Characterisation of European Marine Sites

Essex Estuaries

European Marine Site

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Cover photograph: The River Blackwater at low tide, from the Promenade, Maldon, Essex.

Site Characterisation of European Marine Sites

Essex Estuaries European Marine Site

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2006

A study carried out on behalf of English Nature

By

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It should be noted that the opinions expressed in this report are largely those of the authors and do not necessarily reflect the views of EN.

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1. EXECUTIVE SUMMARY

This report provides an overview of water and sediment quality within the Essex Estuaries European Marine Site (EMS) and examines evidence for their influence on biological condition. Site characterisation has been accomplished by review of published literature and unpublished reports, together with interrogation of summary data sets for tidal waters provided by EA. Key findings are as follows:

- Some of the more extensive water quality issues relate to hypernutrification[1], notably in the Colne and Blackwater and perhaps to a lesser extent, in the Crouch & Roach. Associated influences on DO, turbidity etc. are difficult to establish though there is evidence of macroalgal proliferation in the Blackwater.
- Concentrations and risks from water-borne, toxic contaminants appear to be mostly low, though few sites within the EMS have been monitored comprehensively. (Note: IPPC information, discharge data or riverine loadings were not available and therefore it is not possible to comment on sources).
- Metals are unlikely to pose an acute threat to the site but some, such as Cu and Zn, could contribute to sub-lethal stress, in combination with other contaminants. Earlier evidence points to an important component from antifouling/ anticorrosion uses, particularly for Cu. STW could also contribute to the Zn loadings. Indications are that the risk of EQS excedence has declined during the last decade. Sediment levels are seldom above probable effects levels for any metal but are sometimes above threshold levels. Contamination in the area tends to be relatively low-level and widespread (reflected also in body burdens), with distributions influenced by variety of processes and inputs, both present-day and historical. For some contaminants, this probably includes a significant contribution from the Thames.
- Investigations into the practice of managed retreat indicate that metals such as Fe, Mn and Cd may be released from sediments, whilst Pb, Cr and Cu may increase, following redox and salinity changes that accompany tidal inundation.
- TBT threats are receding, generally, but residual contamination remains a concern near marinas, particularly in sediments. Also there are some anomalous TBT increases in the Blackwater. Further ecotoxicological investigations are needed.
- There is a dearth of information on PAHs in the EMS. The little that is available indicates that PAH enrichment in sediment is consistent with an anthropogenic origin. Though not as high as those recorded in highly industrialised estuaries, concentrations may represent a chronic threat for benthic organisms (>TEL), notably in the Blackwater Estuary.
- Risks from pesticides, herbicides, PCBs, and VOCs appear low, though the data are not sufficient for a rigorous assessment. Pesticides and herbicides in sediments generally indicate low level contamination, with occasional enrichment associated with both urban and agricultural inputs. Episodic pulses of high concentrations of pesticides and herbicides can occur following application to agricultural areas adjacent to tidal waters. This may have consequences for local flora and fauna, though detailed biological effects studies have not been carried out. In the past, sludge dumping and sources in the Thames estuary may have influenced levels of PCBs and DDT in sediment (and perhaps PCBs in biota) at coastal sites (Dengie).
- Bradwell nuclear power station has been the main consented source of radionuclides to the EMS for 40 years, but ceased production in 2002. Monitoring

[1] See Glossary

of samples from the Blackwater Estuary, in 2003, indicated that, as in previous years the radiological impact from authorized disposals was relatively low.

- Little information exists on endocrine disruption caused by oestrogens or xenoestrogens, except for flounder in the Crouch Estuary, where male fish display relatively low levels of feminization (some vitellogenesis but no ovotestis). There are no data for the Colne and Blackwater, or for invertebrates.
- Recent bioaccumulation data are sparse and are largely restricted to mussel sampling at the West Mersea NMMP Shellfish site. Bioaccumulation is not considered a threat here, and the patchy historic data from other sites tend to indicate the same, though bioavailability of most contaminants tends to be above background. Targeted updates on metals, pesticides, PCBs and PAHs, using appropriate monitoring organisms, is suggested, alongside effects studies.
- Invertebrate diversity in Essex Estuaries is considered good, although quantitative biodiversity indices have not been applied widely in the EMS. There are few examples of unambiguous links with environmental quality. One exception is decline and recovery of invertebrates in the Crouch Estuary from the effects of TBT pollution. There is no evidence to link the decline in saltmarsh extent (or condition) to water or sediment quality. Offshore, benthic communities tend to be impoverished due, largely, to the nature of the substrate. Despite historical dumping of sewage sludge in the Barrow and Black Deeps, impact on biota was minor. Because of the dispersive nature of the site, and net transport in a NE direction, significant community-level changes within the EMS, particularly estuaries, seem unlikely.
- Since the submission of bird population baselines for the original SPA designations, internationally important levels of several of the major qualifying species have declined in two of the component estuaries of the EMS. A number of nationally important qualifying species and non-qualifying species also appear to have decreased over the period. It is uncertain whether these trends can be attributed solely to water quality although reduced freshwater inputs and changes to waste water treatment are cited by BTO as factors directly related to EA consents.
- Biological response information, linked to WQ data, is sparse and disjointed for sites within the EMS. Bioassay and biomarker-type studies could provide important contextual information to inform the EMS assessment. Links with the broader biological consequences should also be investigated in the long-term by traditional benthic survey. It is recommended that a programme of harmonised chemical and biological effects monitoring be instigated at the earliest opportunity, focusing on priority contaminants and key biota of the EMS, particularly those which are of major dietary significance for dependent bird populations.
- Review of the available chemical data for the area rarely raises concerns over acute toxicity within the EMS, though the possibility that combined pressures might impair performance of sensitive species and communities cannot be ruled out. At present the (limited) evidence probably does not justify expensive remedial action on sources. However, there is sufficient uncertainty to justify a more targeted and detailed programme of research to measure actual biological impacts at a variety of levels (community, population, biochemical, bioaccumulation in suitable indicators). If such studies indicate deleterious effects, which can be attributed to known causes, then the case for remedial action against key sources would be placed on a stronger, scientifically-sound basis.

CONTENTS

2. INTRODUCTION

This review considers the characteristics of the marine areas of ESSEX ESTUARIES EUROPEAN MARINE SITE (EMS) and how the status of the site is influenced by existing permissions and activities, either alone or in combination. Also considered are possible impacts from other factors such as unconsented activities, diffuse sources and natural processes. This includes activities and consents outside the site itself. The purpose is thus to collate and interpret information relevant to the assessment of water quality impacts and risks to the marine component of the EMS, to ensure that EA and EN are fully informed when making decisions in relation to the scope of appropriate assessment. It should be noted that although efforts were made to gather all available relevant data, the authors are aware that the report is not comprehensive but draws on data sets and publications that were readily available at the time. The opinions expressed are made on the basis of available information (up to 2005). We have emphasised areas where information is lacking, or where we see an opportunity to improve implementation and monitoring to comply with the requirements of the Habitats Directive and to provide a better means of establishing the status of the site.

To achieve this goal, specific objectives were:

- To prepare comprehensive reference lists of previous investigations and existing datasets, including published research and unpublished reports, relevant to an assessment of the effects of water quality on the marine sites and interest features identified.
- To review the existing information, pinpoint key studies, collate and summarize their findings.
- To identify site-specific models predicting pollutant concentrations and their links to impact.
- To prepare a summary of existing datasets (spatial and temporal) on water and sediment quality (e.g. determinands and summary statistics where available).
- To integrate and evaluate biological information, with specific reference to water/sediment quality.
- To conclude if there is any evidence that existing water (or sediment) quality is causing impact and highlight limitations of the available data.
- To identify and recommend further research which will address the limitations of current information and establish cause/effect relationships.

English Nature has provided advice on the European marine site, given under Regulation 33(2) of the Conservation Regulations 1994 (English Nature, 2000). A summary of the interest (or qualifying) features, and conservation objectives, for the site is given in Annex 1. The table below is a summary of the operations which, in the opinion of English Nature, may cause disturbance or deterioration to these interest features. In terms of the current project's emphasis on consents, we will focus on the vulnerability to toxic contamination and non-toxic contamination unless any of the other threats are seen as highly relevant.

Table 1. Summary of the operations, which, in ENs opinion may cause disturbance or deterioration to key interest features of Essex Estuaries European Marine Site. Toxic and non-toxic contamination are the principal threats considered in the current project. (Table adapted from English Nature, 2000)

Standard list of operations which may cause deterioration or disturbance	Pioneer saltmarsh	Cordgrass swards	Atlantic salt meadows	Mediterranean saltmarsh scrub	Estuaries	Intertidal mudflats and sandflats	Internationally important Annex 1 birds	Internationally important assemblages of waterfowl
Physical loss								
Removal (e.g. harvesting, coastal development)	✔	✔	✔	✔	✔	✔	✔	✔
Smothering (e.g. artificial structures, disposal of dredge spoil)					✔	✔	✔	✔
Physical damage								
Siltation (e.g. run-off, channel dredging, outfalls)					✔	✔	✔	✔
Abrasion (e.g. boating, anchoring, trampling)	✔	✔	✔		✔	✔	✔	✔
Selective extraction (e.g. aggregate dredging,)								
Non-physical disturbance							✔	✔
Noise (e.g. boat activity)								
Visual presence(e.g. recreational activity)							✔	✔
Toxic contamination								
Introduction of synthetic compounds (e.g. TBT, PCB's,)	✔	✔	✔	✔	✔	✔	✔	✔
Introduction of non-synthetic compounds (e.g. heavy metals, hydrocarbons)	✔	✔	✔	✔	✔	✔	✔	✔
Non-toxic contamination								
Changes in nutrient loading (e.g. agricultural run-off, outfalls)	✔	✔	✔		✔	✔	✔	✔
Changes in organic loading (e.g. mariculture, outfalls)	✔	✔	✔	✔	✔	✔	✔	✔
Changes in thermal regime (e.g. power station)								
Changes in turbidity (e.g. run-off, dredging)						✔	✔	✔
Changes in salinity (e.g. water abstraction, outfalls)								
Biological disturbance								
Introduction of microbial pathogens					✔			
Introduction of non-native species and translocation					✔			
Selective extraction of species (e.g. bait digging, wildfowl, commercial and recreational fishing)	✔				✔		✔	✔

The key questions, which we have tried to incorporate into our considerations of site characteristics are in line with the Agency's Management System i.e.

- Is there a potential hazard mechanism by which the consent/activity could affect the interest features of the site (directly or indirectly)?
- Is there a probability that the consent/activity could affect the interest features of the site (directly or indirectly)?
- Is the scale and magnitude of any effect likely to be significant[2]?

Clearly if the answer to all three questions is positive a more detailed assessment is likely to be required.

We have also kept in mind similar criteria which EA/EN may need to apply during the review process as outlined in their *Guidance for the Review of Environment Agency Permissions: Determining Relevant Permissions and 'significant effect'* (March 1999):

A. The designated feature is in favourable condition and there is no evidence to suggest existing consents are currently having a significant effect.

B. The designated feature is in favourable condition but there is concern that a water quality problem caused by a consented discharge may be threatening that condition and/or causing a decline in it.

C. The designated feature is in unfavourable condition, but this can be attributed to a factor unrelated to water quality, e.g. vegetation management, and there is no evidence to suggest relevant consents are currently having a 'significant effect'.

D. The designated feature is in unfavourable condition and poor water quality may be or is likely to be responsible.

[2] Examples of 'significant' effects criteria:
- Causing change to coherence of the site
- Causing reduction in area of the habitat
- Causing change to the physical quality and hydrology
- Altering community structure (species composition)
- Causing ongoing disturbance to qualifying species or habitats
- Causing damage to size, characteristics or reproductive ability of qualifying species (or species on which they depend)
- Altering exposure to other impacts
- Causing a reduction in resilience against other anthropogenic or natural changes
- Changing stability of the site/feature
- Affecting a conservation objective

Structure of the report

In **Section 3** we identify sources of information and the generalised approach used in assessing raw data to come to an opinion on water and sediment quality. There is also a critical evaluation of the constraints.

Section 4 of this report describes the main physical, chemical and biological features which shape the character of the site and summarises some of the perceived threats to its favourable status.

Studies which describe the biology and ecology of benthic communities within the site, many of which support the bird populations for which the site is designated, are discussed briefly in **section 5.** This section also provides a resume of trends in bird populations for SPAs, based on British Trust for Ornithology reports.

Section 6 discusses published information on toxic contamination (metals, TBT, petrochemicals, pesticides, PCBs, PAHs, volatile organics) and non-toxic contamination (nutrients, turbidity, dissolved oxygen). Section 6 also presents summary statistics of previously unpublished water quality data, in relation to Environmental Quality Standards and guidelines (listed in Annexes 3-5). This draws on available information provided by the Environment Agency (extracted from WIMS). A synthesis of available information on sediment quality is also given in this section**.**

Section 7 focuses on evidence for biological impacts within the EMS, including reviews of bioaccumulation and biological effects (from the cellular to community level).

A brief description of modelling exercises of direct relevance to the environmental quality status of the site is provided in **section 8**.

Concluding remarks (**section 9**) include a summary of evidence for impact in the Essex Estuaries European Marine Site, together with recommendations for future monitoring and research requirements.

A bibliography containing references used to compile this report is included in **section 10.** A number of annexes are also appended.

3. REFERENCE LISTS AND SOURCES OF INFORMATION

A full list of publications in the open literature has been assembled using the Aquatic Sciences and Fisheries Abstracts (ASFA) and Web of Science information retrieval systems. The National Marine Biological Library in-house data base LIB-WEB has provided additional listings (see accompanying electronic database).

Unpublished reports and data-bases include: Environment Agency, Joint Nature Conservancy Council (JNCC) Coastal Directories Reports, Centre for Environment, Fisheries and Aquaculture Science (CEFAS);

Information, monitoring data and summary statistics provided by the Environment Agency between 2000 and 2005, extracted from WIMS. This does not include recent compliance data and other forms of self-monitoring for Integrated Pollution Control sites, which was not available.

Comparative data for other UK estuaries, including south-west European marine sites (e.g. Exe, Severn, Poole, Fal, Plymouth Sound and Estuaries, Chesil and the Fleet) have been used to draw comparisons with the Essex Estuaries European Marine Site, wherever possible.

Methods: Contaminants and Quality Standards

Water Quality Data

In section 6 we examine published and unpublished information on contaminant trends, together with EA data (from WIMS) on determinands which may influence the Essex Estuaries European Marine Site (EMS). Summary statistics for tidal water quality have been drawn up by the Agency (based on monitoring between 2000 and 2005, rather than annual averages). Raw data has been analysed for some priority contaminants in an attempt to establish further evidence as to whether or not water quality is likely to cause impact, though this was outside the project specification.

It should be noted that much of the data from EA monitoring surveys may be for the purpose of compliance monitoring only. Detection limits are often set with that specific intention in mind, such that the data may be of limited value for environmental behaviour studies. Nevertheless values reported as below detection limits have usually been included in summary statistics since it allows at least a crude assessment of water quality issues. With these caveats in mind the majority of List I and List II (Dangerous Substances) determinands have been screened in the tidal waters data-set, together with other water quality parameters such as nutrients, chlorophyll a and DO. The location of principal tidal water sampling sites is shown in figure 5.

In the absence of extensive site-specific biological effects information, comparisons of water-monitoring results with Environmental Quality Standards (EQS) are used in order to gain a first-order approximation of possible impact on biota. Thus, in the context of the current project, descriptions of ‘threat’ or ‘risk’ to the site from individual contaminants are scaled against the relevant EQS, assuming this to be an appropriate threshold for the protection of aquatic life.

For a number of reasons this is an uncertain supposition. The compliance limits for contaminants and other water quality parameters are themselves based on reviews of general toxicity data for aquatic life, coupled with a safety margin below the lowest reliable adverse effects concentration. The assumption is that below the EQS, adverse biological and ecological effects *are unlikely*. Above the EQS, effects *might be expected to occur* though this will depend on the magnitude and duration of the exposure. The application of EQS values involves uncertainties arising from limited toxicity data, differential responses between chronic and acute toxicity, inter-species variation in sensitivity, and modifying factors within each individual ecosystem (notably, the issue of synergy and additivity discussed below). Sensitivity may also vary between different levels of biological organisation; lower-order effects (molecules and cells) are likely to occur at lower levels of contamination, and in advance of, community and ecosystem-level response. Often this involves a high degree of precaution in setting standards and could give rise to an apparent mis-match between chemical data and measured biological responses, particularly at the level of biological diversity. Conversely, it is also possible that subtle effects may occur at concentrations below the EQS, giving rise to a failure to protect the system. Compliance/non-compliance patterns are therefore not necessarily synonymous with ecological implications: at present the latter can only be gauged by considering a wider array of ecosystem characteristics. EQS values are used here merely to help prioritize some of those sites and contaminants which merit closer investigation. They do not necessarily assure Favourable Condition.

Another drawback to the EQS approach is that it considers the toxicology of contaminants individually, assuming that each is acting independently of others. In reality, some of the more significant discharges contain a range of contaminants which, though they may individually pass the 'EQS test', may pose a greater threat to nature as a result of additive toxicity. The question of synergistic/ antagonistic interactions from outfalls should be a priority for future research.

Data for Harmonised Monitoring Points or the equivalent freshwater site immediately above tidal limits were requested (to characterise riverine inputs) but were not available within the timescale of the current project. Similarly, discharge data for contaminants were requested to gauge the importance of specific industrial and trade effluent point sources. However, as the Agency is in the process of evaluation under stage 3 of the ROC process[3], their recommendation was that further review was unnecessary. Sites of some of the more important outfalls, supplied by the EA are shown in figures 2 and 3 to put the current evaluation in context.

Sediment Quality Data

Sediments are not monitored routinely by the Agency as there are no requirements to do so in the context of managing outfalls. Nevertheless they represent important sinks

[3] All discharges listed in stage 2 ROC documentation are, by convention, considered to be of likely significance either alone or in combination

and sources for chemicals in estuaries and coastal areas and can play an important part in determining site condition. In general, levels of contamination in estuarine sediments decrease significantly towards offshore sites, partly due to distance from major inputs, and partly due to changing characteristics of the sediments. The progression from fine silts rich in binding sites in the upper estuaries, to coarser sediments offshore is usually accompanied by decreasing contaminant loading. Thus, distributions will be governed to a large extent by the hydrodynamic regime in the system and the sorting and redistribution of fines. Sieving and normalisation procedures[4] are required to compensate for such granulometric and geochemical effects, to allow meaningful comparison of contamination levels, particularly for metals (Langston *et al*., 1999; MPMMG, 1998). In the current report we have used our own metals data, for sediments sieved at 100µm, to summarise sediment quality in the EMS (partly for comparability with other reviews on SPAs and SACs). Metals discussed are Hg, Cd, As, Cr, Cu, Ni, Pb and Zn. These data are relatively old (mainly from the late 1980s, except for Foulness and Shoeburyness which were sampled between 1997 and 2001) and are therefore only intended as guidelines. Methodologies have been successfully validated in numerous intercalibration exercises (Langston *et al.,* 1994a). The MBA sediment metals data described here are for the <100µm fraction, without further normalisation (*see footnote for details*), and are in concentrated nitric acid digests. Trends and 'risks' are briefly compared with a limited number of published and unpublished data on sediment analyses.

In comparison with metals, the information on organic contaminants is very limited at sites within the EMS.

At present there are no statutory standards for sediments in the UK. Several guidelines on sediment quality are emerging, and CEFAS has cautiously recommended the Canadian/US effects-based approach (CCME ,1999; Long *et al*., 1995). Threshold Effects Levels (TELs - affecting the most sensitive species) and Probable Effect Levels (PELs - likely to affect a range of organisms) are derived from published toxicity data for a variety of substances in sediments (laboratory and field exposures). TELs are proposed as an Interim Sediment Quality Guideline (ISQG) value. Briefly, their application is used to inform a decision matrix which includes assessing in the context of other biological information, rather than 'stand alone' EQSs. Their application is discussed further in the Habitats Water Quality TAG document

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[4] The need to standardise/normalise sediment measurements: This stems from the fact that chemical composition varies according to the sediment type, irrespective of anthropogenic influence. Thus muds and silts naturally have higher metal loadings than coarse sands because of their larger surface area and more extensive oxyhdroxide and organic coatings (capable of sequestering other chemicals). There are various ways in which this granulometric variance can be overcome, including normalisation to geogenic elements such as Al and Li: this may be particularly useful when comparing sediments of totally different geological backgound. An alternative and more direct technique to minimise the influence of grain size in comparisons is to select particles of similar size – hence the use of particles <100µm for the examples shown on the following pages in the current exercise. A study of microwave-digested Irish Sea sediments conducted in our own laboratory has shown that, following sieving at this mesh size, further normalisation confers no significant additional advantage when comparing metal contaminant trends. Sieving fulfils a further function - to place emphasis on particles which are accepted by benthic organisms. Sieving at 100µm was the preferred option for comparisons made in this project. It is stressed that this is only one of the options for classifying sediments, others may be equally suitable; the point is that some adjustment has to be made for grain size otherwise comparisons are uncontrolled and of little value.

WQTAG078K (EA). As yet these guidelines have not been validated in the UK, though for many List I substances of the Dangerous Substances Directive a 'standstill' provision applies, whereby the concentration of the substance in sediments (and organisms) must not increase with time. Sediment quality is also important under the remit of the Habitats Directive (attainment of Favourable Conservation Status - FCS) which may require improvements to sediments at the site in order to secure long-term sustainability.

There are a number of further caveats to the application of these guidelines, as discussed by Grimwood and Dixon (1997) in the context of List II metals. Foremost are the possible influences due to fundamental differences in sediment geochemistry (as discussed above) and the use of non-indigenous test species in deriving thresholds. It is stressed that current sediment TELs/PELs are for total fraction (unsieved), therefore MBA data (<100µM) may represent the "worst case" assessment (discussed also in footnote[4]). Nevertheless, in the absence of any UK standards, interim guidelines adopted by Environment Canada (CCME 1999; see Annex 7) serve as a rough indication of the risk to biota from sediment contaminants. Hence, their application will help to identify instances where efforts may need to be made to minimise further inputs of these substances to the SPA.

Apart from the limited MBA metals/TBT sediment data, the most frequently used sources for metals and organic contaminants, are those of CEFAS, which along with the EA feed in to the UK National Marine Monitoring Programme (NMMP). Together these provide the information gathered under various EC Directives including Dangerous Substances (76/464/EEC); Shellfish Waters (79/923/EEC), Shellfish Hygiene (91/492/EEC), Urban Wastewater Treatment (91/271/EEC) and Nitrates (91/676/EEC). Many requirements will soon fall within the remit of the Water Framework Directive (2000/60/EC).

NMMP monitoring sites

There is a National Marine Monitoring Programme Shellfish Site (390) at West Mersea and nearby sediment/water quality sites in the outer Thames, 15 km off Foulness (site 466) and at Thames Wharf, a further 15km offshore towards the N. Kent Coast (site 465). Other NMMP sites in the region include the Thames Estuary (Mucking, 455) and, offshore some 20m to the NE at Thames Gabbard (site 475). The locations of these sites are shown in Figure 2.

Bioaccumulation and Biological Effects Indicators

The use of biological samples as indicators of environmental contaminant levels has long been suggested as an essential component of monitoring programmes, in addition to more traditional assessments using water and sediment analyses (Phillips, 1980; Bryan *et al.*, 1985). The incoming water framework directive also places increasing emphasis on biological effects monitoring.

With biological/ecological condition becoming a major diver behind environmental protection the reasons for the inclusion of indicator species (of exposure) are principally two-fold:

1. They accumulate only the biologically available forms of contaminants.
2. They act as temporal integrators of contamination, and average out varying environmental levels during the period of exposure.

In contrast, analyses of water or sediments usually indicate total concentrations of the contaminant, irrespective of whether it is bioavailable or not, and do not define accumulation potential or biological impact. Furthermore, water, and to a lesser extent sediment analyses provide a 'snapshot' of conditions at the time of sampling: they are influenced by tidal and meteorological conditions at that moment, and are not temporally integrated. Since environmental quality targets are most often aimed at the protection of biological resources, the use of organisms which reflect the presence of bioavailable metals is therefore a preferable means of assessing contamination.

As far as we are aware, EA and CEFAS monitoring of bioaccumulation has focused on the analysis of commercial shellfish (mussels and oysters) from one site in the EMS – the NMMP shellfish site at West Mersea - for obvious reasons connected to health of human consumers. Biological sampling to gauge the wider health of the ecosystem has received less attention.

Sampling of bioindicator species has been carried out at a number of sites in the Essex Estuaries EMS by MBA staff, to assess metal and TBT bioavailability. These surveys were not intended as a comprehensive study of the Essex Site but are part of broader programmes (on indicator organisms in UK estuaries; evaluation of temporal and spatial trends in bioavailability; identification of factors which modify accumulation; assessing the effectiveness of pollution control measures). Furthermore, as with sediment data, these results are now comparatively old (mainly from the late 1980s). There are no recent comprehensive biomonitoring datasets for the area, a feature which may need to be considered for the assessment process.

Seaweed *Fucus vesiculosus*, a gastropod *Littorina littorea*, the polychaete *Nereis diversicolor,* suspension feeding bivalves *Mytilus edulis*, *Cerastoderma edule* and deposit-feeding clams *Macoma balthica*, *Scrobicularia plana* were the principle indicators analysed in the Essex Estuaries. Their relative merits as indicators of metal contamination have been described in previous studies (Bryan *et al*., 1985; Langston *et al*., 2004). Because the current synthesis is only intended as general guidance of trends in bioaccumulation, we limit discussion of results to *Scrobicularia and Nereis.*

Biological effects (as opposed to exposure) monitoring has not been extensively undertaken in the EMS. CEFAS have conducted long-term research on TBT impacts on communities within in the Crouch Estuary, and have examined the evidence for endocrine disruption in flounders. Protocols using biomarkers and other biological indicators to measure impacts are only just being trialled in other areas, or are still under development and have not been routinely applied within the EMS. Where relevant information is available, this has been reviewed, though application of biological effects techniques within the Essex Estuaries, alongside targeted chemistry, is suggested as a priority for future assessments (as in most other EMS).

4. THE SITE: FEATURES AND THREATS

The Essex Estuaries European Marine Site (EMS) lies on the east coast of Essex in south-east England, where the average tidal range is ~4.5m. The EMS is a combination of intertidal Special Protection Areas (SPAs) for wild birds (designated under the European Commission Directive on the Conservation of Wild Birds (79/409/EEC), and the intertidal and subtidal marine Special Area of Conservation (SAC), designated under the Habitats Directive. The Essex Estuaries site has been recommended as a Special Area of Conservation (SAC) because it contains habitat types and/or species which are rare or threatened within a European context. The European Marine Site encompasses five SPAs: Colne Estuary, Blackwater Estuary (including Old Hall Marshes), Dengie, Crouch and Roach Estuaries and Foulness, all of which are designated SSSIs (SSSI citations describe the special interests for which the sites was notified in the British context - although this does not include the sublittoral) and Ramsar sites (wetlands of international importance designated under the Ramsar convention). Additionally, parts of the site are National Nature Reserves (NNRs) (Blackwater, Colne and Dengie), and Sites of Importance for Nature Conservation (SINCs). There are several designated shellfish areas within the EMS (see annex 6).

The boundaries of the EMS, which covers an area of 472 km^2, are shown in figure 1. Detailed maps of biological communities and features within the site can be found elsewhere (English Nature, 2000).

The marine SAC, stretches from Lion Point at the mouth of the Colne estuary to the mouth of the Thames at Shoebury Ness. It includes the estuaries of the Colne, Blackwater, Crouch and Roach as well as stretches of open coast off of Foulness and the Dengie. The landward boundary of the site is the high water mark (highest astronomical tide) whilst the seaward boundary is defined by navigation buoys. The five component SPAs are part of the phased Mid-Essex Coast SPA. The phased approach was adopted for a small number of very large sites which are ecologically a single entity, but where their sheer size made the classification process too complex. Thus, the Mid-Essex Coast SPA has been sub-divided into a number of separate phases that have been classified as individual SPAs. Annexes 1 and 2 list the features of the SAC and SPA, respectively.

