



TR131 Sizewell Water Quality Literature Report Edition 2

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

Executive summary

This Edition 2 report includes updated values for Environmental Quality Standards for lead, mercury, nickel, benzo a pyrene, copper and zinc and a corrected value for iron. In addition more detailed assessment of the temperature data and inclusion of nutrient concentrations and a discussion of their assessment as supporting elements in determining status is provided for the Suffolk coastal waterbody.

Both water quality data from monitoring surveys and data from the scientific literature collected in the vicinity of Sizewell nuclear power station and in the Suffolk waterbody or associated waterbodies were evaluated to provide a baseline against which to assess the potential for any impact of future development and operation at this site.

The data from successive Environment Agency surveys focuses on metals in seawater (1989 – 2006) and mostly the concentrations of these are low. During this period only in the case of cadmium was the Environmental Quality Standard exceeded for one location and for two locations for zinc with other locations very close to their respective EQS for zinc and copper. In the case of the latter two metals their use in antifouling of boats may have contributed. However with the exception of copper and zinc for which there are a number of diffuse input sources Nationally From 1990 - 2007 there is evidence of an overall decline in the concentration of a number of metals in riverine and direct discharges to the marine environment.

Other compounds of relevance to power station operation are generally not measured routinely and therefore data on levels within the area are limited to historic studies on power station discharges. These studies indicate relatively low and localised inputs of chlorine produced oxidants and bromoform not exceeding current or indicative standards beyond 1-2 kilometres of the point of discharge.

The thermal input from the power station cooling water discharge is one of the more significant potential affects upon the marine environment off Sizewell. The data for temperature for four sites across the Suffolk waterbody indicate that there is likely to be sufficient margin between the derived 98 percentile baseline temperature for the waterbody (19.4°C) to not result in major areas failing to meet the temperature boundary for Good/Moderate status (20 – 23 °C). The boundary value for the Thames SPA for the Habitats Directive criteria (28°C as a 98 percentile) is also likely to be met with only small areas of exceedance likely within the immediate mixing zone.

The East Suffolk region and the coast in particular is relatively sparsely populated and there are few major industries (although these include ports on the Orwell and Stour and at Lowestoft). The few industrial input sources has led to few major contaminant issues for the Suffolk waterbody and its current chemical status is considered to be Good.

During construction and operation the potential wastewater inputs from the workforce and from permanent staffing of the site will need to be assessed in terms of nutrient inputs but the current background levels have a limited impact because of the presence of relatively high suspended particulate matter (SPM) concentration in the waterbody. The impact of any additional nutrient inputs will need to be assessed against the SPM background and this may limit the extent of any effects.

The Suffolk coast waterbody is designated as heavily modified due to coastal protection and its current overall status is moderate. The waterbody is required to meet good ecological potential by 2027 so future changes that may result from power station construction and operation will need to be assessed against this target.

1 Introduction and Objectives

To provide a baseline against which to assess the potential for any impact of future development and operation of a new nuclear power station at Sizewell this report provides an evaluation of the literature on marine water quality in the adjacent Suffolk waterbody. Data from specific studies as well as from Environment Agency monitoring programmes is reported and discussed in the context of further development at Sizewell.

1.1 Background to the site

Any development at Sizewell that may affect freshwater and/or estuarine and coastal water quality must be considered in relation to the Water Framework Directive designations associated with the site which is located in East Suffolk Zone (ESZ) of the Anglian River Basin District (RBD). In this RBD, only 5% of rivers (by length) meet the requirements for good ecological status (GES) or good ecological potential (GEP). In total, 15% of all surface waters are designated as artificial and 56% of all surface waters are designated as heavily modified. Currently none of the estuaries and transitional and coastal waters meets the requirements for GES or GEP. Lowestoft north and south of Claremont pier and Southwold the Denes and Southwold the Pier are designated as bathing waters.

The European Water Framework Directive (WFD) sets a target of achieving good ecological and chemical status by 2015, unless a waterbody is designated as heavily modified in which case its target is to achieve good ecological potential with a target date of 2027. Therefore significant improvements in water quality in the Anglian RBD are required (Appendix L Water Framework Directive Assessment, 2010).

Sizewell is situated on an area of Suffolk coast covered by the shoreline management plan 2 (SMP2) area which includes ten transitional and coastal waterbodies (Table 1).

Table 1 – WFD waterbodies within the SMP2 area (taken from Appendix L Water Framework Directive Assessment, 2010)

Name of waterbody	Type of waterbody	Reason for designation as Heavily modified waterbody	Current overall status*	Proposed status
Suffolk Coast	Coastal	Coastal Protection, Flood Protection	Moderate	GEP by 2027
Benacre Broad	Coastal	Not designated	Moderate	GES by 2027
Covehithe Broad	Coastal	Not designated	Moderate	GES by 2027
Walberswick Marshes	Coastal	Flood Protection	Good	Remain at GEP
Essex Coast	Coastal	Coastal Protection, Flood Protection	Moderate	GEP by 2027
Harwich Approaches	Coastal	Coastal Protection, Navigation, Dredge Disposal	Good	Remain at GEP
Bure and Waveney	Transitional	Flood Protection, Navigation, Structure	Moderate	GEP by 2027
Blyth	Transitional	Coastal Protection, Flood Protection	Moderate	GEP by 2027
Alde and Ore	Transitional	Flood Protection	Moderate	GEP by 2027
Deben	Transitional	Flood Protection	Moderate	GEP by 2027

1.2 Water Quality parameters of importance to the site

The Water Framework Directive classifies waterbodies on the basis of a number of biological, morphological, physicochemical and chemical quality elements; these are listed below for transitional and coastal waters (Table 2). Five “General chemical and physicochemical elements supporting the biological elements” are included and these are transparency, thermal conditions, oxygenation conditions, salinity and nutrient conditions. The presence of priority and other substances in waterbodies or discharges to them must be managed so that they remain at levels that prevent any deterioration. The monitoring and assessment of the physical and physicochemical quality elements will support the interpretation, assessment and classification of the results arising from the monitoring of the biological quality elements (Best et al., 2007).

Table 2 - Quality elements for transitional and coastal waterbodies

Quality element
Phytoplankton
Macro-algae
Angiosperms
Benthic Invertebrate fauna
Fish Fauna (not included for coastal)
Morphological conditions
Tidal regime
Oxygenation conditions
Transparency
Thermal conditions
Salinity
Acidification status
Nutrient Conditions
Pollution by priority substances
Pollution by other substances

For the Water Framework Directive, certain substances that are regarded as the most polluting were identified in 2001 as Priority and Priority Hazardous Substances by a Decision of the European Parliament and the Council of Ministers (Decision 2455/2001/EC). This first list of substances became Annex X of the WFD. This first list was replaced by Annex II of the Directive on Environmental Quality Standards (Directive 2008/105/EC) (EQSD), also known as the Priority Substances Directive and this was further updated in 2013, DIRECTIVE 2013/39/EU. For these substances (a selected list of those that are potentially of most relevance to Sizewell are shown in Table 3), Environmental Quality Standards are determined at the European level, and these will apply to all Member States. For other substances, standards may be derived by each Member State, and they should lay down, where necessary, rules for their management. This list of compounds or Specific Pollutants is defined as substances that can have a harmful effect on biological

quality, and which may be identified by Member States as being discharged to water in “significant quantities”.

EQSs are concentrations below which a substance is not believed to be detrimental to aquatic life. These were originally developed for the EC Dangerous Substances Directive (76/464/EEC). The concept is now well established and is incorporated into the Environmental Quality Standards Directive(2008/105/EC) which is a daughter directive of the Water Framework Directive (60/2000/EC). EQSs are derived using acute toxicity tests on organisms at different trophic levels. To provide a safety factor, the EQS is set substantially below the concentration observed to have a toxic effect on the test organisms. EQSs vary for each substance and can be different for fresh, estuarine or coastal waters they may also be adjusted for individual waterbodies dependent upon the level of other local factors such as dissolved organic carbon concentration.

In the case of the metals, cadmium, lead, mercury and nickel, the water EQS refer to the dissolved concentration, i.e. the dissolved phase of a water sample obtained by filtration through a 0,45 µm filter or any equivalent pre-treatment, or, where specifically indicated, to the bioavailable concentration.

Table 3 - Environmental Quality Standards (EQS) for other surface waters (TraC Waters) for priority hazardous substances and other pollutants (Directive 2013/39/EU)

Substance	Annual average (AA) EQS other surface waters(µg l ⁻¹)	Maximum allowable concentration (MAC)* EQS other surface waters (µg l ⁻¹)
Cadmium and its compounds	0.2	1.5
Total DDT (sum of four isomers)	0.025	Not applicable
Para, para-DDT	0.01	Not applicable
Lead and its compounds	1.3	14
Mercury and its compounds	-	0.07**
Nickel and its compounds	8.6	34
Naphthalene	2	130
Benzo(a)pyrene	1.7 x 10 ⁻⁴	0.027
Tributyltin compounds	0.0002	0.0015

* The MAC is defined as an annual 95 percentile, (UKTAG 2013) **A biota EQS of 20 µg kg⁻¹ of tissue wet weight is also available and relates specifically to fish

UK Technical Advisory Group (UKTAG) identified a list of substances for consideration as specific pollutants based on their appearance on lists of discharge consents. Substances and their proposed EQS values relevant to subsequent discussion are listed in Table 4.

Table 4 - Environmental Quality Standards (EQSs) for specific pollutants and other substances (reference version UKTAG, 2013, Defra 2014)

Substance	Other Surface Waters (TraC Waters) Annual average (AA) EQS ($\mu\text{g l}^{-1}$)	Other Surface Waters (TraC Waters) MAC-EQS ($\mu\text{g l}^{-1}$)
Ammonia un-ionised	21	Not applicable
Chromium VI (dissolved)	0.6	32
Chromium III	Not applicable	Not applicable
Arsenic (dissolved)	25	Not applicable
Copper (dissolved)	3.76*	Not applicable
Chlorine	Not applicable	10 (total residual oxidants)
Iron (dissolved)	1000	Not applicable
Zinc	6.8 plus ambient background**	Not applicable

*+ $(2.677 \times ((\text{DOC}/2) - 0.5)) \mu\text{g l}^{-1}$ dissolved, where $\text{DOC} > 1 \text{ mg l}^{-1}$ **Ambient Background Concentration (ABC) is an estimate of background levels of zinc based on a low percentile of monitoring data. For saltwater, an ABC of $1.1 \mu\text{g l}^{-1}$ is recommended.

As well as hazardous substances Natural England and Environment Agency also considers physical changes and other disturbance factors that represent hazards for habitats, species and birds (i.e. Table 2).

Nutrient inputs from agricultural areas and sewage discharges can have significant effects upon estuarine and coastal waters. The major concern for increased inputs of nutrients mainly nitrogen (nitrate) and phosphorus (phosphate) is the enhanced growth of attached and planktonic plants which if it reaches excessive levels can lead to oxygen depletion.

Dissolved oxygen standards are also set for coastal waters under the Water Framework Directive as are temperature standards which consider both maximum temperatures as well as the temperature uplift that is considered acceptable over the natural background temperature regimes. These criteria are summarised in Table 5.

