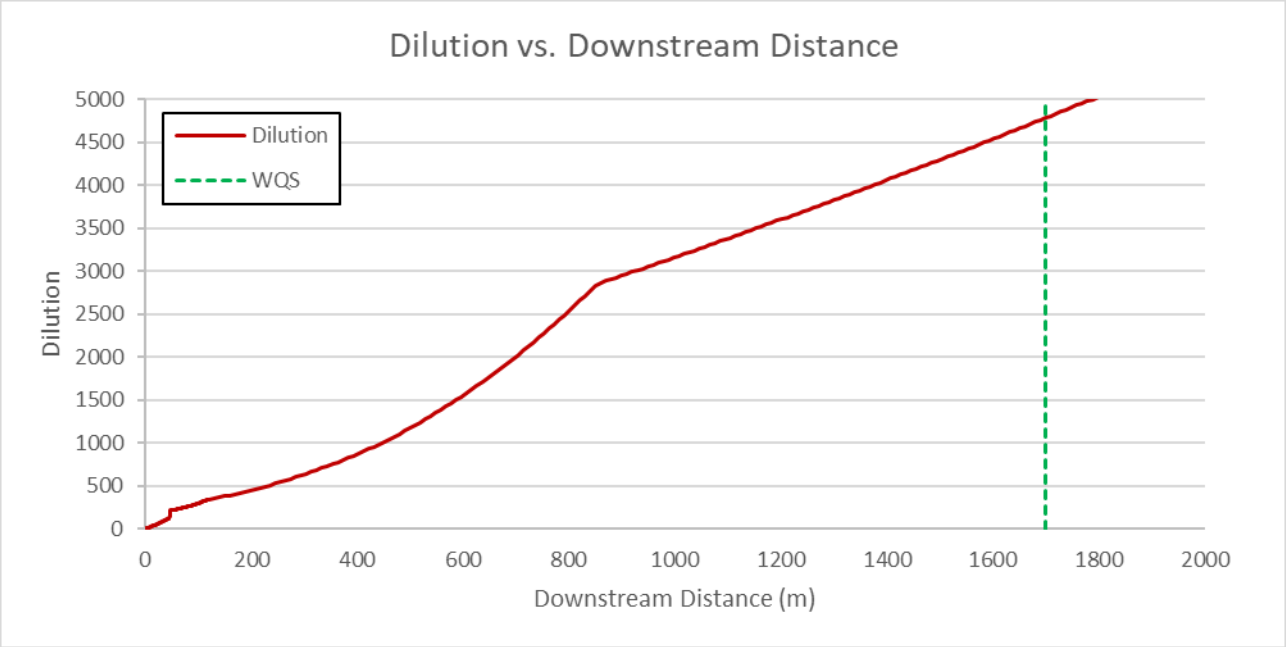


Figure 23: Dilution curve of E.coli for a 30 l $s^{-1}$ discharge at the CDO. Relevant for Case D.

# 18 Appendix E Sizewell seawater monitoring data

Background concentration data for various chemical determinands measured in Sizewell seawater in surveys conducted in 2010 and in 2014/15 and that are referenced as part of the screening assessment are included in the following Tables.

## E1 Various physical and chemical parameters measured at Sizewell for marine water quality that provide site background values

Tables below are taken from BEEMS Technical Report TR189 and supplemented with data from TR314

Table 51: Sizewell spatial survey water sample analysis for Lithium (data from BEEMS TR189)

| Lithium (mg $l^{-1}$) | Lithium (mg $l^{-1}$) | Lithium (mg $l^{-1}$) | Lithium (mg $l^{-1}$) |
|---|---|---|---|
| 0.05 | 0.09 | 0.06 | 0.07 |
| 0.05 | 0.07 | 0.07 | 0.05 |
| 0.07 | 0.06 | 0.09 | 0.07 |
| 0.03 | 0.09 | 0.07 | 0.05 |
| 0.06 | 0.07 | 0.06 | 0.07 |
| 0.06 | 0.03 | 0.08 | 0.07 |
| 0.06 | 0.06 | 0.08 | 0.06 |
| 0.05 | 0.06 | 0.08 | 0.07 |
| 0.07 | 0.06 | 0.07 | 0.07 |
| 0.05 | 0.05 | 0.05 | 0.05 |
| 0.05 | 0.06 | 0.09 | 0.09 |
| 0.06 | 0.05 | 0.05 | 0.07 |
| 0.07 | 0.06 | 0.05 | 0.08 |
| 0.07 | 0.06 | 0.07 | 0.06 |
| 0.07 | 0.07 | 0.05 | 0.07 |
| 0.06 | 0.07 | 0.09 | |
| 0.07 | 0.07 | 0.08 | |
| | | | |
| **Mean overall concentration** | **0.065** | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 52: Sizewell spatial survey water sample analysis for suspended solids (data from BEEMS TR189)

| Site | Suspended solids Surface (mg $l^{-1}$) | Suspended solids near bed (mg $l^{-1}$) |
|---|---|---|
| 1 | 26, 56 | 73 |
| 2 | 102, 48 | - |
| 3 | 187, 53 | 167 |
| 4 | 131,53 | - |
| 5 | (9 -144) | 203 |
| 6 | 58,84 | - |
| 7 | 100,57 | 95 |
| 8 | 96, 69 | - |
| 9 | 68, 65 | 80 |
| 10 | 17, 28 | 75, 778 |
| 11 | (28 – 244) | - |
| 12 | 52, 86 | 53, 115 |
| **Mean concentration** | **74.1** | **182.1** |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 53: Sizewell spatial survey water sample analysis for Biochemical Oxygen Demand (BOD) (data from BEEMS TR189)