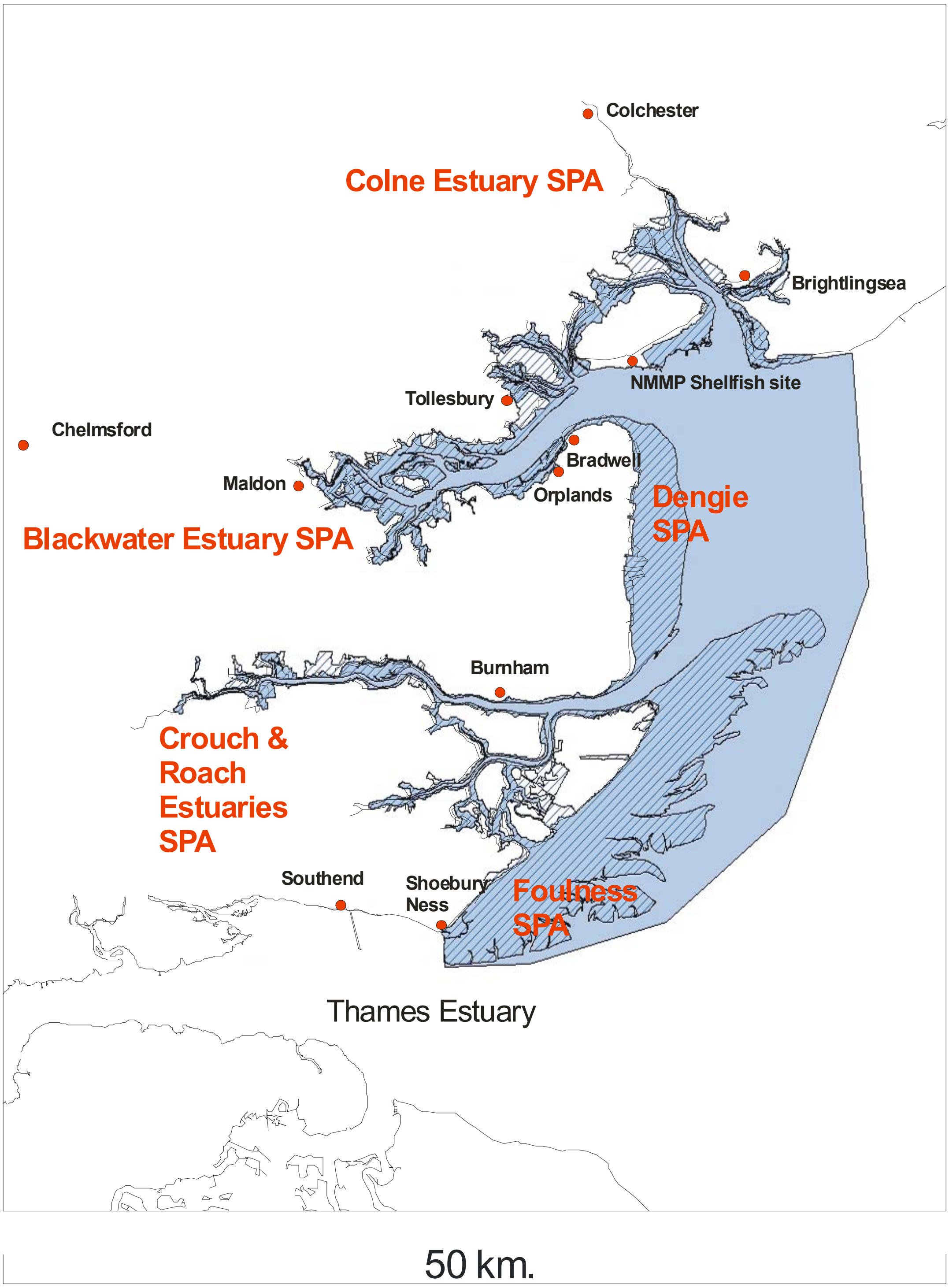

Figure 1: Essex Estuaries European Marine Site showing boundaries of the SAC and component SPAs (hatched areas)

Dengie SPA (phase 1) is a large and remote area of tidal mud-flats and saltmarshes lying at the eastern end of the Dengie peninsula, between the adjacent Blackwater and Crouch Estuaries. The saltmarsh is the largest continuous example of its type in Essex, important for wintering populations of Hen Harrier *Circus cyaneus*, wildfowl and waders. Additionally, the foreshore, saltmarsh and beaches support an exceptional assemblage of rare coastal flora.

The Colne Estuary SPA (phase 2) is a comparatively short and branching estuary, with five tidal arms flowing into the main channel of the River Colne. The intertidal zone is narrow and predominantly composed of fine silt flats, important for a range of wintering wildfowl and waders, and containing mud-flat communities typical of south-eastern English estuaries. The site also includes a wide variety of coastal habitats such as saltmarsh, grazing marsh, sand and shingle spits, which provide a nesting and breeding ground for Little Tern *Sterna albifrons*. Additionally disused gravel pits and reedbeds provide feeding and roosting opportunities for the large numbers of waterbirds that use the site.

The Crouch and Roach Estuaries SPA (phase 3). The Crouch Estuary is a sea inlet extending for approximately 29km, with relatively low freshwater input and corresponding high salinity for much of the year. Water depths rarely exceed 10m except for the outer channel. Much of the bed of the tideway is mud with a high proportion of shell and gravel in upper reaches, grading to sandier sediments of the outer channel. There are sewage inputs upstream, notably at Wickford, Southminster and at Burnham. Direct STW discharges to the estuary amounted to ~17,00 m^3 d^{-1} (maximum dry weather flow) in 1997 (Matthiessen *et al.,* 1998). Upper parts of the Roach receive sewage inputs from Rayleigh and Rochford. Industrial inputs are considered negligible. There are a number of marinas for example at Fambridge, Creeksea and Burnham. Antifouling and anticorrosion devices on boats are likely to be a significant source of Cu, Zn (and in the past TBT). Maintenance dredging of marinas has occurred in the recent past with disposal at Bridgmarsh Island and to land to consolidate sea defences. The influence of this activity is considered minimal (Waldock *et al.,* 1999).

The River Crouch occupies a shallow valley between two ridges of London Clay, whilst the Roach predominantly drains areas of brick earth and loams with patches of sand and gravel. The intertidal zone along the Rivers Crouch and Roach is relatively narrow and runs between sea walls along both banks and the river channel. Despite this limitation, the strip of intertidal mud is used by significant numbers of birds, being important for wintering waterbirds, notably Dark-bellied Brent Goose *Branta bernicla bernicla*.

The Blackwater Estuary SPA (Phase 4), which includes the subsumed SPA of Old Hall Marshes[5], is the largest estuary in Essex and one of the largest estuarine complexes in East Anglia. The Blackwater Estuary is some 23km in length, consists of a number of creeks, river channels and islands and drains an area of ~1200km^2 which is primarily agricultural (arable). The major freshwater inputs are from the Rivers Blackwater and Chelmer which collectively receive waste discharges from a population of ~400,000. Because of abstraction, a major component of the FW flow into the estuary is from the Chelmsford STW (~ 0.6m^3 s^{-1} DWF out of a total 3m^3 s^{-1}) (Emmerson *et al.*, 1997). Because of the relatively low flows river discharge is low in

[5] 'Subsumed' SPAs
Some early SPAs have been significantly extended. As the Birds Directive contains no provision for de-classification, however, it was not possible to declassify a small site simultaneously with the classification of the extended, surrounding area. Some SPAs have therefore been 'double designated'; that is, the area of the original classification has been included within the new extended site classification. In such cases, care has been taken to avoid double counting the areas, and bird populations, for both reporting purposes and in this review

relation to the tidal prism. The Blackwater Estuary, like the Crouch is extensively used for recreational purposes and supports an economically important fishing and shellfish (oyster) industry. There are >6000 leisure craft, three major marinas at Tollesbury, Bradwell and Maylandsea and a number of boatyards.

The Blackwater Estuary SPA comprises a wide diversity of habitats, including mud-flats, shingle, shell banks and offshore islands. The Blackwater Estuary also contains the largest area of saltmarsh in Essex (over 1000 hectares), and is the fifth largest such area in the UK. Consequently the area is very important for bird populations; in summer it is notable for breeding terns and in winter the site hosts a wide range of waterbirds including raptors, geese, ducks and waders. The surrounding terrestrial habitats - the sea wall, ancient grazing marsh and its associated fleet and ditch systems, and semi-improved grassland, are of high conservation interest. Erosion of the saltmarsh is a serious concern (200ha lost between 1973 and 1988) and has resulted in a number of pioneering restoration schemes, including managed retreat, at sites around the estuary.

Hydrography of the estuary was characterized to coincide with the development of the nuclear power station at Bradwell, during the early 1960's (Lowton *et al*., 1966). The estuary is predominantly well mixed with some vertical stratification in the upper reaches and slight lateral gradients in temperature and salinity (both higher on northern shores). In addition to the power station there have been several water quality concerns in the past including elevated levels of Hg, Cd and Pb in sediments of the River Chelmer downstream of Chelmsford, sludge dumping in the outer Thames Estuary, together with issues related to antifouling and anticorrosion on boats (Cu, Zn and TBT) (Emmerson *et al*., 1997).

Foulness SPA (phase 5) is located to the north of the mouth of the Thames estuary and is part of an open coast estuarine system comprising grazing marsh, saltmarsh, intertidal mud-flats, cockle-shell banks and sand-flats. It includes one of the three largest continuous sand-silt flats in the UK. The diversity of high quality coastal habitats in the SPA support important populations of breeding, migratory and wintering waterbirds, notably very important concentrations of Dark-bellied Brent Goose *Branta bernicla bernicla*.

Thus, the large estuaries of the EMS are typical coastal plain estuarine systems with associated open coast mudflats and sandbanks. The 4600ha of saltmarsh make up approximately 10% of the total saltmarsh area of the UK. The surface geology of the catchments are largely characterized by London clay and overlying glacial deposits of chalky boulder clay, sand and gravel. The site as a whole is important as an extensive area of contiguous estuarine habitat with a wide range of characteristic marine and estuarine sediment communities and diverse and unusual marine communities in the lower reaches, including a rich assemblage of sponge communities on mixed, tide-swept substrates. The rich invertebrate fauna includes the reef-building worm *Sabellaria spinulosa*, brittlestar *Ophiothrix fragilis,* crustaceans and ascidians (see section 5).

Threats to water quality

Sources of potential water quality pressures are typical of estuarine and coastal waters throughout the UK and include inputs from sewage effluent, agricultural (and some urban) run-off, landfill leachates and the atmosphere, including incineration (probably small for most contaminants). Shipping, recreational boating and other offshore activities (which in the past has included a significant component from sewage sludge dumping but is now restricted to dredge spoils) add to these land-based sources. All estuaries contain marinas and boatyards and there are small docks (2) for commercial ships in the Colne

The Colne and Blackwater have been classified as an 'At Risk' transitional water by the EA under its risk categorization exercise for the Water Framework Directive. At the time of writing the other estuaries and coastal waters of the EMS had not been classified.

Consented discharges

The river catchments are predominantly rural, with some light industry centred on Colchester. During operation (until 2002) the Bradwell Nuclear Power Station at the mouth of the Blackwater discharged cooling water at a rate of 151,000m3 d^{-1}. Current discharges of significance (small to medium in size) are thought to be mainly via STW, with few direct trade outfalls. The total direct discharges to the area were calculated by Allen *et al* (2000) to be 65,318 m^3/d of which roughly two thirds were sewage effluent and one third trade effluent.

No recent data on contaminants in discharges is available: this would require a major data retrieval exercise outside the scope of this project. The qualitative information provided by EA, on discharges most likely to impact on the system, is plotted in the figures below. The first (figure 2) indicates locations of the Agency's large consents (those in Bands A and B). The subsequent figure 3 is a detailed plot (from EA's Appendix 18 lists) which includes 'surface waters' and discharges to the Colne and Crouch which exceed $10m^3\ d^{-1}$ dry weather flow.

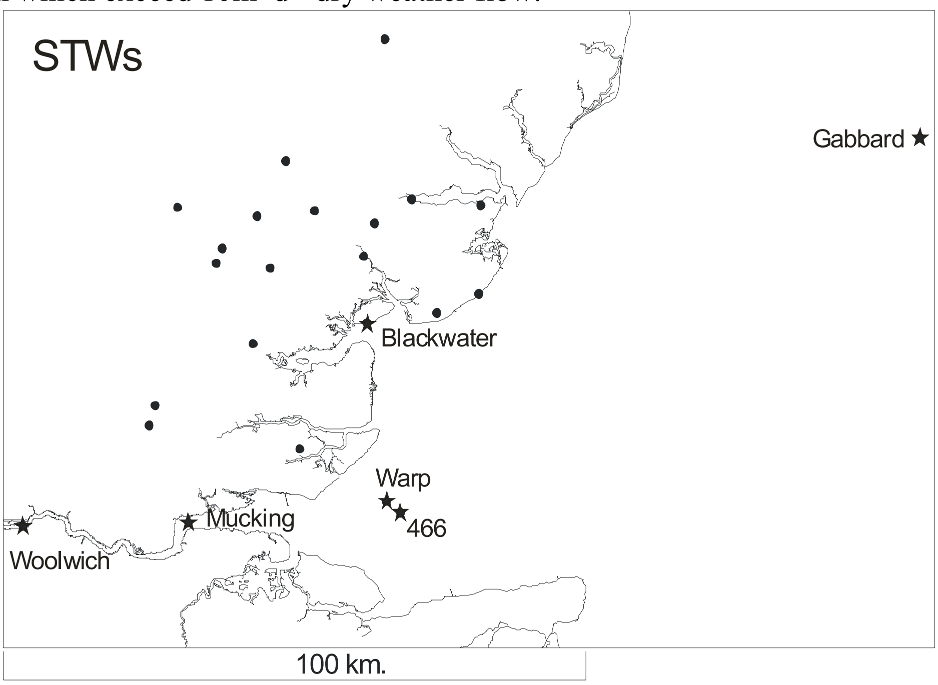

Figure 2. Location of large consents - those in Bands A and B (●). NMMP sites (★)

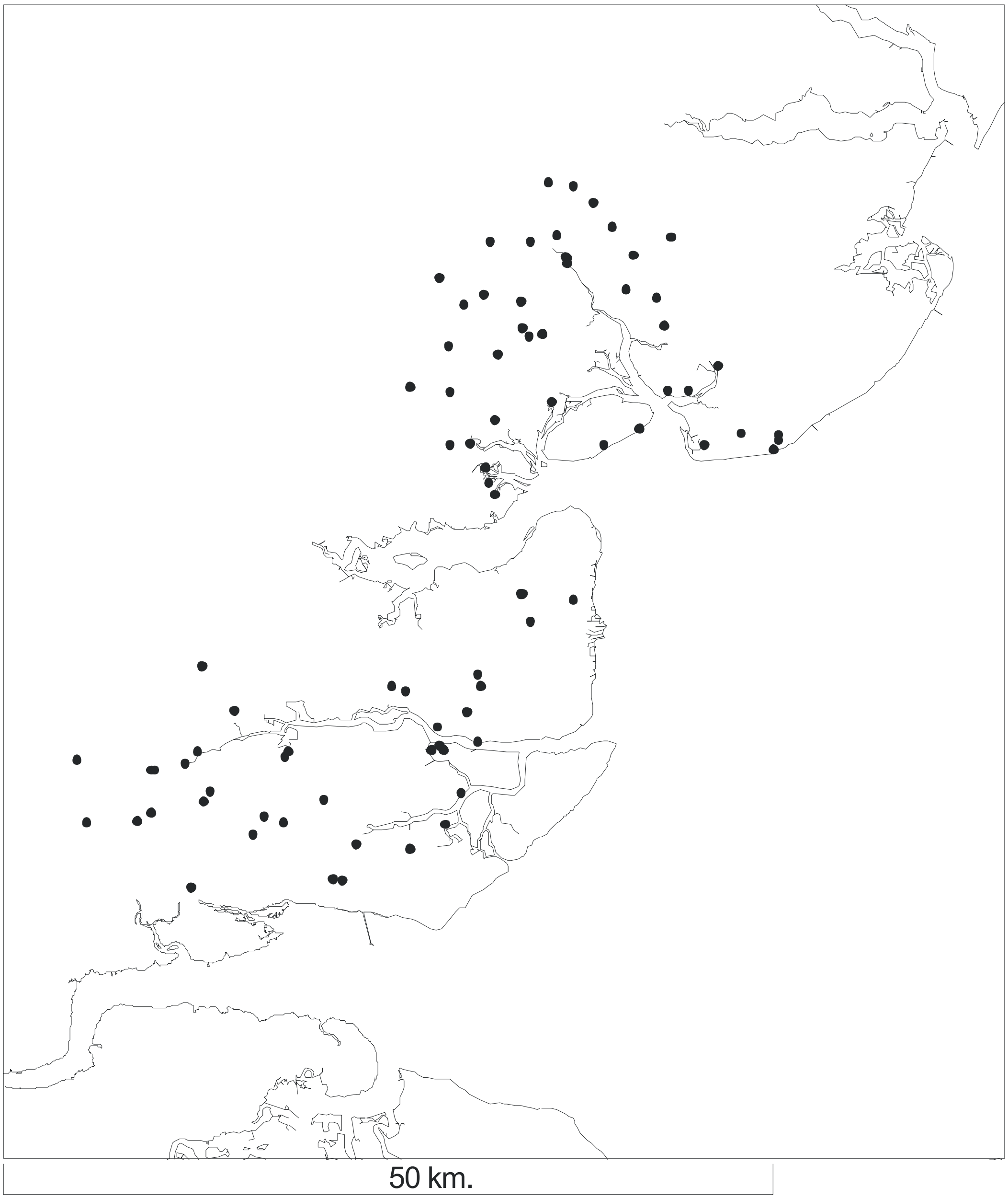

Figure 3. Locations of 'relevant consents' (from Application 18 lists) which include 'surface waters' and discharges to the Colne and Crouch which exceed $10m^3\ d^{-1}$ dry weather flow (data for the Blackwater not available).

An earlier but useful indication of the location and size of STW inputs to the Essex Estuaries, is shown in figure 4 below (from Allen *et al*., 2000). These are outfalls which are situated within the EMS itself and does not include discharges into rivers above the tidal limits.

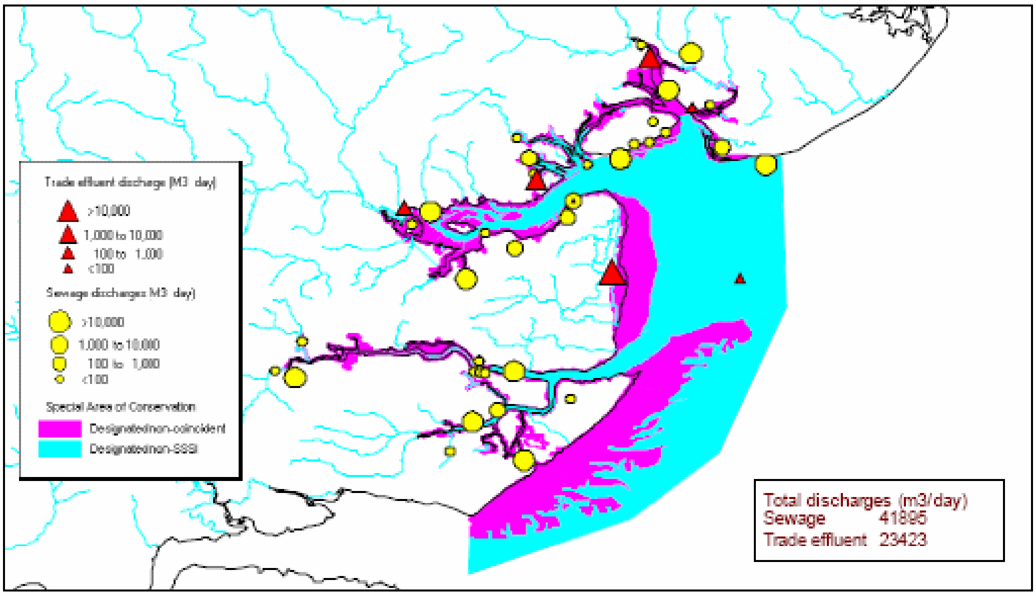

Figure 4. 1999 data on consented discharges within the boundary of the Essex Estuaries EMS. (Reproduced from Allen *et al*., 2000).

Essex Estuaries is one of the minority of marine SACs with disposal sites located in or near the site (others include Morecambe Bay, Fal and Helford, Solway Firth, The Wash, Moray Firth, Flamborough Head, Thanet Coast, Solent Maritime, Severn Estuary, and the Pembrokeshire Islands). The amount and frequency of maintenance dredgings disposed within or near these sites varies greatly: whilst more than 125,000 tonnes were dumped in the outer Thames Estuary in 2001, the figures for 2002-3 indicated there were no licensed disposals in the EMS (CEFAS, 2005a). More than 3.5 million tonnes was disposed off Harwich in that period, though it seems unlikely this would impact on the Essex Estuaries EMS.

Large quantities of sewage sludge, principally from STWs in the Greater London region, was dumped in the Barrow Deep and Black Deep off the Essex coast between 1967 and 1998. In the final year of licensed disposal, some 2.4million tonnes from Crossness, Beckton, Deephams and Riverside STWs were disposed of in this way (4.3 m t in 1997, probably closer to the average tonnage dumped each year). Water movements transported this material around the East Barrow Sand, and some deposition of solids occurred, particularly during calmer summer months. During Autumn and Winter storms, however, the finer fractions were less conspicuous - testimony to the generally dispersive site nature of the Outer Thames Estuary (MAFF, 1993). Because of the hydrodynamics of the system and the net sediment transport in a NE direction it is unlikely that fines and their associated contaminant loadings have impacted the EMS extensively though some influence is perhaps inevitable, as with the other sources of contaminants input directly to the North Sea via the Thames Estuary.

Litter in beam trawls has been reported to be significant off the mouth of the Thames Estuary in comparison to a number of other offshore sites surveyed (Cefas, 2000). However these surveys were conducted in the late 1990s, at a time when large amounts of sewage sludge were being disposed at the Barrow Deep. Presumably this impact has now been reduced considerably.

5. STUDIES ON BIOLOGICAL COMMUNITIES

EN (2000) and the JNCC Information Sheets on Ramsar Wetlands[6] probably contain the most recent descriptions of the ecological features of the Essex Estuaries Marine Site, including species lists and distributions. A number of diverse and unusual marine communities are reported to occur in the EMS, although no quantitative diversity indices have been applied to the Essex Estuaries. In a general description of the SAC Allen *et al.,* (2001) notes that the area has rich intertidal and subtidal invertebrate communities, including sponges, annelids, molluscs, echinoderms, crustaceans and ascidians. The reef-building worm *Sabellaria spinulosa* is also reported to occur in the Colne and on mixed sediments on the seabed of the open coast (EN, 2000: Colne Estuary Partnership, 2002). The Maplin Sands are host to nationally-important eelgrass beds and high densities of the hydroid *Sertularia* spp. The SAC contains the most southerly breeding ground in the North Sea of the common seal (in the Crouch), and the breeding ground of a herring subspecies (in the Blackwater). The Colne is the last known UK site of a coregonid fish known as the houting (*Coregonus oxyrinchus*).

Phytoplankton and microphytobenthos

Phytoplankton communities are considered to be light-limited in the turbid conditions of the EMS estuaries (notably in the Colne) therefore extensive blooms do not generally occur. Kocum *et al*., (2002a) studied phytoplankton communities in relation to nutrient and light availability at four sites within the Colne estuary (Hythe, Wivenhoe, Aldboro Point and Brightlingsea) (see also section 6.2). Phytoplankton biomass showed significant seasonal and spatial variation with higher biomass recorded during summer and lowest values measured at all sites between January and March. Biomass was greater at the head of the estuary (Hythe) than at the seaward end (Brightlingsea) in almost all months throughout the year. Flagellates, largely euglenophytes, were the major components of the phytoplankton in all seasons, although their relative abundance decreased in winter.

Kocum *et al*., (2002a) also reported evidence of some resuspension of benthic epipelagic diatoms (e.g. *Cylindrotheca closterium, Nitzschia* spp. and *Navicula* spp.) in the water column throughout the year, particularly at the 2 upper sampling sites (Hythe and Wivenhoe). The abundance of benthic diatoms was highest during spring and decreased in the summer, but generally diatoms were relatively rare (compared to flagellates) within the phytoplankton.

In contrast to the phytoplankton, the microphytobenthos (MPB) colonising the extensive intertidal mudflats of the marine site are much less light-limited as they are exposed to light for long periods during tidal emersion. As a major component of intertidal sediment microbial communities in terms of biomass and production (Admiraal 1984, Underwood & Paterson 1993, Underwood & Kromkamp 1999), the microphytobenthos (MPB) constitute a primary source of fixed carbon for food webs and provide a food source for estuarine fauna, notably deposit feeders (Admiraal 1984; Heip *et al*. 1995; Underwood & Kromkamp 1999).

Underwood (2002) considered that these benthic microalgae are the major primary producers in the Colne estuary. They form extensive biofilms on the sediment, particularly between Colchester and Alresford, as they migrate to the sediment surface at low tide to photosynthesise, utilising nitrate, ammonium and phosphate.

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[6] http://www.jncc.gov.uk

Thornton *et al.,* (2002) investigated biomass, species composition and primary production of the MPB along environmental gradients in the Colne Estuary between 1996 and 1998. Diatoms dominated the MPB at all sites (>90% cell numbers), though other groups were present at lower densities, including euglenoids and cyanobacteria. Seventy-five taxa of benthic diatoms were identified in the samples counted, and 10 different assemblage types were identified, with the majority of species that characterising the assemblages belonging to the genus *Navicula.*

Navicula salinarum and *Nitzschia frustulum* were principally associated with low temperatures (<5°C) in winter and relatively high concentrations of nitrate and ammonium. *Navicula diserta, N. cryptotenella, N. salinarum* and *Gyrosigma littorale* were also associated with higher concentrations of nutrients, whilst assemblages dominated by *Gyro-sigma limosum* were associated with high nutrients, and low temperatures but also low salinity. *Navicula phyllepta,* and *Fallacia pygmaea* were characteristic of summer assemblages, and the authors note that in the Colne the distribution of *N. phyllepta* may be limited to the lower estuary due to the toxic effects of high ammonium concentrations associated with the low salinity area at the head of the estuary. Competition between species may also be important (Thornton *et al.,* 2002).

Patterns and species-specific variation in migratory rhythms of the MBS in the Colne have recently been the subject of a detailed study (Underwood *et al*., (2005). Biofilm composition was found to vary, not only spatially, but also on a small temporal scale (daylight photoperiod). For instance, in the early morning, biofilms consisted of smaller *naviculoid* and *nitzschioid* taxa, or euglenoid species, but by midday, *Gyrosigma balticum* and *Pleurosigma angulatum* were dominant. Individual cells of all taxa declined significantly after midday, notably *Plagiotropis vitrea* which disappeared from surface layers. Species composition was found to change continuously toward the end of the photoperiod, and there was a general increase in cell numbers toward dusk, with *G. balticum* dominating in diatom-rich biofilms. In Euglena-rich biofilms, initial dense surface films of euglenids became progressively dominated by smaller diatoms.

These sequential micromigrations of MPB in the upper layers of biofilms are seen as a strategy to avoid photoinhibition, and the ability to vertically migrate through the sediment is considered to be a major reason for the success of MPB (Underwood *et al*., 2005).

Zooplankton

Between 1993-1997, CEFAS carried out spring plankton surveys in the Blackwater Estuary. The study was carried out principally in relation to the spawning population of herring *Clupea harengus* (see section on fish, below)*;* however, the macro- and meso-zooplankton of the estuary were described in the final report (Fox *et al.,* 1999). The dominant macrozooplankton were herring larvae, ctenophores (almost exclusively *Pleurobranchia pileus*), mysids, chaetognaths amphipods isopods and other *Natania* spp., whilst *Acartia* and other harpacticoid copepods dominated the mesozooplankton. Other common organisms in the plankton were polychaete larvae, cirripede nauplii and bryozoan, gastropod and bivalve larvae. Seasonal and geographical trends in the distribution of plankton were also identified. Distribution plots, and full listings of occurrence and abundance of planktonic organisms are contained in the final report (Fox *et al.,* 1999).

Macroalgae

There appear to be very few recent accounts of the macroalgal communities within the EMS. Any recent references are generally allude to the nuisance mats of *Enteromorpha* which can blanket intertidal sediments in many of the component estuaries (See also section 6.2.3). For instance, Reading *et al*., (2000) report growth mats of *Enteromorpha* at Tollesbury saltmarsh, adjacent to the Old Hall Marsh on the northern shore of the Blackwater Estuary, in relation to poorer establishment of *Salicornia* spp., and in a recent discussion document, Defra (2003) state that Colne and Blackwater Estuaries are suffering from increased occurrences of these algal mats.

Batters (1894) provided one of the earliest accounts of the marine algae of Essex, and the seaweeds of the Blackwater estuary are described in Howard (1964) and Milligan (1965). There are also two other very early publications (Baker, 1912, 1916) which describe the brown seaweeds of the saltmarsh, again, in the Blackwater estuary). These publications could provide a baseline (albeit somewhat spatially limited) with which to assess temporal change in macroalgal communities, should an up-to-date survey be carried out.

Saltmarsh

One fifth of the total area of British saltmarsh occurs in East Anglia, with the Essex Estuaries European marine site containing approximately 8% of the UK saltmarsh resource (about 3,500 hectares). Saltmarshes are areas of high biological productivity. They provide a rich source of nutrients to the mudflats, sandflats and subtidal areas that constitute wildfowl grazing sites. Saltmarsh is also important as a sediment store for the estuary system as a whole, and it provides roosting sites for waders and wildfowl at high tide. Saltmarshes also function as flood defences, absorbing wave energy and forming a natural buffer between land and sea. This latter function helps to protect the marsh surface itself from erosion (EN, 2000).

The saltmarshes of south-east England have been eroding rapidly for about the last 50 years, at a continuing rate of about 40 ha $year^{-1}$. Much of the loss of vegetation has been from the pioneer zone (table 2) (Hughes & Paramor, 2004).