Table 5 - Summary of legislation primarily triggered by the direct and indirect impacts of thermal plumes (prolonged elevated temperatures) adapted from the Bella Earth Project Report, (2008)

POTENTIAL ACTION BY POWER PLANT : Thermal Discharge				
Activity	Measurement	Threshold	Consequence	Directive
Thermal plume, increase in temperature	Temperature of surface water	Set against WFD status thresholds exceedance, e.g. not >(defined value) for more than 2% of time	Temperature and DO part of the ecological classification. Potential to directly impact on the health of biological elements. Classification integrated into overall ecology. Failure of temperature or DO results in failure of water body	WFD assessment from 2009. Will continue to 2030
Decrease in Dissolved Oxygen (DO)	DO monitoring (high frequency)	DO value no less than 4 mg l ⁻¹ for more than 5% of time		WFD assessment from 2009. Will continue to 2030
Fish behaviour, fish mortality	Sub-metrics under the fish classification scheme in WFD	Failure of ecological quality ratios (EQR) in the overall sub-metrics	Changes in fish behaviour relating to migration patterns and spawning are identified in the fish classification scheme. Change in fish species composition must relate to a pressure	WFD assessment from 2009. Will continue to 2030
Benthic invertebrates	Limited data on the effect of temperature on benthic invertebrates		More information needed	
Change in phytoplankton community	Sub-metrics under the marine plant classification scheme	Exceedance threshold based on 30% deviation of natural population (community indicators)	Significant deviation in community composition is part of the normative definitions and will be identified in the phytoplankton classification tools	WFD assessment from 2009. Will continue to 2030
Impact on SPA/SAC biological element	Listed under the Habitats Directive	Measurable change in a protected species or conservation area	Modification of pressure as to eliminate the impact on the high conservation species or area	Habitats Directive

2 Approach

2.1 Data handling

The contaminants of importance to the marine environment are described in this section and in each case an attempt has been made to provide data from studies that consider each of the relevant contaminant groups. Because the East Suffolk coast is relatively sparsely populated and there are few major industries the area has been chosen as a reference site for several studies when comparisons to the more industrialised estuaries e.g. on the North East coast of the UK are the subject of study. Literature data "Contains Environment Agency information © Environment Agency and database right" for contaminants measured. The main locations from which data were available are shown in Figure 1.

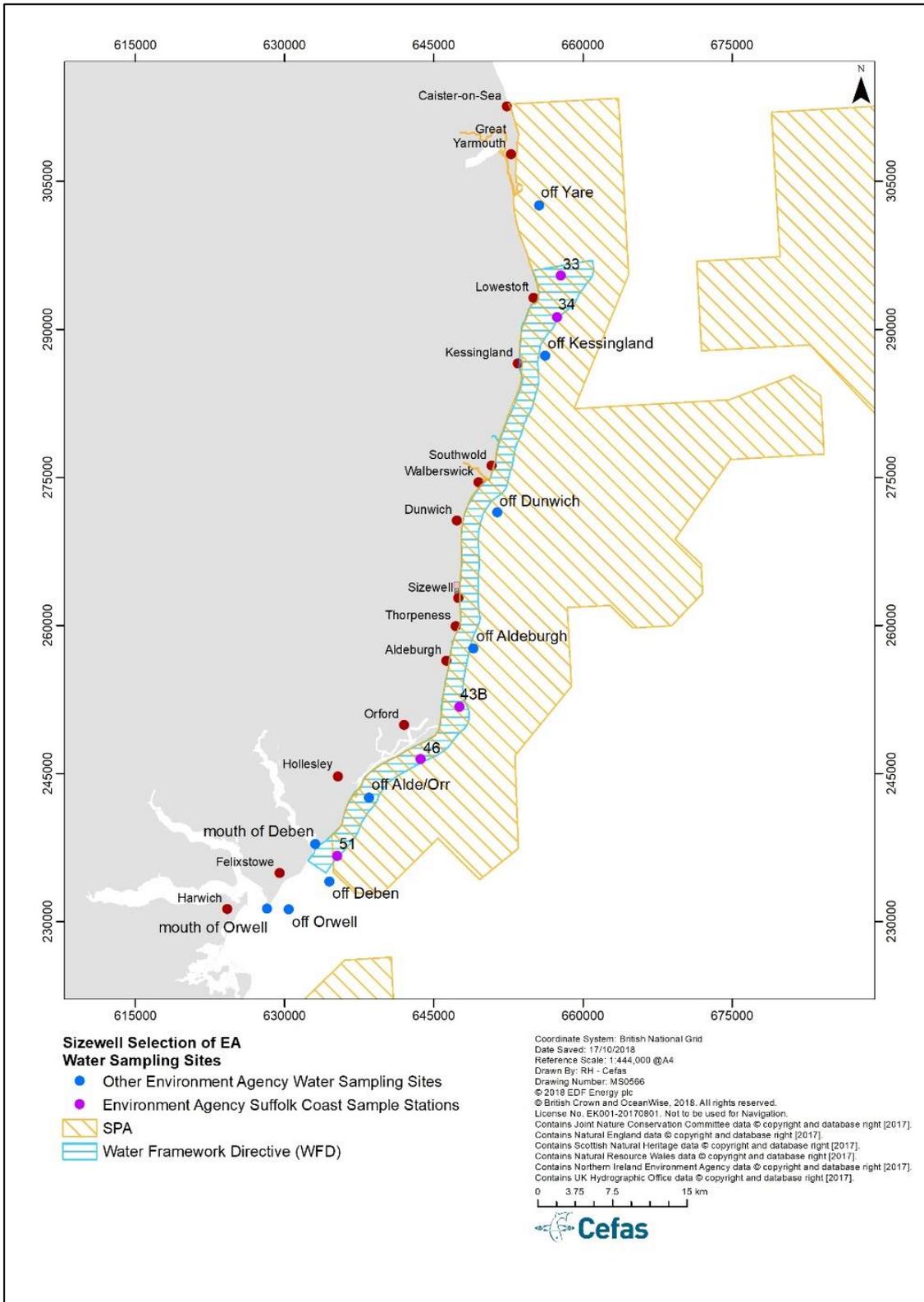


Figure 1 - The Environment Agency (EA) sampling stations for which water quality data were available are shown in relation to Sizewell Power Station and major towns on the Suffolk coast. The numbered sample locations are the Suffolk Waterbody sampling points and the Suffolk Waterbody is delimited by 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 Environment Agency sampling points are shown as blue circles.

3 Results

3.1 General Literature data

Dissolved and sediment associated metals

Metals enter the aquatic environment as a result of various processes. On the East coast of the UK the main sources are geological weathering, leaching of fertilizers, atmospheric deposition, animal excretion and the discharge of human sewage. Other sources include leaching from dumps and surface runoff e.g. from roads which contain metals that are present as a result of the abrasion of metal in the road surface and from vehicle lubricants and components. Metals enter estuaries both from feeder rivers and from direct discharges. These metals tend to be trapped in estuaries and accumulate in sediments. Physical disturbance and changes in physicochemical processes may make the metals in sediments more or less available for accumulation by marine organisms.

Cadmium

Cadmium has been used in the manufacture of wide range of products such as batteries, paints, plated metal etc, which may have entered waste streams for disposal to landfill in the past. A number of diffuse sources of cadmium release to controlled waters exist: numerous contributory discharges to STWs; deposition from air directly and indirectly via land; from agricultural land subject to fertiliser and biosolids applications; and from cadmium containing wastes that have been recycled or disposed to landfill. Based on monitoring data in 2006 none of the 39 waterbodies sampled in the Anglian region failed the Cadmium EQS (Pollution reduction plan for Cadmium, 2008).

Although cadmium does not break down in the environment physical and chemical processes that modify its mobility, bioavailability, and residence time in different environmental media may affect it. In both fresh and saltwaters, particulate matter and dissolved organic material may bind a substantial portion of the cadmium, and under these conditions cadmium may not be bioavailable due to this binding (Callahan et. al., 1979, Kramer et. al., 1997).

Studies conducted on the Orwell in 1999 showed elevated cadmium concentrations in sediments associated with salt marshes near the mouth of the estuary with values around 0.8 – 2.0 $\mu\text{g g}^{-1}$ dry weight. Sampling of different estuarine organisms from the Orwell showed mussels, *mytilus edulis* to have the highest tissue concentrations of cadmium, 1.43-2.85 $\mu\text{g g}^{-1}$ dry weight (Wright and Mason, 1999).

Chromium

Although chromium occurs naturally it also enters the environment through emissions from the metallurgy and metal-finishing industries, e.g. chromium compounds are used in ferrochrome production, electroplating, pigment production, and tanning and from its use as a chemical intermediate. These industries, the burning of fossil fuels and waste incineration are sources of chromium in air and water. In surface waters, chromium exists in two oxidation states, chromium (III) and chromium(VI) or hexavalent chromium, but the more thermodynamically stable state is Cr(VI). Almost all the hexavalent chromium in the environment arises from human activities. In the hexavalent oxidation state Cr (VI), chromium is relatively stable in air and pure water, but is reduced to the trivalent state CR (III) when it comes into contact with organic matter in biota, soil, and water. Chromium(III) is less toxic than Cr(VI) and its low solubility in water limits its bioavailability.

The UK EQS values derived in 1984 (Mance et. al., 1984a) were for total dissolved chromium and the freshwater standards were banded according to water hardness. The data available for the effects of chromium on marine species indicated that the acute toxicity of hexavalent chromium was extremely variable. Fish appeared to be considerably less sensitive than invertebrates, although fish larvae were reported to be susceptible to chromium contamination. The limited information available did not entirely support the view that trivalent chromium was less toxic than the hexavalent form. Because of this and the possibility of transformation between the two species, the EQS was defined as being for total chromium. The

standard was based on a chronic lowest adverse effects value of $30 \mu\text{g l}^{-1}$ for a polychaete worm. This value was halved to give the annual average standard. The EQSs were subsequently revised and new values are shown in Table 3 and 4.

Chromium is found in sediments and can pose a hazard to sediment dwelling organisms at concentrations above 52.3 mg kg^{-1} (CCREM, 1987).

Laboratory experiments on annelids, crustaceans and molluscs have resulted in bioconcentration factors (BCFs) in the range 158 to 596 for annelids and 383 to 620 (based on dry weights) for molluscs and crustaceans in the laboratory (Hunt and Hedgecott, 1992a). Much lower BCFs have been calculated in the field: 0.46 to 15 for annelids and <1 for molluscs and crustaceans. These BCFs indicate that chromium is unlikely to bioaccumulate under field conditions.

Copper

Uses of copper include electrical wiring and electroplating, the production of alloys, corrosion of copper piping, and roofing material. A number of copper-based products are also manufactured as antifouling paints, pesticide formulations, and preservatives for wood. Major industrial sources include mining, smelting, refining and coal-burning industries. Copper is also an essential nutrient and so is present in human and animal wastes. However, a few investigations (e.g. Young et. al., 1979; Claisse and Alzieu, 1993) have concluded that boat traffic may be a significant source of estuarine copper through its use in anti-fouling paints. The latter source has become more important since the late 1980s when the use of tributyltin-based antifouling paints on small vessels was phased out in Britain and elsewhere.

Copper may be present in a natural water system in a number of forms, either dissolved in solution, as a precipitate or absorbed to organic matter (Mance et. al., 1984b).

The high concentrations of particulate matter in most estuaries will facilitate the removal of copper from solution by adsorption to suspended particles, which in turn may be deposited and accumulate in sediments. Estuarine sediments are thought to be the most important depositional site for particulate copper transported from rivers, although remobilisation may occur when sediment is disturbed.

A review of toxicity data for copper to freshwater and saltwater organisms found that invertebrates exhibited slightly greater sensitivity to divalent copper than fish species tested (Mance et. al., 1984b). There were also indications of the moderation of toxicity in the presence of natural and manufactured chemicals such as humic acids and detergents.

A study of copper loadings from different sources in the Orwell, Ore and Deben indicated that docks (2874 kg y^{-1}), marinas and estuarine sewage works were important sources for the Orwell and marinas also made a significant contribution to the copper loading in the Deben and Ore although there was no data for sewage works for the latter two sites. The concentration range for copper for the three estuaries was $0.5\text{-}75 \mu\text{g l}^{-1}$ (Matthiessen et. al., 1999). The copper concentration measured in the sediments of the Orwell estuary in 1999 were $> 20 \text{ ug g}^{-1}$ dry weight at 15 of the 20 sites sampled. Higher concentrations were associated with port and marina facilities. The highest tissue concentrations of copper for organisms sampled from the Orwell estuary were recorded for the periwinkle *Littorina littorea*, 86.5 ug g^{-1} dry weight. The tissue concentration range for eight species sampled was $6.61\text{-}98.6 \text{ ug g}^{-1}$ dry weight (Wright and Mason, 1999).