| BOD (mg $l^{-1}$) | BOD (mg $l^{-1}$) | BOD (mg $l^{-1}$) | BOD (mg $l^{-1}$) | BOD (mg $l^{-1}$) |
|---|---|---|---|---|
| <2 | <2 | 8.5 | <2 | <2 |
| 2 | 5.5 | <2 | <2 | <2 |
| <2 | <2 | <2 | <2 | <2 |
| <2 | <2 | 3 | <2 | <2 |
| <2 | <2 | <2 | 2 | <2 |
| <2 | <2 | <2 | <2 | <2 |
| 7.5 | <2 | 3 | 2 | <2 |
| <2 | <2 | <2 | <2 | <2 |
| <2 | <2 | <2 | <2 | <2 |
| 8.5 | <2 | <2 | <2 | <2 |
| <2 | <2 | <2 | <2 | <2 |
| <2 | 4.5 | <2 | <2 | <2 |
| 3.5 | 1 | <2 | <2 | |
| <2 | <2 | <2 | <2 | |
| **Overall Mean** | **2 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 54: Sizewell spatial survey water sample analysis for Chemical Oxygen Demand (COD) (data from BEEMS TR189)

| COD (mg $l^{-1}$) | COD (mg $l^{-1}$) | COD (mg $l^{-1}$) | COD (mg $l^{-1}$) | COD (mg $l^{-1}$) |
|---|---|---|---|---|
| 230 | 270 | 200 | 230 | 300 |
| 200 | 280 | 200 | 200 | 250 |
| 185 | 320 | 260 | 200 | 240 |
| 240 | 250 | 240 | 190 | 300 |
| 2.5 | 125 | 210 | 240 | 230 |
| 2.5 | 320 | 280 | 220 | 230 |
| 1100 | 120 | 270 | 230 | 200 |
| 240 | 210 | 220 | 230 | 50 |
| 230 | 235 | 280 | 220 | 250 |
| 180 | 2.5 | 150 | 240 | 210 |
| 280 | 2.5 | 200 | 225 | 240 |
| 280 | 975 | 50 | 190 | 280 |
| 270 | 250 | 210 | 280 | |
| 370 | 210 | 205 | 200 | |
| **Overall Mean** | **239 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 55: Sizewell spatial survey water sample analysis for aluminium as Al (dissolved) (data from BEEMS TR189)

| Aluminium (mg $l^{-1}$) | Aluminium (mg $l^{-1}$) | Aluminium (mg $l^{-1}$) | Aluminium (mg $l^{-1}$) | Aluminium (mg $l^{-1}$) |
|---|---|---|---|---|
| <0.01 | 0.01 | 0.01 | 0.01 | 0.02 |
| <0.01 | 0.06 | 0.17 | <0.01 | 0.01 |
| 0.01 | 0.01 | <0.01 | <0.01 | 0.02 |
| <0.01 | <0.01 | 0.01 | <0.01 | 0.01 |
| <0.01 | 0.01 | 0.02 | 0.01 | 0.01 |
| 0.02 | <0.01 | 0.01 | <0.01 | <0.01 |
| <0.01 | <0.01 | 0.01 | <0.01 | 0.01 |
| <0.01 | 0.01 | 0.01 | <0.01 | 0.01 |
| <0.01 | <0.01 | <0.01 | <0.01 | 0.01 |
| <0.01 | <0.01 | 0.01 | 0.01 | <0.01 |
| 0.01 | 0.04 | 0.01 | 0.01 | |
| 0.01 | <0.01 | 0.02 | 0.01 | |
| 0.01 | <0.01 | 0.01 | 0.01 | |
| 0.02 | <0.01 | 0.01 | 0.01 | |
| **Overall Mean** | **0.012 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 56: Sizewell spatial survey water sample analysis for manganese as Mn (dissolved) (data from BEEMS TR189)

| Manganese (mg $l^{-1}$) | Manganese (mg $l^{-1}$) | Manganese (mg $l^{-1}$) | Manganese (mg $l^{-1}$) | Manganese (mg $l^{-1}$) |
|---|---|---|---|---|
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | 0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | 0.009 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | <0.002 |
| <0.002 | <0.002 | <0.002 | <0.002 | |
| <0.002 | 0.003 | <0.002 | <0.002 | |
| <0.002 | 0.002 | <0.002 | <0.002 | |
| **Overall Mean** | **0.002 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 57: Sizewell spatial survey water sample analysis for chloride (data from BEEMS TR189)

| Chloride (mg $l^{-1}$) | Chloride (mg $l^{-1}$) | Chloride (mg $l^{-1}$) | Chloride (mg $l^{-1}$) | Chloride (mg $l^{-1}$) |
|---|---|---|---|---|
| 14200 | 12500 | 11900 | 13600 | 12300 |
| 14200 | 15100 | 16900 | 10200 | 14300 |
| 14800 | 16400 | 15700 | 16300 | 15500 |
| 15400 | 16000 | 16400 | 11500 | 13000 |
| 10100 | 12400 | 14400 | 12900 | 15300 |
| 14800 | 11000 | 13800 | 14900 | 10600 |
| 17200 | 13400 | 13100 | 13600 | 10600 |
| 17000 | 15100 | 18000 | 16400 | 14500 |
| 15600 | 11600 | 14000 | 13400 | 16600 |
| 13200 | 10400 | 11300 | 15800 | 13700 |
| 16400 | 14000 | 17300 | 14300 | 10700 |
| 16200 | 14200 | 15200 | 11600 | 12900 |
| 15000 | 17800 | 12900 | 16600 | |
| 14800 | 13000 | 12700 | 14200 | |
| **Overall Mean** | **14,128 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 58: Sizewell spatial survey water sample analysis for sulphate (data from BEEMS TR189)