Table 2. Proportions (%) of saltmarsh lost from the estuaries of Essex EMS from 1973 – 1988 (data from Burd, 1992) and from 1988 – 1997/8 (data from Coastal Geomorphology Partnership, 2000)

	1973 - 1988		1988 – 1997/8
	Total % loss	Pioneer zone % loss	Total %loss
Colne	12	53	7
Blackwater	23	74	7
Crouch	26	25	11

Saltmarsh losses are thought to be particularly high in south-east England because of isostatic land subsidence, which leads to a rise in relative sea level of about 1·5mm year (Shennan 1989). The processes affecting the dynamics of such subsiding coastlines are seen to augur what may occur more generally, with the anticipated acceleration of sea level rise due to global warming; hence the problems and management responses on the Essex coastline have attracted international interest and, sometimes, conflicting views (Hughes & Paramour, 2004; Morris et al., 2004).

There could be a number of factors contributing to saltmarsh decline, notably disruption in patterns of erosion or accretion resulting from coastal realignment, and erection of coastal defence structures and channel dredging. A degree of smothering by storm-washed sediment is a natural occurrence in this dynamic system, although increased storm activity due to climate change may be an additional factor. In a study of changes along the open coast marshes of the Dengie peninsula, Pethick (1992) showed that storms resulted in 1.5 metre waves at the marsh edge and they caused a lowering of some 50 mm in the mean surface level of the marsh. These losses were recovered within two years, but given that the marsh would have continued accreting in this period, the full recovery was only achieved five years after the storm. The implication here is that if there is an increase in the frequency of extreme events, increasingly shorter periods will be available for recovery (Boorman, 2003).

There are currently several managed realignment schemes within the Essex Estuaries EMS, i.e. the Blackwater (includes Northey Island, Tollesbury, Orplands and Abbots Hall) and the Colne, the Dengie Peninsula, the Roach and the Crouch, and more are planned (Halcrow Group, 2002). Managed realignment (also known as retreat or set-back), where sea walls are breached or neglected to allow land to become intertidal, is currently a favoured EA/EN option for saltmarsh restoration and flood defence. Salt marsh created in this way can, it is hoped, significantly reduce wave energy allowing the provision of more sustainable sea defences. There are however differing opinions about the causes of erosion and the prospects for development of new saltmarsh at managed realignment sites.

It has generally been considered that the majority of saltmarsh erosion in England is due to coastal squeeze, where sea walls have been built as sea/flood defences and have prevented a landward migration of saltmarsh in response to sea level rise (Burd 1992; English Nature 1992; Covey & Laffoley 2002). However, the pattern of vegetation loss, mostly of pioneer zone species, is considered by some to be opposite to that predicted by coastal squeeze, where the upper marsh plants should disappear first (Hughes & Paramor, 2004). Results from recent laboratory and field studies on saltmarshes at Tollesbury have been put forward as offering an alternative explanation for saltmarsh erosion which could give pause for thought over similar managed realignment schemes in future (Hughes & Paramour, Paramour & Hughes 2004). These authors suggest that much of the loss of Tollesbury saltmarsh is due to lateral erosion of the internal creeks, exacerbated by bioturbation and herbivory by the infaunal polychaete *Nereis diversicolor* (see also the section on seagrass). Hughes & Paramor, (2004) note that much of the loss of vegetation has been of pioneer zone species (table 2) and earlier studies in Essex have suggested that, if protected from bioturbation and herbivory by the *N. diversicolor* and, to a lesser extent, the amphipod *Corophium volutator* these plants can exist at lower elevations than they normally occur (Gerdol & Hughes 1993; Smith, Hughes & Cox 1996; Hughes 1999; Hughes *et al.* 2000; Paramor & Hughes 2004).

Hughes *et al*., (2000) proposed that there are two stable states on the upper mudflats. One state is dominated by plants, including *Zostera* spp., which prevent colonisation by burrowing infauna, and the other is dominated by infauna, which prevent colonisation by plants. There is some evidence that *Nereis*, has increased in abundance over the last few decades (Hughes, 1999), although the reasons for this are not certain. It is argued therefore that a shift from the plant-dominant to the infauna-dominant state could explain the relatively recent regional-scale losses of pioneer zone vegetation. The erosion of the seaward edge of some marshes may also be due to increased wave action, and increased tidal current speeds in estuaries, following the loss of intertidal seagrasses since the 1930s through wasting disease (Paramor & Hughes, 2004).

These suggestions - that the causes of saltmarsh loss are not necessarily related to sea level rise - have led Hughes and Paramor (2004) to call into question the management realignment option as the most appropriate means of saltmarsh creation, not least because many realignment areas were considered unlikely to develop vegetation. However, Morris et al (2004), in a paper reflecting EN's position, claim that recolonisation of saltmarsh plants has been successful at some realignment sites (albeit at rates slower than expected)[7], whilst at the same time rebutting suggestions put forward by Paramour and Hughes (2004) that bioturbation and herbivory by infauna are a principal cause of erosion, preventing saltmarsh development. Morris *et al* (2004) argue that the use of matting by Paramour and Hughes, to exclude infauna from experimental plots, may have had direct effects on both sediment accretion and colonization by halophytes.

Clearly this controversial debate signals the need for more large-scale research into the processes of marsh protection and regeneration, over a wider range of sites.There is no doubt, however, that for saltmarsh to continue to adjust to changing sea level (including at management realignment sites) there must be sufficient sediment supply for the development of pioneer species. Methods available for creating new marshes, and for reducing/ reversing marsh erosion therefore include use of dredged material for strategic intertidal recharge, and might include also transplantation of intertidal seagrasses and, controversially, exclusion of the infauna, (Paramor & Hughes, 2004).

Foreshore recharge is currently in operation at Mersea Island and Cobb Marsh Island and thus affects the Blackwater and Colne Estuaries. Recharge is an attempt to stabilise and build up the foreshore with sediments able to withstand local erosive forces. Sediment polders (a grid of stakes and brushwood) constructed at the shoreline encourage deposition of silt and have been placed at various locations along the coast (BTO, 2004).

[7] Some 6ha of the 21ha Tollesbury site were reported to be colonised by *Salicornia europaea* agg. (L) six years after realignment in 1995; see also other examples in Garbutt et al., (2003)

Saltmarsh Communities

Apart from very early studies of Essex Estuaries saltmarsh communities (e.g. Baker, 1912; Baker & Bohling, 1916; Wake, 1978; Whitehead, 1911), studies of the Essex Estuaries saltmarshes have generally focussed on the mechanisms, causes and implications of their loss, and the restoration/remediation schemes that have been initiated to ameliorate major impacts.

Again, EN (2000) and the JNCC Information Sheets on Ramsar Wetlands[8] probably contain the most recent descriptions of the ecological features of the saltmarshes within component Estuaries of the EMS, including species lists and distributions.

In the Blackwater Estuary, sandy areas of beach have a typical plant community of sand couch *Elymus farctus*, marram grass *Ammophila arenaria* and the very local sea-holly *Eryngium maritimum*, and the frosted orache *Atriplex laciniata* occurs on the drift line (JNCC, 2004). Generally, the lower saltmarshes of the various estuaries support plants such as glassworts *Salicornia* spp. including the nationally scarce *S. perennis* and *S. pusilla* and the invasive cord grass *Spartina anglica,* which are salt tolerant and are able to withstand prolonged tidal immersion. Upper shore plants include sea purslane *Atriplex portulacoides*, sea aster *Aster tripolium*, sea lavender *Limonium vulgare* and common saltmarsh grass *Puccinellia maritima*. The nationally scarce lax-flowered sea lavender *Limonium humile* is sometimes interspersed among the more common species, although the main site for this is at Old Hall. Saltmarsh grasses usually associated with higher marsh zones, such as stiff saltmarsh-grass *Puccinellia rupestris* and Borrer's saltmarsh grass *Puccinellia fasciculata*, grow behind the sea walls.

Generally, the transition from saltmarsh to grassland has been truncated by sea defences, although Ray Island, in the Strood Channel of the Blackwater Estuary is one of the few sites in Essex where there is no wall. Rare upper saltmarsh plants such as golden samphire *Inula crithmoides* and shrubby sea blite *Suaeda vera* occur mainly at the base of sea walls. These are Mediterranean plants at the northerly limit of their distributions.

S. vera also colonises unstable shingle along the drift line and large populations are present at West Mersea and Osea Island. The higher wave energies at creek mouths and the upper reaches of the estuary promote the deposition of shingle and sand. Where this has occurred on the foreshore at West Mersea the county rarity, sea spurge *Euphorbia paralias*, has been discovered along with sea mayweed *Tripleurospermum maritimum,* which has a limited distribution in Essex.

The most extensive remaining stand of the native small cord-grass *Spartina maritima* in the UK and possibly in Europe is located at Foulness Point and covers approximately 0.17 ha. Other smaller stands of *S. maritima* are found elsewhere in the estuary complex, notably in the Colne estuary, where it forms a major component of the upper marsh areas.

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[8] http://www.jncc.gov.uk

Prior to 1870, *S. maritima* was the only *Spartina* species in Great Britain. It was never common, but formed a significant community of the low marsh around the Wash, Thames basin and the Solent. Accidental introduction of *S. alterniflora*, a native of America, to Southampton Water and its subsequent crossing with the native plant resulted in the appearance of a fertile common cord-grass *S. anglica*, which is now the most widespread *Spartina* species in the UK (Davidson *et al.,* 1991). Reasons for its success are numerous, and include a suite of biological properties related to its hybrid origin, and notably, its occupancy of a formerly vacant niche on intertidal mudflats to seaward of the previous limit of perennial vegetation (Gray, 1986). There are no records to indicate the current distribution and extent of *S. anglica* in Essex Estuaries EMS, although JNCC indicate its presence in all the component estuaries. Its establishment does not appear to have altered the general ecology of saltmarshes (EN, 2000).

S. anglica is generally considered to be a negative conservation feature of sites where it occurs, and at several sites in the UK the loss of mudflat was considered to be serious enough to justify attempts to control it. However, in some areas, such as the Dee Estuary, it can act as a pioneer species for the formation of Atlantic salt meadow (Dargie 2001) and it was deliberately introduced to some estuaries, such as the Exe Estuary, on the south coast, where it was intended to help stabilise the Dawlish Warren saltmarsh, and the spit, and to protect the embankment in the estuary (Proctor, 1980). In the Blackwater estuary, the smooth cordgrass *Spartina alterniflora* was introduced and is currently well established at three locations (EN, 2000).

A distinctive feature of the Blackwater saltmarsh is the intricate network of drainage channels which, in addition to their habitat value, perform an important coastal protection role, dissipating tidal energy and lowering the force of waves (BEMP, 1996). These channels may be important in determining the composition of saltmarsh vegetation; a recent study on the natural marsh at St Peters Flat on the Dengie Peninsula found soil drainage to be an important factor, acting against species such as *Halimione portulacoides,* which is intolerant of poor drainage, and favouring those more tolerant of moisture and anaerobic soil conditions e.g. *Puccinellia maritima*. It may therefore be the case that the poor drainage on restored marshes in this region has an influence, not only on sediment erodibility and extent of bare mudflat but also upon the saltmarsh vegetation communities (Crooks *et al.,* 2000).

Seagrass

Eelgrass (*Zostera* spp.) beds are relatively rare in Europe, and although they probably occur mostly in UK estuaries, they are by no means common (Davison & Hughes, 1998). There are three species of *Zostera* in the UK, *Zostera marina, Z. angustifolia, Z. noltii*, all of which are considered nationally scarce. Because eelgrass is able to establish where few algal species can grow, eelgrass beds influence both the physical and chemical environment around them. By reducing current speeds within the beds, a substrate of stable, poorly sorted sediments develops which encourages suspended material to deposit out. In this manner, the plants stabilise the substratum. The beds are an important source of organic matter and a rich source of food, particularly for wildfowl (wigeon, Brent goose). Eelgrass beds also provide shelter and surface attachment for many species, such as fish (flatfish, cuttlefish, squid) and molluscs (*Rissoa membranacea, Akera bullata*) as well as algae, diatoms, anemones and stalked jellyfish *Haliclystus auricula*). The stems and leaves of the eelgrass plants provide refuges for young fish and crustaceans (shrimps, crabs etc). Thus, the beds have a high conservation value. In the 1930s a wasting disease reduced the size of many subtidal UK eelgrass beds dramatically.

Evidence suggests that eelgrass beds in the UK are declining. Its decline nationally may have serious consequences for the rich and diverse fauna, often associated with beds of *Zostera* and fine algae, and for the waders and wildfowl which feed amongst the beds. Wildfowl (ducks and geese) are among the few animals which graze directly upon *Zostera* and are able to digest its leaves. In Britain, *Zostera* is an important constituent of the diet of two sub-species of Brent geese *Branta bernicla*, wigeon *Anas penelope*, mute swans *Cygnus olor*, and whooper swans *C. cygnus*. Teal *Anas crecca* are reported to consume eelgrass seeds (Tubbs & Tubbs, 1983). This is particularly relevant in SPAs and SACs, where rare and nationally important species occur, notably in the Essex Estuaries EMS, i.e. Brent geese in Foulness and Blackwater SPAs, Teal in the Blackwater Estuary SPA. Davison and Hughes (1998) have produced a comprehensive overview of dynamics and sensitivity characteristics of *Zostera* in UK SACs, and although it is difficult to speculate on the exact cause of the decline, table 3 summarises natural events and human activities which may be contributing factors.

Zostera beds in the Essex Estuaries marine site, are limited to Maplin Sands, and two sites on the Blackwater. The shallow water *Zostera* bed on Maplin Sands covers 58 hectares (Irving, 1998) and is particularly important for its associated animal communities. This and sites linked to the Mid-Essex Coast SPAs in the Thames estuary, support some 40 to 50% of the UK population of dark-bellied brent geese in November (EN 2000).

All three species of eelgrass once formed large beds in the Blackwater Estuary but are now relatively scarce (JNCC, 2004) - a strip extending along the northern shore of Osea Island, and a discrete patch at St Lawrence Bay (English Nature, 1998).

As noted above loss of saltmarsh vegetation in south-east England is a significant conservation and coastal defence problem. The losses of vegetation in the EMS began in the 1930s with the loss of intertidal *Z. marina*, due to wasting disease, (although it has been suggested that plants only succumb when they are stressed by other factors) and has continued more recently. Some preliminary trials at establishing *Zostera nolteii* in some estuaries of Essex have not been successful. Hughes *et al.,* (2000) investigated the possibility that the infauna, particularly the polychaete *Nereis diversicolor*, may restrict natural colonisation by *Zostera* and reduce the success of transplanting trials. Field experiments were carried out transplanting *Z. noltii* into several areas. i) where *Nereis* were common ii) close to an established seagrass bed and iii) into two other estuaries. The transplants which were protected from the effects of the polychaetes by netting had a higher survival rate, lower index of root damage and greater biomass at the end of the experiments than those that were unprotected (Hughes *et al.,* 2000).

Table 3. Natural events and human activities which may be contributing factors to the decline of *Zostera spp.* (adapted from Davison and Hughes (1998)

Natural events -*Zostera* beds are spatially dynamic, and subject to a number of naturally-occurring factors which can cause changes in coverage at a range of scales.
• Extreme weather conditions such as violent storms or heavy floods can denude eelgrass beds over wide areas. Plants can also be killed or damaged by severe frosts. • Wasting disease is the most important factor observed to cause long-lasting declines in the number and extent of *Zostera* beds. The most severe outbreak of this disease took place in the early 1930s, and recovery from this is still incomplete. The disease-causing agent is the fungus *Labyrinthula macrocystis*. This is probably continually present at low levels, but undergoes occasional epidemic outbreaks for reasons which are not fully understood. *Labyrinthula* does not appear to cause disease in conditions if salinity is low, so that the intertidal/estuarine *Zostera* species (*Z. angustifolia* and Z. *noltii*) are much less susceptible than *Z. marina*, which prefers subtidal marine conditions. • Wildfowl grazing can remove a high proportion of the available *Zostera* biomass (over 90% in some cases), but beds can normally withstand this grazing pressure unless under stress from some other factor. • Declines in populations of epiphyte grazers can indirectly affect the health of *Zostera* beds by allowing increased growth of fouling algae. Nutrient enrichment or other forms of anthropogenic pollution are the factors most likely to bring about such changes.

Human activities - A large proportion of the UK's population lives on or adjacent to the coast. As a result, pollution, development and recreation pressures are increasingly affecting the coastal environment, and their impacts can be especially acute in the shallow bays, estuaries and lagoons where *Zostera* biotopes most commonly occur.
• Coastal development can have adverse effects on *Zostera* beds by causing increased sediment erosion or accretion (depending on the nature of development), and by causing increases in water turbidity. • There is little evidence of harm caused by heavy metals or antifoulants, but runoff of terrestrial herbicides has been shown to affect growth and survival of *Zostera* plants. • Eelgrass beds are not highly sensitive to chronic oil pollution (e.g. refinery effluent). However, when exposed to major oil spillages, the associated fauna appear to be more susceptible to damage than the *Zostera* itself. The chemical dispersants used to control oil spills are more harmful to Zostera than the oil alone, and should not be used in these biotopes. • Excessive nutrient enrichment can cause damage to eelgrass beds by a variety of mechanisms, the most important of which are metabolic imbalance, proliferation of phytoplankton, epiphtyic or blanketing algae, and increased susceptibility to wasting disease. • Eelgrass beds are not physically robust biotopes, and can be degraded by trampling, mechanical bivalve harvesting, dredging and other forms of disturbance. • Two non-indigenous plants, the cord-grass *Spartina anglica* and the brown alga *Sargassum muticum* have colonized eelgrass beds in the UK, mainly in the south of England. To date, there is no firm evidence of either species competing significantly with *Zostera* or displacing it in the absence of other adverse environmental factors. • Disturbance by wildfowlers may cause local increases in numbers of ducks and geese on *Zostera* beds, and hence higher grazing pressure on the eelgrass. • Human-induced climate change may have significant long-term effects on the distribution and extent of *Zostera* beds. Possible significant effects include higher temperatures and increased frequency and severity of storms.

In laboratory experiments, *Nereis* reduced the survival of *Z. noltii* by grasping the leaves and pulling them into their burrows. Results indicated that herbivory and disturbance by *N. diversicolor* is partly responsible restricting the distribution of *Z. noltii*, and may have been important in limiting the success of previous transplanting experiments. Hughes *et al*., (2000) proposed that there are two stable states on the upper mudflats of the EMS (see 'saltmarshes' above). The authors considered that managing these two states could be the key to re-establishing the early successional stages of saltmarsh development (Hughes *et al*., 2000).

Although *Zostera* species are fast-growing and relatively short-lived, they can take a considerable time to recover from damaging impacts - if recovery is possible at all. Holt *et al.,* (1997) estimated that *Zostera* species recoverability is within the range of five to ten years but, in many cases, recovery may take longer. This is borne out by the slow or apparent lack of recovery from the 1920s to mid-1930s wasting disease epidemic.

A reduction in the biomass is an early indication of stress in seagrass beds (EN 2000) and it is important to identify possible stressors in the vicinity of seagrass beds. Table 4 summarises the key factors which may limit or facilitate seagrass bed recovery in marine SACs and elsewhere.

Table 4. Summary of major factors believed to influence the capacity of *Zostera* beds to recover after disturbance or destruction (from Davison and Hughes 1998).

Factors that may limit bed recovery	Factors that may facilitate bed recovery
Removal of habitat	Artificial transplantation
Unstable substrata	Stable substrata
Fragmenting and destabilized *Zostera* beds, caused by factors such as changes to coastal processes, physical damage or stochastic weather events	Stable *Zostera* beds
Reduced rhizome growth, seed production, germling success and seedling development into patches	Increased rhizome growth, seed production, germling success and seedling development into patches
Reduced light penetration, caused by increased turbidity, eutrophication, some forms of pollution, or epiphyte smothering	Improvements in light penetration, caused by reductions in turbidity, eutrophication, pollution, epiphyte and algal smothering
Nutrient enrichment	Reductions of, or limited increases to, nutrient inputs
Declines in epiphyte grazer populations Unusual increases in wildfowl grazing pressure	Healthy and stable epiphyte grazer populations Wildfowl grazing activities may prevent excessive sediment build up in *Zostera* beds
Competition with non-native species, *Spartina* sp. and *Sargassum muticum*	Absence of non-native species, *Spartina* sp. and *Sargassum muticum*
Environmental stress, (e.g. extreme temperatures or pollutants), which may increase the susceptibility to wasting disease infection	Absence of environmental stresses and low populations of *L. macrocystis*, the causative fungal pathogen for wasting disease

Meiofauna

Prompted by the OSPAR Convention requirement to examine alternative options to conventional marine disposal of dredged material (OSPAR Commission, 1998), Cefas have recently carried out a field manipulation experiment on intertidal mudflats in the Crouch Estuary (Bridgemarsh Island) designed to investigate processes affecting the development of a meiobenthic nematode community in response to the intertidal placement of different types of sediment (Schratzberger *et al.*, 2004). This is interesting in that the use of dredged material for 'intertidal recharge' is one of the management options proposed by Paramor &Hughes (2004) in response to saltmarsh erosion (see section above on saltmarsh).

In the Cefas study, natural assemblages of nematodes were exposed to different types of substrate, including a defaunated control sediment with natural sand and organic content, sediment with elevated sand content and sediment with elevated organic content for a period of 12 months.

Results showed that differences in the composition of nematode communities were mainly determined by changes in the abundance of the chromadorid species *Chromadora macrolaima* and *Ptycholaimellus ponticus* the comesomatid *Sabatieria punctata* and the microlaimid *Molgolaimus demani*. *C. macrolaima* and *P. ponticus* quickly recolonised both the defaunated control and the high sand content treatment. Nematode numbers in the organically enriched sediment equalled those in the ambient control after 3 months, and after 1 year were significantly greater. Slower colonisation rates of *S. punctata* and *M. demani* were observed in the defaunated control, but numbers in the high sand content treatment were equal to that of the ambient control after 3 months. In contrast, numbers of these species had not recovered in the sediment with elevated organic content by the end of the experiment.

Thus, results showed that assemblage structure in all treatments, notably the defaunated control and the high sand content, became increasingly similar to the ambient sediment over time, although recovery to ambient condition had not occurred 1 year after the set-up of the experiment. Overall, the experiment indicated that characteristics of 'recharge' sediment, particularly the organic content compared to native substrate, strongly influence the composition of colonising assemblages (Schratzberger *et al.*, 2004). This will obviously have implications should the use of dredged material to supplement eroding saltmarsh and create new habitat become standard practice.

Macrofauna

There are several designated shellfish areas within the EMS (see annex 6). Oysters, *Ostrea edulis* and *Crassostrea gigas,* clams *Tapes philippinarum* and *Mercenaria mercenaria*, cockles *Cardium edule* and mussels *Mytilus edulis* are all farmed within the marine site. The Blackwater estuary has a long history as a shellfish resource, particularly for oysters, and also a long history of problems and set-backs. The oyster fishery was severely affected by the harsh winter of 1963, and later the disease bonamiasis (resulting from a protistan parasite *Bonamia ostreae* which infects the blood cells of the native oyster *Ostrea edulis*) in the 1980s (BEMP, 1996). Around this time TBT (antifouling) inputs from boats were high and caused serious effects to the local oyster industry and on benthic invertebrate communities, but since TBT-use was banned on small vessels in 1987, these communities are reported to have recovered (Rees *et al.,* 1999, 2000; Waldock *et al*., 1999).

Predatory gastropods such as the native Sting winkle *Ocenebra erinacea* and more notably the introduced American oyster drill *Urosalpinx cinerea,* have also caused problems for the shellfisheries over the years, as they prey on small oysters and oyster spat. *U. cinerea* is believed to have been introduced from the east coast of the USA at the end of the 19th Century, associated with American oysters *Crassostrea virginica*. It was first recorded from the Essex oyster grounds in 1927 (Orton & Winckworth, 1928) and by the mid-20th Century, had become a serious threat to the shellfisheries of Essex. In 1953, 55 -58% of the oyster spat settling in Essex oyster beds were destroyed by *U. cinerea* (Hancock, 1954).

TBT (tri-butyl tin) antifouling paints used on ships and leisure craft in the early 1980s may also have had some adverse effects to *O. edulis,* (stunted growth and reduced reproductive capacity[9]), however, *U. cinerea* are much more highly susceptible to TBT, and the development of 'imposex' depleted populations of the American oyster drill on the Essex oyster beds since the early 1970s (Gibbs, Spencer & Pascoe 1991) (see also section 'TBT' in section 6). However, live specimens and egg cases of *U. cinerea* continue to occur amongst oysters collected outside the Blackwater and Crouch estuaries, in deeper water offshore, and some breeding enclaves may still survive off Whitstable, (Gibbs, Spencer & Pascoe 1991). This may once again pose a threat to the marine site oyster populations once there have been sufficient reductions in concentrations of TBT.

Natural populations of *O. edulis* are now reported throughout the Blackwater Estuary, especially in the outer reaches to the south of Mersea Island. Sheltered areas, such as the Salcott and Pyfleet Channels provide ideal grounds for 'growing on' oysters brought in as spat, and Pacific oysters *Crassostrea gigas*, which grow to market size more rapidly are also cultivated in Blackwater. The principal Pacific oyster fishery is considered to be in Goldhanger Creek on the north shore of the inner estuary (BEMP, 1996).

The river Crouch was the subject of several biological studies in the 1950s, also in relation the commercial interest in *O. edulis* stocks and the need to establish reasons for their long-term decline (e.g. Waugh, 1957; Shelbourne, 1957; Knight-Jones, 1952). With the recognition of the wide environmental consequences that the use of TBT might have, and its subsequent ban on small craft, various studies were initiated to examine any changes to benthic communities arising from the anticipated reduction in TBT concentrations. Rees *et al.,* (1999) and Waldock, *et al.,* (1999) reported on this work up until 1992 and found evidence of an increase in the diversity of infauna and epifauna.

More recently (1997) Rees, *et al.,* (2001) carried out a survey of the epifauna of the Crouch. Comparisons with very early surveys (Fitch, 1891 and Crouch, 1892) indicated significant long-term changes in the dominant species, most notably high densities of the gastropod mollusc *Crepidula fornicata,* recorded at the innermost sampling sites, and a decline in numbers of *O. edulis.*

C. fornicata, more commonly known as the slipper limpet, is known to have been introduced to Essex between 1887 and 1890 from North America, in association with

[9] akbap.org.uk

the American oyster *Crassostrea virginica* (Crouch 1894, 1895; Orton 1912; Fretter and Graham 1981) and spread fairly rapidly (Franklin and Pickett 1974). Its success in this country is probably due to a lack of predators and the unusual method of reproduction (which relies upon individuals settling upon each other to form breeding 'stacks' as they develop from males to females). A pelagic larval stage aids the spread of the species, once introduced.

Reports suggest that high densities of *C. fornicata* can modify the nature and texture of sediments in some bays (Ehrhold *et al.,* 1998) and where *Crepidula* stacks are abundant, few other bivalves or other filter-feeding invertebrates can live amongst them. This is due to spatial competition, trophic competition and alteration of the substratum (the pseudofaeces of *C. fornicata* smother other bivalves and render the substratum unsuitable for larval settlement) (Fretter and Graham, 1981; Blanchard, 1997). In this way, *C. fornicata* has become a serious pest on oyster beds and has caused many traditional oyster fisheries to be abandoned (e.g. in the Norman Gulf, France) (Blanchard, 1997). However, De Montaudouin *et al*. (1999) showed that *Crepidula* had no major influence on the local density or diversity of smaller coexisting macroinvertebrates and did not affect the growth of 18 month old oysters.

There is no evidence to associate the long-term decline of *O. edulis* in the Crouch directly to the establishment of *C. fornicata.* As indicated above, there have been several other influencing factors, which may have acted alone or in combination.

In the shorter term (5 years), Rees, *et al.,* (2001) report a substantial increase in the numbers of young adult *O. edulis* at inner estuarine sites, which is noted with special interest in view of a possible link to declining TBT concentrations (see also section 6).

Rees, *et al.,* (2001) also found high densities of ascidians in the inner Crouch during the survey of 1997. Six ascidian taxa were recorded here, compared with only two ten years earlier. Species included *Ascidiella aspersa* and *Ascidia conchilega. Ciona intestinalis* was also, for the first time, relatively common in this region. There were also increases in numbers of scale worms Polynoidae, spider crab *Macropodia rostrata,* and gastropod *Gibbula* spp. In the mid estuary a substantial increase in numbers of (mainly juvenile) *Buccinum undatum* was also noted (Rees, *et al.,* 2001)

The distributions and densities of several species including the shore crab *Carcinus maenas,* starfish *Asterias rubens* and sand goby *Pomatoschistus minutes* did not appear to have changed over a five year period, however the authors noted a complete absence of the sun star *Crossaster papposus,* previously common in mid-estuary.

Migratory shrimps *Paleamon montagui* and *Crangon crangon* were widespread in the estuary, although in much reduced numbers to those recorded in previous surveys. Perhaps because of this reduction, numbers of shrimp predators *Agonus catahractus* the armed bullhead, and *Myoxocephalus scorplus* the bullrout, had also declined.