Lead

With the exception of Lead nitrate and chlorate, and, to a much lesser degree, chloride, the salts of lead are poorly soluble in water; it also forms stable organic compounds (Nordic Council of Ministers, 2003). Lead is primarily used in its elemental form but is also used for manufacture of lead oxide and alkyl lead (Pollution reduction plan for lead, 2008). Tetraethyllead and tetramethyllead were used extensively as fuel additives and the latter breaks down in the environment to form trialkylalkylleads. In contrast to tetraethyllead and tetramethyllead, trialkyl compounds are less volatile and more readily soluble in water. Lead still has a variety of uses e.g. it is used in batteries, lead sheet and as lead oxide as a PVC stabiliser.

Entry into the aquatic environment occurs through releases through atmospheric deposition from the the burning of coal and oil, through abrasion of lead containing products in domestic properties and through inputs via sewer which represents the largest source (Pollution reduction plan for lead, 2008).

Following a review of toxicity data in 1992 a standard of $10 \mu\text{g Pb l}^{-1}$ was proposed for saltwater but Lead becomes predominantly associated with particulates and so the transport of lead in estuaries and coastal waters is closely linked with the movement of particles. A study of the Orwell estuary in Suffolk showed sediment lead concentrations of $> 50 \mu\text{g g}^{-1}$ at four of the sites on the Southern shore and one on the Northern shore of the outer estuary. Canadian interim marine sediment quality guidelines (CCREM, 1998) for lead recommend a threshold value of 30 mg kg^{-1} above which biological effects may be expected.

Bioconcentration factors (BCFs) of ~3000 times have been demonstrated in some studies using bivalves but generally fish had lower BCFs and in studies of organisms from contaminated sites BCFs were also lower than those found in the laboratory. Samples of a number of organisms collected from the Orwell estuary showed similar concentrations of lead for mussels *Mytilus edulis*, $3.15 \mu\text{g g}^{-1}$ dry weight (range 0.81-5.93); cockles *Cerastoderma edule* $4.6 \mu\text{g g}^{-1}$ dry weight (range 0.95-9.99) and the polychaete worm *Nereis diversicolor* 3.26 (range 0.31-4.81), (Wright and Mason, 1999).

Iron

Iron is the fourth most abundant element in the Earth's crust. While it is naturally released into the environment from weathering, it may also be released into the aquatic environment through human activities, such as burning of coke and coal, sewage, iron related industries and the corrosion of iron and steel (CCREM, 1999).

In unpolluted oceanic seawater, concentrations of iron between 2.8 to 29 ng l^{-1} and 224 to $1,228 \text{ ng l}^{-1}$ have been reported, although higher concentrations may be found in estuarine waters.

On reaching saltwater, suspended iron oxyhydroxides are rapidly precipitated such that at salinities of 10 ppt or greater, the vast majority of the iron present occurs in particulate form and is effectively removed from solution. In anoxic marine waters, ferrous iron is mobilised from sediments and diffuses into the water column.

Data reviewed on the toxicity of iron to fresh and saltwater species proposed the same EQSs for iron in solution of $1000 \mu\text{g l}^{-1}$ (as an annual average). Due to a lack of data, the derivation of an objective EQS based on iron toxicity was not considered possible. Therefore, the above values are based on observations of general water quality at various estuarine and marine sites. A further review in 1998 considered the current annual average of $1,000 \mu\text{g l}^{-1}$, was still appropriate (Whitehouse et. al., 1998).

Marine organisms accumulate iron but also rapidly excrete it in clean water conditions. Normally, tissue concentrations of iron are related to the water and sediment concentrations, but there is considerable variability. Tissue concentrations vary seasonally, being lower in winter and spring than in summer and autumn and furthermore tissue and shell concentrations increase with increasing salinity (Whitehouse et. al., 1998). The bioaccumulation of iron by marine organisms does not appear to pose a hazard to higher trophic levels.

Mercury

Mercury is a metal, which is liquid at normal temperatures and pressures. It is present in the environment in three oxidation states and as inorganic (mercuric II chlorides, sulphides, hydroxides and oxides) or organic (e.g.methylmercury) form.

Atmospheric pollution from industrial production is probably low however the burning of fossil fuels is a source of mercury. Although the use of mercury is decreasing, high concentrations of the metal are still present in sediments associated with previous industrial applications of mercury.

Dissolved mercury associates with organic matter and particles suspended in the water column (up to 95%, Eurochlor, 1999) and so will accumulate in sediments. Methylation of sediment bound mercury produces methylmercury and this form of mercury is bioavailable and is a hazard to aquatic life (Campbell et al., 1986).

Based on 21 data for fish and amphibians, 68 data for invertebrates and 35 data for algae, a PNEC for inorganic mercury of 470 ng l⁻¹ was derived. Based on 11 data for fish, 9 data for invertebrates and 7 data for algae a PNEC for organic mercury of 10 ng l⁻¹ was estimated. For inorganic mercury, worst case concentrations in coastal waters and estuaries found recently are up to 170 ng l⁻¹ giving a safety margin up to 3 between PEC and PNEC. For organic mercury, a worst case PEC of 8.5 ng l⁻¹ was calculated leading nearly to no safety margin (Eurochlor, 1999).

In terms of bioconcentration of mercury from water Slooff et al., (1995) give inorganic mercury BCF values of 190-5,300 l kg⁻¹ for molluscs. Values derived for mussels (*Mytilus edulis*) and recalculated as part of a risk assessment for mercury gave a BCF of 13,061 l kg⁻¹ dry weight. For methylmercury the reported BCF = 99,500 l kg⁻¹, based on recalculated data (Eurochlor, 1999).

Bioconcentration of methylmercury from sediments was estimated to be between 10 (*Fucus vesiculosus*) and 100 (*Mytilus edulis*) times higher than sediment concentrations at the sites they were collected from in the Mersey estuary (Langston et al., 1995).

Data for the Orwell estuary indicate that for six out of a total of 20 sites sampled the sediment concentration of mercury was > 0.4 ug g⁻¹ dry weight at four of 20 sites sampled and for a range of organisms sampled the tissue concentration range was 0.04 -0.59 ug g⁻¹ dry weight. The highest tissue concentration was recorded for the cockle, *Cerastoderma edule* and the lowest for the seaweed *Enteromorpha* spp.

Nickel

Nickel is used extensively as a metal in alloys, as a plating material, in batteries manufacture and as a catalyst. Nickel releases to controlled waters reported to the Pollution Inventory for 2006 totalled 66 tonnes, mostly from sewage treatment works, the chemical industry and metal production and processing plants. Disused metal mines are thought to be a potentially significant source of release to water. Abrasion and erosion of nickel-containing products in households and commercial premises results in a diffuse source of nickel discharged to STWs and to land and water directly. Abrasion of road materials that incidentally contain nickel – such as slag, ashes and waste products – constitutes a diffuse source of runoff to surface water directly or to STWs for treatment. Road runoff will also contain nickel from engine oil containing an accumulation of abraded engine parts, dust from worn brake linings, wear losses from tyres, products of car body corrosion and deposits from vehicular emissions. It has been estimated that the rate of nickel released from abrasion of urban road surfaces in England is about 21 kg per kilometre per year (Luker and Montague, 1994). The distribution of deposited nickel between land, surface runoff to water and surface runoff to STWs is unknown.

Twenty sites sampled on the Orwell estuary in 1999, had Nickel concentrations of 9 - >30 ug g⁻¹ dry weight. The highest concentrations of Nickel in the sediments were measured in samples from the Southern shore near the mouth of the estuary close to areas of port development but on the Northern shore the higher concentrations were closer to the riverine end of the estuary but again this is probably associated with the location of port facilities. Tissue concentrations of a range of organisms sampled from the Orwell ranged between mean values of 9.06 -96.2 ug g⁻¹ dry weight. The highest tissue concentration was recorded for the polychaete worm *Arenicola marina*.

Zinc

Zinc is used in coating to protect iron and steel, in alloys for die casting, in brass, in strips for dry batteries, in roofing and in some print processes. It may enter the aquatic environment through natural or anthropogenic sources, including sewage and industrial discharges.

Concentrations of zinc have been measured in water, sediments and biota as part of the National Monitoring Programme at sites throughout the UK in estuaries and coastal waters (MPMMG, 1998). The biggest loading of zinc calculated for the Orwell, Deben and Ore was from sewage treatment works discharges, 1638

kg yr⁻¹ on the Orwell (Matthiessen et. al., 1999). Zinc loading from marinas on the Deben and Orwell were about a quarter of this value.

Zinc is transported in natural waters in dissolved form and associated with suspended particles (Mance and Yates, 1984). In river water, zinc is predominantly present in the dissolved form. In estuaries, where concentrations of suspended particles are higher, a greater proportion of the zinc is adsorbed to suspended particles (CCREM, 1999). In seawater, much of the zinc is found in dissolved form as inorganic and organic complexes. In a survey of six Essex and three Suffolk estuaries there were 74 out of 138 year/location combinations where the concentration of zinc exceeded 10 µg l⁻¹. The marine EQS is 6.8 µg l⁻¹ (expressed as a dissolved annual average concentration).

A review of toxicity data for zinc to freshwater and marine organisms showed that invertebrates were generally more sensitive than fish, while, effects on marine macro and microalgae were noted at concentrations slightly lower than those reported for invertebrates (Mance and Yates, 1984). The toxicity and bioaccumulation of zinc are also reported to be greater at lower salinity (Hunt and Hedgecote, 1992b).

Zinc accumulates in sediments and can pose a hazard to sediment dwelling organisms at concentrations above 124 mg kg⁻¹ (CCREM, 1999) particularly as it is readily bioaccumulated. Several species of crustacean are able to regulate the uptake of zinc but, at higher concentrations, this process appears to breakdown leading to an influx of zinc. Zinc concentrations measured in sediments of the Orwell estuary were > 40 µg g⁻¹ dry weight for all 20 sites sampled with maximum concentrations > 120 µg g⁻¹ dry weight at sites associated with port activities (Wright and Mason, 1999). The tissue concentrations for zinc measured in eight species ranged between 40-269 µg g⁻¹ dry weight with the algae *enteromorpha sp* at the low end of the range and the polychaete worm *Nereis diversicolor* showing the highest concentrations particularly associated with moorings and marinas.

Arsenic

Arsenic is a commonly present element with metalloid properties. Its chemistry is complex and there are many different compounds of both inorganic and organic arsenic. Arsenic enters the aquatic environment from natural diffuse sources and from anthropogenic point and diffuse sources.

A data review of the aquatic toxicity of arsenic to freshwater and saltwater organisms derived an annual average EQS value of 50 and 25 µg l⁻¹, (expressed as a dissolved concentration) for freshwater, and coastal and estuarine waters respectively (Mance et. al., 1984c). Although limited data on the toxicity of arsenic to marine organisms was considered as part of this review it was concluded that invertebrate species appeared more sensitive than vertebrate species and in particular during the larval stages.

Arsenic is found in sediments and can pose a hazard to sediment dwelling organisms at concentrations above 7.24 mg kg⁻¹ according to the Canadian interim marine sediment quality guidelines (CCREM, 1999). Sediment concentrations of arsenic measured in the sediments of the Orwell estuary were > 18 µg g⁻¹ dry weight at 18 of 20 sites sampled which exceed the level at which Canadian guidelines suggest that biological quality could be compromised. A range of marine organisms has been found to accumulate arsenic from sediments and the water column, including bivalve molluscs and macro algae. While these species appear to accumulate arsenic to quite high levels, a large proportion may be present as arsenobetaine which is a water soluble compound that poses little hazard to organisms that ingest it (Smith and Edwards, 1992). Arsenic is bioconcentrated in organisms but is not biomagnified in food chains and so bioaccumulation is unlikely to be a problem in marine organisms.