| sulphate (mg $l^{-1}$) | sulphate (mg $l^{-1}$) | sulphate (mg $l^{-1}$) | sulphate (mg $l^{-1}$) | sulphate (mg $l^{-1}$) |
|---|---|---|---|---|
| 2570 | 2700 | 2750 | 2830 | 2630 |
| 2620 | 3900 | 2610 | 2610 | 2800 |
| 2860 | 1730 | 2650 | 2670 | 3200 |
| 3060 | 2730 | 2500 | 2600 | 2820 |
| 2720 | 2890 | 2690 | 2630 | 3130 |
| 2610 | 2750 | 2530 | 2780 | 2700 |
| 2500 | 2630 | 3810 | 2600 | 2620 |
| 2810 | 2930 | 2870 | 2730 | 2760 |
| 2590 | 2970 | 2630 | 2780 | 2820 |
| 2670 | 2760 | 3150 | 2710 | 3240 |
| 2620 | 2700 | 2630 | 3080 | 2940 |
| 2630 | 2530 | 2780 | 2780 | 2690 |
| 2460 | 2800 | 3110 | 2760 | |
| 2810 | 2610 | 3240 | 2870 | |
| **Overall Mean** | **2778 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 59: Sizewell spatial survey water sample analysis for sodium

| Sodium (mg $l^{-1}$) | Sodium (mg $l^{-1}$) | Sodium (mg $l^{-1}$) | Sodium (mg $l^{-1}$) | Sodium (mg $l^{-1}$) |
|---|---|---|---|---|
| 10200 | 10500 | 11300 | 10600 | 11400 |
| 10300 | 11500 | 9640 | 10200 | 10100 |
| 9880 | 288 | 9950 | 11200 | 9510 |
| 11000 | 11300 | 9200 | 10200 | 11000 |
| 10700 | 11400 | 10900 | 10400 | 9430 |
| 10400 | 11600 | 10600 | 10800 | 9610 |
| 10200 | 11100 | 11400 | 10200 | 11800 |
| 11500 | 9870 | 10900 | 11100 | 10500 |
| 9740 | 10100 | 11200 | 9570 | 10300 |
| 11000 | 11100 | 11700 | 11400 | 9830 |
| 9790 | 9580 | 11500 | 9400 | 11600 |
| 9970 | 10100 | 10100 | 10100 | 11200 |
| 9460 | 11600 | 11600 | 11600 | |
| 11000 | 10100 | 8900 | 10000 | |
| **Overall Mean** | **10,400 mg $l^{-1}$** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 60: Sizewell spatial survey water sample analysis for Dissolved oxygen, salinity and pH. All data part of survey in 2009/10 and reported in TR189

| Station | Date acquired | Time (GMT) | Sample depth (m) | Dissolved oxygen (%) | Salinity (PSU) | pH |
|---|---|---|---|---|---|---|
| 1 | 07/04/2010 | 11:40 | 0 | | | |
| 1 | 07/04/2010 | 11:50 | 4.8 | | | |
| 1 | 06/12/2010 | 12:30 | 0 | 98.5 | 34.4 | 8.4 |
| 2 | 25/02/2010 | 09:45 | 0 | 106 | 31.8 | 7.86 |
| 2 | 15/12/2010 | 12:10 | 0 | 99.2 | 34.1 | 7.97 |
| 3 | 25/02/2010 | 10:38 | 0 | 108 | 31.9 | 7.93 |
| 3 | 25/02/2010 | 11:25 | 3.7 | 107 | 31.8 | 7.95 |
| 3 | 06/12/2010 | 11:40 | 0 | 100.8 | 34 | 8.2 |
| 4 | 25/02/2010 | 12:20 | 0 | 119 | 32.2 | 7.89 |
| 4 | 15/12/2010 | 12:40 | 0 | 100.2 | 34.1 | 8.08 |
| 5 | 25/02/2010 | 13:05 | 0 | 118 | 32.3 | 8.03 |
| 5 | 25/02/2010 | 13:25 | 4.4 | 112 | 32.2 | 7.93 |
| 5 | 02/03/2010 | 11:00 | 0 | 91.7 | 31.5 | 7.42 |
| 5 | 02/03/2010 | 12:00 | 0 | 98.3 | 32 | 7.96 |
| 5 | 02/03/2010 | 13:00 | 0 | 93 | 31.8 | 7.93 |
| 5 | 02/03/2010 | 14:00 | 0 | | 31.9 | 7.9 |
| 5 | 02/03/2010 | 15:00 | 0 | | 32.6 | 7.96 |
| 5 | 02/03/2010 | 16:00 | 0 | | 32.5 | 8.01 |
| 5 | 02/03/2010 | 17:00 | 0 | | 32.4 | 7.97 |
| 5 | 02/03/2010 | 18:00 | 0 | | 32.2 | 8.04 |
| 5 | 02/03/2010 | 19:00 | 0 | | 32.2 | 7.92 |
| 5 | 02/03/2010 | 20:00 | 0 | | 32.3 | 7.88 |
| 5 | 02/03/2010 | 21:00 | 0 | | 31.9 | 7.87 |
| 5 | 02/03/2010 | 22:00 | 0 | | 32 | 7.92 |
| 5 | 02/03/2010 | 23:00 | 0 | | 31.9 | 7.87 |
| 5 | 08/04/2010 | 17:30 | 0 | 104.5 | 33.4 | 8.16 |
| 5 | 21/04/2010 | 09:45 | 0 | 102.9 | | 8 |
| 5 | 19/05/2010 | 08:45 | 0 | 102.9 | 33.7 | 8.15 |
| 5 | 07/06/2010 | 11:10 | 0 | 108.3 | 33.9 | 8.23 |
| 5 | 22/06/2010 | 09:15 | 0 | 99.3 | 32.8 | 8.12 |
| 5 | 06/07/2010 | 01:20 | 0 | 103.4 | 32.2 | 8.06 |
| 5 | 20/07/2010 | 13:45 | 0 | 94.3 | 33 | 8.02 |
| 5 | 11/08/2010 | 09:20 | 0 | 98.4 | 34.2 | 7.77 |
| 5 | 18/08/2010 | 10:15 | 0 | 97.6 | 34.3 | 7.85 |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