Generally, results from this survey prompted Rees, *et al.,* (2001) to conclude that there had been an underlying improvement in environmental conditions following the TBT ban in 1980, however, the continued absence of neogastropods suggested that recovery was still incomplete at the population level.

Semi-quantitative studies of epifauna sampled by beam trawl at offshore sites at the mouth of the Thames Estuary and off the Essex coast have been used to characterize key biological elements. Dominant taxa included *Paguridae, Electra pilosa, Asterias rubens, Pandulus montagui, Sertularia cupressina, Balanus crenatus* and *Crangon crangon* (Cefas, 2000)

There have been a series of studies into various aspects of the amphipod *Corophium volutator* in estuaries of the marine site, notably in relation to the abundance of benthic diatoms (e.g. Smith e*t al.,* 1996; Gerdol & Hughes, 1994a,b,). Hughes & Gerdol, (1997) found *C. volutator* to be twice as abundant in creeks and semi-enclosed bays of the Blackwater and Crouch Estuaries, than on the open mud flats. This was attributed to their dispersal behaviour of swimming on the flood tide, which would sweep the amphipods into these areas where the tide rises but does not flow laterally. On the open mud flats, displacement of swimming amphipods by the flood tide further upstream and into semi-enclosed areas would occur. Their dispersal behaviour places *C. volutator* in the creeks and bays within the saltmarsh vegetation, where their bioturbatory feeding habits may be responsible, in part, for the significant loss of pioneer zone vegetation that occurs there (Gerdol & Hughes, 1993).

Borrow dykes, present throughout most of the marine site, are broad ditches behind a sea wall, created when the soil was 'borrowed' or dug out to create the wall itself. These dykes now receive drainage from the land, and excess water is sluiced to the estuary. Many are brackish, the salinity of individual dykes depending on the amount of saltwater ingress from the estuary (via leaky sluices or seepage through sea walls) and many house dense reedbeds which can provide valuable habitats.

Wilson & Wilson (2001) investigated these unusual habitats and found the salinity in approximately half of the dykes sampled (throughout the EMS) was high enough to support marine and specialist lagoon species including several very varied mollusc species. The lagoon seaslug *Tenellia adspersa*, previously only recorded from very few sites in Britain (e.g. Bristol Channel and the Fleet lagoon in Dorset) (Wilson, 2005) was found at five of the sampling sites. This is a very small nudibranch which may be more common but is easily overlooked[10]. Mud snails *Hydrobia* spp were relatively common in the dykes, notably *Hydrobia (Ventrosia) ventrosa* which was widespread throughout the area, *H. neglecta* was recorded in Essex for the first time. Other marine molluscs identified during the survey were *Littorina littorea, Limapontia depressa, Tergipes tergipes, Eubranchus exiguous, Ovatella myostosis, Cerastaderma edule, C. glaucum, Macoma balthica, Abra tenuis* and *Scrobicularia plana. Potamopyrgus jenkinsi* was the most often recoded.

Non-molluscan protected specialist lagoon species were also recorded including Starlet sea anemone *Nematostella vectensis,* Lagoon sand shrimp *Gammarus insensibilis* and the Bembridge beetle *Paracymus aeneus*, an extremely rare water beetle currently known from only two sites in England[2].

Using similar methods to those described for a study on meiofauna (see above section) Cefas have recently carried out a field manipulation experiment, on intertidal mudflats in the Crouch Estuary (Bridgemarsh Island), designed to investigate the

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effects of increased sediment organic content (from 0.9% to 2.8%) and sand content (from 12.0% to 47.0%) on macrofaunal recolonisation and sediment properties on the intertidal mudflat (Bolam *et al*., 2004). They found that the natural macrofaunal communities at the site were numerically rich (35,000 individuals m^2), dominated by tubificid oligochaetes, cirratulids *Tharyx sp*., and at the surface by the gastropod *Hydrobia ulvae*.

In the study, manipulated sediments were placed in plots on the mudflats and colonising macrofauna (and sediments) sampled after 1 week, and 1, 3, 6, and 12 months. Macrofaunal recovery was found to be affected by the increased sand content in the short-term (1 month), whilst increased organic content delayed recovery of total numbers of individuals, species number and diversity.

There was a gradual increase in the abundance of those species present in ambient sediments, rather than a distinct successional sequence. Over 85% of the individuals were represented by five species; these were, (in order of overall abundance) the oligochaete *Tubificoides benedii* , gastropod *H. ulvae*, and the polychaetes *Streblospio shrubsolii, Tharyx sp* .and *Hediste diversicolor.*

H. diversicolor and *H. ulvae* recovered to ambient levels within the first week of the experiment. Data analysis showed that community assemblages in the low organic content sediments had recovered after 12 months, while high organic content treatments were still significantly different to those of the controls. The authors proposed that the opportunistic nature of the ambient assemblage and dispersive nature of the dominant species, explained the relatively rapid recovery and lack of successional sequence observed in all sediment types.

The placement of dredged material on intertidal mudflats can, in theory, fulfil two goals; ‘beneficial use’ of the dredged material, since ocean disposal of industrial waste and sewage sludge has been phased out, and ‘intertidal recharge’ to create or augment mudflats and saltmarshes.

Results from this study indicate that the organic content of fine-grained dredged material must be taken into account during the licensing process of such schemes, if invertebrate recolonisation is a concern, as variations in organic content can lead to altered communities with potential implications for the functioning of the created habitat (Bolam *et al*., 2004).

Fish

There are several fisheries within the EMS, which are economically important to the area. The largest inshore fishing fleet between Lowestoft (Suffolk) and Brixham (Devon) is based in West Mersea, and smaller fleets are also based in the Blackwater, Colne and the Crouch & Roach estuaries (Pawson *et al.,* 2002).
The largest catches are of sprat and herring in winter and dover sole in spring. Other fish caught at different times of year include cod, whiting, plaice, dab, flounder, bass, grey mullet, thorn back ray, eels and smelt (Blackwater Management Plan, 1996). Fyke-netting activity in the estuary has declined due to a scarcity of eels (Pawson *et al.,* 2002).

The Colne and nearby creeks are fishing grounds for sprat, sole, plaice, ray, bass, mullet, eels, cod, whiting and herring. In the Crouch and Roach fisheries include herring, sprats, mullet (mostly golden-grey), bass, rays and sole. Maplin Sands provides a popular fishing ground for both trawlers and netters. In the summer months the Blackwater Estuary is an important nursery ground for various fish including plaice, sole, turbot, cod and notably, bass. Bass populations are centred around Bradwell, and the area around the warm effluent discharged from Bradwell power station remains closed to bass fishing despite the facility having ceased operating early in 2002 (Pawson *et al.,* 2002).

The Blackwater was one of estuaries included in a study into patterns of bass (*Dicentrarchus labrax*) recruitment from nursery areas around England and Wales (Pickett *et al.,* 2004). The aim of the study was to clarify the dispersal patterns of juvenile and adolescent bass and reveal the potential benefits of protecting juvenile bass by closing nursery areas. Bass were tagged and released in 11 nursery areas around England and Wales between 1988 and 1994.

This study was partly prompted by concerns that the main benefits of the UK bass management measures might profit offshore fleets (Pickett *et al*., 1995). This concern was heightened by reports of large catches of bass (boosted by recruitment of the exceptional 1989 year-class) taken by offshore pair-trawlers (ICES, 2003). This study found no evidence to support this presumption, since results from recapture showed that the majority of tagged fish were recruited to the local fisheries near to the nursery areas in which they were tagged. However, the Blackwater and other southeastern estuaries appear to be a net "exporters" of recruiting bass, and local fisheries therefore benefit the least from recruitment of bass derived from other areas (Pickett *et al*., 1995).

The Blackwater Estuary is also particularly important for a spring-spawning population of herring *Clupea harengus*. Various aspects of the herring larvae, and the fishery in the estuary have been studied over the last four decades, notably in the 1980's in relation to Bradwell Nuclear Power Station and the effects to the population of entrainment, and thermal pollution (Coughlan, 1980: Coughlan *et al.,* 1980; Dempsey & Henderson, 1981; Bamber & Henderson, 1981). More recently, studies have focussed on the ecology of, and management options for, the spawning herring stock.

The inshore migration of the Blackwater herring begins in early October, with shoals of fish concentrated within 10 miles of the east Anglian coast in preparation for the following spring spawning. The principal spawning site is Eagle Bank at the mouth of the Blackwater estuary and limited spawning has taken place at Stone, opposite Osea Island (Fox, 2001).

Fox *et al.,* (1999a) carried out spring surveys of herring larvae in the estuary between 1993-1997 and found high concentrations centred around Osea Island toward the end of March 1993. Shortly afterwards, numbers declined and low concentrations of larvae were reported toward the seaward end of the estuary. In early April, another hatch occurred and larvae continued to be caught until mid May when very low catches determined the end of sampling. Sampling patterns were standardised to coincide with similar tidal conditions for the subsequent years' surveys, although spawning and distribution patterns only varied by a few days (Fox *et al.,*1999a)

A further study (Fox *et al.,* (1999b) focussed on the feeding ecology of herring larvae in the Blackwater Estuary. Findings indicated that the larvae showed preference for *Acartia* spp, (early and adult stages) and gastropod larvae whilst copepod nauplii were negatively selected. Concentrations of potential prey in the water were low compared with literature estimates of the requirements of herring larvae, and tidally-induced turbidity in this estuary reduced the effective water depth available for visual feeding by herring larvae by up to 50%. However, no clear effect of this on feeding success could be shown and the authors considered that high levels of tidally-induced turbulence may be beneficial in enhancing the encounter rates between the larvae and their prey.

Later studies examined long-term trends in stock recruitment of the Blackwater herring in relation to larval production (Fox, 2001). During the 1960s, pair trawling over the spawning grounds had reduced the stock bringing it to an all-time low in 1977. This resulted in closure of the herring fishery for three years. Subsequently stocks continued to recover and in 1992 yielded the highest spawning stock biomass (SSB) recorded. This recovery has been partly attributed to a decline in consumer interest in herring, resulting in low economic value of the stock and thus reduced fishing effort (Roel *et al.,* (2004). The study carried out between 1994-1997 (Fox, 2001) was prompted because the high SSB generated relatively weak year-classes. The study compared data for larval production, growth and loss rates, with historical estimates from 1979. Fox (2001) reported that estimated levels of larval production were higher than those of 1979, and post-hatching growth rates were similar. However, mortality rates appeared to be greater. An inverse relationship was found between estimated survival of larvae from hatching to 2 years, and the level of total larval production. These results suggested that year class strength was density-dependant and/or affected by environmental processes that operated after hatching, rather than at the egg stage.

For several years, the herring fishery has been regulated by an annual total allowable catch (TAC) based on results from an age-structured stock assessment model, which were adjusted each year. The annual assessment relied on data provided by a scientific survey, because of the decline in fishing effort, and sporadic nature of commercial catch information. Given the low economic value of the herring stocks, the possibility of setting a fixed TAC (thus negating the need for annual scientific surveys) and the consequences of such a strategy were investigated. Roel *et al.,* (2004) used modelling to examine various management options for the Blackwater herring in order to ensure sustainability. Four options were considered; annual revision (maintaining the *status quo*), fixed TAC for 3-5 years, fixed TAC for 3-5 years with an additional constraint that limited maximum TAC change after the fixed period, and fixed 20-year TAC. The (modelled) outcomes indicated that a fixed 3-year TAC with a limit of 40% variability after this period appeared to be the most appropriate choice for sustainable stock management. This option was forecast to keep the stock within safe biological limits and compared well with the existing approach of annual TAC revision in terms of yield and risk, and was thus recommended (Roel *et al*., 2004)

Birds

The mudflats and salt marshes in and around Essex Estuaries are of great ecological value for feeding and roosting birds. Full description of their distributions is beyond the scope of the current report. Table 5 lists bird species of European importance that used the component estuaries for breeding, roosting or overwintering, at the time the component SPAs were classified.

Table 5. Birds of European importance in Essex Estuaries European Marine Site at the time the component SPAs were classified (from EN, 2000)

Species	Common name	SPA
Sterna albifrons	Little tern*	1,2,5
Pluvialis squatarola	Grey Plover	1,2,4,5
Branta bernicla bernicla	Dark-bellied Brent Goose	1,2,3,4,5
Calidris alpina	Dunlin	1,2,4,5
Limosa limosa	Black-tailed Godwit	1,2,4
Phalacrocorax carbo	Cormorant	1,2
Tringa totanus	Redshank	1,2,5
Tadorna tadorna	Shelduck	1,2,5
Anas crecca	Teal	1
Bucephala clangula	Goldeneye	1,2
Charadrius hiaticula	Ringed plover	1,2,5
Numenius arquata	Curlew	1,2,5
Cygnus olor	Mute Swan	2
Calidris alba	Sanderling	2
Calidris canutus	Knot	4,5
Limosa lapponica	Bar-tailed Godwit	4,5
Sterna sandvicensis	Sandwich tern*	5
Sterna hirundo	Common Tern*	5
Recurvirostra avosetta	Avocet*	5
Haemotopus ostralagus	Oystercatcher	5

[1] Blackwater [2] Colne. [3] Crouch & Roach [4] Dengie [5] Foulness
*Breeding populations of internationally important Annex 1 species

Ravenscroft & Beardall, (2003) examined the importance of freshwater flows over estuarine mudflats for wintering waders and wildfowl on the mudflats of Essex Estuaries, including the Blackwater. They found that numbers of shelduck, wigeon, pintail and redshank were present around flows on most counts on all 3 estuaries in the study. Grey plover were recorded in consistently greater numbers around flows, but the small populations during most counts did not permit statistical analysis (likewise brent goose, ringed plover and black-tailed godwit). Counts revealed that some species, such as redshank, occurred at high densities around freshwater flows throughout the tidal cycle. There was also a large turnover of birds using flow areas - flocks of waders such as knot alighted briefly in flows before dispersing. Some species such as ducks and curlew vacated mudflats 2–3 h after low water and would gather around the mouth of flows. Others, such as ringed plover used flows more as high water approached, and were joined by other species such as lapwing (*Vanellus vanellus*) and gulls. Overall, the findings indicated that large proportions of bird populations of several species occurred around these flows. The attraction of freshwater to waterbirds, especially ducks and geese, is considered to be largely for preening and drinking, close to their feeding grounds.

Ravenscroft & Beardall, (2003) concluded that freshwater abstraction could impact large numbers of birds and that this impact should be considered in granting proposals to alter or remove discharges within estuaries. Reduced freshwater input was identified as a factor in the Blackwater Estuary (BTO 2004) (see below).

The British Trust for Ornithology carried out a (level 1) review of species trends in SPAs over the 5, 10 and 25 year time periods up to 2000 using data collected as part of the Wetland Bird Survey (WeBS) (Armitage *et al* 2002). SPAs where species are declining at a rate of greater than 25% over a specified time period when the larger-scale regional or national trends indicate stable or increasing population sizes are targeted as being of concern. Population declines of between 25% and 50% are flagged as 'Medium Alerts' and declines of greater than 50% as 'High Alerts'. Alerts are intended as advisory measures triggering further investigation.

The report, produced for the Environment Agency, English Nature and the Countryside Council for Wales summarises statistics for 17 evaluated species in the Blackwater Estuary SPA, 10 in the Colne Estuary SPA, 1 in the Crouch & Roach Estuaries, 8 in the Dengie SPA and 13 in Foulness SPA.

Table 6 lists the species, and the relevant component SPAs, for which alerts were triggered. In total there were 8 high alerts and 9 medium alerts for species throughout the EMS. Specific details regarding the time-span for which these alerts were issued, for individual species/estuaries, are available within the level 1 Report account (Armitage *et al* 2002).

Table 6. Species and relevant component SPAs for which alerts were triggered and the level of the alert (from Armitage *et al.,* 2002)

	Common name	**SPA***
High Alert	Great crested Grebe	1, 2,4
	Goldeneye	1
	Shoveler	1
	Cormorant	2
	Dark-bellied Brent Goose	**2**
	Shelduck	2
	Black-tailed Godwit	2,5
	Knot	**5**
Medium Alert	Great Crested Grebe	1,4
	Cormorant	1,4
	Dark-bellied Brent Goose	**1**,2,5
	Teal	1
	Ringed Plover	**1**,2
	Shelduck	2, 5
	Curlew	5
	Bar-tailed Godwit	5
	Knot	**5**

[1] Blackwater [2] Colne. [3] Crouch & Roach [4] Dengie [5] Foulness
*Species for which a site qualifies under EC Birds Directive (79/409/EEC) by supporting populations of international importance (usually > or equal to 1% of a species' international flyway population) are highlighted in bold in the SPA column.

As a result of the level 1 review, Armitage *et al*., (2002) identified two of the component estuaries in the marine site, the Blackwater Estuary SPA and the Colne Estuary SPA, as requiring further investigation. Results of these investigations (level 2) have recently been published and describe distributions and changes for the alerted species and look at possible factors in the population decline (BTO, 2004).

To briefly summarise some principal findings of this report: Two of the medium alerts triggered in the Blackwater Estuary were for species that are internationally important in the SPA in winter – Dark-bellied Brent Goose and Ringed Plover.

Amongst possible reasons for the decline of these and other bird populations in the estuary, the reduction in freshwater inputs to the estuary due to increased water abstraction[11] was considered a plausible factor (BTO, 2004). Ravenscroft & Beardall (2003) reported close links between the densities of estuarine waders and wildfowl and freshwater flows on several estuaries in East Anglia including the Blackwater. .

Perhaps even more relevant are changes to waste water treatment that have occurred in the Blackwater: improvements were made to the sewage discharges at West Mersea in 1992 and 1996, which may be a factor in the five-year Alerts triggered for five species (Great Crested Grebe, Cormorant, Dark-bellied Brent Goose, Teal and Shoveler) and the 10-year Alerts for Goldeneye and Ringed Plover. Water quality improvements predictably reduce the organic loading and consequently the available nutrients for estuarine invertebrates. This may have had a knock-on effect in that the availability of some prey items for waterfowl and waders may also be reduced, causing alterations to the feeding numbers and distributions of certain species.

Changes to waste water treatment are also considered to be important in the Colne Estuary: sewage effluent discharge (e.g. Colchester) has contributed to the hyper-nutrification of the Colne (Thornton *et al*. 2002), and changes in water operations may be relevant to waterbird declines, either through direct pollution or by altering the biotic environment. EA information indicates that effluent output almost doubled at Thorrington and West Mersea in 1989, although, as noted above, improvements to effluent quality occurred in 1992 and 1996 at West Mersea. Secondary treatment was introduced at Jaywick in 2001, although these changes would not affect the Alerts considered here. Discharges of organic matter from Tiptree to Salcott Creek were reduced in 1988. Unfortunately, less information is available regarding discharges at Colchester and the inner estuary. Thus, BTO (2004) consider it possible that sewage changes in 1988/89 could have contributed to 10-year declines of Cormorant, Dark-bellied Brent Goose, Shelduck and Ringed Plover in the Colne Estuary, particularly as these species were found in large concentrations in the vicinity of Mersea Island.

An overall reduction in organic loading at Tiptree may have resulted in the area becoming impoverished in terms of waterbird prey, and a similar effect could be involved in 5-year declines, due to the 1992 and 1996 improvements at West Mersea.
Other factors that may have contributed to bird population declines on the Blackwater and the Colne include disturbance (visitor pressure and an increase in general disturbance erosion caused by cycling and walking along the Blackwater Estuary, and

[11] Water is abstracted during the winter for summer spray irrigation for the farmland adjacent to the Blackwater Estuary

a gradual increase in water-based leisure pursuits, such as jet skiing, and a similar increase in moorings in the Colne), the creation of new wetlands (notably Tollesbury Wick reserve on the Blackwater, where bird counts are carried out from the edge, potentially missing birds located in the central part of the reserve), and the use of adjacent sites. With regard to the latter, it should be noted that there are declines in the bird populations of both estuaries, although movements between the coastal feeding areas of the Blackwater Estuary and Abberton Reservoir are common for some species i.e. Great Crested Grebe, Cormorant, Shoveler and Goldeneye which may, in part, explain the short-term declines reported for these species.

Changes in surrounding land-use patterns (the increasing use of farmland) could partly explain the declines in Dark-bellied Brent Geese on the Blackwater Estuary. Cold weather influxes may be an important factor in the five, ten and 25-year Alerts triggered for Goldeneye.

More general factors considered include national and regional declines: annual changes in Ringed Plover populations in the Blackwater over the 25-year period were related to annual changes in the species' national population (as recorded by WeBS). Also, there has been a recent decline in numbers of Ringed Plover wintering on non-estuarine coasts in the UK (Rehfisch *et al*., 2003). A regional decline noted for this species might be related to the impacts of concurrent changes to waste water discharges on invertebrate prey. Treatment to the majority of discharges within the region was upgraded during the 1990s as a result of the UWWTD. Changes in Shelduck numbers on the Colne Estuary over the 5-year period were related to annual changes in the species' regional population (as recorded by WeBS).

In the longer term, erosion of saltmarsh and mudflats, largely as a result of coastal squeeze and hard flood defence systems, has resulted in the loss of suitable feeding and roosting sites for wildfowl and waders. This may also affect waders, notably the Ringed Plover and it is possible that this species' decline could, in part, be explained by the loss of this habitat, as birds are forced to move elsewhere to find suitable feeding and / or roosting areas (BTO, 2004).

Table 7 summarises the factors that are potentially related to declines in the Blackwater and Colne Estuaries. However it is probable that the declines result from a combination of these (often inter-related) factors rather than any one in isolation.

Table 7. Factors that may be related to the decline of bird populations in the Blackwater and Colne Estuaries (from BTO, 2004).

	Blackwater Estuary	Colne Estuary
Use of adjacent sites	●	●
Changes in adjacent agricultural land use	●	●
Reduced freshwater inputs[1]	●	
Saltmarsh loss / other erosion	●	●
Sediment changes		●
Changes to waste water treatment[2]	●	●
Other human disturbance	●	●

[1]waders and some wildfowl may be affected by reduced invertebrate food supplies
[2] wildfowl may be affected by the reduction of directly edible matter from discharges and waders and some wildfowl may also be affected by reduced invertebrate food supplies
NB Factors that may be directly related to EA consents are highlighted in bold.

Interestingly, eutrophication and *Enteromorpha* growth were not listed as possible factors in the decline of bird populations, although the problems of algal mats, resulting from eutrophication, have been the subject of much research. Previous studies have linked growth in algal mats to a loss of waders, as such mats restrict the availability of invertebrate prey (Cabral *et al*. 1999, Raffaelli *et al*. 1999, Lopes *et al*. 2000, Lewis & Kelly 2001). It might therefore be useful to study the extent to which areas covered by *Enteromorpha* are avoided by waterbirds, particularly on the Blackwater Estuary, where the problem appears to greatest (see section 6.2).

Perhaps also related to the decline in the Brent geese population is the loss of *Zostera* beds in the marine site (see section on seagrass, above). Since the occurrence of wasting disease and the consequent decline of *Z. marina* beds, the relative importance of the different *Zostera* species in Brent geese diet has shifted. Although *Zostera noltii* has replaced *Z. marina* as the preferred food, and is considered by Davidson and Hughes (1998) to provide the main source of energy for Brent geese overwintering in Britain, *Z. noltii* is also declining. Conversely, Burton (1961) studied dark-bellied Brent geese on the Essex coast in the late 1950s and early 1960s and found that they fed almost entirely on *Z. noltii* <u>and</u> *Enteromorpha*. Charman (1975) also found that when Brent geese had exhausted the *Zoster*a stock along the Essex coast, they had to move onto less preferred food sources, including *Enteromorpha* and saltmarsh plants, and then onto less traditional food sources such as inland pastures and winter cereals. This may indicate that although seagrass beds are now limited to two areas of the EMS, alternative food sources are generally available for this species.

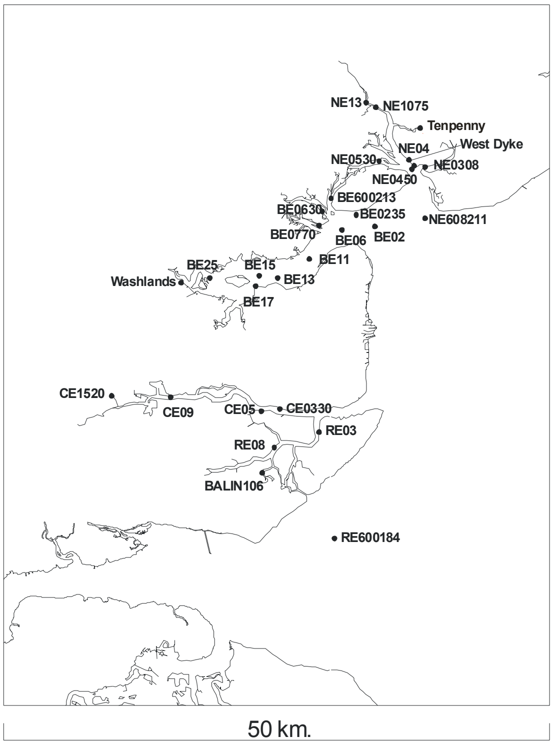
Amongst recommendations for further research, BTO (2004) include:

- Quantification of temporal and spatial changes in water quality, particularly concerning discharges from West Mersea, Tiptree, Tollesbury, Bradwell-on-Sea and Maldon
- Quantification of any changes in water temperature and quality following the decommissioning of Bradwell Power Station in 2002
- An assessment of the effects wetland creation and management of Tollesbury Wick Nature Reserve on the numbers and distribution of Dark-bellied Brent Geese
- Investigations into the effects of dredging in the estuary on the numbers and distributions of individual species (compared with non-dredged estuaries).
- Investigations into the threat posed by re-suspension of TBT from sediments through coastal erosion, soft-engineering schemes and recreational disturbance (through ecotoxicological studies and also through the quantification of any changes in the concentration of this chemical, particularly in the vicinity of the three managed realignment schemes and in areas where recreational disturbance is at its highest such as Bradwell, Maldon, Tollesbury, St Lawrence and West Mersea).
- Quantification of organic loading changes for the Colne (Thorrington, Tiptee, West Mersea, and perhaps most importantly, Colchester).
- Further consideration of the effects of water abstraction issues and reduced freshwater flow

Thus BTO, (2004) have considered various factors in relation to the observed declines in bird populations in the EMS.

6. TRENDS IN WATER AND SEDIMENT QUALITY

The main tidal water sites sampled for water quality, and the number of times visited[12] by EA between 2000-5, are shown in Figure 5.



Code	Site	n (2000-5)
Colne		
NE0308	BRIGHTL'SEA CK HARD	65
NE04	OFF BATEMANS TOWER	66
NE0450	COLNE BATEMANS TOWER	227
NE0530	PYEFLEET CHANNEL	48
NE1075	COLNE WIVENHOE	59
NE13	COLNE ROWHEDGE FERRY	66
NE608211	BEACH HEAD BOUY	48
Blackwater		
BE02	SE OF WEST MERSEA	9
BE0235	WEST MERSEA BEACH	114
BE06	MERSEA QUARTERS	9
BE0630	SALCOTT CHANNEL	48
BE0770	TOLLESBURY N CHANNEL	48
BE11	B'WATER SE OF TOLLESBURY	55
BE13	THE STONE	8
BE15	GOLDHANGER CREEK	59
BE17	B'WATER STANSGATE	66
BE25	B'WATER DECOY POINT	22
BE600213	STROOD CHANNEL	48
Dengie		
NE608211	DENGIE BEACH HD BY	48
Crouch		
CE0330	CROUCH BURNHAM YC	67
CE05	CROUCH ESSEX MARINA	68
CE09	CROUCH FAMBRIDGE	60
CE1520	FENN CREEK	66
Roach		
BALIN106	BARLING HALL CREEK	7
RE03	ROACH MONKTON QUAY	63
RE08	ROACH PAGLESHAM	67
Foulness		
RE600184	N SEA MAPLIN SAND	46

Figure 5. Tidal waters and freshwater inputs for which sampling data (2000-2004) is available (data source EA).

6.1 Toxic contaminants

6.1.1 Metals

Water

Results are discussed here on a metal by metal basis, based on EA statistics for tidal waters (2000-2005).

Arsenic

The EQS for As in fresh waters is 50µg l^{-1} (annual average, dissolved - AD). The mean values for upper-most tidal sites of the Essex Estuaries indicate relatively little

[12] Not every determinant measured on each occasion.

enrichment with As compared with background values for As in freshwaters (1-2 µg l^{-1}); riverine inputs are therefore unlikely to cause excedence in the EMS.

The pattern of dissolved As in tidal waters across the EMS is shown in figure 6. Averages were scarcely above background and were invariably below the EQS for tidal waters (25 µg l^{-1} AD). Occasionally, higher individual concentrations were recorded - as at Paglesham on the Roach and Stansgate on the Blackwater – but on this evidence, for the sites monitored, dissolved As appears to pose little direct threat (risk of EQS failure low).