Organotin

The tributyltin compounds act as fungicides, disinfectants, and microbiocides. They are used in water cooling towers, wood preservatives, hard surface disinfectants for farm premises; as materials preservatives in textiles, carpet backing, sponges, rope, fiberfill, foam, paper, and building materials (e.g., drywall, joint compound, grout); metal working fluids; and petrochemical injection fluids (EPA, 2008). The most well known use of tributyltin (TBT) in terms of environmental impacts was as an antifouling paint. On 1st July 1987 the use of TBT as an antifouling paint used on fish farming equipment and boats of <25 metres waterline length was banned. A ban on TBT use on all ships hulls began 1st January 2008. As a consequence of these bans the environmental concentration of tributyltin concentration in the water column of

estuaries has shown a downward trend (Dowson et. al., 1993, Matthiessen et. al., 1999). A survey of Essex and Suffolk estuaries in 1993 showed that for the Alde the concentration of TBT in sediments ranged from $<3 - 653 \text{ ng g}^{-1}$ in spring 1991 to $<3 - 186 \text{ ng g}^{-1}$ in spring 1992 a similar decrease was shown for the sites monitored on the Alde $<3 - 466$ in spring 1991 to $<3 \text{ ng g}^{-1}$ in spring 1993

Pesticides

Studies of pesticides in seawater have generally indicated a decreasing trend following the successive introduction various control measures (e.g. Power et. al., 1999).

Flounder from several UK estuaries were shown to have depressed acetyl- and butyl-cholinesterase (ChE) activity in muscle which is a response frequently associated with exposure to carbamate and organophosphate pesticides (Kirby et. al., 2000). Kirby et. al., have shown that flounder sampled from several locations on the Mersey, Tees, Humber, Tyne and Tamar estuaries in 1997 showed significant ChE inhibition compared with fish from the Alde, concentrations of up to eight organophosphates and six carbamate insecticides were above detection limits in all the surveyed estuaries except the Alde, so it is assumed that they were at least contributing to the observed effects.

The introduction of pesticide-based biocides to replace tributyltins for antifouling has led to an increase in the presence of compounds such as diuron and the triazine herbicide irgarol in the freshwater and marine environment. Studies have detected a variety of these compounds in Norfolk and Suffolk broads as well as the rivers Bure and Yare (Lambert et. al., 2006).

Hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are hydrophobic organic chemicals which are derived from the combustion of petroleum products and from various industrial processes. Many of these compounds are stable and highly toxic, some are carcinogens and others are mutagenic. The major route of entry of PAHs to the aquatic environment is through atmospheric deposition either directly onto the water surface or via material washed off of hard surfaces such as roads. PAHs are also present in various oils and are particularly enriched during use in lubricating oils.

PAHs are frequently considered as a group however they differ markedly in solubility, volatility and other physico-chemical characters which affect their distribution, bioavailability and toxicity.

Exposure to UV light present in natural sunlight may also increase the toxicity of PAHs and this effect has been widely reviewed (Arfsten et. al., 1996). Phototoxicity occurs in the presence of UV light by the formation of free radicals and oxidation of parent PAHs to more toxic forms which may damage a variety of macromolecules.

PAHs may also become more toxic due to enzymatic transformation within an organism to form highly reactive compounds which bond with protein and DNA; this may give rise to mutations which ultimately induce tumour formation or birth defects. Only certain PAHs are metabolically activated, not all organisms have the enzyme systems which metabolise them to the more reactive form and cellular DNA repair mechanisms vary between species.

An extensive survey analysed for 15 PAHs in water from UK estuarine and offshore locations between 1993 -1995 (Law et.al., 1997). The data showed that offshore sites had generally undetectable levels but 23 sites had total PAH concentrations greater than $1 \mu\text{g l}^{-1}$, these included the Thames and Great Ouse as well as a number of industrialised Northern estuaries. The PAHs detected in one sample on the Tees included, naphthalene, acenaphthene, fluorene and phenanthrene. Analysis of 15 PAHs in sediments around in estuaries, coastal and offshore waters (Woodhead et. al., 1999) also highlighted high concentrations in the more industrialised Northern estuaries. However relatively high concentrations of a number of PAHs including anthracene, pyrene, benz a anthracene, chrysene, benz e pyrene were measured in sediments from the River Blackwater in Essex. Four of the concentrations detected at this site were above predicted effects thresholds based on Canadian sediment quality guidelines (CCREM, 1999).

Studies that included a survey of the River Alde as well as industrialised estuaries in Southampton and at various Northern locations (Kirby et. al., 1999) showed that relative to fish from the Alde estuary those present in the industrialised sites had elevated levels of enzymes that are induced following exposure to PAHs. These data further support the conclusion that with the exception of a few point sources there are relatively low inputs of these contaminants in this area

Inorganics

Ammonia

Ammonia is excreted by animals and plants and is a product of the decomposition of all organisms. The intensification of agricultural practices and increased density of human populations in towns and cities has led to higher inputs of ammonia to rivers and estuaries. In the marine environment both ammonia in its ionised NH_4 and unionised NH_3 form may contribute to toxicity although it is the unionised form that is the most toxic. Ammonia may be lost from water by volatilisation or under aerobic conditions may be oxidised to nitrite and then nitrate. Various water quality parameters influence the toxicity of ammonia mainly by increasing the proportion of the most toxic, unionised, form of ammonia. The pH of seawater has the most influence on ammonia toxicity, increasing it by 1 unit (e.g. pH 7 to 8) at 10°C produces about a 10 fold increase in NH_3 concentration while increasing the temperature by 10°C (10 to 20°C) approximately doubles the NH_3 concentration. Increasing salinity from 0.5 to 32 ppt at 10°C reduces the NH_3 concentration by about 15% (Eddy, 2005).

In a survey of water quality and nutrients in lowland rivers in Suffolk (Howden et. al., 2009) the mean NH_4 concentration and range for the period 1981-2005 were 1.75 (0.01-70), 0.78 (0.0-39) and 0.11(0.0-11.9) mg l^{-1} for the Loathingland Hundred river that joins the coast at Lowestoft and the Rivers Blyth that joins at Southwold, and the Ore/Alde that joins the sea approximately 16 kilometres South of Aldeburgh. The two higher mean values would result in unionised ammonia concentrations above the recommended standard for unionised ammonia 0.021 mg l^{-1} $\text{NH}_3\text{-N}$ (assuming a salinity of 30 psu, a temperature of 18°C and pH of 8).

Chlorine

Chlorine is used in the manufacture of a wide variety of products but its use in preventing biofouling in cooling water systems is the main focus here. Different chlorine dosing regimes may be used in cooling water systems with some focussing on the main settlement period of bivalve molluscs and others applying low level continuous chlorine dosing e.g. 0.5-1.5 mg l^{-1} (Jenner et. al., 1997). When chlorine is added to seawater it oxidises the bromide ions present to form the free oxidants - hypobromous acid and hypobromite. These free oxidants remain undissociated and more effective as a biocide in seawater. A number of non-oxidising chlorination byproducts (CBPs) may also result when chlorine is added to seawater and combines with organic matter:

- (i) Volatile organohalogenes – predominantly trihalomethanes e.g. bromoform and bromochloromethanes.
- (ii) Semi-volatile bromoacetonitriles
- (iii) Non volatile bromoacetic acids, bromophenols and other CBPs

In addition to the above, the presence of ammonia in seawater may also lead to the formation of monochloramine (NH_2Cl) and bromamines (NH_2Br , NHBr_2 , NBr_3) (Taylor, 2006).

The main environmental concerns regarding cooling water chlorination are therefore the potential for any toxicological effects of residual oxidants much beyond the immediate discharge to the sea. Concerns regarding CBPs consider toxicity but are also focussed on their persistence and potential to bioaccumulate.

Studies conducted in 1981 to develop a model of chlorine decay used the discharge from the Sizewell A power station to validate the model (Davis and Coughlan, 1983). Samples of the plume were taken along a transect based on diminishing temperature (dilution) away from the point of discharge. Two sampling occasions in September 1991 are described for which the total residual oxidants (TRO) produced by chlorine addition have an initial concentration at the point of discharge of 0.05 and 0.1 mg l^{-1} decreasing to 0.01 mg l^{-1}

within 1000 and 1500 metres respectively. A similar study conducted in August 1993 (Jenner et. al., 1997), measured a TRO concentration of 0.02 mg l⁻¹ at 375 metres from the discharge, this is comparable to the results described by Davis and Coughlan for the lower initial discharge concentration. At the time of the 1993 study, Sizewell was operating on a once through with a flow of 45 m³ s⁻¹ and a chlorination regime from April to November of up to 1 mg l⁻¹ dosed at the strainer outlet to maintain a residual of 0.2 mg l⁻¹ at the condenser inlet.

During the plume studies in 1993, bromoform concentration was also measured in the discharge. The initial seawater concentration of bromoform was just over 9.85 µg l⁻¹ and decreased by 966 metres from the discharge to 2.35 µg l⁻¹ which is below the proposed reference level concentration for bromoform of 5 µg l⁻¹ as a Maximum allowable concentration (MAC), Taylor, 2006. Under WFD, the MAC values are assessed as an annual 95 percentile (UKTAG, 2013). Compared to an existing standard for chloroform (trichloromethane, CHCl₃) 2.5 µg l⁻¹ as an annual average if this is considered an equivalent the concentration of bromoform measured within 1 km of the discharge is just below this value (Common Implementation Strategy, Substance datasheet 32, trichloromethane, 2005).

At this time of the other CBPs analysed for (Table 6), dibromoacetonitrile, was the only one detected and this was only present in a single plume sample at a concentration of 0.21 µg l⁻¹(Jenner et. al., 1997).

Table 6 - CBPs analysed for but below detection in Sizewell discharge in 1993 study

Haloforms	Detection Limits
chloroform, bromodichloromethane, chlorodibromomethane	<0.1µg l ⁻¹
Haloacetonitriles	Detection Limits
dichloroacetonitrile, dibromoacetonitrile	<0.1µg l ⁻¹
Halophenols	Detection Limits
2,4-dichlorophenol	0.4µg l ⁻¹
2,4,6-trichlorophenol	<0.6µg l ⁻¹
2,4-dibromophenol	<0.02µg l ⁻¹
2,6-dibromophenol	<0.1µg l ⁻¹
2,4,6-tribromophenol	<0.05µg l ⁻¹

Nutrients

Nutrient additions to estuaries and coastal water bodies occur both naturally as a result of geological weathering, atmospheric deposition and nitrogen fixation by plants, but growth of human populations has led to increasing inputs of nutrients from sources such as agriculture, wastewater treatment plants, urban run-off, and consumption of fossil fuels. As a result, nutrient inputs have increased to many times their natural levels to the point that eutrophication is now regarded as one of the greatest threats to coastal ecosystems. Eutrophication is defined as 'the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned' (CEC 1991a, Urban Waste Water Treatment Directive, 91/271/EEC).

In a tidal estuary, the potential for nutrient enrichment to have an impact on the waterbody is determined by the water residence time, the tidal regime and growth rates of primary producers (Painting et. al., 2007). Elevated nutrient concentrations can lead to increased primary production by phytoplankton (indicated by concentration of chlorophyll a (Chl-a) and macroalgae that may in turn impact upon dissolved oxygen (DO) levels, pH and turbidity. These changes may lead to the development of localised hypoxic or anoxic conditions, and indirect effects such as changes in biological community structure and mortality of fish or benthic animals. Nitrogen (as nitrate, nitrite and ammonium) and phosphorus are the primary nutrients that contribute to increased plant growth and potential for eutrophication. However, nutrient enrichment or hyper-nutritification does not necessarily result in undesirable effects and therefore does not always result in eutrophication (de Jonge and Elliott 2001, Tett et al., 2007, Painting et al., 2007).

In terms of inputs to the Suffolk coastal waterbody, Howden et al., 2009 conducted a review and analysis of Environment Agency water quality data from 60 sampling sites for the period between August 1981 and October 2005. The analysis indicated that the long-term average nitrogen concentration for rivers in this area (7.85 mg l^{-1}) was similar to but slightly lower than that for other agriculturally-impacted eastern UK rivers such as the Great Ouse and Thames. Although the River Deben was described as having one of the highest nutrient inputs of Suffolk and Essex estuaries, its high turbidity is thought to limit growth of phytoplankton and macroalgae (Nedwell et. al., 2002).