| | | | | | | |
|---|---|---|---|---|---|---|
| 5 | 09/09/2010 | 10:00 | 0 | 94 | 33.8 | 7.07 |
| 5 | 14/09/2010 | 10:45 | 0 | 97.2 | 30.2 | 8.27 |
| 5 | 28/09/2010 | 10:50 | 0 | 97.2 | 33.6 | 8.16 |
| 5 | 14/10/2010 | 10:15 | 0 | 96.9 | 32.2 | 8.06 |
| 5 | 15/11/2010 | 12:00 | 0 | 108.8 | 32.3 | 8.03 |
| 5 | 06/12/2010 | 11:08 | 0 | 100.9 | 34.1 | 8.17 |
| 5 | 15/12/2010 | 11:40 | 0 | 99.8 | 34.1 | 8.12 |
| 5 | 17/01/2011 | | 0 | 102.7 | 34.4 | 8.12 |
| 5 | 31/01/2011 | 14:00 | 0 | 112.6 | 33.5 | 8.02 |
| 5 | 14/02/2011 | 13:30 | 0 | 112.5 | 34.5 | 8.1 |
| 6 | 07/04/2010 | 13:15 | 0 | | | |
| 6 | 15/12/2010 | 11:15 | 0 | 98.7 | 34.2 | 8.05 |
| 7 | 08/04/2010 | 16:15 | 0 | 105 | 33.2 | 8.15 |
| 7 | 08/04/2010 | 16:30 | 7 | 107.8 | 33.1 | 8.14 |
| 7 | 31/01/2011 | 11:20 | 0 | 101.3 | 33.4 | 8 |
| 8 | 08/04/2010 | 15:45 | 0 | 109.4 | 32.8 | 8.11 |
| 8 | 31/01/2011 | 10:55 | 0 | 101.7 | 33.3 | 8.05 |
| 9 | 08/04/2010 | 14:00 | 0 | 101.8 | 32.7 | 8.12 |
| 9 | 08/04/2010 | 14:30 | 5 | 104.4 | 33.4 | 8.1 |
| 9 | 17/01/2011 | 14:30 | 0 | 100.1 | 34.1 | 8.13 |
| 10 | 07/04/2010 | 10:20 | 0 | | | |
| 10 | 07/04/2010 | 10:30 | 11 | | | |
| 10 | 31/01/2011 | 09:20 | 0 | 99.4 | 33.3 | 7.89 |
| 10 | 14/02/2011 | 11:00 | 10 | 103 | 35.2 | 8.03 |
| 11 | 07/04/2010 | 13:45 | 0 | | | |
| 11 | 21/04/2010 | 10:45 | 0 | 100.8 | | 7.99 |
| 11 | 19/05/2010 | 09:45 | 0 | 100.1 | 33.8 | 8.18 |
| 11 | 07/06/2010 | 10:35 | 0 | 109.1 | 34 | 8.23 |
| 11 | 22/06/2010 | 09:45 | 0 | 95 | 33 | 8.16 |
| 11 | 06/07/2010 | 10:00 | 0 | 94.3 | 31.6 | 8.02 |
| 11 | 20/07/2010 | 13:00 | 0 | 102.6 | 33.1 | 8.09 |
| 11 | 11/08/2010 | 10:08 | 0 | 97.8 | 34.3 | 8.01 |
| 11 | 18/08/2010 | 11:10 | 0 | 97.3 | 34.5 | 7.98 |
| 11 | 09/09/2010 | 10:45 | 0 | 95.3 | 34.3 | 7.22 |
| 11 | 14/09/2010 | 10:15 | 0 | 97.4 | 30.2 | 8.29 |
| 11 | 28/09/2010 | 10:00 | 0 | 98.5 | 33.8 | 8.14 |
| 11 | 14/10/2010 | 11:15 | 0 | 95.8 | 32.3 | 8.05 |
| 11 | 15/11/2010 | 11:15 | 0 | 97.6 | 32.2 | 8.08 |