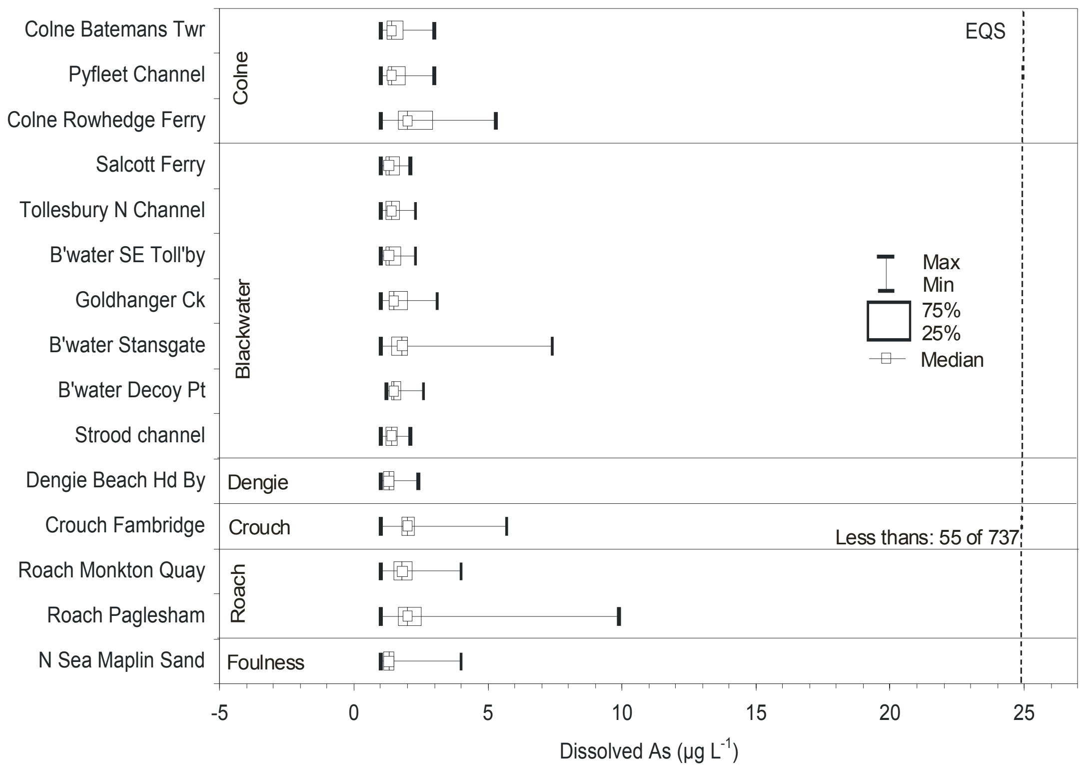


Figure 6. Concentrations of dissolved As (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

Cadmium

The pattern of dissolved Cd in tidal waters of the EMS is plotted in figure 7.
The EQS for Cd in fresh waters is 5µg l^{-1} ('total' Cd). In the upper tidal section of the estuaries median values for dissolved Cd were below this threshold by almost two orders of magnitude. Riverine inputs are unlikely to cause excedence in the EMS, though occasional high values have occurred - as at Blackwater Decoy Point. It has been suggested that in addition to discharges, run-off and drainage from agricultural land may contain Cd due to its association with some forms of phosphate fertilizer (Chang *et al*., 2001).

Elevated Cd concentrations were occasionally measured at more saline sites, including Foulness and Dengie Beach Head Buoy, and may reflect greater solubility of Cd in marine conditions due to complexation with chloride ions. However, median

Cd concentrations at all sites were invariably below the EQS for tidal water (2.5 µg l^{-1} dissolved); half of all tidal water values were below DL . For the sites monitored, dissolved Cd appears to pose little direct threat, and the overall risk of EQS failure is considered low.

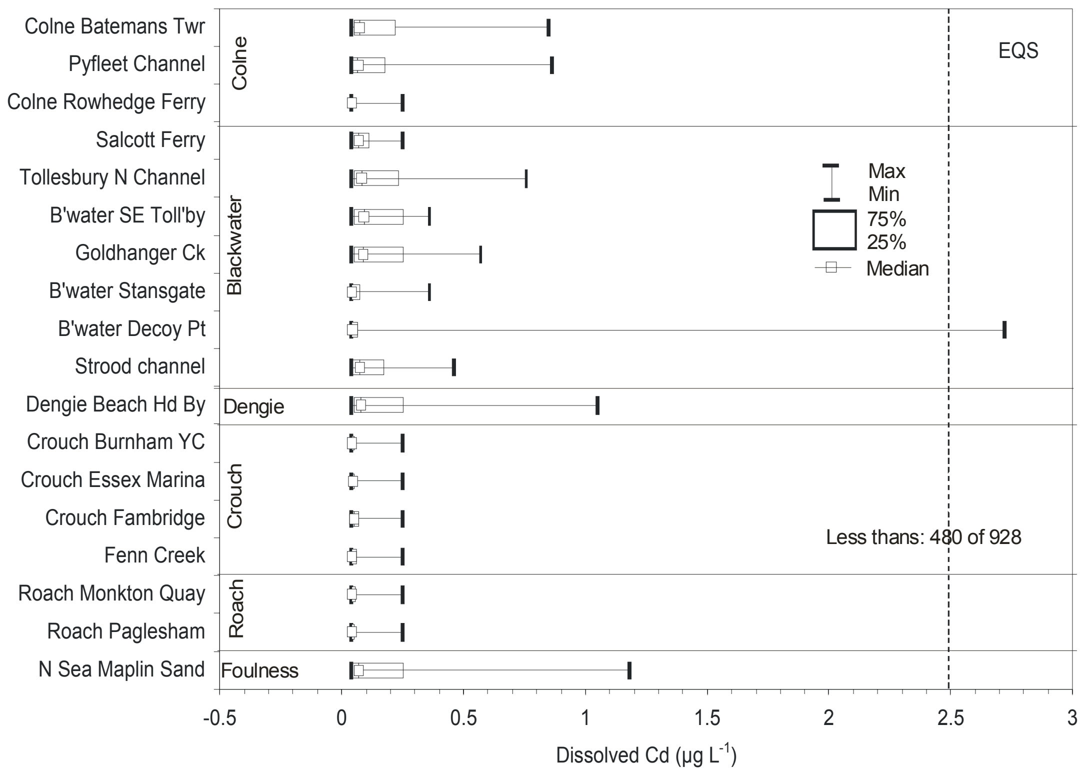


Figure 7. Concentrations of dissolved Cd (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005; proportion of values <DL indicated).

Chromium.

The EQS for Cr in fresh waters suitable for salmonids ranges between 5 and 50µg l^{-1} depending on hardness. Median concentrations in upstream tidal waters sites in the Essex EMS are not exceptional (figure 8) therefore substantial inputs from rivers (above the tidal limit) are not a likelihood.

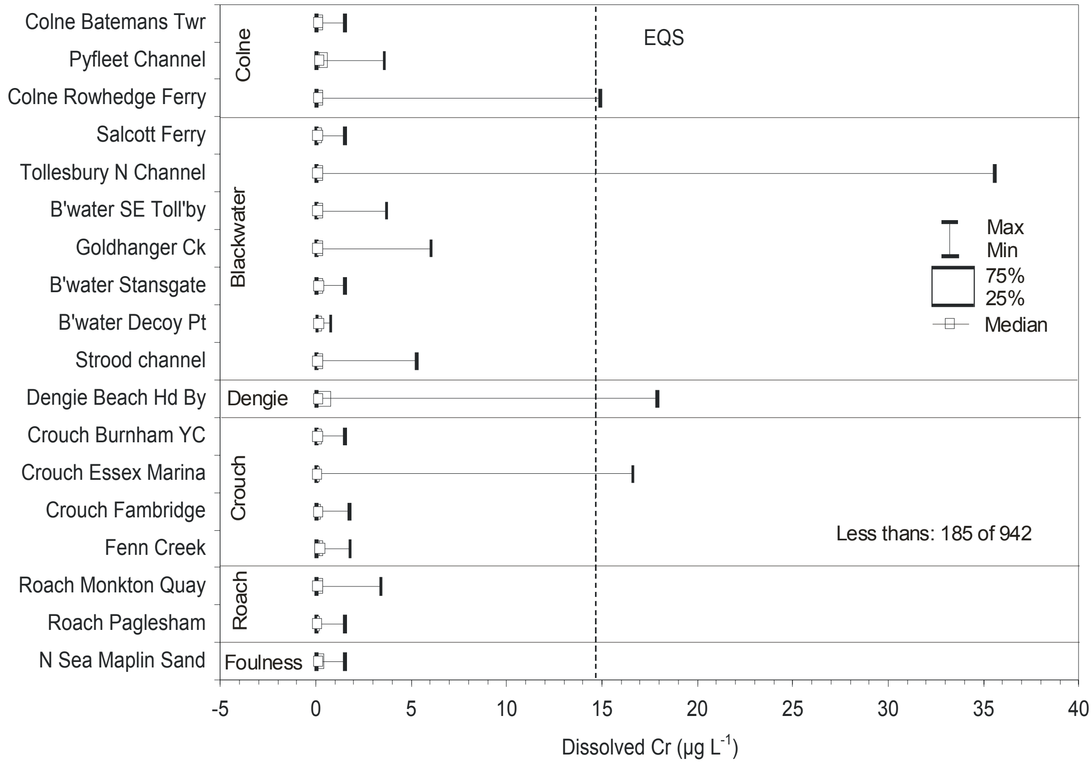


Figure 8. Concentrations of dissolved Cr (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

The pattern of Cr in estuarine waters (figure 8) indicates that median values for all sites were lower than the EQS (15 µg l^{-1}) by more than an order of magnitude and would comply with the standard. Some 20% of values were below detection limits. Occasional elevated levels of dissolved Cr, above the standard, were recorded at several sites (for example Tollesbury N Channel). Although there is little evidence to suggest there is a risk of failure of the EQS, or that Cr concentrations in tidal waters would be acutely toxic, some forms of Cr cause deleterious effects in invertebrates and fish at concentrations close to the EQS for saline waters. Cr is also considered genotoxic. Investigations might be considered to assess whether high values are possible transient sources or anomalies.

Copper

The EQS for Cu in freshwater is in the range 1 - 28 µg l^{-1} depending on hardness; that for saline waters is 5 µg l^{-1}. At the majority of sampling sites ~2-3 µg l^{-1} dissolved Cu appears to be the normal (Figure 9). The risk of compliance failure at the majority of tidal waters monitoring sites appears to be low. Elevated levels above the EQS benchmark are occasionally recorded, notably in the Crouch and Blackwater (highest value in the Strood Channel). Antifouling paints and sediments may be a source of Cu in addition to some outfalls.

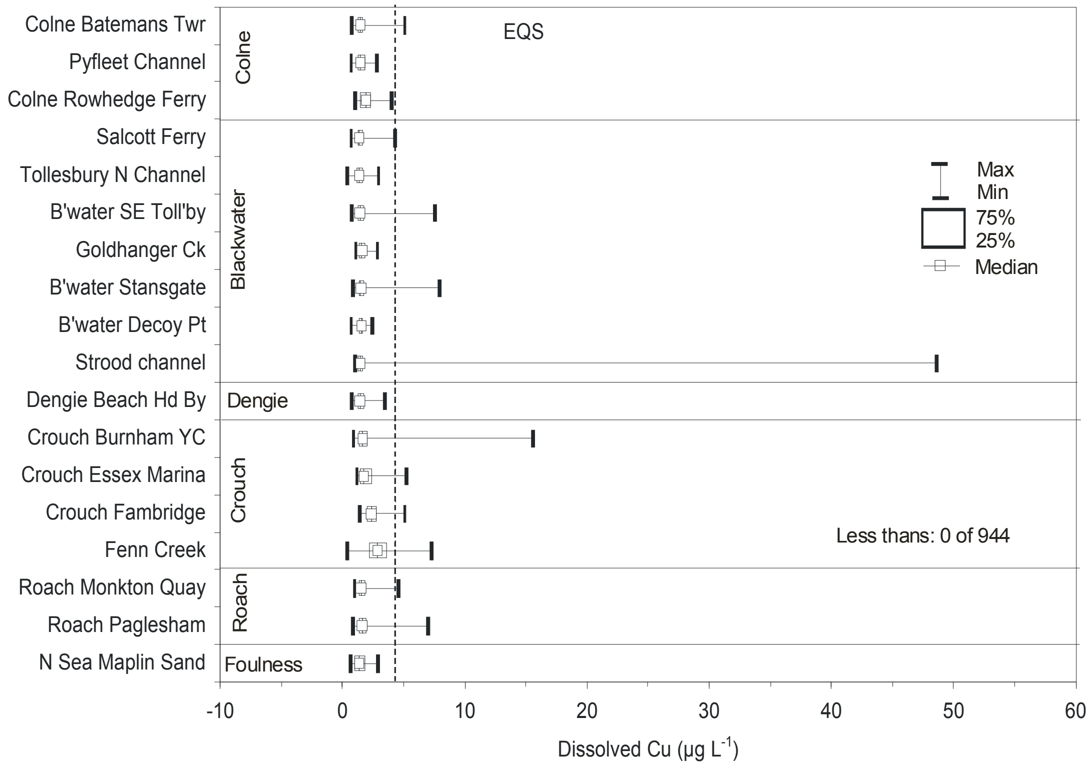


Figure 9. Concentrations of dissolved Cu (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

Comparison with 1992-1996 EA data from a similar suite of sites suggest the most recent picture represents an improvement in overall conditions, since values exceeding the Cu EQS were frequent in the upper part of the Crouch ten years ago, in addition to sites in the upper and lower Colne and Blackwater (Matthiessen *et al.*, 1999). In total 21.7% of site/year combinations failed to meet the EQS between 1992-1996. Sites where Cu concentrations exceeded EQS and estimates of inputs from major sources are shown in Figure 10(reproduced from Matthiessen *et a*l., 1999).

Loadings estimates for Cu to the Essex Estuaries were also carried by Matthiessen *et al* (1999), plotted in figure 11. Although these were approximations made on data collected a decade ago they indicated that boating and shipping activity were likely to be the dominant source, and dockyards in the Colne may have dominated inputs in this estuary. STWs tended to be much less significant for Cu and were greatest in the Blackwater. Riverine inputs were estimated to be small, though no accurate information was available for the Crouch and Roach. It was noted that while Cu concentrations in the Bradwell power station discharge were unlikely to be very high, its sheer volume could have contributed to EQS excedences sometimes observed in the vicinity of West Mersea. The highest Cu concentration in the current data set (~50 µg l^{-1}) was in a sample from the Strood Channel to the north of West Mersea.

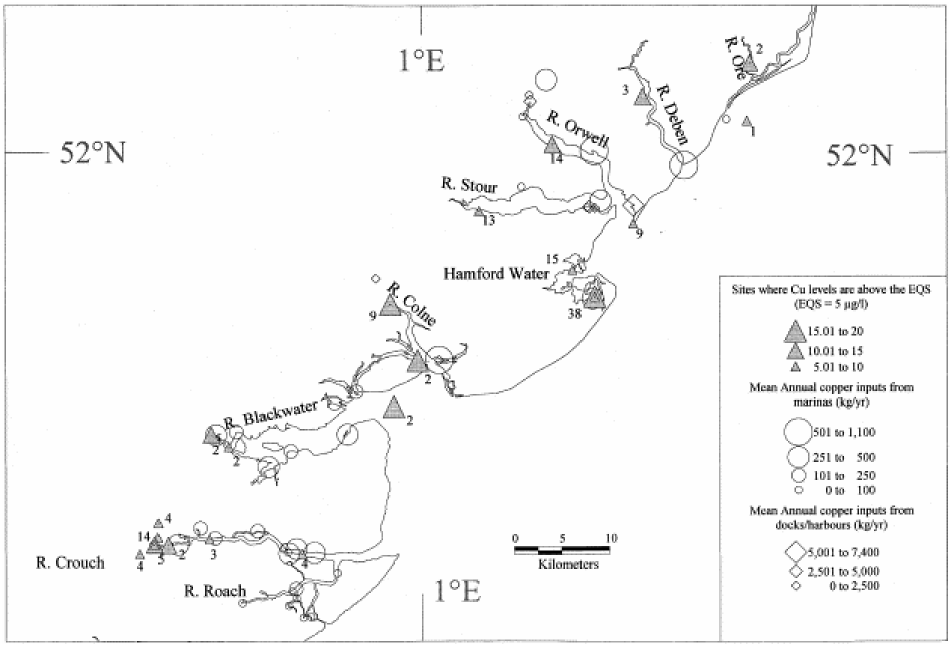


Figure 10. Mean annual copper inputs from marinas and docks (kg Cu/yr), and mean annual copper concentrations (1992-1996) at sites that exceeded the EQS (5 µg Cu l^{-1}). Values attached to the triangular symbols for concentrations indicate the number of observations which contributed to the respective annual means. (Reproduced from Matthiessen et al., 1999, with permission, Elsevier).

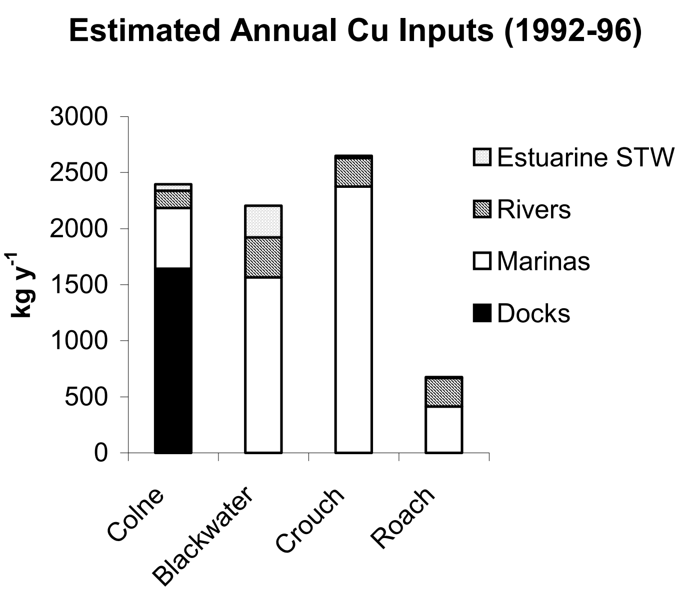


Figure 11. Estimated annual Cu inputs to the Essex Estuaries for the period 1992-1996 (plotted from data in Matthiessen et al., 1999). Riverine inputs for the Crouch and Roach are estimates, based on an average for the other two rivers.

More comprehensive discharge data and evaluation are needed to assess possible current influences on estuarine water quality and sublethal impacts on biota, since Cu is one of the most toxicologically significant of metals.

Iron

The EQS for dissolved Fe in fresh and seawater is 1000µg l^{-1}.

Because of the relatively small number of data for this parameter, the pattern of dissolved Fe concentrations in tidal waters of the EMS is difficult to discern (Figure 12). Median values for the three sites sampled are <10 µg l^{-1} - well below the EQS (1000 µg l^{-1}) - and half the analyses are below detection limits (hence 'Low Risk' of non-compliance). Despite the limited data set, it is unlikely that Fe would represent a threat to marine biota at the site. Iron is not a listed substance, is relatively insoluble in sea water, and tends to reside in particulate and colloidal phases.

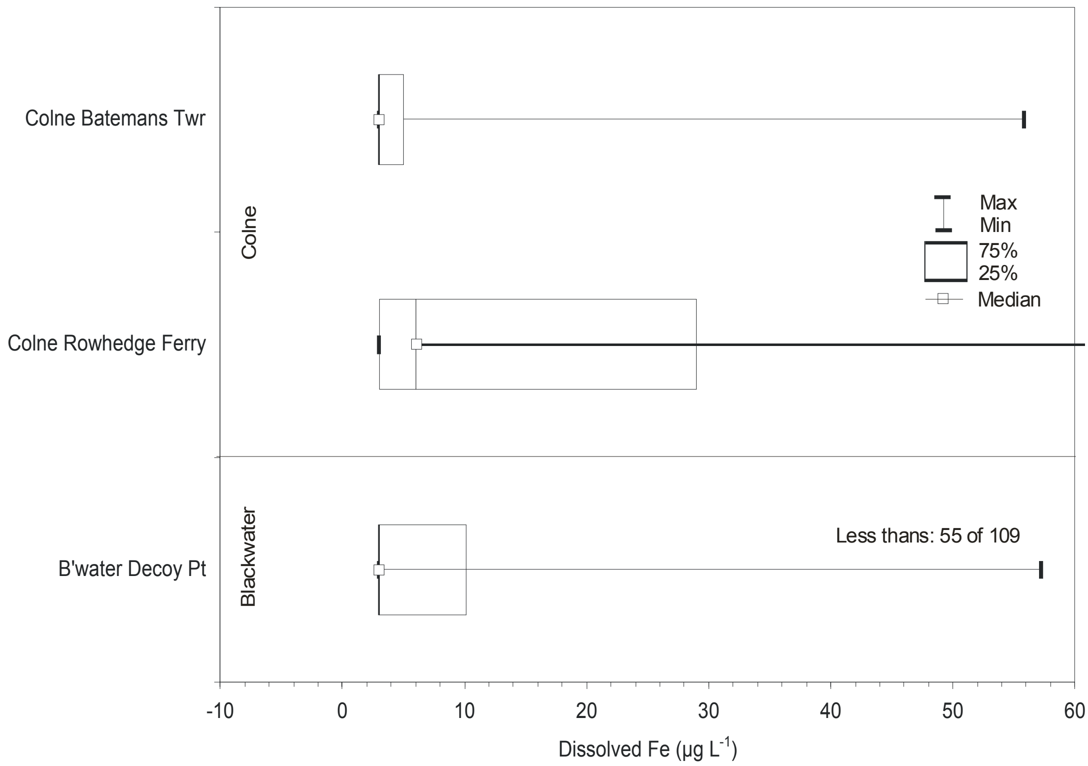


Figure 12. Concentrations of dissolved Fe (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

Only one freshwater source of Fe was offered in the EA database, that into the upper Blackwater from 'WASHLANDS AT OUTLET AND POINT C' with an average 1739 µg l^{-1} – above the EQS. Manganese concentrations were also elevated at this location (176 µg l^{-1}): there are two EQS for Mn under the Surface Water Abstraction Directive (SWAD, category A1 and category A2) of 17.5 and 35 µg l^{-1}, respectively. This input of metals may require further investigation.

Nickel

The distribution of dissolved Ni in tidal waters suggests little in the way of significant freshwater loadings, though some inputs from the Rivers Colne and Crouch is indicated by the elevated median values at Rowhedge Ferry and Fenn Creek, respectively (Figure 13). There is little evidence of anthropogenic contribution to dissolved Ni levels at sites further downstream. Medians are below the EQS for saline water (30 µg l^{-1}). Maximum values are also well within EQS, though the occasional value at Fenn Creek comes close. The risk of non-compliance is 'Low'. Very few of the data involve detection limit-derived values.

On this evidence it is unlikely that Ni would represent a threat to marine biota at the site, though the source of Ni to the upper Crouch and Colne may be worth investigating.

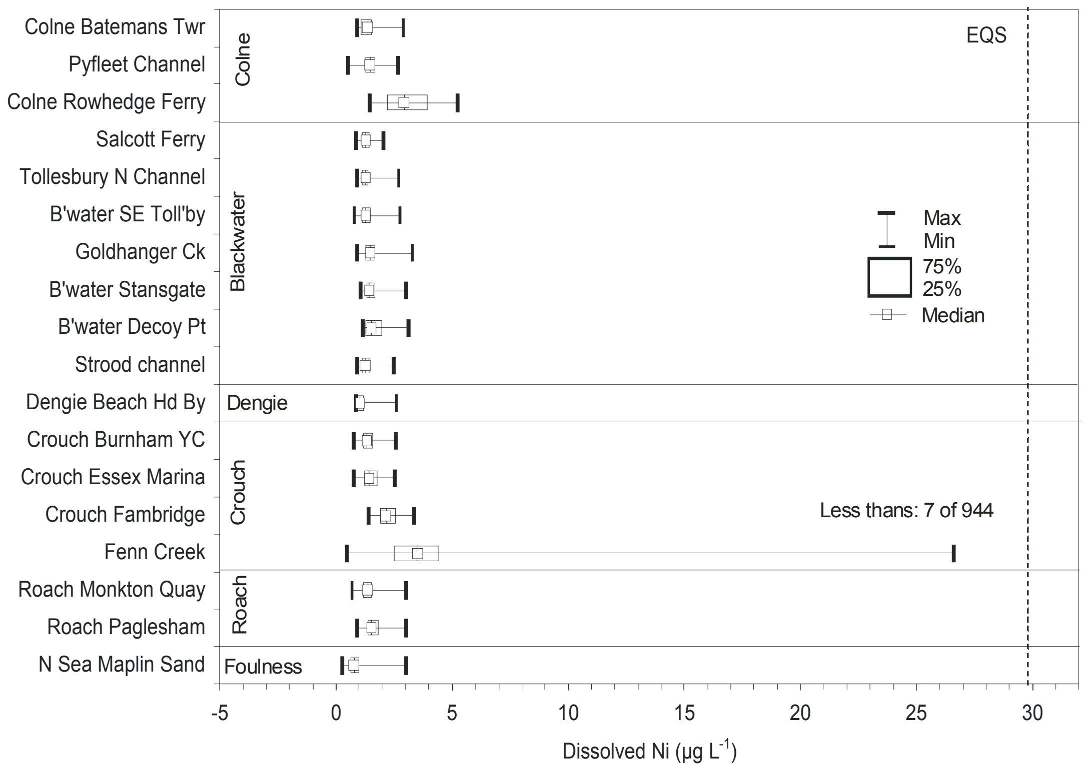


Figure 13. Concentrations of dissolved Ni (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

Lead

Median values for dissolved Pb in tidal waters (figure 14) do not exceed ~1µg l^{-1} (generally more than two orders of magnitude below the EQS of 25 µg l^{-1} in saline waters, with ~20% of values below DL). There is little indication of any consistent anthropogenic contribution to EMS, either directly from point sources, or via rivers: occasional 'outliers' are observed, in the Roach and, notably, at Burnham, which may be worth further investigation. Overall, however, the characterization of risk of non-compliance within the EMS is 'low'.

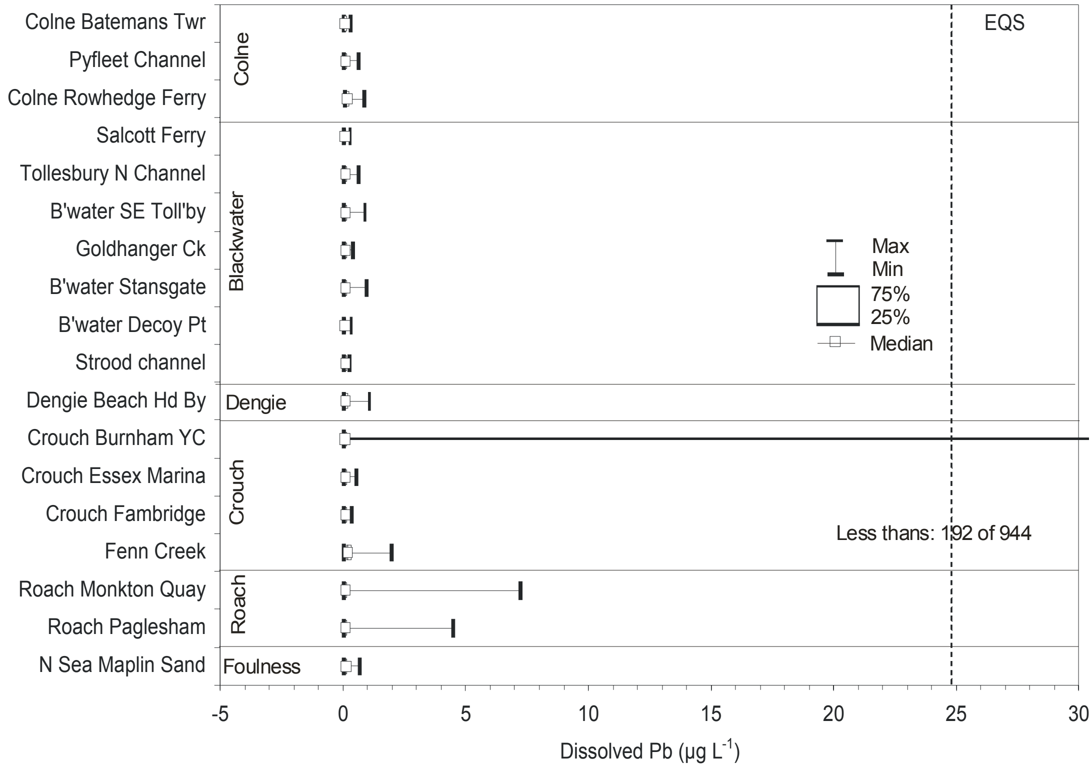


Figure 14. Concentrations of dissolved Pb (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

Zinc

The distribution of dissolved Zn in tidal waters suggests freshwater loadings are not excessively high, though some inputs from the Rivers Colne and Crouch may be indicated by the slightly elevated median values at Rowhedge Ferry and Fenn Creek, respectively (figure 15). Based on an EQS for saline waters of 40µg l^{-1} (AD) all sites would appear to be compliant, though with a safety factor of only ~ four-fold at Fenn Creek, and with maximum values occasionally approaching and sometimes exceeding the EQS. Correspondingly, risk of non-compliance is probably 'low'. It is unlikely that Zn would represent an acute threat to most marine biota at the site. However since values at some sites are elevated, and would not meet an earlier proposed[13] revised standard, it may be useful to look in further detail at possible sources of Zn and biological effects.