For coastal and marine waterbodies the EU objectives for the protection and maintenance of water quality have been set under various Directives and Conventions. Directives include the Urban Waste Water Directive (UWWTD, CEC 1991a), the Nitrates Directive (ND, CEC 1991b), the Habitats Directive (HD, CEC 1992), the Water Framework Directive (WFD, CEC 2000) and the Marine Strategy Framework Directive (MSFD, CEC 2008). Conventions include the Oslo Paris Convention (OSPAR 2003a, b) and the Helsinki Commission (HELCOM, Andersen and Laamanen 2009). Methodologies developed across Europe and in the Mediterranean (e.g. the Trophic Index, TRIX, Vollenweider et al., 1998) all assess the impacts of nutrient enrichment using measurements of key indicators such as concentrations of nutrients, Chl-a and DO in the water column (Devlin et al., 2011). Some of the more recent Directives (WFD, MSFD) and OSPAR include the additional identification of secondary impacts and undesirable disturbance to the ecosystem (e.g. low DO events, toxic algal blooms).

The OSPAR Strategy to Combat Eutrophication seeks to achieve 'a healthy marine environment where eutrophication does not occur'. The strategy requires that the eutrophication status of the maritime area be identified through the OSPAR Common Procedure (COMP, OSPAR Commission 2005), and the original target year was 2010. The first application of the OSPAR COMP by Contracting Parties was for the period 1996–2000 inclusive (OSPAR *Commission 2003*); the second application was for the period 2000-2006 (OSPAR *Commission 2008*), and the third application (2006-2014) is due in 2017. Under OSPAR, water bodies are classified as Problem Areas or Non Problem Areas.

The MSFD aims to reach or maintain Good Environmental Status (GES) in marine waters by 2020. Eleven descriptors of the state of the environment have been defined, including Descriptor 5 on human-induced eutrophication. Assessments under the MSFD are anticipated to be broadly similar to those under OSPAR.

The WFD requires the classification of all surface waterbodies into one of five ecological status classes: High, Good, Moderate, Poor or Bad (UKTAG 2008). Development of the UK nutrient standards was based on the offshore values established for OSPAR and aligned with freshwater reference values assuming conservative behaviour between nutrients and salinity. At present, coastal and transitional waters are assessed using only the winter value for concentrations of dissolved inorganic nitrogen (DIN, the sum of nitrate, nitrite and ammonia, measured in micromoles per litre, μM), as DIN is recognised as the primary driver of eutrophication. Impacts of dissolved inorganic phosphorus and other limiting nutrients are under investigation.

The relationship between the OSPAR Common Procedure and the WFD has been set out by OSPAR (2005). The boundary between OSPAR's Problem Areas and Non Problem Areas is the boundary between the WFD classes of Good and Moderate. The United Kingdom Technical Advisory Group (UKTAG, 2008) used this to define offshore thresholds and reference conditions for the WFD, and derived standards for coastal and transitional waters. These standards are related to salinity, and provide values for UK offshore, coastal and transitional waters (normalised for salinity). For assessment of ecological status, coastal waters extend to 1 or 3 nautical miles off the coast for England and Wales respectively, or have a salinity of 30 to 34.5. Transitional waters (estuaries) are generally described by a salinity of less than 30. The boundaries for WFD and MSFD assessments overlap in coastal waters (Figure 2). However, in coastal waters, the WFD assessment tools are used, and the MSFD is applied to descriptors which are not covered by the WFD (e.g. noise, litter, aspects of biodiversity).

WFD assessments of nutrients are based on winter (November to February inclusive) values of DIN, taking account of mitigation of impacts by light limitation. The concentration of suspended particulate matter (SPM) is used as a surrogate for light (UKTAG, in prep), and is used to designate waterbodies as "clear" or "not clear". "Clear waters" are described as waters with an annual mean of SPM of $<10 \text{ mg l}^{-1}$. "Not clear" waters are described as waterbodies with an annual mean SPM of $>10 \text{ mg l}^{-1}$. Not clear waters are grouped further by the mean annual SPM value, and described along a continuous gradient of "intermediate" ($10 < \text{SPM} < 100 \text{ mg l}^{-1}$), "turbid" ($100 < \text{SPM} < 300 \text{ mg l}^{-1}$) or "very turbid" ($>300 \text{ mg l}^{-1}$) conditions. The average winter DIN

concentration from waterbodies designated as *clear* is assigned to one of five classes along the WFD boundary conditions (high to bad) based on the value of the normalised winter DIN. For *not clear* waterbodies (annual average SPM >10 mg l⁻¹), the 99th percentile of winter DIN is assigned to one of four classes. Although values are classified into five classes overall, they are only *reported* to Europe as one of three classes, i.e. high, good or moderate. The poor and bad classes are indicative only and used internally. At present, the nutrient assessment does not calculate a numerical ecological quality ratio (UKTAG, in prep).

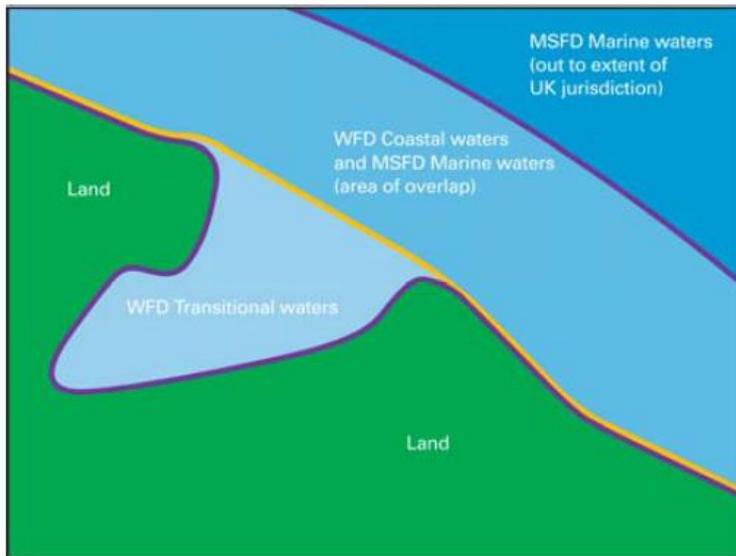


Figure 2 - Diagram to indicate overlap between the WFD and the MSFD. The WFD applies to estuaries and coastal water bodies out to 12 nm (baseline + 12 nm) for biological status, and 12 nm for chemical status. The MSFD applies to marine waters and includes coastal waters not addressed by the WFD or other Community legislation, as well as the full extent of the territorial waters of Member States (HM Government 2012).

The WFD DIN tool requires the measurement of 4 separate statistics: mean winter DIN, mean salinity, 99th percentile winter DIN, Mean annual SPM (which can be derived from other measures such as turbidity or light extinction). The assessment is made in a conditional stepwise procedure:

- Step 1. Compare mean winter DIN against OSPAR derived criteria
- Step 2. Compare mean winter DIN against salinity derived thresholds for *clear waters*
- Step 3. Compare 99th percentile DIN against SPM derived thresholds for *non-clear waters*

Winter DIN boundary (or threshold) values agreed by UKTAG (2008, in prep) for classifications of nutrient status as High, Good or Moderate are shown in Tables 7 and 8, below. For offshore waters, boundary values were set based on the OSPAR threshold of 15 μM between Non Problem Area and Problem Area (OSPAR 2003, Foden et al., 2011), which is equivalent to the WFD boundary between Good and Moderate (15 μM = 0.21 mg l⁻¹).

For 'clear' coastal waters (normalised to salinity 32), the winter DIN boundary between High/Good is 12 μM and between Good/Moderate is 18 μM (UKTAG 2008). These thresholds are equivalent to 0.168 and 0.252 mg l⁻¹ expressed as nitrogen (N). For 'Not clear' waterbodies, the 99th percentile of the winter DIN values are used. Winter DIN thresholds for classification of these waterbodies (Table 8) are based on a sliding scale, depending on the mean annual SPM value measured in each waterbody. For the Suffolk coastal waterbody, the current classification under the WFD indicates that SPM is intermediate, and that nutrients are Moderate and could potentially be Poor.

Table 7 - For clear waterbodies: boundary/threshold values for offshore, coastal and transitional waters (from UKTAG, in prep) used by the WFD (Anon, 2008). These values were set based on the OSPAR threshold of 15 μM for Problem Area vs Non Problem Area (OSPAR 2003, Foden et al., 2011), equivalent to the WFD boundary between Good and Moderate. For coastal and transitional waters, winter DIN values are normalised to salinity 32 and 25 respectively.

Area	Assessment	Salinity	Dissolved Inorganic Nitrogen (Winter mean, μM)	
			High / Good boundary	Good / Moderate boundary
Offshore	OSPAR	More than 34.5	10	15
Coastal (at salinity 32)	OSPAR WFD	30 - 34.5	12	18
Transitional (at salinity 25)	OSPAR WFD	Less than 30	20	30

Table 8 - For 'Not clear' waterbodies: winter DIN thresholds (μmol) for classification of waterbodies as Good, Moderate, Poor or Bad using the 99th percentile of the winter DIN values. Thresholds are shown on a sliding scale, depending on the mean annual SPM value measured in each waterbody.

Annual SPM (mg l^{-1})	DIN Threshold (99%ile) for Good / Mod	DIN Threshold (99%ile) for Mod / Poor	DIN Threshold (99%ile) Poor / Bad
10	31.2	169.5	372.0
25	43.4	181.6	384.1
50	63.7	202.0	404.5
75	84.0	222.3	424.8
100	104.3	242.6	445.1
125	124.6	262.9	465.4
150	144.9	283.2	485.7
175	165.2	303.5	506.0
200	185.5	323.8	526.3
225	205.8	344.1	546.6
250	226.1	364.4	566.9
275	246.4	384.7	587.2
300	266.7	405.0	607.5

Dissolved oxygen

The presence of dissolved oxygen at sufficient levels in all waterbodies including estuaries and coastal waters is essential to the survival and normal functioning of biological communities. In the marine environment chronic and acute oxygen deficiency occurs when levels fall between 2.0 and 6.0 $\text{mg l}^{-1} \text{O}_2$ and below 2.0 $\text{mg l}^{-1} \text{O}_2$ (levels $<2.0 \text{ mg l}^{-1}$ defined as hypoxic), respectively (OSPAR, 2005). Dissolved oxygen levels in parts of the marine environment have shown rapid change since the 1950s, and there is strong evidence that hypoxia in coastal areas is becoming more frequently linked to human activities (Diaz and Rosenberg, 2008). Anoxic or 'no-oxygen' conditions occur when levels fall below 0.2 $\text{mg l}^{-1} \text{O}_2$. These low oxygen levels can have adverse effects on marine organisms.

Oxygen depletion may occur over a number of timescales influenced by both seasonal and anthropogenic factors (Kemp et al., 2009). The solubility of oxygen varies with salinity, temperature and pressure (Garcia

and Gordon, 1992) and an increase in water temperature will lead to a decrease in oxygen saturation. The other major factor controlling dissolved oxygen concentration is biological activity: photosynthesis producing oxygen while respiration and nitrification consume oxygen. The proposed provisional Water Framework Directive standards for dissolved oxygen reflect these issues, while remaining generally compatible with previous recommendations. They are all 5%ile, i.e. they should be exceeded for 95% of the time Table 9.

Table 9 - Dissolved oxygen standards for transitional and coastal waters (Best et al., 2007)

WFD Status	Freshwater 5%ile (mg l ⁻¹)	Marine 5%ile (mg l ⁻¹)	Objectives
High	7.0	5.7	Protects all life stages of salmonid fish
Good	5.0–7.0	4.0–5.7	Resident salmonid fish
Moderate	3.0–5.0	2.4–4.0	Protects most life stages of non-salmonid adults
Poor	2.0–3.0	1.6–2.4	Resident non-salmonid fish, poor survival of salmonid fish
Bad	2.0	1.6	No salmonid fish. Migration survival of resident species

Survey data (2001- 2005) for the OSPAR East Anglia marine region did not indicate dissolved oxygen depletion, with a reported mean value of 7.87 mg l⁻¹ and range 2.55 – 10.90 mg l⁻¹ (n=382) (Foden et al., 2010)

Temperature

The effect of thermal inputs from power station cooling water upon the temperature regime of the receiving water must be assessed with respect to baseline conditions for the water body concerned. In this summary, temperature records from sources relevant to the Sizewell power station have been collated into time-series for the previous 48 years. Individuals on behalf of Cefas, councils, companies and other organisations have obtained records of coastal sea surface temperatures, for some stations, of more than 100 years duration. Approximately half of the stations started recording coastal temperatures in the mid 1960s. There are 30 stations in England and 8 stations in Wales and the Isle of Man where 25 out of 38 are still in operation. These datasets include records for Lowestoft, Southwold, Sizewell Power station. Near surface temperature and salinity samples have also been collected by ferries, the most recent, the Stena Partner, along 52°N between Harwich (formerly Felixstowe) and Rotterdam, from August 1970 onwards. Throughout the year, at weekly intervals, temperature data are recorded and water samples are taken at 9 standard station positions across the Southern Bight of the North Sea. The dataset for the end member location for this transect approximately 8 nautical miles offshore from Felixstowe was included with the above datasets to derive 98% values and to produce the data plot.