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| | | | | | | |
|---|---|---|---|---|---|---|
| 11 | 06/12/2010 | 10:30 | 0 | 97.5 | 34.2 | 7.97 |
| 11 | 15/12/2010 | 10:38 | 0 | 97 | 34.1 | 8.13 |
| 11 | 17/01/2011 | 16:00 | 0 | 99.8 | 34.6 | 8.12 |
| 11 | 31/01/2011 | 10:10 | 0 | 98.5 | 33.4 | 8.04 |
| 11 | 14/02/2011 | 12:45 | 0 | 101.4 | 34.5 | 8.08 |
| 12 | 08/04/2010 | 11:30 | 0 | 108.6 | 33.2 | 8.07 |
| 12 | 08/04/2010 | 13:00 | 17 | 102.5 | 33.6 | 8.11 |
| 12 | 17/01/2011 | 12:20 | 14 | 100.3 | 34.2 | 8.21 |
| 12 | 17/01/2011 | 12:20 | 0 | 100.1 | 34.3 | 8.1 |
| | | | | **Dissolved oxygen (%)** | **Salinity (PSU)** | **pH** |
| | | | **Mean** | **101.63** | **33.12** | **8.02** |
| | | | **95percentile** | **112.40** | **34.50** | **8.23** |
| | | | **5 percentile** | **94.30** | **31.72** | **7.83** |
| | | | **50 percentile** | **100.20** | **33.30** | **8.05** |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

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Table 61: Sizewell spatial survey water sample analysis for Dissolved inorganic nitrogen (DIN) $\mu g l^{-1}$ Shaded cells show the winter DIN values used to derive the site background. All data part of survey in 2014/15 and reported in TR314

| Sampling period | SZ B intake | SZ C intake/outfall | SZ B outfall | WFD site SZ3 |
|---|---|---|---|---|
| SIZE01/14 | 468, 441, 441 | 447, 440 | 463, 462, 427 | 483 |
| SIZE02/14 | 367, 358, 379 | 349, 346, 364 | 385, 357, 357 | 399 |
| SIZE03/14 | 349, 322, 287 | 280, 307, 307 | 316, 312, 287 | 321 |
| SIZE04/14 | 213, 190, 179 | 199, 227, 192 | 211, 188, 182 | 183 |
| SIZE05/14 | 57, 29 | 53, 34 | 64, 43, 42 | 85 |
| SIZE06/14 | 106, 81, 78 | 77, 76, 70 | 85, 88, 92 | 109 |
| SIZE07/14 | 116, 120, 118 | 120, 116, 113 | 134, 130, 134 | 99, 111 |
| SIZE09/14 | 91, 71, 76 | 87, 78, 63 | 94, 95, 90 | 95, 88 |
| SIZE10/14 | 200,213, 217 | 183, 183, 185 | 190, 218, 217 | 214, 206 |
| **SIZE11/14** | **337** | **337** | **364** | **367** |
| **SIZE11/14** | **358** | **316** | **363** | **371** |
| **SIZE11/14** | **339** | **312** | **340** | **-** |
| **SIZE01/15** | **-** | **-** | **-** | **298, 312** |
| **SIZE02/15** | **419** | **399** | **409** | **427** |
| **Winter 99% values** | **426** | | | |
| **Winter Mean** | **357** | **Annual Mean** | **232** | |
| **Winter Min** | **298** | | | |
| **Winter Max** | **427** | | | |

# CEFAS REPORT TR193 SZC DISCHARGES HI TYPE ASSESSMENT SUPPORTING DATA REPORT

## NOT PROTECTIVELY MARKED

Table 62: Sizewell spatial survey water sample analysis for phosphate (PO4-P) $\mu gl^{-1}$. All data are part of survey in 2014/15 and are reported in TR 314

| Sampling period | SZ B intake | SZ C intake/outfall | SZ B outfall | WFD site SZ3 |
|---|---|---|---|---|
| SIZE01/14 | 71, 61, 121 | 46, 42 | 99, 60, 56, 71 | 47 |
| SIZE02/14 | 33, 110, 27 | 29, 38, 26 | 28, 27, 25 | 32 |
| SIZE03/14 | 19, 19, 17 | 20, 20, 18 | 23, 21, 20 | 22 |
| SIZE04/14 | 1, 6, 9 | 4, 8 | 4, 11, 10 | 11 |
| SIZE05/14 | 42, 210 | 37, 33 | 64, 36, 4 | 127 |
| SIZE06/14 | 30, 79, 13 | 116, 13, 12 | 44, 39, 17 | 100 |
| SIZE07/14 | 19, 17, 17 | 33, 18, 17 | 21, 21, 20 | 18, 20 |
| SIZE09/14 | 18, 13, 13 | 17, 12, 11 | 16, 14, 17 | 17, 15 |
| SIZE10/14 | 22, 19, 22 | 24, 22, 23 | 25, 23, 23 | 27, 25 |
| SIZE11/14 | 24, 26, 26 | 26, 25, 25 | 26, 26, 27 | 189, 38 |
| SIZE01/15 | | | | 24, 25 |
| SIZE02/15 | 28 | 26 | 27 | 29 |
| | | | | |
| **Overall Mean** | **33.5** | | | |

# 19 Appendix F Dissolved inorganic nitrogen loading over the first three years of the construction period

The construction schedule profile developed for HPC was adapted to include information about groundwater dewatering known for SZC construction. The discharge rates for groundwater and for treated sewage effluent from site coupled with the mean DIN contributions from groundwater 1021$\mu gl^{-1}$ and treated sewage effluent 5000$\mu gl^{-1}$ were used to derive DIN loading for the first three years of construction (Table 62). Based on a maximum discharge rate (unlikely to persist for whole period) DIN loading for the first three years is also shown to provide an upper bounding value (Table 63).