[13] Following a review of toxicity data, Hunt and Hedgecott (1992) proposed a more stringent EQS to DoE of 10 µg l^{-1}, based on the lowest, most reliable NOECs (7 – 20µg l^{-1}) though this has yet to be adopted.

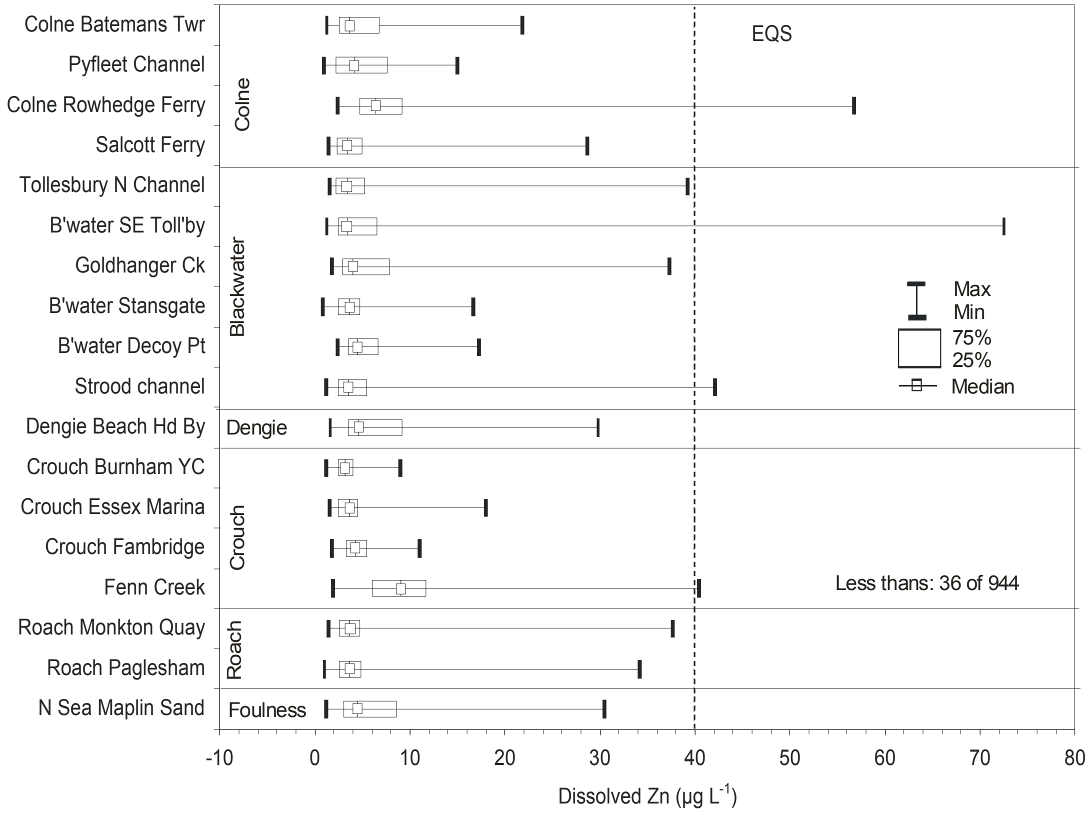
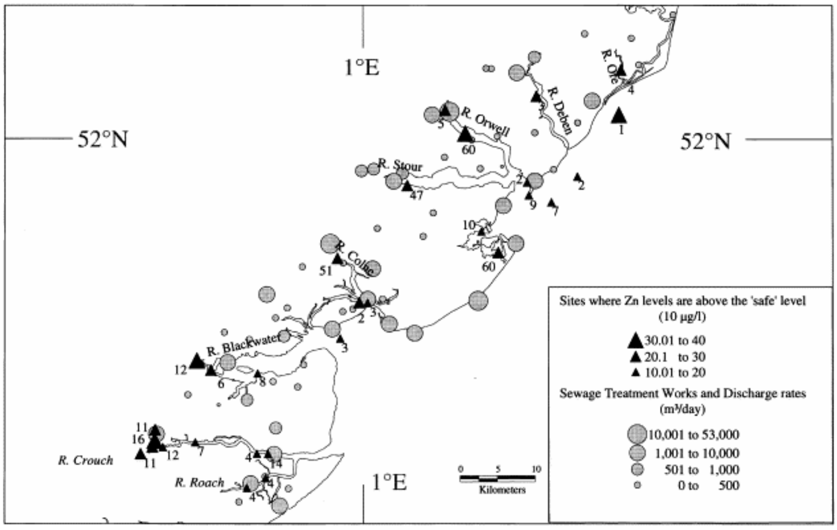


Figure 15. Concentrations of dissolved Zn (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

As with Cu, comparison with 1992-1996 EA Zn data from a similar suite of sites suggests that the most recent picture may represent an improvement in conditions. In the earlier data, sites in the upper and lower Colne, Blackwater and Crouch, together with the Roach (a total of 54% of site/year combinations) exceeded the perceived Zn safe level (Figure 16 from Matthiessen *et al.,* 1999). However, Mattiessen and co-workers based their 'safe level' on the provisional, more stringent guideline of 10 µg l^{-1} recommended by Hunt and Hedgecott (1992). Figure 16 illustrates the location of these earlier 'at risk' sites in relation to STW (top) and boating/shipping sources (bottom) of Zn; like Cu there is no clear correspondence between location of inputs and EQS excedence.

Despite the overall improvement, maximum values for Zn in the current data set are not dissimilar to those observed a decade earlier.



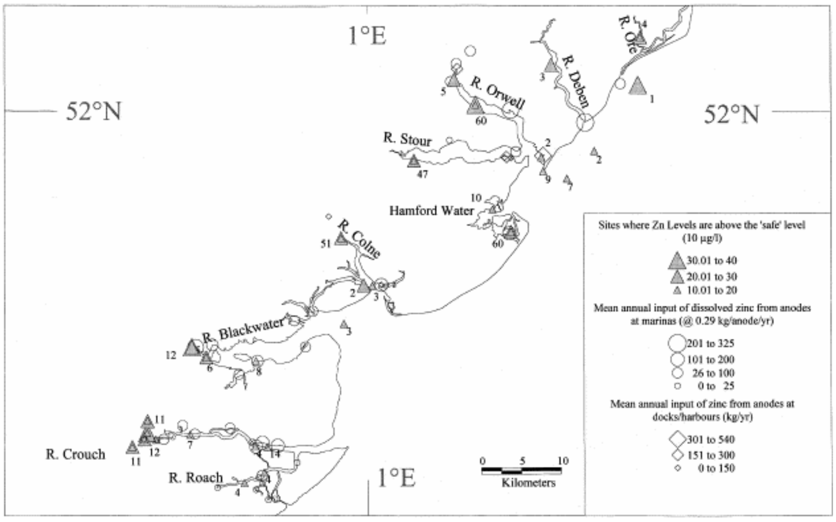


Figure 16. Mean annual zinc inputs (kg Zn/yr) from STW (top) and marinas and docks (bottom), and mean annual Zn concentrations (1992-1996) at sites that exceeded the 'safe level' (10 µg Zn l^{-1}). Values attached to the triangular symbols for concentrations indicate the number of observations which contributed to the respective annual means. (Reproduced from Matthiessen et al., 1999, with permission, Elsevier).

Loadings estimates for Zn to the Essex Estuaries were carried by Matthiessen *et al* (1999) and are plotted in figure 17. Although these are approximations made on data a decade ago they indicate that riverine and STW can outstrip other sources in the Colne and Blackwater. Nevertheless boating and shipping activity are likely to represent a significant proportion of total sources, except, perhaps in the Roach estuary. Zn pyrrithione is used as a booster biocide to replace TBT in antifouling paints and the use of sacrificial anodes on boats and structures, to reduce corrosion, can be expected to lead to a measurable increase in concentrations of Zn in marinas, ports and nearby estuarine waters.

Estimated Annual Zn Inputs (1992-96)

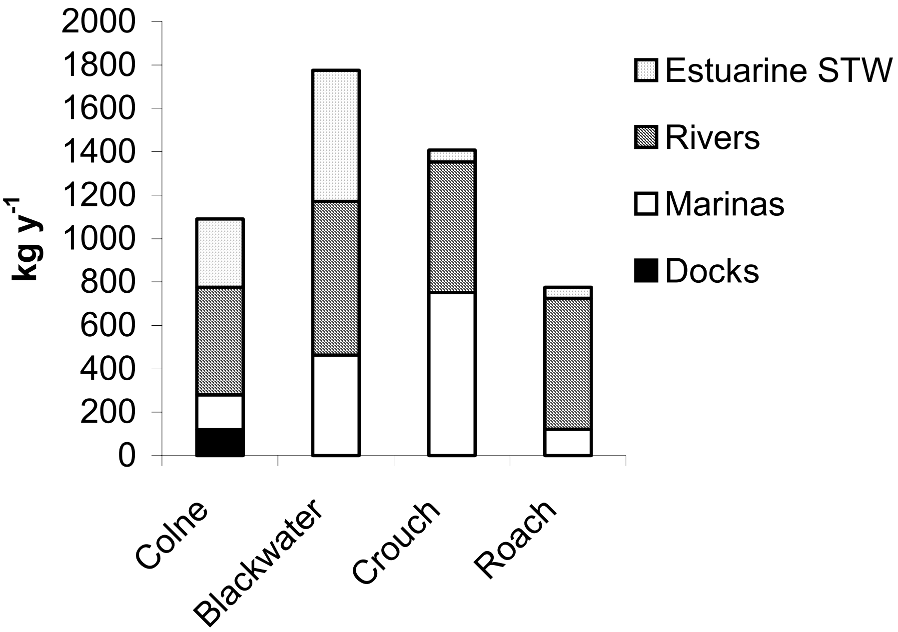


Figure 17. Estimated annual Zn inputs to the Essex Estuaries for the period 1992-1996 (plotted from data in Matthiessen et al., 1999). Riverine inputs for the Crouch and Roach are estimates , based on an average for the other two rivers.

More comprehensive discharge data and evaluation are needed to assess possible current influences on estuarine water quality.

Mercury

The pattern of dissolved Hg in estuarine water along the tideway is plotted in figure 18. Median values are invariably below the EQS (0.3 µg l^{-1}) by an order of magnitude or more (hence a low risk of failure) and generally represent detection limit values (71% of values are in this category). A small number of 'outliers' appear to exceed the benchmark notably in the Crouch and Roach and upper Colne, but do not significantly impact on median values. The risk of non-compliance is therefore considered low. However, in view of the toxicological importance of Hg further investigations could be considered (including bioaccumulation). Sediment data for Hg indicate that 'probable effects levels' are not exceeded, though at many sites concentrations are at levels where impacts cannot be excluded (see later in this section).

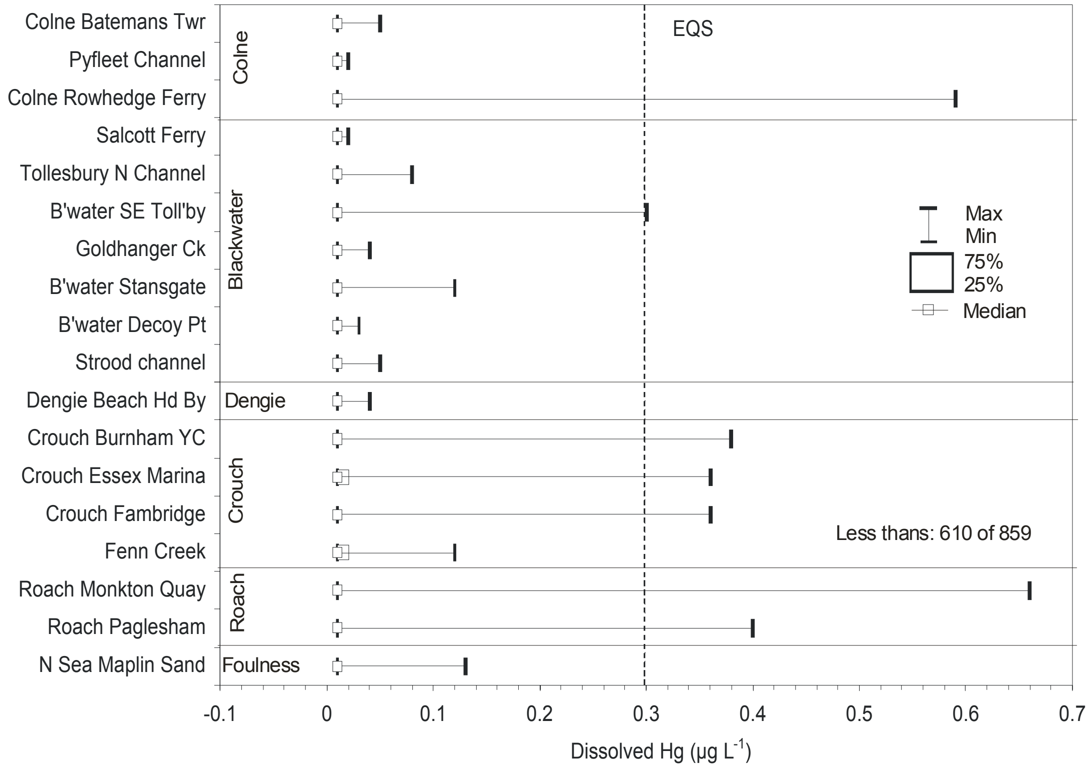


Figure 18. Concentrations of dissolved Hg (µg l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2000-2005).

The situation regarding water quality therefore suggests that anthropogenic contributions of metals to tidal waters are relatively low, based on median values. Unfortunately, the absence of discharge information prohibits a detailed appraisal of impacts on tidal water quality. There appears to be evidence of periodic enrichment with a number of metals, but the frequency of these events suggests little risk of EQS compliance failure.

There are some indications of recent improvements in water quality, and reductions in anthropogenic loadings. In the past, for example, metal levels in the Colne Estuary at locations near Colchester metal were considered to be higher than expected from background geochemistry, implying contribution from discharges (Thornton *et al* 1975). Between 1992 and 1996 Cu and Zn failures were frequent across the EMS (Matthiessen *et al* 1999), whereas this no longer appears to be the case.

Sediments

Particulate metals can form an important part of the loading discharged to estuaries. Scavenging of dissolved metals also occurs, so that the sediments of the estuary provide an integrated record of contamination history. Because of their larger surface area and greater density of organic and oxyhydroxide binding sites, contamination loadings will be highest in fine fractions and lowest on coarser substrates.

There are various techniques to compensate for the effects of granulometry (discussed earlier in 'Methods'). For the appraisal of spatial distributions the focus here is on the MBA data set which relates to analysis of inter-tidal surface sediment fines (<100µm, nitric acid digest, see Figures 19 and 20). The reason for this choice is partly that contemporary information on bioaccumulation in benthic invertebrates is also available for the same sites (section 7) – of relevance for bird populations. Furthermore, though the data are rather old, they appear to represent the most spatially extensive coverage of the EMS available, for metals. Because of the persistence of sediment bound contaminants, the age of the data may in fact not detract significantly from the present day picture. Some recent sediment analyses from the EA database are shown in figure 21 for comparative purposes.

With the possibility of biological effects in mind, we have represented data for metals in inter-tidal sediments from the Essex Estuary EMS in map form, classifying sites according to the interim sediment guideline criteria for each metal (figures 19 and 20). Green bars denote sites where no harm to biota is predicted (below ISQG's / TELs), grey bars denote sites where effects cannot be excluded (between ISQG's/TELs and PEL's) and red bars represent sediment concentrations where harmful effects might be expected (above PEL's).

For two metals, Cd and Zn, concentrations in sediments throughout the entire EMS fall below the ISQG/TEL value (green bars in figures 19 and 20) i.e. biological effects would not be expected. For Ag (results not shown) only one site (Wivenhoe) would exceed the ISQG/TEL of 0.73 $\mu g\ g^{-1}$, by a small margin.

For As, Cu, Hg, Ni, and Pb (and to a lesser extent Cr), sediment samples from the majority of sites displayed concentrations between ISQG's/TELs and PEL's – where biological effects cannot be excluded (grey bars in figures 19 and 20).

Lowest values for the majority of metals were those at open coastal sites in the Foulness SPA, and at Osea Island and Stansgate in the Blackwater. These may represent sediments of a slightly coarse nature with a low organic content, and lower metal complexation capacity (most metals in the data set co-vary significantly with % organic matter). Strong gradients were not a feature for any of these elements, though Cd was somewhat elevated at Maldon at the head of the Blackwater.

PEL values (where effects would be expected) were not exceeded for any of the metals tested.

It is stressed that these are guideline values only. Where sediments exceed the TEL this is generally by a relatively small margin. Effects due to these metals, if they occurred, would largely be chronic rather than acute. Furthermore, the data are in some cases almost 20 years old and may not be representative of conditions now. (However, comparisons with more recent EA samples suggest that concentration ranges have remained similar in relation to toxicity guidelines - figure 21; furthermore, MBA and EA data for sediment from Stansgate in the Blackwater Estuary are remarkably alike, despite a time difference of 20 years and different methodologies and size fractions: see figure 22). Re-survey of the MBA sites, using the same techniques would be useful to establish the current status and to evaluate both spatial and temporal trends. This is seen as a particularly important issue in terms of meeting standstill provisions for sediments under the Dangerous Substances Directive, and attainment of Favourable Condition (Habitats Directive), and may partially drive any requirement to minimise further inputs via aqueous discharges.

Thus, although these sediment classifications do not themselves constitute direct evidence for possible effects (or lack of), the results indicate that sediments *could* be of significance for the biota of the EMS. The absence of values exceeding the PEL implies this risk category is likely to 'low to medium' rather than 'high'.

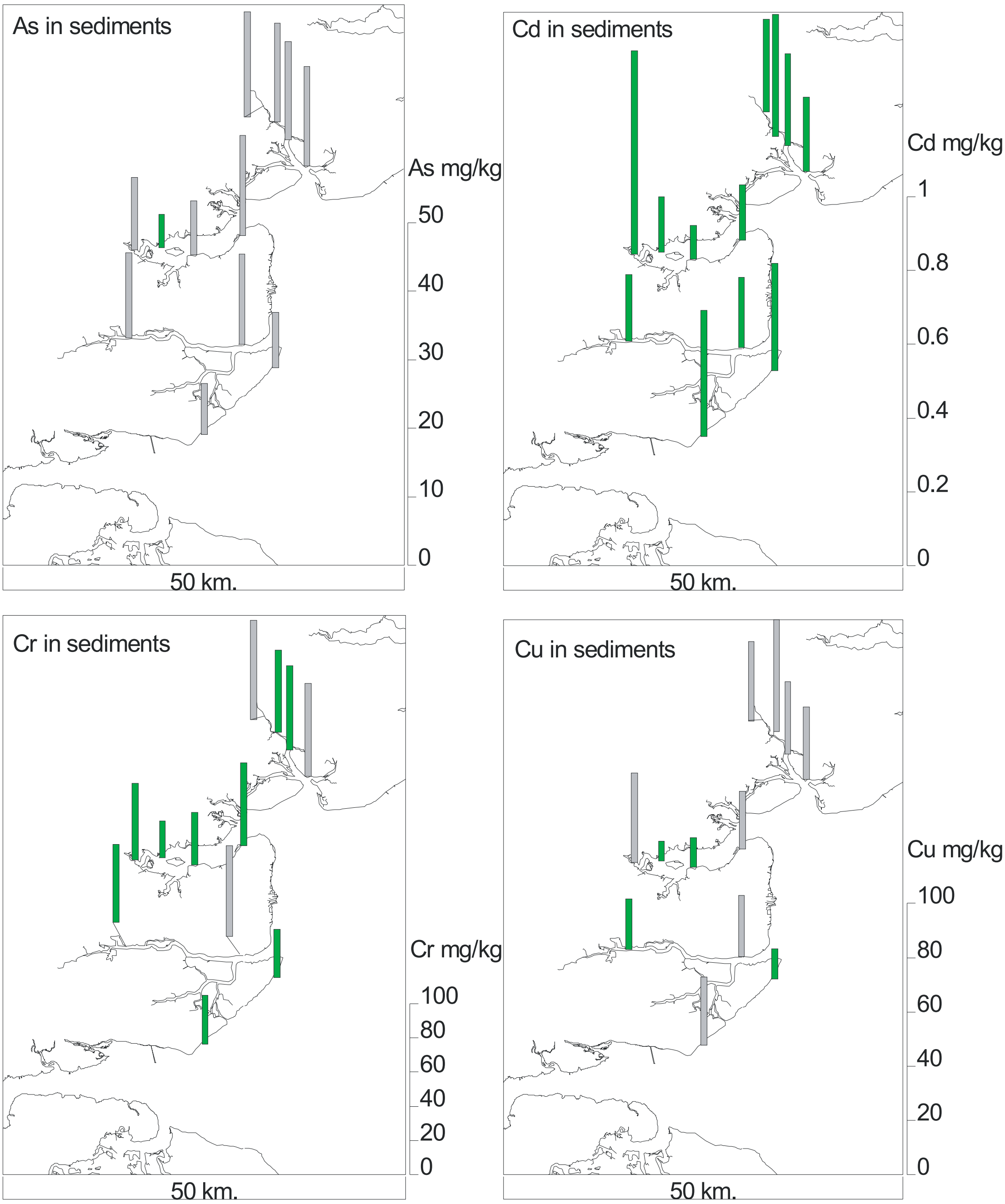


Figure 19. Arsenic, Cadmium, Chromium, and Copper in sediment (<100µm fraction). Classification of the Essex Estuaries EMS based on interim marine sediment quality guidelines/Threshold Effect Levels (ISQG/TEL) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source MBA: mainly from late 1980s except Foulness, Shoeburyness -1997-2001).

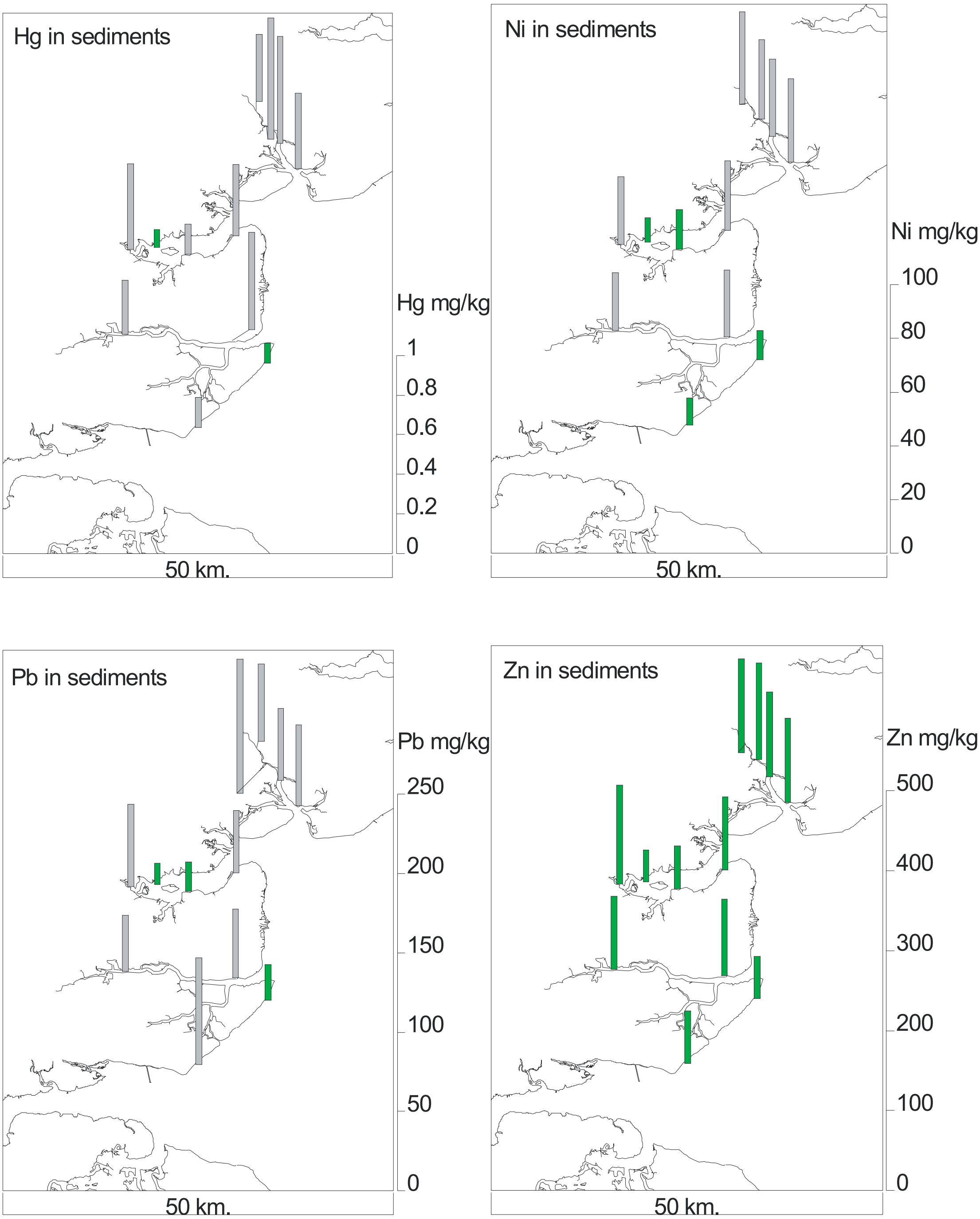


Figure 20. Cadmium, Lead, Zinc and Mercury in sediment (<100µm fraction). Classification of the Essex Estuaries EMS based on interim marine sediment quality guidelines/Threshold Effect Levels (ISQG/TEL) and probable effect levels (PEL's) (from CCME 1999). Red = effects expected; Grey = possible effects cannot be excluded; Green = no harm to the environment expected. (Data Source MBA: mainly from late 1980s except Foulness, Shoeburyness -1997-2001)

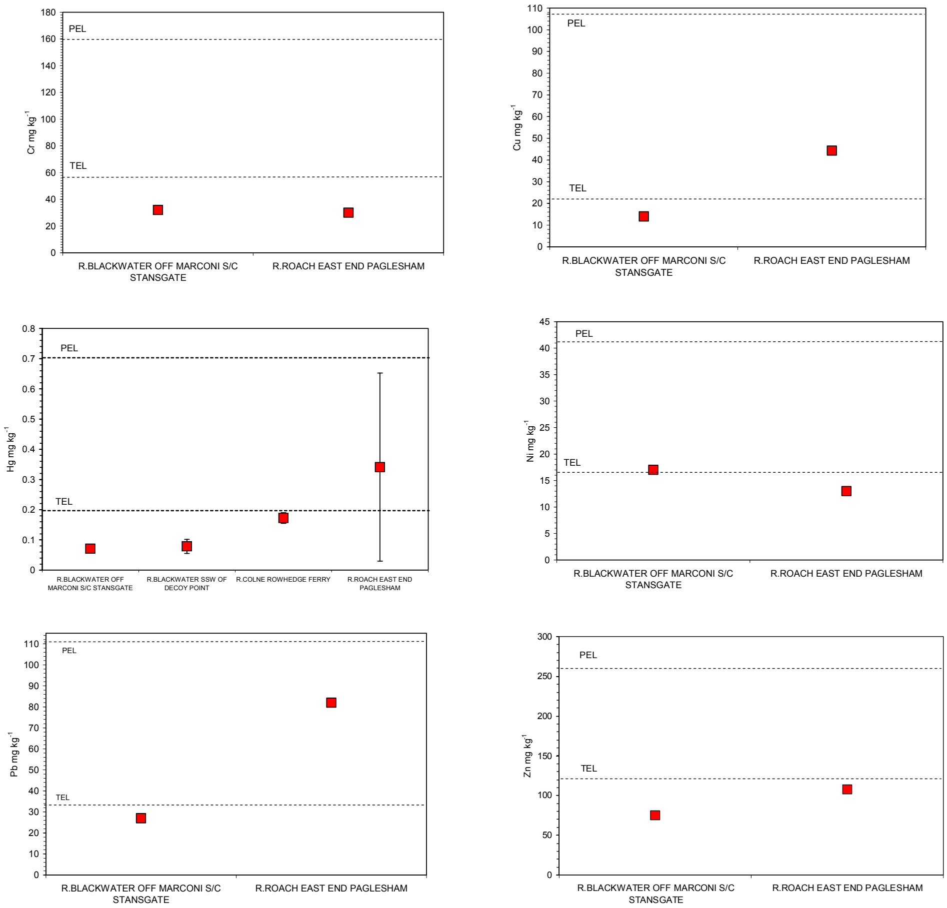


Figure 21. Recent data for metals in Essex Estuary sediments in relation to PEL and TEL (data source: Environment Agency –averages for samples collected between 2000 and 2005).