Data Collection

Cefas observers record coastal sea surface temperatures using calibrated thermometers approximately 6 – 14 times per month, usually close to the time of high water. Other organisations record sea surface temperature ranging from daily values to monthly means. The Cefas instruments are calibrated at Lowestoft to an accuracy of ±0.1°C. The accuracy of other instruments is not known, but is thought to be at least to an accuracy of ±0.2°C.

The ferry route observers record offshore sea surface temperature from the ships main seawater pipe using a calibrated thermometer 4 times a month. The temperatures are recorded to at least an accuracy of $\pm 0.2^{\circ}\text{C}$. The seawater samples are taken from the sea water main pipe to the harbour pump about 1.5 metres inboard.

Quality Assurance

Quality assurance checks are applied to the data for each station by comparing the current dataset with either a 5 or 10 year running mean for each month. The data is first tested to see whether it is normally distributed i.e. whether all the data are close to average. The standard deviation is calculated to see how tightly the data are clustered around the mean; three standard deviations are then calculated to account for 99% of the data. If the data is outside of this range (3 standard deviations) then the value is flagged and removed from subsequent analysis.

Derivation of temperature statistics

Figure 3 shows the range of temperature data for four locations in the Suffolk coastal waterbody from 1963 – 2013. Yearly averages are only derived from those years which have a complete set of monthly values. Table 10 and Figure 4 show the locations from which the temperature datasets were acquired.

The annual range of temperature for this region (based on mean monthly data for 2009 to 2013, Table 11) is bounded by a lower limit above 2 percentile of readings of 3.5°C and an upper limit of 98 percentile of all readings of 19.4°C , with occasional values exceeding these limits.

The main concern regarding water temperature elevation from cooling water input to Suffolk coastal water is that exceedance of specific standard values may result, or there may be an impact on the biology to the extent that (as this area is classified as heavily modified based on coastal protection) good ecological potential cannot be attained under the Water Framework Directive, or that protected species or habitats are impacted. Taking account of the most recent temperature data covering the five year period between 2009 - 2013 the 98 percentile is 19.4°C . Heat excess from the discharge of cooling water from Sizewell will need to be considered in relation to this value.

Table 10 - Data Summary _Sizewell PS & Adjacent Areas: Long Term Surface Sea Water Temperature Observations_ 1966 - 2013

SITE	SOURCE	SAMPLE TYPE	FREQUENCY	TIME SERIES
Lowestoft (52.450°N; 1.750°E)	(Source: CEFAS)	CTSLOWESTO CTN – CEFAS LOGGER	Monthly mean Daily Mean	1966 - 2013 2010 -2013
Southwold (52.316°N; 1.683°E)	(Source: CEFAS)	CTSSIZEWEL/ CTSSIZEWEL	Daily Mean	1967 - 2013
Sizewell PS (52.216°N; 1.633°E)	(Source: EDF, British Energy Generation Ltd., BNFL, CEGB)	CTMSIZEWEL/ CTSSIZEWEL	Daily Mean	1966 - 2013
Felixstowe- Rotterdam_ Pos 2 ((52.033°N; 1.666°E)	(Source: CEFAS)	FERRY ROUTE	Weekly	1970-2010

Table 11 - Summary statistics for the Suffolk Waterbody based on mean monthly temperature data for the period 2009 – 2013 (details in Appendix)

Measurement statistic	Temperature °C
Mean	11.43
Max	19.9
Min	3.0
98%	19.39
2%	3.53

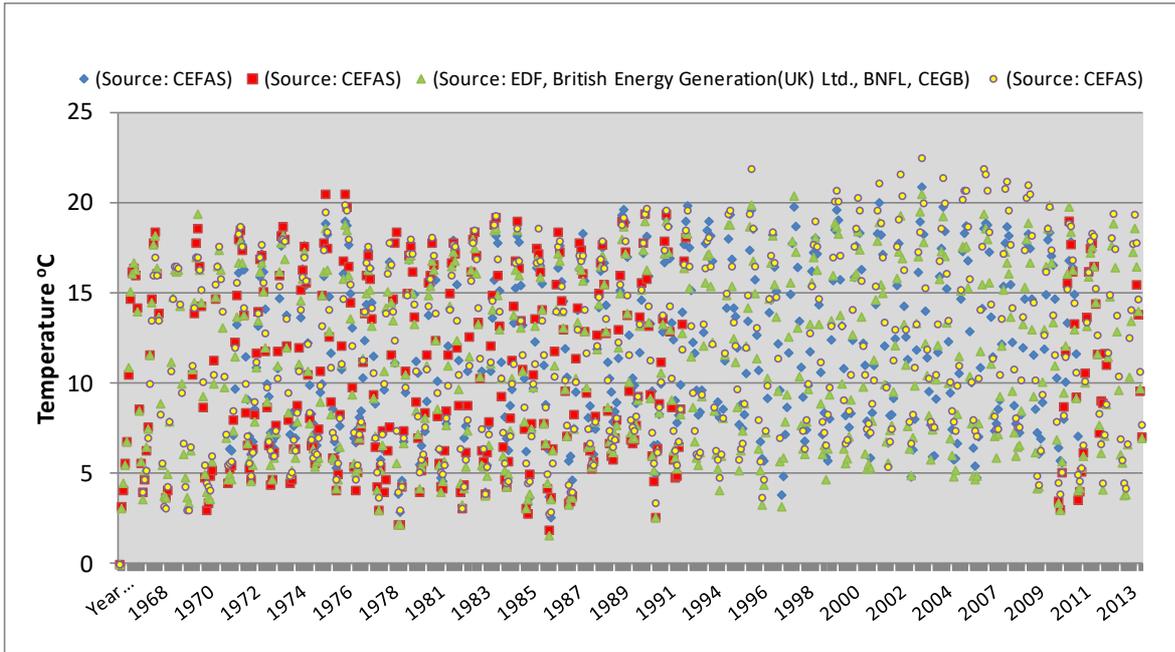


Figure 3 Monthly Sea Temperatures (°C) for four locations in the Suffolk coastal water 1966 – 2013

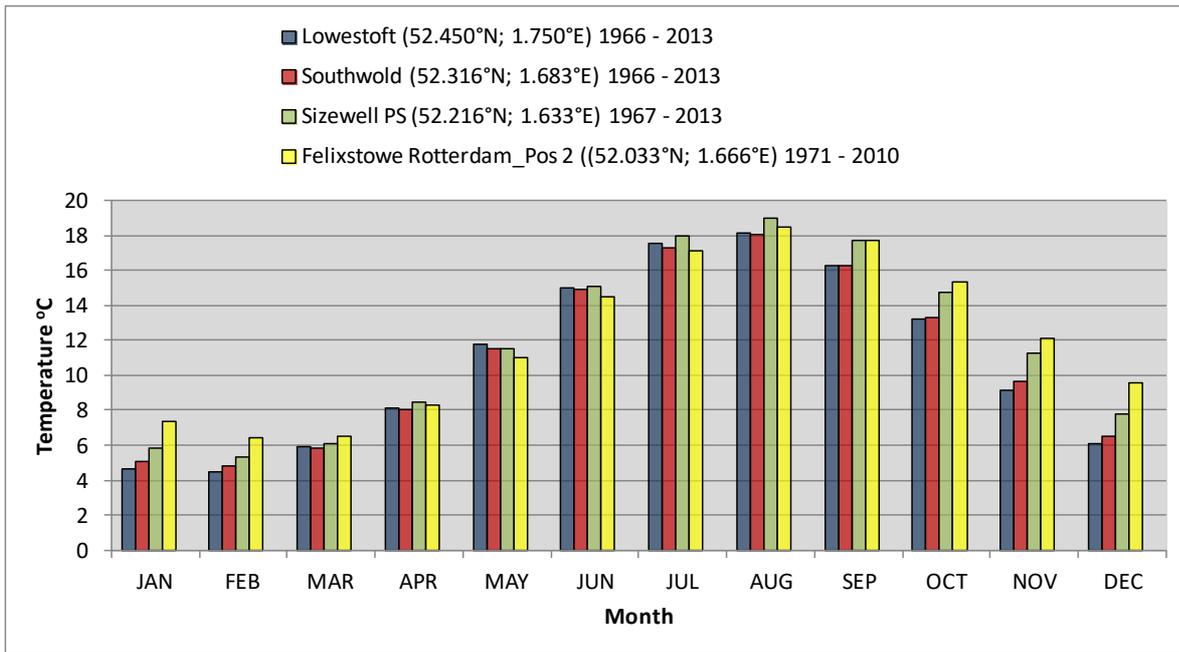


Figure 4 - Monthly mean Sea Temperatures (°C) for four locations in the Suffolk coastal water 1966 -2013

3.2 Environment agency data

This section describes Environment Agency monitoring surveys for compliance. The sites for which data are reported include a range of sites along the Suffolk coast of which those marked in bold in the Tables are within the Suffolk waterbody. However monitoring sites specifically associated to the Suffolk waterbody are only identified for the nutrient monitoring data. The data for dissolved metals covers the period 1989 to 2006 but the nutrients and inorganics data includes samples collected between 1991 and the early part of 2014. The EQS are derived from Directive 2013/39/EU as regards priority substances, cadmium, lead, nickel and mercury (Table 12).

Table 12 - Mean water column concentration (as annual average) for cadmium, lead and mercury from Environment Agency surveys 1989-2006. The years covered for specific determinands are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Sampling Location	Chemical mean concentration $\mu\text{g l}^{-1}$ as annual average	Concentration Range $\mu\text{g l}^{-1}$
	Cadmium dissolved AA EQS 0.2	1993-2006
Mouth of Orwell	0.104 (0.124 ¹)	<0.02-0.7(2.59 ¹)
Off Orwell	<0.25	<0.25
Off Deben	0.21 (0.29 ¹)	<0.25 – 0.42(1.11 ¹)
Mouth Deben	NA	NA
Off Alde/Ore	<0.25	<0.25
Off Aldeburgh	<0.25	<0.25
Off Dunwich	<0.25	<0.25
Off Kessingland	<0.25	<0.25
Off Yare	<0.25	<0.25
	Lead dissolved AA EQS 1.3	1991-2006
Mouth of Orwell	0.99	<0.02 -18
Off Orwell	-	<1.25
Off Deben	-	<1.25
Mouth Deben	NA*	NA
Off Alde/Ore	-	<1.25
Off Aldeburgh	-	<1.25
Off Dunwich	-	<1.25
Off Kessingland	-	<1.25
Off Yare	-	<1.25
	Mercury dissolved MAC-EQS 0.07	1991-2006
Mouth of Orwell	-	<0.01 -0.16
Off Orwell	-	<0.01 -0.27
Off Deben	NA	NA
Mouth Deben	NA	NA
Off Alde/Ore	-	<0.01-0.09
Off Aldeburgh	-	<0.01
Off Dunwich	-	<0.01-0.12
Off Kessingland	-	<0.01
Off Yare	-	<0.01
	Nickel dissolved AA EQS 8.6	1991-2006
Mouth of Orwell	1.56	<1-3.77
Off Orwell	1.49	<1-4.9
Off Deben	0.69	0.4-1.13
Mouth Deben	NA	NA
Off Alde/Ore	0.80	0.37-0.96
Off Aldeburgh	1.20	<3-4.1
Off Dunwich	0.69	0.34-0.88
Off Kessingland	-	<0.01
Off Yare	-	<0.01

*NA – not analysed; ¹ At the mouth of the Orwell a single value of 2.59 $\mu\text{g l}^{-1}$ was recorded in September 1993 and a single value of 1.11 $\mu\text{g l}^{-1}$ was recorded Off the Deben, for comparison the mean has been derived with and without these values included

For some compounds as detection limits have improved earlier data often includes higher limits of detection. Values below detection are halved and included in the calculation of the mean. More values below detection are present from earlier dates which reflects improved analytical methods. These data are compared to environmental quality standards (EQS) to provide an indication of the potential for biological effects.