Table 63: DIN loadings calculation for SZC over first three years of the construction period based on average discharge concentrations and flow rates

| Construction Year | DIN loading calculation based on sewage discharge rate 13.3$ls^{-1}$ | Calculated total |
|---|---|---|
| Year 1 | 365*24*3600[1](1021[2]*33.69[3]+5000[4]*13.31[5])/(1000*1000000)[6] | 3183.61 |
| Year 2 | 365*24*3600(1021*22.46+5000*13.31)/(1000*1000000) | 2822.06 |
| Year 3 | 365*24*3600(1021*18.82+5000*13.31)/(1000*1000000) | 2704.95 |

[1]calculation of per second discharge; [2] mean DIN for treated sewage effluent; [3] average dewatering rate, litres per second for year one; [4] ammoniacal nitrogen as N from treated sewage effluent; [5] treated sewage effluent flow rate per second; [6] conversion from micrograms to kilograms.

Table 64: DIN loadings calculation for SZC over first three years of the construction period based on maximum discharge flow rate of 30$lsec^{-1}$

| Construction Year | DIN loading calculation based on sewage discharge rate 30$ls^{-1}$ | Calculated total |
|---|---|---|
| Year 1 | 365*24*3600[1](1021[2]*33.69[3]+5000[4]*30[5])/(1000*1000000)[6] | 5815.3 |
| Year 2 | 365*24*3600(1021*22.46+5000*30)/(1000*1000000) | 5453.7 |
| Year 3 | 365*24*3600(1021*18.82+5000*30)/(1000*1000000) | 5336.6 |

[1]calculation of per second discharge; [2] mean DIN for treated sewage effluent; [3] average dewatering rate, litres per second for year one; [4] ammoniacal nitrogen as N from treated sewage effluent; [5] treated sewage effluent flow rate per second; [6] conversion from micrograms to kilograms.

# 20 Appendix G Coliforms, intestinal enterococci in construction discharge and compliance with bathing water and shellfish standards

For the construction discharge following either sewage treatment at a secondary or tertiary (UV) level the distance from the discharge point, at which enough dilution occurs to be below relevant microbiological standard levels, has been estimated using CORMIX for Case D ($30ls^{-1}$) sewage discharge and Case D1 ($72ls^{-1}$). Results are shown below in Table 64.

Table 65 Estimate of minimum distance from point of discharge at which microbiological standards for bathing waters are met following different levels of sewage treatment for the construction discharge from SZC

| Species | Standard cells/ 100ml | Discharge concentration cells / 100ml | $2^{ndry}$ treatment 2 log reduction | Dilution required to meet bathing water standard | Maximum potential distance from the discharge at which meets bathing water standard 30 l $s^{-1}$ | 72 l $s^{-1}$ | UV treatment reduction 5.4 log reduction | Dilution factor required for discharge to meet bathing water standard | Maximum distance from the discharge at which it meets bathing water standard |
|---|---|---|---|---|---|---|---|---|---|
| *E.coli* | 500 | 240,000,000[1] | 2400000 | 4800 | ~1.7 km | ~3.1 km | 955.5 | 1.9 | <1 m pass immediately on discharge, for both cases. |
| Entero-cocci | 200 | 13,600,000 | 136000 | 680 | ~66 m | ~460 m | 54.1 | 0.3 | <1 m pass immediately on discharge, for both cases. |

[1]Cell numbers/100ml are based on data in support of the Hinkley Point C development (pers. Comm. EDF);

# 21 Appendix H Calculations of values associated with FRR

This section provides the supporting data and calculation used to derive the values for the FRR assessment. The values are the outputs of the modelling approach used to estimate impingement at SZB, and are the same values that form the basis of the final impingement estimates. In TR339, only the final annual estimates for SZB are given.

Mean daily estimated numbers impinged were modelled using data collected during onshore impingement sampling between 2009 and 2017 at SZB. Full details of the modelling approach are given in BEEMS Technical Report TR339. The mean estimated daily values for SZB were raised to predicted daily values for SZC on the ratio of the pumping capacities of the two stations (i.e. the SZB daily mean values were multiplied by (131.86/51.5 cumecs). Next, survival through the FRR was considered, as some of the more robust species will not all die during their passage through the cooling water systems. In this case, the same values of FRR survival that were used to predict impingement at SZC were used (BEEMS Technical Report TR406). The resulting numbers lost to impingement were converted to weight using the mean length of each species in the impingement samples and published length-weight relationships. The size distributions used are shown in BEEMS Technical Report TR339. The length-weight relationships used were taken from Silva *et al.* (2013).

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Table 66 Sizewell B mean daily estimated number of fish impinged

| Species | MeanJAN | MeanFEB | MeanMAR | MeanAPR | MeanMAY | MeanJUN | MeanJUL | MeanAUG | MeanSEP | MeanOCT | MeanNOV | MeanDEC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sprat | 57550 | 14569 | 5879 | 813 | 1203 | 494 | 825 | 467 | 600 | 483 | 1470 | 6820 |
| Herring | 2001 | 5508 | 12363 | 6412 | 362 | 75 | 462 | 282 | 608 | 1337 | 1378 | 2174 |
| Whiting | 5296 | 3684 | 2552 | 1354 | 685 | 258 | 314 | 430 | 1337 | 1734 | 2416 | 3944 |
| Bass | 1354 | 3415 | 1085 | 30 | 7 | 3 | 3 | 8 | 9 | 25 | 206 | 1414 |
| Sand gobies | 292 | 98 | 101 | 56 | 42 | 11 | 71 | 334 | 1393 | 848 | 840 | 805 |
| Sole | 19 | 105 | 398 | 794 | 650 | 131 | 110 | 150 | 471 | 224 | 127 | 31 |
| Dab | 296 | 140 | 85 | 78 | 21 | 10 | 21 | 35 | 475 | 283 | 279 | 195 |
| Anchovy | 0 | 0 | 33 | 33 | 318 | 318 | 111 | 111 | 6 | 6 | 3 | 3 |
| Thin-lipped grey mullet | 316 | 316 | 52 | 52 | 1 | 1 | 0 | 0 | 1 | 1 | 73 | 73 |