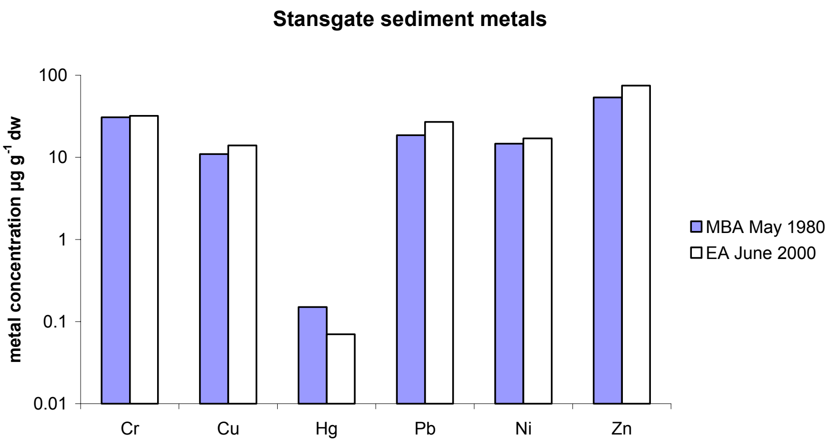


Figure 22. Comparison of MBA and EA sediment metal data for Stansgate, Blackwater Estuary; samples taken 20 years apart, using differing methods and size fractions (MBA <100µm; EA <63µm?).

Sediment metal concentrations from five sites along the Blackwater, and at Dengie, reported by Emmerson *et al* (1997) display similar trends to EA and MBA data. The mean values at each site, for data collected between 1992 and 1995, generally fell within the classifications described above, with respect to TELs and PELs (Table 8). Thus, a number of values, depicted by shaded cells, were above threshold levels - mainly in the upper estuary, but occasionally further downstream (and, for Hg and Ni, even at Dengie). In all other cases values were below TEL (effects not expected). Values did not exceed the PEL (where effects are expected to occur). The STW at Chelmsford was cited as a possible source of Cu and Zn to the upper estuary sediments at Beeleigh by Emmerson *et al* (1997).

Table 8. Sediment metals (µg g^{-1} dw) in samples from the Blackwater estuary, 1992 -1995 (from Emmerson et al., 1997)

	Beeleigh	Maldon	Limbourne	Tollesbury Wall	Tollesbury Fleet	Orplands	Dengie
Cd	<0.1	1.71	0.1	<0.1	0.38	0.19	0.08
Cr	61	54	56	37	32	65	35
Cu	77	40	25	16	16	25	18
Pb	60	64	54	31	42	39	27
Hg	0.43	0.51	0.3	0.11	0.26	0.33	0.27
Ni	33	29	32	31	32	39	19
Zn	190	138	96	68	63	94	64

values above TEL: possible effects cannot be excluded

To put Essex Estuary sediment data in context table 9 compares average sediment-metal data for the EMS with other estuaries, including the relatively uncontaminated Avon (MBA data). Clearly, concentrations of metals such as As, Cu and Zn, are low in comparison with those derived from mining sources in parts of SW England (Restronguet, Table 9). In fact with the exception of Ag and perhaps Hg there is only a small degree of enhancement in Essex sediments, relative to the 'baselines' as represented by the Devon Avon. Of the three Essex estuaries, samples from the Colne appear to have slightly higher sediment metal levels, probably because of a higher organic content.

Table 9. Metals in intertidal sediments (µg g^{-1} dry wt): typical values for Essex Estuaries and other estuarine sites in England (MBA data)

site	Ag	Cu	Zn	Pb	Cd	Cr	Ni	As	Hg	%OM	Ref
Colne	0.67	31	112	58	0.26	52	31	15	0.47	4.2	2
Blackwater	0.66	18	77	31	0.24	37	19	9	0.26	2.5	2
Crouch	0.51	20	94	39	0.19	49	23	13	0.38	3.4	2
Foulness	0.16	18	59	45	0.32	28	10	8	0.13	2.1	2
Poole	0.82	50	165	96	1.85	49	26	14.1	0.81		1
Tamar	1.22	330	452	235	0.96	47	44	93	0.83		1
Severn	0.42	35	242	84	0.63	55	33	8.4	0.44		2
Restronguet	3.76	2398	2821	341	1.53	32	58	1740	0.46		1
Avon (Devon)	0.06	19	98	39	0.3	28	23	13	0.12		2

[1] Bryan and Langston, 1992; [2] own unpublished data; OM-organic matter estimated by weight loss on ignition at 400°C

Dredging of sediments is carried out to maintain navigational channels and other purposes and samples of this material are analysed by CEFAS as part of its legislative responsibility to prevent disposal of hazardous materials to sea. Results for the Crouch from 1996 indicate that metal concentrations in the dredged materials were largely within accepted guidelines and that the metal component should pose little direct toxic threat following disposal (table 10). Most samples tend to be either just below or just above TEL levels, with Hg and Pb exceeding the PEL in one sample (presumably influenced by localized sources).

Table 10. Metals in dredged materials (µg g^{-1} dry wt), Crouch Estuary 1996 (Data source (CEFAS, 2000)

n	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
4	7.8-9.6	0.1-0.86	28-107	9.5 – 52	0.06-0.96	14.5-17	16.5-159	29.4-50.2

n = number of measurements

The origins of metals loadings in sediments includes both geogenic (natural) and anthropogenic components (usually sequestered from overlying water) and selective extraction techniques can help to separate their relative contributions and sometimes their sources. Using such techniques enrichment of, for example, Cd has been observed in the more soluble phases of Essex saltmarsh sediments (from Two Tree Island, near Southend, and the more open coastal marshes on the Dengie peninsula) with gradients suggesting transport northwards from sources in the Thames Estuary, as far as the Blackwater (Fletcher *et al* 1994a,b; O'Reilly Wiese *et al*, 1995, 1997a,b). This dispersal may occur either in dissolved or fine particulate forms. Our own data for Ag suggests a similar long-distance transport mechanism, dominated by inputs from the Thames. Leggett *et al* (1995) has also indicated that the Thames contributes a much greater amount of metals (and PCBs) to the North Sea than that released from the Essex sub-region.

Trends in porewaters and partitioning behaviour in saltmarsh areas (from the Thames northwards along the Essex coast) confirm that Cd was amongst the more soluble metals, and that , as with solid phases, gradients of the dissolved form decline as a function of distance from the Thames (O'Reilly Wiese *et al*., 1997b). Other metals (Cu, Ni, and particularly Pb, Mn and Fe) tend to partition more strongly with particulate phases, mediated partly by redox changes. Overall, concentrations in pore waters were not considered by O'Reilly Wiese *et al* (1997b) to be unusually high relative to background values elsewhere, or to constitute a threat to salt marsh biota. Even so, some of the Cu values in pore water (4-14 µg l^{-1} at Dengie) are above LOEC for sensitive species and some bioaccumulation of metals from this enriched phase seems possible.

Attempting to build a picture of temporal trends in metal inputs to the region by examining chronological records in sediment cores has proved to be equivocal. Apart from cores where marked transition of sediment type occurred, only slight variations in metal concentration with depth were observed at three Essex Estuary sites (one in the Crouch, two in the Blackwater) studied by Fletcher *et al*, (1994b), and another on the Dingle peninsula (O'Reilly Weise *et al* 1995), implying fairly constant deposition and accretion rates or substantial mixing. There was also little indication of redox-related change on total metal levels. Time-based profiles from Dengie, derived from accretion rates using ^{137}Cs dating (34-46mm y^{-1}), also indicate little variation in metals with depth, compared with industrialized sites elsewhere in Europe, where marked increases in metals - up to five-fold - coincide with industrialisation since the turn of the 19th / 20th Century (Callaway *et al.*, 1998). Thus there appears to no clear signature of major anthropogenic increases in recent history in any of the Essex saltmarsh cores studied to date.

Organic pollutants – Organotins, PCBs, pesticides, herbicides, PAHs, alkylphenols, VOCs etc.

6.1.2 TBT and other booster biocides

Use of tributyltin (TBT) antifouling on boats less than 25m in length was prohibited in 1987, following the discovery of significant effects, notably in oysters and neogastropods. Larger vessels (essentially the commercial fleet and Navy) were still entitled to use TBT paints, at least until 2003 when recommendations from IMO for a total ban were implemented in the EU by Council directive 2002/62/EC. The Cefas laboratory at Burnham have conducted a number of long term studies on TBT in the Crouch Estuary to monitor the effectiveness of the 1987 legislation (water, sediment and faunistic changes). Because this estuary is dominated by small boat inputs, legislation in 1987 was extremely effective in reducing inputs of TBT, as reflected in concentrations in water. At the time of the ban, mean summer concentrations of 40-50 ng l^{-1} were not uncommon in parts of the inner estuary (Figure 23), with concentrations in marinas sometimes more than an order of magnitude higher (Waite *et al.*, 1991; Rees *et al*, 2001). Concentrations dropped significantly following legislation, as indicated in Figure 23. By 1998, during the peak yachting season (April-October), concentrations in the estuary were generally around the EQS but in Burnham marina concentrations rose to ~6 ng l^{-1} at the start of the season indicating some (illegal) usage (Thomas *et al.*, 2001). During the off-peak season, concentrations were at or below the EQS (2ng l^{-1}).

Reductions in TBT concentrations in water were not so evident, initially, in the Blackwater (in fact concentrations peaked in 1988, figure 23) and remained higher than in the Crouch, probably as a result of illegal usage.

There are no rigorous TBT time series for other parts of the EMS though it is expected that reductions seen in the Crouch have been mirrored in other small-boat dominated sections of the site.

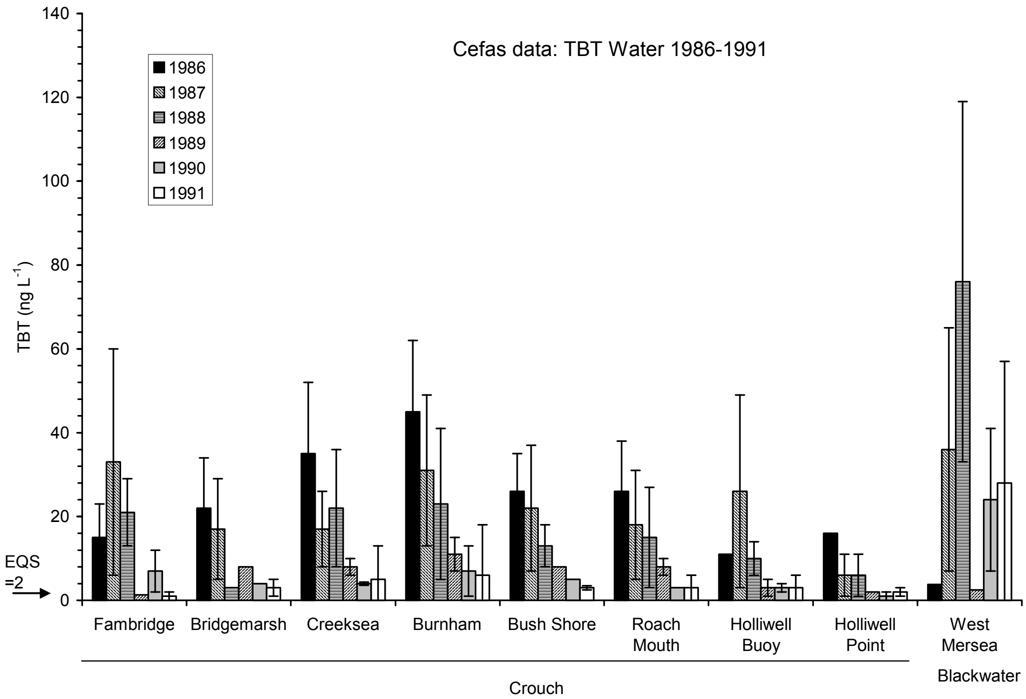


Figure 23. TBT concentrations in water, Crouch and Blackwater, following TBT legislation in 1987 (plotted from Cefas data, MAFF 1993)

EA TBT data between 2000 and 2005 consists of a small number of samples from the Blackwater Estuary (Figure 24) taken in autumn 2002. Median values are all below the EQS benchmark, though the latter is in fact set as a Maximum Allowable Concentration. Clearly at Decoy Point in the upper estuary the EQS was exceeded by a considerable margin on one occasion. Excedence has also occurred to a lesser extent, SE of Tollesbury.

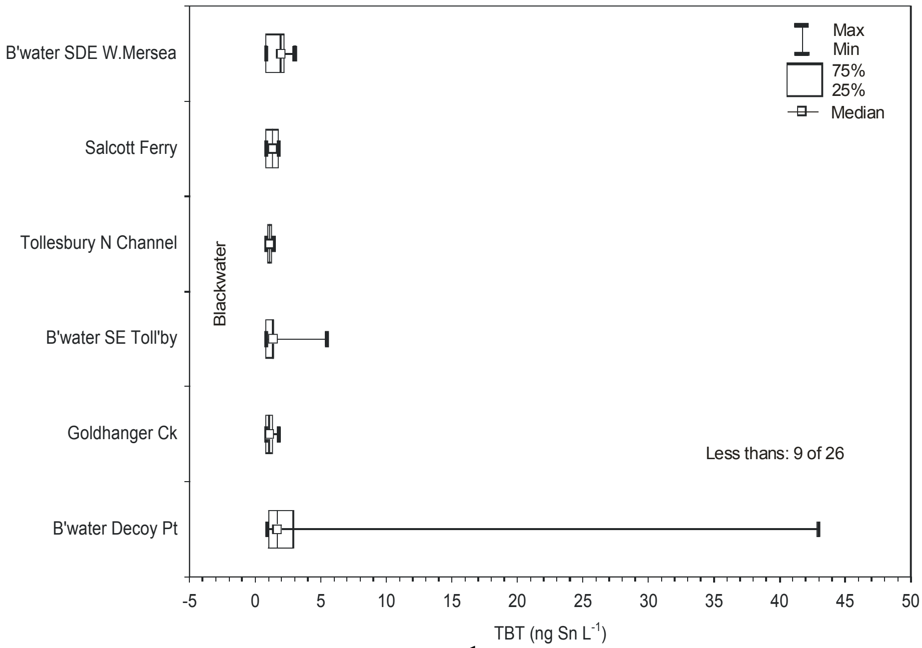


Figure 24. Concentrations of TBT (ng l^{-1}) in tidal waters, Essex Estuaries EMS. Data source EA. (median, min and max, and percentiles, 2002).

The threat from TBT is therefore receding, at least in terms of acutely toxic levels in water, but has not entirely disappeared. The persistence of sediment-bound TBT is also potentially a source of long-term chronic impact in poorly flushed areas, particularly for deposit feeders, and could be exacerbated by release of more generally available dissolved and suspended forms during dredging activities.

There are no CCME guidelines for TBT, though OSPARCOM (1994;2000) has set a provisional ecotoxicological guideline values of 0.000005µg g^{-1} - 0.00005µg g^{-1}. It is felt however that this guideline (which has no legal significance and is intended as a guide for further work) may be set too low. Other TBT 'guidelines' are set for dredge spoil samples as part of the procedure for licensing for disposal at sea (CEFAS, 2000, 2005). At concentrations above 0.1 µg g^{-1} dry weight (Action Limit 1) disposal will be condoned only if conditions and amounts are considered to be of low concern. Above 1µg g^{-1} dry weight (Action Limit 2) it is unlikely that disposal at sea would be permitted.

MBA data from the late 1980s (when TBT inputs from small boats were probably at their peak) indicate that most Essex sites were above the (over precautious?) OSPARCOM provisional ecotoxicological guidelines, but were below the more realistic lower action limits of 0.1 µg g^{-1} dry weight (see Figure 25). TBT in later samples from Shoeburyness and Foulness (1997-2001)) were considerably lower, perhaps indicative of declining inputs and perhaps also reflecting the lower organic content of these fairly marine sediments. These data suggest TBT is unlikely to be a major risk to the biota of the EMS, though up-to-date sampling would be useful for confirmation given the toxicological significance of this compound.

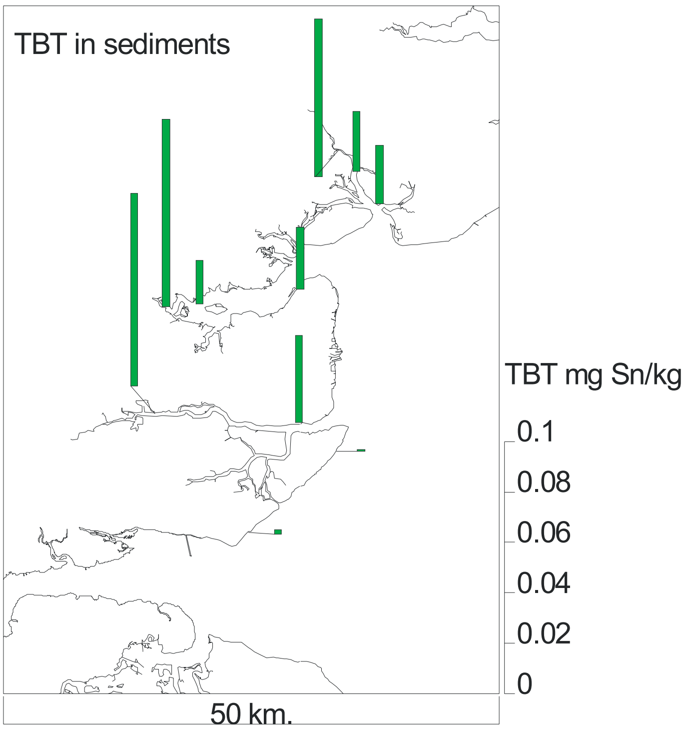


Figure 25. Historical data (1987 - 2001) for tributyltin (TBT as Sn) in sediment of Essex Estuaries (<100µm fraction). Green = below Action Limit 1 (Data Source MBA)

TBT was measured in sediments from the Crouch, Roach and Colne in 1990 by Dowson *et al* (1992) and again between 1990 and 1992 (Dowson *et al*., 1993) and results are generally consistent with the ranges shown in figure 25, though higher concentrations were indicated at marinas/boatyard sites (table 11). Sediments were not sieved or corrected for grain size however, and some of the variation, may be a reflection of granulometry.

Table 11. TBT concentrations (mg kg^{-1}, as Sn) in bulk sediments, Essex estuaries, summer 1990 (data source Dowson et al., 1992)

	n	Typical TBT range	Marina/boatyard
Colne	4	<0.003 – 0.02	
Blackwater	5	<0.003 -0.03	2.8 (Tollesbury)
Crouch and Roach	5	<0.003 – 0.04	1.29 (Paglesham)

Trend data for sites near marinas and slipways on the Blackwater (Tollesbury) and Roach (Paglesham) suggest there were some substantial reductions in TBT loadings in the immediate aftermath of the legislation (data from Dowson *et al*, 1993 summarised in figure 26). A similar conclusion was drawn for sediment data for the inner Crouch as indicated by results in Waldock *et al* 1999, described below. However there may be an indication from these data of the approach of a much slower decline in the long-term.

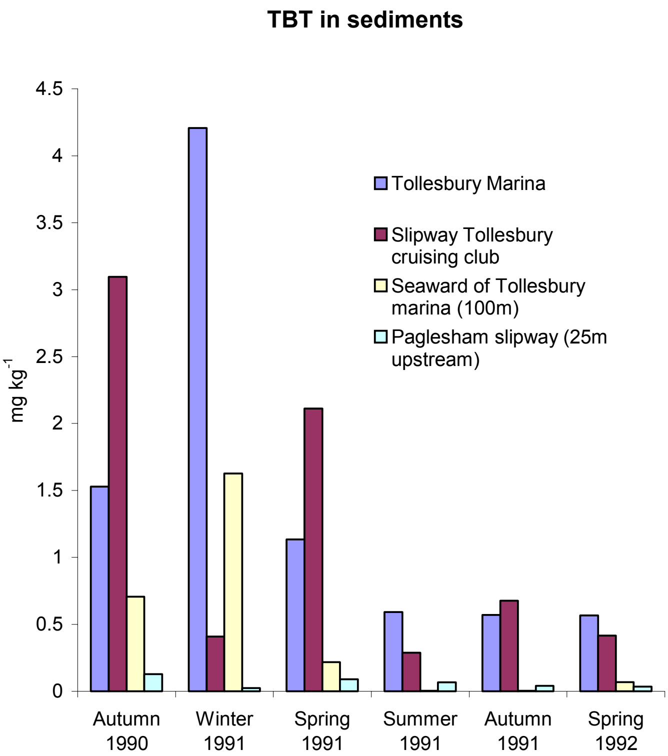


Figure 26. Trends in TBT concentrations in sediments near marinas in the Blackwater (Tollesbury) and Roach (Paglesham) estuaries, following legislation in 1987 (drawn from data in Dowson et al., 1993)

A further indication of (TBT) sediment quality in the region may be inferred from surveys undertaken for the National Monitoring Programme (e.g. MPMMG, 1998, 2004) and from analysis of dredge spoil samples as part of the procedure for licensing for disposal at sea (CEFAS, 2000, 2005a). Sample dredgings from the Crouch and Blackwater in 1994 (0.16-0.17µg g^{-1} ww) were not dissimilar in TBT content to the intertidal samples shown in Figure 25 (MBA data). Sediments from the Thames were in the range 0.06-0.26 µg g^{-1} ww. However, according to EA's database, samples from West Mersea Beach (NMMP shellfish site), taken in 2000, had a mean value of 545 (min= 363, max = 836) µg g^{-1} dry weight (as TBT cation) i.e. above 0.1 µg g^{-1} dry weight (Action Limit 1).

Maximum TBT concentrations in sediment samples from disposal sites off the Essex coast at South Falls and Barrow Deep between 1998 and 2004 were at or close to detection limits (<0.001 and 0.003 µg g^{-1}, respectively) whilst those at Roughs Tower off the Stour and Orwell were slightly higher (0.073 µg g^{-1}), though still significantly below Action Levels (CEFAS, 2005a) and unlikely to represent a risk to the EMS.

Thus, despite some reduction, TBT concentrations in sediment are relatively persistent, but are probably only of significance in the vicinity of marinas and boatyards where caution may be raised, particularly if they are to be dredged. Processes including physical resuspension and bioturbation could remobilise these sinks. Furthermore, TBT in such contaminated sediments is likely to be bioavailable, and potentially harmful, to deposit-feeders and infauna (Langston and Burt, 1991).

Long-term monitoring of TBT in the Crouch Estuary by Cefas has been accompanied by observations which link improvements in environmental quality with biological recovery. This is discussed in greater detail in section 7.

Alternative Booster biocides

Since the ban on TBT on small boats, a range of alternative booster biocides have been used, though only Irgarol 1051 and Diuron have been detected extensively. Analysis of waters collected in the peak yachting season from five sites in the Crouch estuary, in 1998, yielded median values for Irgarol 1051 of between 3-11 ng l^{-1} and, for Diuron, 11-44 ng l^{-1} (Thomas *et al*, 2001). Higher median values were observed inside Burnham Marina (24 and 88 ng l^{-1}, respectively). Out of season, lower concentrations of these compounds were observed in the estuary (<1 - 9.4 ng l^{-1} Irgarol; 0.6 - 4.5 ng l^{-1} Diuron). It is difficult to assess risks for these compounds due to a lack of toxicity data, and though impacts are predicted to be less severe than TBT, localized effects cannot be dismissed (Thomas *et al*., 2004). An EQS of 2µg l^{-1} (annual average) has been proposed by EA for diuron, whilst lowest chronic (phytotoxic) effects of Irgarol are reported to occur at ~ 0.15 µg l^{-1} (with lowest NOEC[14] in the region of 0.02-0.05µg l^{-1}), which would suggest acute effects in the Crouch are unlikely, except perhaps in marinas. The possibility of additive contributions to chronic stress-effects with other contaminants is, however, unknown. A further risk arises from bioaccumulation; algae and angiosperm (including *Zostera noltii*) are known to concentrate them relative to water (even though these compounds

[14] NOEC – no observable effects concentrations (see Scarlett *et al*., 1997, 1999; Thomas *et al*., 2004)

are phytotoxic) and are a potential route of entry into food chains (Thomas *et al.*, 2004).

Surveys of booster biocides in the Blackwater Estuary (1998/1999) indicate that concentrations of Irgarol vary from below the limit of detection (<0.15µg l^{-1}) at most estuarine sites, to 0.68 µg l^{-1} in marinas and mooring areas during the yachting season (above NOEC for species such as *Enteromorpha* -0.022 µg l^{-1}, and even *Zostera* – 0.2 µg l^{-1}) (Voulvoulis *et al.*, 2000). Diuron and Dichlofluanid, another biocide used in antifouling paints, were largely undetectable in water, whilst Chlorothalonil was measured up to 1.38 µg l^{-1} but was apparently independent of seasonal boating activity – possibly indicating agricultural inputs of this fungicide (sprayed mainly in spring). Little is known of the toxicity of this compound, though reduced shell growth occurs in Eastern oysters (one of the more sensitive species tested) at a concentration of 7 µg l^{-1}. A provisional benchmark of 0.36 µg l^{-1} set by the Canadian Environment Ministry would be exceeded by a number of the Blackwater samples (Voulvoulis *et al.*, 2000).

Concentrations of Irgarol (3 to 223 µg kg^{-1}), dichlofuanid (<5 – 688 µg kg^{-1}) and chlorothalonil (<4 – 46.5 µg kg^{-1}) have been determined in sediments of the Blackwater estuary, with the latter compound peaking after spring spraying (agricultural inputs dominate, with fairly uniform concentrations in sediments throughout the estuary). As in water, concentrations of the booster biocides Irgarol and dichlofuanid in the sediments peaked in October (especially at marina sites), presumably representing the build up from inputs from antifouling. Diuron, in contrast, could not be detected (Voulvoulis *et al.*, 2000). Analysis of cores from three sites in the Blackwater suggests fairly uniform concentration profiles of Irgarol, chlorothalonil and dichlofuanid, with depth, down to around 30cm (Voulvoulis *et al.*, 2000).

The practice of hosing boats can result in temporary point sources of antifouling compounds, whose transient concentrations could reach high levels in water, along with entrainment of paint particles within sediments. Fortunately, most of these compounds are now banned on small boats, therefore concentrations are likely to decrease in future.

6.1.3 Pesticides and herbicides

There is a lack of qualitative evidence on the distribution, bioavailability and toxic effects of organic micropollutants on the flora and fauna of the Essex Estuaries EMS. There are concerns, nevertheless, that they may contribute, along with inorganic contaminants, towards loss of condition of features at the site including stimulation of the observed saltmarsh depletion over the last 50 years (Scrimshaw and Lester, 1996).

In general terms STW are likely to act as important sources of such contaminants, along with run-off from agricultural land, some non-crop areas (weed control in urbanized locations) and landfill sites. Seasonal pulses are to be expected for many agrochemicals. Because a number of less polar compounds tend to be hydrophobic, their fate is often determined by partitioning in sediments, where many (particularly organochlorines) may be highly persistent and bioavailable to deposit-feeding organisms, and subsequently to higher members of the food chain. Others such as the herbicides simazine and atrazine and chlorphenoxy acids (CPH, e.g. mecoprop) are more hydrophilic and therefore tend to be encountered more frequently in the water courses of eastern England. For this reason a number of herbicides may be considered as a *potential* threat to sensitive salt marsh plants (as well as algae and *Zostera*) in the region. As yet these threat remain largely unquantified, other than from extrapolations to threshold values for a few compounds at a small number of sites.

EA has analysed pesticides and other POPs in a small subset of tidal waters sites since 2000 (figure 27). The agricultural nature of the catchments feeding into the EMS, suggests that application of pesticides and other agrochemicals may have been widespread. However, because of the controls on organochlorines, there are now very few values for such pesticides (or later alternatives such as organophosphates) in the EA data base which exceed the detection limits. Table 12 summarises this data in relation to EQS values. It should be noted that some of the detection limits are close to, and occasionally above, the EQS.

Rigorous classification of risk is therefore not feasible. Arguably, however, because of the large number of 'less than values' and absence of positive data which exceed EQS, the risk of excedence would be anticipated to be low.