The Water Framework Directive requires that Member States identify Specific Pollutants and set standards for them. Specific Pollutants are toxic substances that are discharged in significant quantities into the water environment. Previous work by the UKTAG has led to standards for 19 Specific Pollutants. For substances

classified as specific pollutants the EQS are referenced from UKTAG, 2013. Measured values for selected substances classified as Specific pollutants are shown in Table 13.

Table 13 - Mean water column concentration of arsenic, chromium, copper and zinc from Environment Agency surveys $\mu\text{g l}^{-1}$ 1989-2006. The years covered for specific determinands are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Sampling Location	Chemical mean concentration or MAC $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
	Arsenic dissolved AA EQS 25	1991-1999
Mouth of Orwell	1.13	<1.0 -1.5 (n=7)
Off Orwell	1.14	<1.0 -2.4 (n=12)
Off Deben	0.65	<1.0 -1.4 (n=6)
Mouth Deben	NA*	NA
Off Alde/Ore	0.95	<1.0 -1.3 (n=13)
Off Aldeburgh	0.97	<1.0 -1.4 (n=14)
Off Dunwich	1.03	<1.0 -1.3 (n=14)
Off Kessingland	1.04	<1.0 -2.2 (n=18)
	Chromium VI dissolved AA EQS 0.6	1989-2006
Mouth of Orwell	0.68	<1.5 -13 (n=116)
Off Orwell	-	<1.5 (n=13)
Off Deben	<1.5	<1.5 (n=6)
Mouth Deben	NA	NA
Off Alde/Ore	1.57	<1.5 – 4.9 (n=20)
Off Aldeburgh	0.87 ¹	<1.5 – 2.4 (n=13)
Off Dunwich	1.09 ¹	<1.5 - 3.2 (n=14)
Off Kessingland	0.86 ¹	<1.5 – 2.1 (n=22)
	Copper dissolved AA EQS 3.76 ²	1989-2005
Mouth of Orwell	3.32	0.63 -4.88 (n=128)
Off Orwell	1.41	<0.25 -2.2 (n=12)
Off Deben	1.27	0.76 -1.5 (n=6)
Mouth Deben	NA	NA
Off Alde/Ore	2.03	0.87-1.08 (n=20)
Off Aldeburgh	1.12	<1-1.4 (n=15)
Off Dunwich	1.51	<0.25-7.47 (n=14)
Off Kessingland	1.08	0.84-1.45 (n=4)
	Zinc dissolved AA EQS 6.8 ³	1989-2006
Mouth of Orwell	15.8	<1-131 (n=127)
Off Orwell	5.3	<4-11.4 (n=11)
Off Deben	2.84	<4-4.98 (n=6)
Mouth Deben	NA	NA
Off Alde/Ore	7.31	<4-33 (n=14)
Off Aldeburgh	4.28	<4-9.8 (n=14)
Off Dunwich	7.19	<4-26.6 (n=16)
Off Kessingland	5.25	<4-14.6 (n=4)

NA – not analysed; ¹These means are based on relatively few measured values with the majority of values below detection ²The copper EQS includes a modification of the standard when dissolved organic carbon > 1mg l⁻¹. ³Zinc dissolved plus Ambient Background Concentration (ug/l) - Ambient Background Concentration (ABC) is an estimate of background levels of zinc based on a low percentile of monitoring data. For zinc in saltwater, an ABC of 1.1 $\mu\text{g/l}$ is recommended. ABC is the environmental concentration expected where no (or only minor) anthropogenic inputs are present.

All of the metals data relates to samples collected between 1989 and 2006 and there is no clear trend in concentrations measured and values below detection are interspersed with high values. For the concentrations of metals in seawater from various sites within the Suffolk Waterbody zinc exceeded its EQS at the mouth of the Orwell and Off the Alde/Ore although high values were also measured in samples Off Dunwich. Chromium concentrations were also high at the mouth of the Orwell and in a few samples at other sites. For other determinands for sample points outside the waterbody cadmium exceeds its EQS value Off

the River Deben. 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 numbers 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 would need to be taken account of in assessing the EQS.

Zinc is a naturally occurring substance and is ubiquitous in aquatic environments where it tends to occur at higher concentrations than most metals. Therefore to best assess compliance with the environmental quality standard (EQS) we need to take account of ambient background concentrations (ABCs); the EQS applies only to the additional contribution over and above the ambient background level (i.e. the value at which toxic effects occur, ignoring contributions from background concentrations).

Saltwater ABCs have been derived by assessing around 43000 samples and a low percentile has been used to exclude significant anthropogenic influences. There are limited variations in ABCs around most of the UK coast, and hence a national ABC value of $1.1 \mu\text{g l}^{-1}$ dissolved zinc is recommended for all coastal and estuarine waters.

From 1990 - 2007 there is evidence of an overall decline in the concentration of a number of metals in riverine and direct discharges to the marine environment (Charting Progress 2, 2010). However data collected in 2007 for metals concentrations under the EU Shellfish Waters Directive were used for comparison against the relevant standards for dissolved metals in water. Copper is the trace metal with the greatest number of results above the EQS (6%) in this survey and zinc the next highest at 1.5% above EQS (Charting Progress 2, 2010). Both these metals have a range of input sources from the urban environment which mean that their concentration in surface waters may not show any significant decline over the next decade.

Within the Suffolk Waterbody data for designated sampling points was available for measured values of ammonia, unionised ammonia and dissolved inorganic nitrogen and these are shown in Tables 14 - 18. The chlorophyll concentration measured from the some of the same sampling points is shown in Tables 19 and 20.

On account of its greater toxicity unionised ammonia (NH_3) has a specific EQS value set ($21 \mu\text{g l}^{-1} \text{NH}_3\text{-N}$). In some cases direct measures of unionised ammonia have been made but the percentage of NH_3 may also be calculated from the ammonium ion (NH_4^+) concentration based on knowledge of seawater pH, salinity and temperature. Overall the mean NH_4^+ concentrations measured are similar at all of the sampling sites and are relatively low (Table 14). For example the EQS value of $21 \mu\text{g l}^{-1} \text{NH}_3\text{-N}$. (at pH 8, 32 ppt salinity at 20°C) corresponds to a total ammonia concentration of c. $670 \mu\text{g l}^{-1} \text{NH}_4\text{-N}$. The mean unionised ammonia concentration is relatively high at several sites and exceeds the EQS most notably just off Lowestoft (Table 16).

For dissolved inorganic nitrogen (DIN) the SPM levels in the Suffolk Waterbody should be taken account of in deriving a reference 99 percentile value indicative of status (see Table 8). SPM in the vicinity of Sizewell can be considered to be around 50mg l^{-1} this is a conservative value based on monitoring data in the vicinity of Sizewell in 2010 (Beems TR 189). Based on an SPM of 50mg l^{-1} the 99 percentile DIN should be at or below $63 \mu\text{mol}$ for Good/Moderate status. Reference to data on the Mean dissolved inorganic nitrogen concentrations (Table 17) indicates that all sites sampled within the waterbody meet the threshold value which means they would be considered at Good/Moderate status.

For nearshore waters, where the level of production may be expected to be higher, $15 \mu\text{g l}^{-1}$ chlorophyll is adopted as the reference value (implying a background value of $10 \mu\text{g l}^{-1}$, Devlin et al., 2007). The 90th percentile chlorophyll concentrations during the growing season (March to September) should remain below thresholds set for the high/good ($10 \mu\text{g l}^{-1}$) and good/moderate boundaries ($15 \mu\text{g l}^{-1}$) for type specific conditions (UKTAG, 2014). Reference to the data in Table 19 and 20 indicates that values at most sites meet the high/good threshold and all the good/moderate.

Table 14 - Mean water column concentration of ammonia filtered as N from Environment Agency surveys. The years covered are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Sampling Location	Mean concentration $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
	Ammonia total	1992 - 2007
Mouth of Orwell	-	-
Off Orwell	32	<2.0 – 110 (n=77)*
Off Deben	20	<5.0 – 144 (n=188)
Mouth Deben	23	0.0 – 180 (n=218)**
Off Alde/Ore	24	<2.0 – 111 (n=79)
Off Aldeburgh	27	<7.0 – 146 (n=78)
Off Dunwich	31	<7.0 – 141 (n=79)***
Off Kessingland	35	<5.0 – 156 (n=82)
Off Yare	-	-

*1992 -2005; **2002-Q12014; ***1992-2004

Table 15 - Mean water column concentration of ammonia un-ionised as N from Environment Agency surveys. The years covered are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Approximate Sampling Location	Mean concentration $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
	Ammonia un-ionised AA EQS 21	1997-2014*
Mouth of Orwell	2.6	<1.0 – 11.2 (n=47)
Off Orwell	-	-
Off Deben	-	-
Mouth Deben	1.3	<1.0 – 5.0 (n=48)
Off Alde/Ore	-	-
Off Aldeburgh	-	-
Off Dunwich	-	-
Off Kessingland	-	-
Off Yare	-	-

* Samples taken up to beginning of April 2014

Table 16 - Mean water column concentration of ammonia un-ionised as N (filtered) from Environment Agency surveys for the Suffolk waterbody sites the years covered are shown at the second row right-hand column.

Approximate Sampling Location	Approximate location	Mean concentration $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
		Ammonia un-ionised AA EQS 21	1991 - 2013
North Sea NO. 51	Off Deben	9.0	<7.0 - 25.2 (n=5)*
North Sea NO. 46	Just above Alde/Orr	6.8	<7.0 - 20.0 (n=5)*
North Sea NO. 43B	Just below Aldeburgh	17.5	<7.0 - 108 (n=54)
North Sea NO. 34	Just below Lowestoft	25.9	<1.0 - 114 (n=20)
North Sea NO. 33	Just Off Lowestoft	26.9	<5.0 – 122 (n=60)

* 2012-2013

Table 17 - Mean (and 99 percentile) water column concentration of Winter (November – February) dissolved inorganic nitrogen (DIN: nitrate, nitrite and ammonia) $\mu\text{g l}^{-1}$ as N (and μmol) from Environment Agency surveys. The years covered are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Approximate Sampling Location	Mean (and 99 percentile) concentration $\mu\text{g l}^{-1}$	Mean (and 99 percentile) concentration μmol	Concentration range $\mu\text{g l}^{-1}$
			1992-2007
Mouth of Orwell	-		-
Off Orwell	449 (833)	32 (60)	190 - 865 (n=20)
Off Deben	373 (578)	27 (41)	183 - 582 (n=23)
Mouth Deben	-	-	-
Off Alde/Ore	335 (462)	24 (33)	132 - 463 (n=23)
Off Aldeburgh	305 (529)	22 (38)	<9.0 - 545 (n=21)
Off Dunwich	333 (562)	24 (40)	118 - 565 (n=23)
Off Kessingland	-	-	-
Off Yare	415 (781)	30 (56)	102 - 799 (n=20)

Table 18 - Mean (and 99 percentile) water column concentration of Winter (November – February) dissolved inorganic nitrogen (DIN: nitrate, nitrite and ammonia) from Environment Agency surveys $\mu\text{g l}^{-1}$ as N (and μmol) for the Suffolk waterbody sites the years covered are shown at the second row right-hand column.