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## NOT PROTECTIVELY MARKED

Table 67 Sizewell C mean daily estimated number of fish impinged

| Species | MeanJAN | MeanFEB | MeanMAR | MeanAPR | MeanMAY | MeanJUN | MeanJUL | MeanAUG | MeanSEP | MeanOCT | MeanNOV | MeanDEC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sprat | 147350 | 37301 | 15052 | 2081 | 3081 | 1265 | 2113 | 1196 | 1537 | 1237 | 3764 | 17461 |
| Herring | 5124 | 14103 | 31654 | 16418 | 927 | 193 | 1184 | 721 | 1556 | 3422 | 3527 | 5567 |
| Whiting | 13560 | 9432 | 6534 | 3466 | 1753 | 661 | 803 | 1101 | 3424 | 4439 | 6187 | 10097 |
| Bass | 3468 | 8743 | 2777 | 77 | 17 | 7 | 7 | 21 | 24 | 65 | 529 | 3621 |
| Sand gobies | 748 | 250 | 258 | 144 | 108 | 27 | 182 | 856 | 3568 | 2170 | 2150 | 2062 |
| Sole | 50 | 268 | 1019 | 2032 | 1665 | 335 | 282 | 384 | 1205 | 573 | 325 | 78 |
| Dab | 757 | 358 | 217 | 200 | 54 | 25 | 53 | 89 | 1216 | 724 | 713 | 499 |
| Anchovy | 0 | 0 | 86 | 86 | 815 | 815 | 284 | 284 | 14 | 14 | 7 | 7 |
| Thin-lipped grey mullet | 810 | 810 | 132 | 132 | 1 | 1 | 0 | 0 | 3 | 3 | 186 | 186 |

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Table 68 Sizewell C mean daily estimated weight of fish impinged with an FRR fitted

| Species | mean weight at length (kg) | MeanJAN | MeanFEB | MeanMAR | MeanAPR | MeanMAY | MeanJUN | MeanJUL | MeanAUG | MeanSEP | MeanOCT | MeanNOV | MeanDEC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sprat | 0.009 | 1389.17 | 351.67 | 141.90 | 19.62 | 29.05 | 11.93 | 19.92 | 11.28 | 14.49 | 11.66 | 35.49 | 164.62 |
| Herring | 0.095 | 484.56 | 1333.65 | 2993.45 | 1552.60 | 87.64 | 18.21 | 111.93 | 68.18 | 147.17 | 323.66 | 333.59 | 526.46 |
| Whiting | 0.050 | 343.96 | 239.27 | 165.74 | 87.92 | 44.47 | 16.76 | 20.38 | 27.94 | 86.85 | 112.59 | 156.95 | 256.13 |
| Bass | 0.095 | 166.92 | 420.82 | 133.66 | 3.71 | 0.84 | 0.36 | 0.35 | 1.03 | 1.14 | 3.11 | 25.45 | 174.31 |
| Sand gobies | 0.001 | 0.23 | 0.08 | 0.08 | 0.04 | 0.03 | 0.01 | 0.05 | 0.26 | 1.08 | 0.66 | 0.65 | 0.62 |
| Sole | 0.015 | 0.15 | 0.81 | 3.08 | 6.13 | 5.02 | 1.01 | 0.85 | 1.16 | 3.63 | 1.73 | 0.98 | 0.24 |
| Dab | 0.018 | 2.85 | 1.35 | 0.82 | 0.75 | 0.20 | 0.09 | 0.20 | 0.34 | 4.58 | 2.73 | 2.69 | 1.88 |
| Anchovy | 0.008 | 0.00 | 0.00 | 0.70 | 0.70 | 6.69 | 6.69 | 2.33 | 2.33 | 0.12 | 0.12 | 0.06 | 0.06 |
| Thin-lipped grey mullet | 0.041 | 16.66 | 16.66 | 2.71 | 2.71 | 0.03 | 0.03 | 0.00 | 0.00 | 0.07 | 0.07 | 3.83 | 3.83 |
| Total daily biomass kg | | 2404.49 | 2364.30 | **3442.14** | 1674.20 | 173.98 | 55.08 | 156.02 | 112.52 | 259.14 | 456.32 | 559.67 | 1128.14 |

[1]Maximum daily biomass value for March used for calculation (3442.1kg); [2]Average biomass period April to September (405.2kg)

### 21.1.1 Calculation of moribund biomass FRR and potential nutrient input and influence on dissolved oxygen levels

The total biomass of moribund biota that potentially may be discharged from the FRR has been estimated based on the level of abstraction (pump rates) for the planned Sizewell C intakes and the information on seasonal distribution of species and length weight distribution of the species impinged for the existing Sizewell B (BEEMS TR339 and TR381). The derived Sizewell C data indicate that the highest biomass discharged occurs during the winter months December to March but is also high in April. Mean annual daily biomass is 1065.5kg per day and from April to September is 405.2 kg per day. Estimates of tissue concentration for nitrogen and phosphorus from several studies are shown in Table 68.

Table 69 Phosphorus and nitrogen concentration data for fish tissue derived from several studies and which is used in calculation of potential nutrient loadings released during decay of dead fish released from the FRR

| Nutrient | g/kg | Percentage (wet weight) | Average daily biomass (405.2kg) nutrient content April to September | Average Daily biomass (1065.5kg) nutrient content (January - December) | Literature Source |
|---|---|---|---|---|---|
| P content | 3.74-4.7 (ww) | 0.47 | (405.20/100) x0.47=1.9kg | - | Storrebakken et al., 2000 |
| P content | | 0.45-0.5 | 2.03kg | 5.33 | Gende et. al., 2004 |
| P content | | 0.64 (dry weight) | 0.93kg | - | Wang et. al., 2013 |
| N content | | 3.2-3.5 | 14.18kg | 37.29 | Walker et. al., 2011 |
| N content | | 3.4 | 13.78kg | - | Gende et. al., 2004 |
| N content | | 7.4 (dry weight) | 10.79kg | - | Wang et. al., 2013 |

The April to September period represents a time when sea temperatures and light levels at depth are increasing and phytoplankton growth is also increasing. At this time nutrients also become more limited in supply and become a limiting factor for algal growth. Average daily biomass from April to September is 405.2kg and multiplying this by the maximum estimates of phosphorus and nitrogen (Table 68) give maximum daily loadings of 14.18kg N and 2.03kg P per day.

Based on the mean annual biomass (1065.5kg) maximum daily predicted loadings are 37.29kg N and 5.33 kg P per day.

Un-ionised ammonia is calculated for the same period April to September as increasing temperatures and increasing growth and reproduction of species make this a more critical period. The ammonia, $NH_4$ concentration derived from a study of cod tissue is used to derive an equivalent value for fish biomass:405.20 (kg) fish biomass x 125 =
Total mg $NH_4$-N (50650). This value was used in the un-ionised ammonia calculator along with average background conditions for Sizewell (pH 8.05, salinity 33.3 and temperature 11.43°C) to derive an equivalent un-ionised ammonia value= 926,824 µg $NH_3$-N
Volume litres required to dilute this mass of $NH_3$-N to the EQS of 21 $\mu g l^{-1}$ $NH_3$-N minus natural background: 926,824/(21-1.6)=47,774.4 litres
47,774.4 litres is equivalent of an area of 109.4m x 109.4m x depth 4m = 1.20ha

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Using the same biomass loading a similar calculation was made but adjusting the average background conditions for the un-ionised ammonia calculator to 98$^{th}$ percentile temperature (19.4°C),95$^{th}$ percentile pH (8.23) and 50$^{th}$ percentile salinity to represent extreme summer conditions. This adjustment results in un-ionised ammonia load of 2,513,781 µg NH3-N and an equivalent area exceeding the EQS of 193.8m x193.8m=3.76ha that exceeds the EQS.

For the winter period the highest biomass discharged is predicted for March (3442.14kg). This loading is equivalent to $NH_4$-N of 3442.14 x 125= 430,267.5 mg $NH_4$-N. The total $NH_4$ loading generated when used as a source value in the un-ionised ammonia calculator with an average temperature for March of 6.09°C and 5$^{th}$ percentile salinity and 50$^{th}$ percentile pH= 5,198,601µg $NH_3$-N. This loading of un-ionised ammonia is equivalent to an area exceeding the EQS of 258.8m x258.8m=6.70ha.

For BOD calculation the annual daily average is used = 1065.5kg biomass
BOD generated from this biomass is estimated based on an equivalent value of 3.5g BOD/kg dry weight of fish tissue (Stigebrandt *et al*., 2001).
The estimate of BOD load per day is:

(i) 1065.5 x (3.5 x dry weight/wet weight conversion 0.36)= 1342.5kg BOD

Total oxygen reduction potential based on OSPAR information BOD of 1.5mgl$^{-1}$ = 0.5mgl$^{-1}$ oxygen reduction:

(i) 1342.5/1.5=447.5kg/ day $O_2$ reduction
(ii) Based on a background dissolved oxygen level 6.96mg/l this is equivalent to oxygen present in 64,297 m$^3$.
(iii) The daily volume exchange of 10% of GSB = 36,380,000 m$^3$
(iv) 64,297/36,380,000=0.18% of daily exchange
(v) Also, daily reaeration contributes 3.2gm$^{-2}$d$^{-1}$

Therefore, reaeration over 447.51/0.0032= 139846.87m$^{-2}$ (13.98ha) would meet this daily oxygen demand.

For BOD calculation the maximum daily value (March) is used = 3442.1kg biomass
BOD generated from this biomass is estimated based on an equivalent value of 3.5g BOD/kg dry weight of fish tissue (Stigebrandt *et al*., 2001).
The estimate of BOD load per day is:

(ii) 3442.1 x (3.5 x dry weight/wet weight conversion 0.36)= 4337.1kg BOD

Total oxygen reduction potential based on OSPAR information BOD of 1.5mgl$^{-1}$ = 0.5mgl$^{-1}$ oxygen reduction:

(vi) 4337.1/1.5=1445.7kg/ day $O_2$ reduction
(vii) Based on a background dissolved oxygen level 6.96mg/l this is equivalent to oxygen present in 207,715 m$^3$.
(viii) The daily volume exchange of 10% of GSB = 36,380,000 m$^3$
(ix) 207,715/36,380,000=0.57% of daily exchange
(x) Also, daily reaeration contributes 3.2gm$^{-2}$d$^{-1}$

Therefore, reaeration over 1445.7/0.0032= 451,780m$^{-2}$ (45.2ha) would meet this daily oxygen demand.