Approximate Sampling Location	Mean (and 99 percentile) concentration $\mu\text{g l}^{-1}$	Mean (and 99 percentile) concentration μmol	Concentration range $\mu\text{g l}^{-1}$
North Sea			1992 - 2013
NO. 51 Off Deben	277 (351)	20 (25)	185 – 352 (n=4)*
NO. 46 Just above Alde/Orr	253 (348)	18 (25)	211 – 351 (n=4)*
NO. 43B Just below Aldeburgh	272 (486)	19 (35)	121 – 502 (n=20)
NO. 34 Just below Lowestoft	394 (585)	28 (42)	88 – 586 (n=9)
NO. 33 Just Off Lowestoft	397 (699)	28 (50)	97 – 723 (n=15)

*2012 – 2013

Table 19 - Mean water column concentration chlorophyll $\mu\text{g l}^{-1}$ March – September from Environment Agency surveys. The years covered are shown at the second row right-hand column. Sample locations shown in bold are within the Suffolk waterbody

Approximate Sampling Location	Mean (90 percentile) concentration $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
	Chlorophyll	1992-2005
Mouth of Orwell	4.8 (8.0)	0.4 – 9.5 (n=22)
Off Orwell	4.0 (6.7)	<1.0 -11.1 (n=63)*
Off Deben	4.3 (7.8)	<1.0-15.4 (n=134)
Mouth Deben	5.4 (9.6)	<1.4 – 18.6 (n=108)**
Off Alde/Ore	-	-
Off Aldeburgh	-	-
Off Dunwich	3.8 (6.5)	0.3 – 11.5 (n=46)
Off Kessingland	-	-
Off Yare	5.1 (9.1)	<0.3 – 33.0 (n=58)

*1992-1994; ** 2002-2013;

Table 20 - Mean water column concentration of chlorophyll March – September from Environment Agency surveys $\mu\text{g l}^{-1}$ for the Suffolk waterbody sites the years covered are shown at the second row right-hand column.

Approximate Sampling Location	Approximate location	Mean (90 percentile) concentration $\mu\text{g l}^{-1}$	Concentration range $\mu\text{g l}^{-1}$
		Chlorophyll	1992 - 2013
North Sea NO. 51	Off Deben	7.3 (9.2)	<1.5 – 45.0 (n=45)*
North Sea NO. 46	Just above Alde/Orr	4.3 (7.7)	<1.25 – 9.9 (n=18)*
North Sea NO. 43B	Just below Aldeburgh	4.3 (8.0)	<0.8 – 15.2 (n=42)
North Sea NO. 34	Just below Lowestoft	6.4 (13.2)	1.5 – 18.5 (n=24)
North Sea NO. 33	Just Off Lowestoft	5.3 (10.1)	<0.3 – 19.7 (n=56)

*2012 – 2013

4 Discussion and Conclusions

With a particular focus on data relevant to the Suffolk coastal waterbody within which the Sizewell nuclear power station is situated this review has gathered information on various chemical and physical parameters that can influence water quality.

The concentration of contaminants is relatively low by comparison to levels present in estuaries and coastal waters associated with more industrialised areas although port activities on the Orwell have probably contributed to increased metal inputs to sediments.

Shipping and boating activity have led to a legacy of contamination from antifouling compounds particularly tributyltins in sediments and currently to the input of copper and zinc which are again localised to areas of highest activity. It is likely that this contribution is responsible for the elevated concentration close to and in a few cases for zinc (mouth of the Orwell and off the Alde/Orr) exceeding respective EQS for these metals in seawater samples collected from a range of sites on the Suffolk coast from 1989-2006. The cadmium EQS was exceeded for the Mouth of the Orwell and this is potentially linked to sewage works inputs. Upgrades to a number of sewage works that discharge to the Orwell were however due to be completed by 2005 (Stour and Orwell Estuaries management plan, 2004) so improvement in this parameter would be expected.

Other compounds of relevance to power station operation are generally not measured routinely and therefore data on levels within the area are limited to historic studies on power station discharges. These studies indicate relatively low and localised inputs of chlorine produced oxidants and bromoform not exceeding current or indicative standards beyond 1-2 kilometres of the point of discharge.

The thermal input from the power station cooling water discharge is one of the more significant potential affects upon the marine environment off Sizewell. The data for temperature for four sites across the Suffolk Waterbody indicate that there is likely to be sufficient margin between the derived 98 percentile baseline temperature for the waterbody (19.4°C) to not result in major areas failing to meet the temperature boundary for Good/Moderate status (20 – 23 °C). The boundary value for the Thames SPA for the Habitats Directive criteria (28°C as a 98 percentile) is also likely to be met with only small areas of exceedance likely within the immediate mixing zone.

This location is relatively free of major industrial operations and emissions but agriculture does have a significant influence on water quality and in particular has contributed to the elevation of nutrient concentrations in rivers and estuaries in the region.

During power station construction and operation there will be increased numbers of people on site with associated production of wastewater that may require discharge to the marine environment. Wastewater discharge will make a contribution to nutrient concentrations within the local marine environment and this will need to be assessed against the current status of the Suffolk waterbody.

Assessing the status of coastal waters with respect to specific discharges of nutrients is mandated under the Urban Waste Water Treatment Directive (Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment) which now forms a basic measure under the Water Framework Directive. The Water Framework Directive requires an assessment of Good Ecological Status for those 'biological quality elements' that are sensitive to particular pressures. In the case of the coastal water off Sizewell, the relevant biological quality elements are phytoplankton and macrophytes (including macroalgae) for which the EA has developed classification tools. In addition, nutrients are a 'supporting element' for the biology and nutrient standards have been set for waters moderated for the inherent turbidity of the water (different standards are set for turbid waters that are less likely to respond to the nutrient pressure). These coastal waters are also subject to assessment under the OSPAR Common Procedure which will be the primary method for assessment under the Marine Strategy Framework Directive.

The coastal waters of East Anglia are enriched by nutrients derived from a number of sources including Urban Waste Water discharges but predominantly from riverine inputs which include agricultural sources. While the wider marine waters of the southern North Sea have been assessed as non-problem areas (OSPAR) for eutrophication there are coastal water bodies (within the 1 nm of WFD) that are assessed as

moderate status resulting from the level of nutrients. The Suffolk Coastal water body is Moderate status for DIN and High Status for the biological quality element phytoplankton.

In summary the natural background temperatures at Sizewell meet the required standards under the Water Framework Directive for Good status and also the requirements under the Habitats Directive for the outer Thames SPA with sufficient temperature margin relative to additional predicted inputs from power station development to have a low likelihood of resulting in large areas of exceedance. Contaminant inputs to the waterbody are limited and general trends in the Southern North Sea indicate declining inputs of most metals and a number of organic chemicals. Against this background the main chemical inputs from the new build power station are chlorine produced oxidants and bromoform. Data for these substances associated to the Sizewell B discharge indicate relatively limited areas of exceedance of the relevant EQS or surrogate concentrations. During construction and operation the potential wastewater inputs from the workforce and from permanent staffing of the site will need to be assessed in terms of nutrient inputs but the current background levels have limited impacts because of light limitation on phytoplankton growth due to the higher suspended particulate matter (SPM) levels present in the waterbody. The impact of additional nutrient inputs will need to be assessed against the SPM background and this may limit the extent of any effects.

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6 Appendix

Table 21 - Monthly mean seawater temperature distribution 1966 – 2013 for four sites in the Suffolk coastal waterbody

Monthly Mean Sea Temperature for LOWESTOFT at 52 27 N, 1 45 E												
1966 - 2013	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
mean	4.6	4.5	5.9	8.1	11.8	15.0	17.6	18.2	16.3	13.2	9.1	6.1
count	28	29	29	28	28	28	27	26	27	26	25	25
sd	1.4	1.5	1.5	1.1	0.9	0.9	1.1	1.0	0.8	0.9	0.9	1.0
98%	7.78	7.79	9.45	10.80	13.69	16.80	19.73	20.15	17.74	14.46	11.24	7.40
2%	2.63	2.07	3.76	5.85	10.50	13.60	15.86	16.75	14.86	11.15	7.90	3.77
2010 - 2011												
mean	3.8	4.1	5.6	9.7	12.7	15.9	18.3	17.8	16.4	13.9	10.4	5.4
count	2	2	2	2	2	2	2	2	2	2	2	2
sd	0.4	1.5	0.8	1.3	1.3	0.5	1.1	0.0	0.2	0.8	1.7	2.6
98%	4.02	5.09	6.16	10.56	13.64	16.20	18.98	17.78	16.50	14.40	11.59	7.21
2%	3.5	3.1	5.1	8.8	11.8	15.6	17.5	17.7	16.2	13.3	9.3	3.6
Monthly Mean Sea Temperature for SOUTHWOLD at 52 19 N, 1 41 E												
1966 - 2013	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
mean	5.0	4.8	5.8	8.0	11.5	15.0	17.3	18.1	16.3	13.3	9.7	6.5
count	48	48	48	48	48	48	48	48	48	48	48	48
sd	1.2	1.3	1.3	1.2	1.0	0.9	1.1	1.1	0.9	0.9	1.0	1.0
98%	7.01	7.43	7.90	10.11	13.36	16.53	19.55	20.41	18.02	15.33	11.41	8.32
2%	3.05	2.16	3.59	5.89	9.75	13.29	15.09	16.29	14.47	11.45	7.80	4.48
2009 - 2013												
mean	4.7	4.1	5.8	9.0	12.0	15.4	18.1	18.7	16.5	13.6	10.2	6.4
count	30	30	30	30	30	30	30	30	30	30	30	30
sd	1.1	0.8	1.2	1.4	1.2	1.2	1.1	0.8	0.1	0.7	1.0	1.4
98%	6.12	5.26	7.06	10.10	13.17	16.47	19.70	19.82	16.68	14.46	11.59	7.65

2%	3.46	3.07	4.01	6.81	10.46	13.58	17.25	17.66	16.31	12.90	9.24	4.26
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Monthly Mean Sea Temperature for SIZEWELL PS at 52 13 N, 1 38 E

1967 - 2013	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
mean	5.9	5.3	6.1	8.5	11.6	15.1	18.0	19.0	17.7	14.8	11.3	7.8
count	46	46	46	46	46	46	46	46	46	46	46	45
sd	1.4	1.5	1.4	1.4	1.4	1.4	1.4	1.5	1.5	1.5	1.7	1.5
98%	8.23	7.81	8.32	10.94	14.10	17.60	20.91	21.96	20.70	18.43	14.63	10.87
2%	3.19	2.80	2.99	5.96	9.75	12.81	15.89	16.77	15.58	12.20	8.39	4.96

2009 - 2013

mean	5.3	4.5	5.9	8.7	12.4	15.3	18.2	19.1	17.5	14.4	11.2	7.2
count	5	5	5	5	5	5	5	5	5	5	5	5
sd	0.9	0.5	1.2	1.3	0.8	0.9	0.5	0.6	0.5	0.6	1.0	1.3
98%	6.59	5.27	7.13	10.08	13.35	16.20	18.84	19.77	18.15	15.22	12.61	8.31
2%	4.53	3.93	4.28	6.79	11.51	14.15	17.76	18.35	16.89	13.78	10.43	5.13

Monthly Mean Sea Temperature for Felixstowe Rotterdam_Pos 2 (52.033°N; 1.666°E)

1971 - 2010	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
mean	7.3	6.4	6.5	8.3	11.0	14.5	17.2	18.5	17.7	15.4	12.1	9.6
count	36	39	38	38	38	36	38	37	35	39	36	37
sd	1.5	1.5	1.4	1.0	1.2	1.5	1.2	1.0	1.0	1.1	1.1	1.5
98%	9.88	9.14	8.68	10.18	13.19	17.73	19.08	20.25	19.03	17.22	14.00	12.14
2%	3.90	3.28	3.41	6.62	9.15	11.61	14.62	16.77	15.64	13.44	9.97	6.83

2005 - 2010

mean	7.2	6.4	6.5	8.1	11.3	14.6	17.7	18.4	17.8	16.0	12.8	9.4
count	6	6	6	6	6	6	6	6	6	6	6	6
sd	0.5	0.9	1.4	1.0	0.9	0.7	0.6	0.3	0.9	1.1	1.2	1.5
98%	7.77	7.75	8.47	9.09	12.18	15.35	18.54	18.88	18.80	17.25	14.60	11.38
2%	6.50	5.53	4.88	6.74	10.15	13.48	16.85	17.94	16.56	14.73	11.61	7.25