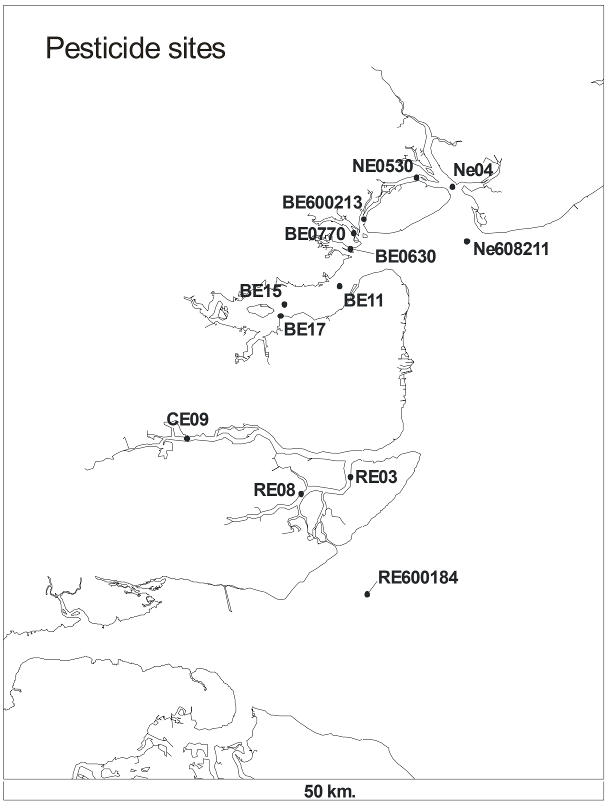


Figure 27. EA sites for which analyses of pesticides in tidal waters have been performed

Table 12. Summary of total numbers of tidal water samples analysed for selected pesticides and herbicides in the Essex Estuaries EMS, and proportions below detection limits (data source EA, 2000-2005)

Determinand	type	No. samples	Detection limit µg/L	% of values <DL	EQS (µg/L)	
ALDRIN	OC insecticide	632	0.001 - 0.01	100%	0.01 AT	
HCH ALPHA	OC insecticide	635	0.002 - 0.022	100%	0.02 AT (s/w)	0.1 AT (f/w)
HCH BETA	OC insecticide	629	0.002 - 0.021	100%	0.02 AT (s/w)	0.1 AT (f/w)
HCH DELTA	OC insecticide	632	0.001 - 0.01	100%	0.02 AT (s/w)	0.1 AT (f/w)
HCH GAMMA	OC insecticide	639	0.002 - 0.022	100%	0.02 AT (s/w)	0.1 AT (f/w)
DIELDRIN	OC insecticide	636	0.001 - 0.01	100%	0.01 AT	
PARATHION { ETHYL}	OP insecticide	632	0.004 - 0.01	100%		
DDT (OP)	OC insecticide	629	0.0059 - 0.05	100%		
DDE (PP)	OC insecticide	632	0.001 - 0.01	100%		
DDT (PP)	OC insecticide	627	0.001 - 0.05	100%	0.025 AT(sum of DDE,DDT,TDE)	
TDE (PP)	OC insecticide	635	0.001 - 0.05	100%		
ENDRIN	OC insecticide	626	0.001 - 0.05	100%	0.005 AT	
SIMAZINE	triazine herbicide	65	0.003 - 0.03	17%	2 AT (+Atrazine)	
ATRAZINE	triazine herbicide	65	0.003 - 0.03 ?	46%	2 AT (+Simazine)	
ISODRIN	OC insecticide	626	0.001 - 0.01	100%	0.005 AT	

Because of the limited number of 'real' values in the database it is not possible to discuss spatial or temporal trends. The only information (>DL) is for simazine and atrazine at a single site (Rowhedge Ferry in the Colne Estuary). Median values here were within the combined EQS of 2µg l^{-1} but a single sample produced high[15] values for both compounds (Figure 28). Sources are not known.

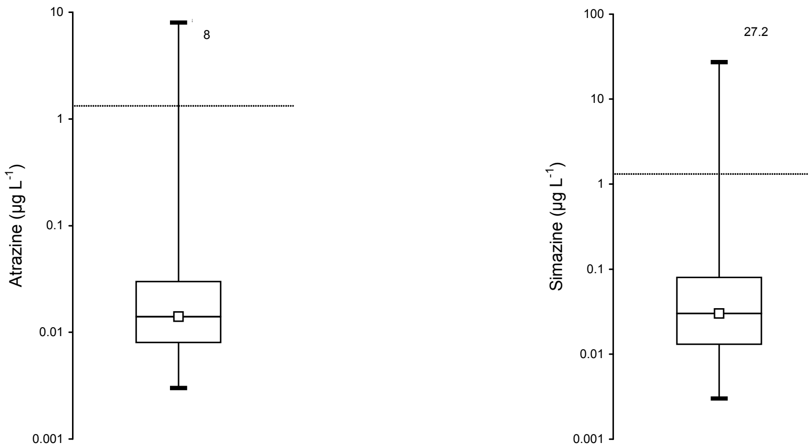


Figure 28. Summary statistics for Atrazine and Simazine at Rowhedge Ferry, Colne Estuary (data source EA, 2000-2005). Combined EQS is 2µg l^{-1}.

Simazine and atrazine have been determined in the past in water samples from near Barrow Deep, but at concentrations (2.1 and 2.2 ng l^{-1}, respectively) which were considerably below the combined EQS of 2µg l^{-1}(Cefas, 1993). This also applies to samples from Mucking in the Thames Estuary collected between 1993-1996 (3.2 and 3.5 ng l^{-1}). During the same period elevated concentrations of γHCH were determined on occasions at Thames Warp (up to 14 ng l^{-1}) and Gabbard (9.9 ng l^{-1}) which could have reflected the extensive disposal of sewage sludge at that time, though these were

[15] Though values of 8 and 27.2 µg l^{-1} are recorded in the database, these appear to be anomalous; it is possible that units may be wrongly labelled i.e. ng l^{-1} rather than µg l^{-1}?.

also below the EQS (Cefas, 1998). Sediments at Thames Warp were indicated to contain measurable amounts of dieldrin (0.23-0.5 $\mu g\ kg^{-1}$ dry wt), but again below guideline values (<TEL).

A detailed study of mecoprop (MCPP) has traced the pathways and timescales of transfer to adjacent saltmarsh, following spring application on adjacent agricultural land (Fletcher *et al*., 1995, 2004). Despite the short half-life of this relatively hydrophobic chlorophenoxy acid herbicide (CPH), a significant rainfall event shortly after application was shown to cause rapid leaching from soil and a corresponding large pulse in drainage systems leading to the marsh, with concentrations up to 386 $\mu g\ l^{-1}$ - significantly above UK guidelines for surface water (10$\mu g\ l^{-1}$). Downstream, at the marsh itself, levels up to 25.8$\mu g\ l^{-1}$ were detected where, normally, they were <0.05$\mu g\ l^{-1}$. Concentrations of mecoprop in adjacent mudflat sediments increased from below detection limits (<0.2$\mu g\ kg^{-1}$) up to 9.9 $\mu g\ kg^{-1}$ one week after the rainfall event, penetrating to depths > 40cm. Concentrations in vegetated marsh sediments (further from the discharge point) did not quite reach such high levels (increasing from 0.4$\mu g\ kg^{-1}$ ~5$\mu g\ kg^{-1}$) but appeared to be more persistent (lower flushing rate, greater adsorption capacity). Though the test site was to the north of the EMS (near Hamford Waters) it demonstrates the potential for episodic impact on saltmarsh systems arising from herbicide and pesticide application. Studies in the US have suggested that, in similar scenarios, run-off containing 2,4-D (another chlorophenoxy acid herbicide) could threaten eel grass beds and associated epiphytes (Mayer and Elkins, 1990; Fletcher *et al*., 1994).

Direct threats from triazine herbicides seem less likely based on the available concentration data and EQS values, though additive responses cannot be ruled out. Laboratory and field trials have shown that triazine herbicide exposure reduces photosynthesis and growth of epipelic diatoms and saltmarsh plants (at applications of ~ 7 $\mu g\ l^{-1}$ simazine and higher): this in turn can decrease sediment stability (erosion thresholds) by suppressing the production of extracellular polymers and reduced development of a cohesive biofilm (Mason et al., 2003). Although such concentrations are probably above those normally encountered in Essex waterways (figure 28), periodic pulses may occur locally following herbicide application (as illustrated for mecoprop, above). Effects on plant behaviour and physiology could, therefore, contribute subtely to the acceleration of saltmarsh disappearance.

Further investigations need to be undertaken to establish whether or not the presence of high levels of herbicides (e.g. following spring application) represents a real risk to the Essex marshes during critical periods of the life cycle of the indigenous flora, notably during germination and early development.

For many pesticides and other organic micropollutants adsorption on to solids - during wastewater treatment and within estuaries - will significantly influence their fate and transport. In the recent past, the practice of disposal of large amounts of sewage sludge in to the mouth of the Thames estuary (almost half of the UK total disposal at sea) would have represented a potential means of redistributing these contaminants, in addition to direct pathways from agricultural and urban (non crop) application. Recent EA data sets contain no information on these compounds in sediments on which to base a realistic impression of distribution and sources of contamination, and hence consequences for benthic fauna of the EMS.

An inventory of some organic micropollutants in sediments from four sites within the EMS, dating back to the period 1991-95, is summarised in table 13 (from data in Leggett *et al*., 1995; Scrimshaw *et al*., 1994, 1996; Fletcher *et al*., 1994). Concentrations are within the μg kg^{-1} range, but tend to be highly variable. Depth profiles from consolidated (vegetated) marsh areas at these sites (excluding the more energetic open coast site at Dengie) have indicated subsurface maxima (at depths of 10-30cm) of pesticides (and PCBs, see section 6.1.4) and occasionally some herbicides, which may be indicative of peak historical inputs, following proliferation of their use in the 1940s, and, presumably, recent curtailment. In contrast peaks tend to be less distinguishable in adjacent areas of intertidal muds, where sediments are better mixed[16]. As with metals, the majority of organic pollutants tend to be trapped more efficiently in sediments from vegetated marsh areas, rather than more open mudflats, due to their greater adsorptive capacity. This pattern of enrichment is most frequently observed for the more hydrophilic compounds (some CPH and triazine herbicides) which tend to be desorbed from the frequently-inundated intertidal sediment, especially at less sheltered sites such as Dengie (Scrimshaw *et al*., 1994; Fletcher *et al*., 1994).

The sparse data on pesticides and herbicides in sediments are indicative of low level but extensive contamination throughout the area. There are one or two examples which exceeded the PEL or TEL by a small margin (shaded in Table 13) indicating that impacts are possible (for example MCPP[17], 2,4-D and other members of the CPH herbicide group at S. Woodham Ferrers, probably of an agricultural origin). Elevated concentrations of DDT at Dengie are perhaps surprising given its open coastal location: given that PCB is also elevated here (section 6.1.4), possible sources are unlikely to be purely agricultural and may include a component from offshore sludge dumping grounds. Differences in sediment properties such as particle size and organic content do not *per se* explain the enrichment in organochlorines at this site (Scrimshaw *et al*., 1996; *see* also section 6.1.4).

In addition to the S. Woodham Ferrers example cited above, sediment contamination with herbicides - particularly MCPP and others in the CPH group – has also been indicated to reflect predominantly agricultural usage at Salcott, in the Blackwater, albeit at lower concentration (Fletcher *et al*., 1994). However, sediment samples near storm water overflows in the Thames at Southend demonstrate that urban run-off can sometimes represent an important source of certain CPH herbicides (2,4-D, 2,4-DP, MCPA, but not MCPP). Saltmarsh sediment concentrations of simazine and atrazine[18] were also high here, and at Tollesbury. Apportioning sources of herbicides is in most cases difficult since they tend to be fairly ubiquitous, low level contaminants and have been used for urban as well as agricultural application.

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[16] One exception is the occurrence of mecoprop (MCPP) in a core from Salcott (Fletcher *et al* 1994)

[17] MCPP is considered to be the most commonly applied chlorphenoxy acid (CPH) herbicide in the Essex region (Scrimshaw and Lester 1996).

[18] Before restrictions on usage, some 77% of atrazine applied in the UK was for non-agricultural purposes. In contrast simazine is used principally in orchards and for peas and beans, which are not extensively cropped in Essex, hence the lower levels compared with atrazine (see Fletcher *et al* 1994).

Elevated concentrations of some pesticides, including the DDT group, were also observed in saltmarsh sediment near Southend (Figure 29), indicative of larger inputs to the Thames, perhaps from STW and landfill. Dieldrin and γHCH were more evenly distributed across the Essex saltmarsh sites shown in table 13, suggestive of diffuse agricultural origins. Clearly, though, the number of Essex sites sampled is inadequate for a rigorous assessment of sources. If no recent data exists, it is recommended that a small sediment sampling programme be conducted incorporating some of the priority compounds such as γ-HCH, DDT isomers, 'drins', herbicides and perhaps other selected organic micropollutants.

Table 13. Concentrations of pesticides, herbicides and other organic micropollutants in saltmarsh sediments (adapted from Leggett et al., 1995; Scrimshaw et al., 1994, 1996; Scrimshaw and Lester, 1996)

Compound	Sediment concentration µg kg^{-1}					
	S. Woodham Ferrers	Dengie	Tollesbury	Salcott	ISQG[a]	PEL[b]
ALDRIN	<0.1 - 1	<0.1 – 1.3	<0.1	<0.1 – 0.2		
DIELDRIN	<0.1 – 1.4	<0.1 – 0.7	<0.1 – 0.8	<0.1 – 1.2	0.71	4.3
HCH GAMMA	<0.1 – 0.5	<0.1 – 2.2	<0.1	<0.1 – 0.8	0.32	0.99
DDT (OP+PP')[c]	<0.1 – 4.3	<0.1 – 17.9	<0.1 – 0.5	<0.1 - 1	1.19	4.77
MCPP: MECOPROP	<0.1 – 31.4		<0.1 – 5.8	<0.1 – 13.4		
2,4 D[d]	<0.1 – 177		<0.05 – 44.9	<0.05 – 14.7		
ATRAZINE	<0.1 – 15.3		<2 – 44.3	<1		
SIMAZINE	<2		<2 – 10.1	<2		
PCB TOTAL	<1 – 5.2	1 - 242	<1- 2.2	1 – 5.8	21.5	189

[a] interim marine sediment quality guidelines (ISQG) and [b]probable effects levels (PEL) (CCME, 1999)
[c] guideline = pp' DDT;

6.1.4 PCBs

Of several hundred analyses of PCBs (congeners 28,52,101,118,128, 138,153,156, 170, 180) performed on tidal water samples by EA between 2000 and 2005 the vast majority (>99%) were below detection limits of ~ 0.001 µg l^{-1} (Table 14). This reflects the somewhat hydrophilic nature of PCBs. Of the few positive values the highest was 0.016 µg l^{-1} congener 138 (measured in the Strood Channel in 2002).

Table 14. Summary of total numbers of samples analysed for PCBs and proportions below detection limits (data source EA, 2000-2005)

Determinand		No. samples	values above DL	median µg/L	max µg/L
PCB CONGENER 028	µg/L	647	0	0.0010	0.0100
PCB CONGENER 052	µg/L	646	0	0.0010	0.0100
PCB CONGENER 101	µg/L	647	0	0.0010	0.0100
PCB CONGENER 105	µg/L	102	0	0.0010	0.0010
PCB CONGENER 118	µg/L	647	0	0.0010	0.0100
PCB CONGENER 128	µg/L	102	0	0.0010	0.0010
PCB CONGENER 138	µg/L	640	3	0.0010	0.0161
PCB CONGENER 153	µg/L	647	1	0.0010	0.0100
PCB CONGENER 156	µg/L	102	2	0.0010	0.0010
PCB CONGENER 170	µg/L	102	2	0.0010	0.0010
PCB CONGENER 180	µg/L	638	2	0.0010	0.0100

Arguably, because of the large number of 'less than values' and absence of positive data, the risk of impact from PCBs, at least in the aqueous phase, would be anticipated to be low.

Because most medium- highly chlorinated congeners partition strongly to particulates it may be preferable to monitor PCBs in the sediment phase. However, there are no recent results for PCBs in sediments in the EA database for Essex Estuaries. Rigorous assessment of recent distributions within the EMS is therefore not feasible.

The limited published information on sediment-bound PCBs in the area includes NMMP sites, together with the rather old data (1991-95) for the four Essex saltmarsh systems listed in table 13; a longitudinal transect along the Thames (1991); and for the outer Thames estuary some 15km of the Foulness coastline (1990). At the latter location concentrations were below detection (<0.2 $\mu g\ kg^{-1}$) in sands, increasing to 0.81$\mu g\ kg^{-1}$ in muds (expressed as the sum of individual congeners, 4.3$\mu g\ kg^{-1}$ expressed as Arochlor 1254). These muds were classified as being 'slightly contaminated' under the OSPAR criteria at the time (MAFF, 1993). More recent provisional ecotoxicological assessment criteria are set at 1-10 $\mu g\ kg^{-1}$ (for the ICES 7 congeners) and in the 1999-2001 NMMP surveys of sediments (<2mm fraction) the Thames (24.6 $\mu g\ kg^{-1}$) was amongst the highest of the UK sites sampled. This value would marginally exceed ISQG/TEL for sediments. The earlier transect along the Thames Estuary (Cefas,1997) showed that concentrations in sediments increased upstream, from relatively low concentrations outside the mouth (~ 1.2$\mu g\ kg^{-1}$ for the ICES 7 congeners) to a maximum at Grays (132 $\mu g\ kg^{-1}$). The majority of samples in the estuary were classified as moderate-heavily contaminated. No sediments from the EMS were sampled, however.

Data for the four Essex saltmarsh sites suggest that low level contamination of sediments with PCBs was widespread a decade ago (table13). Guidelines were occasionally exceeded (table13), most notably so at Dengie (Bridgewick Farm). Since these sites are primarily in agricultural areas, and the most significant source is likely to be sediment deposition, it is likely that some of these PCB loadings originated from mass movement from the outer Thames Estuary including the sludge dumping grounds in the Barrow Deep and Black Deep, which are several km off the Dengie peninsula (Legget *et al*., 1995, Scrimshaw *et al* 1996). Elevated PCB concentrations in saltmarsh sediments from the Thames Estuary at Two Tree Island, near Southend, (Figure 29) were also thought to be indicative of inputs from STW and trade wastes in the area, and upstream (supported by results from multivariate statistical treatment of the data by Scrimshaw and Lester, 2001). Source estimates for UK PCB inputs to the southern North Sea made more than 10 years ago showed that sludge dumping to the North Sea (half of which was off the Thames), and directly from the Thames estuary itself, accounted for 21-577 kg a^{-1} and 47-63 kg a^{-1}, respectively (Klamer *et al*., 1991).

Estuarine sites in the Blackwater and Crouch are less likely to be impact by movements of contaminated sediments from the outer Thames and offshore as they are ebb-flow dominated systems, with little marine sediment input (Scrimshaw *et al*., 1996). We have no recent information on PCB sources, nevertheless, one-off, contemporary monitoring of these persistent compounds in the Essex estuaries seems justified to establish recent distribution patterns and risks from PCBs in the system.

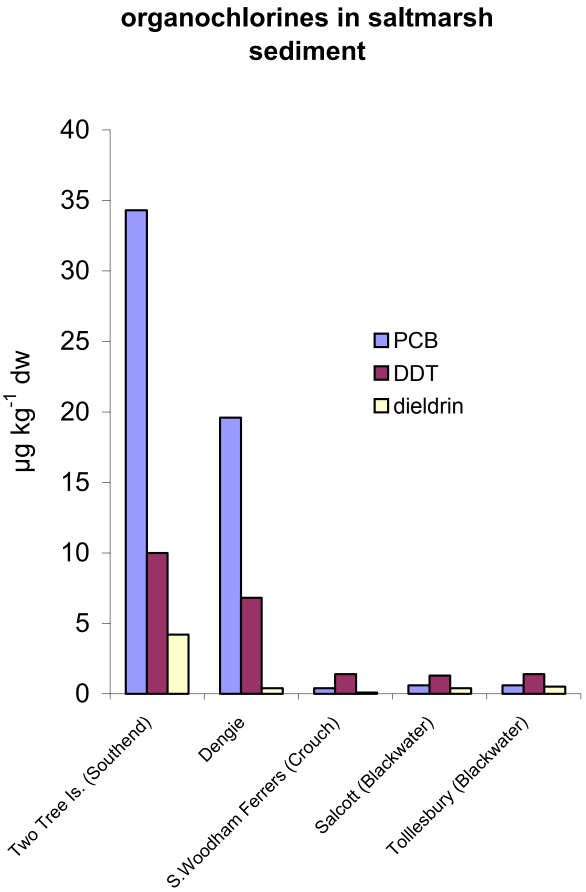


Figure 29. Mean values of organochlorines in Essex saltmarsh sediments, 1990-1995 (data source, Scrimshaw and Lester 2001).

6.1.5 Alkylphenols and other endocrine disruptors

Alkylphenols

Alkylphenol polyethoxylates (APEs), based mainly on nonyl- and to a lesser extent, octyl-phenols, are non-ionic surfactants which have been used for domestic and industrial purposes for more than half a century and are suspected of some involvement in endocrine disruption (see review by Langston *et al.,* 2005). Some 6000 tonnes of APEs enter UK rivers and estuaries annually from various sources including sewage outfalls, industrial discharge and agricultural run-off (Blackburn *et al.,* 1999). Currently this broad group of compounds are utilized in paper and pulp mills, textile processing, resin and paint manufacture and other chemical applications. Up until 1996 they were also a component of waste discharged in to the sea from off-shore oil/gas production platforms, where they were used in lubricant and surfactant formulations (Olsgard & Gray, 1995). About 33% of APEs are (or were, until recent voluntary EU restrictions) employed as ingredients in household and industrial detergents which are mainly disposed of into the sewer system and are biodegraded in the environment or in sewage treatment works, via stepwise loss of ethoxy groups to more hydrophobic, recalcitrant metabolites including nonylphenol (NP), octylphenol (OP). The partition coefficient for these alkyl phenols is of the order of 10^3 indicating moderate affinity for sediment, though they are likely to occur in both solid and dissolved phases.

Alkylphenols are therefore common contaminants in STW and may be present in $\mu g\ l^{-1}$ concentrations in effluent samples and $mg\ g^{-1}$ quantities in sludges and sediment. In the vicinity of discharges (particularly in rivers) concentrations may reach several hundred $\mu g\ l^{-1}$ – levels sufficient to threaten native organisms (Blackburn and Waldock, 1995). At highly contaminated sites such as the lower Tees estuary, total extractable APE (intermittently >80$\mu g\ l^{-1}$) are implicated as contributing to poor water quality in acute toxicity bioassays with marine copepods *Tisbe battagliai* and oyster embryo *Crassostrea gigas* (Thomas *et al.*, 1999, 2001). At the sublethal level, though having less strong oestrogenic properties than natural hormones, APs have been shown to induce vitellogenin (VTG) synthesis and testicular abnormalities in male eelpout (Christiansen *et al.*, 1998) and inhibit the settlement of barnacle larvae (Billinghurst *et al.*, 1998), together with other effects (see review by Langston *et al.,* 2005) . These sublethal effects occur at levels of ~1 $\mu g\ l^{-1}$ i.e. at concentrations sometimes observed in the field. Both OP and NP have been included in Water Framework Directive lists of priority substances. An Environmental Quality Standard (EQS) of 1 $\mu g\ l^{-1}$ (as annual average; maximum allowable concentration 2.5 $\mu g\ l^{-1}$) has been set for NP and OP to control these substances in European waters. For NP mono and di-ethoxylates, MACs of 3.3 and 4.3 $\mu g\ l^{-1}$ (combined value of 7.6 $\mu g\ l^{-1}$).

No published results could be found for Essex estuaries though data from the 1990s from Southend imply levels in nearby coastal waters are comparatively low. At Southend, NP was below the detection limit of 0.2 $\mu g\ l^{-1}$ in water and 0.1 $\mu g\ g^{-1}$ in sediment (Blackburn *et al.,* 1999). Water samples from the Barrow Deep off the Essex Coast in 1998 contained NP in the range 0.1-1.4 $\mu g\ l^{-1}$, NPE in the range 0.83-1.0 $\mu g\ l^{-1}$ and OP in the range 0.01-0.02 $\mu g\ l^{-1}$ which were below the EQS (CEFAS, 2001). Comparable concentrations (0.076, 0.85 and 0.06 $\mu g\ l^{-1}$ for NP, NPE and OP, respectively), were reported in samples collected in 1999 from Thames Wharf (NMMP site 465 in the outer Estuary, off Foulness) by Cefas (2003a).

Alkylphenol concentrations in sediments (<2mm fraction) from the Barrow Deep, Thames Wharf and Thames Disposal site G9 (off the Essex coast) were largely below detection limits (<0.19 $\mu g\ g^{-1}$ NP, <1.0 $\mu g\ g^{-1}$ NPE and <0.01-0.06 $\mu g\ g^{-1}$ OP) (Cefas, 2001, 2003a) and, for NP, fell below the PNEC[19] value of 2 $\mu g\ g^{-1}$ proposed by Environment Canada. However, these detection limits were higher than suggested PNECs from sub-lethal tests, which may be as low as 0.039 $\mu g\ g^{-1}$ and 0.0065 $\mu g\ g^{-1}$ for NP and OP, respectively (Duft *et al*, 2003 Zulkosky, *et al* 2002).

Bisphenol A

Bisphenol A (BpA) is used in plastics manufacture and a variety of other applications and, like APs, has been suspected of inducing oestrogenic effects in marine fauna. In 1999, low concentrations (0.06-0.08 $\mu g\ l^{-1}$) were found in water samples from Thames Wharf and other sites off the Essex coastline, whilst levels in sediments were below detection limits (< 0.03 $\mu g\ g^{-1}$dw, Cefas, 2003a). There is no EQS for BpA and a PNEC of 1.6 $\mu g\ l^{-1}$ has been derived from conventional ecotoxicological tests.

[19] Predicted no effects concentration

However a LOEC of 1.0µg l^{-1} has been suggested, based on sperm development in fathead minnow, resulting in a tentative PNEC revision down to 0.1 µg l^{-1}: offshore samples would fall below this conservative threshold, though not by a large margin.

Phthalates and phthalate esters

These are difficult compounds to analyse because of widespread contamination and are not routinely monitored, though they include some compounds which have been suggested to induce endocrine disruption. Data for water, sediments and biota from the Crouch Estuary have therefore been included here, despite being dated (Waldock, 1983). The most ubiquitous compounds were di-iso-butyl (DiBP), di-n-butyl (DnBP) and di-2-ethylhexyl (DEHP) phthalate and their distributions are summarized in figure 30. Of these, DEHP was present in highest concentrations, notably in digestive glands of the deposit-feeding clam *Scrobicularia plana* (and to a lesser extent oysters), implying that sediments/food may be an important pathway for bioaccumulation of these compounds in estuarine biota. There was no indication of biomagnification in these results, as dab tended to have lower concentrations than molluscs; this may be the result of higher rates of xenobiotic metabolism in the fish. Compared to other published data on these compounds, concentrations for the Crouch do not appear to represent a major threat. Environmental standards are not available for phthalates (DEHP is an EU priority chemical under review), therefore it is not possible to describe the risk status. To do so would require more recent data to compare with effects thresholds.

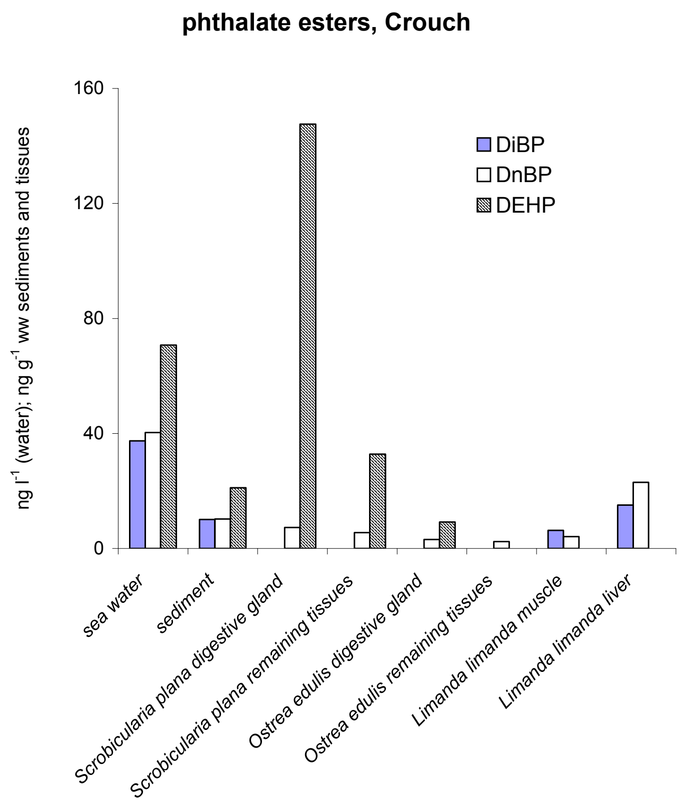


Figure 30 . Phthalate esters in samples from the Crouch Estuary (data from Waldock 1983). Note; only values significantly above blanks have been used.

Oestrogens

Although oestrogen levels are generally recognised as being biologically active near to some STW sources, their analysis is difficult and there are no data on concentrations in natural waters of the EMS,. Published values for oestradiol in the middle section of the Thames estuary ranged from <0.03–7 ng l^{-1}; estrone (E1) tended to be present at slightly higher concentrations close to sewage discharges (up to 17 ng l^{-1}) (Xiao *et al*. 2001). These values coincide with a small degree of VTG levels in male flounder sampled from Gallions Reach area of the Thames, though cause and effects have not been established unambiguously.
In summary, endocrine disruption in flounder from the outer Thames (and Crouch) has been recorded by Cefas but at relatively low levels, and in the absence of other evidence, it is unlikely that ED compounds are having major deleterious effects on reproduction in this species (see section 7). However, the database supplied by EA (2000-2005) contain no information on APEs, phthalates or other (xeno)estrogens near to sources - where effects are most likely to occur - and it may be advisable to sample more sites within the estuaries of the EMS (freshwater inputs, discharges, tidal waters and sediments) in conjunction with more extensive surveys of endocrine disruption.

Pharmaceutical compounds

Pharmaceutical compounds are an emerging concern since many (e.g. antibiotics) are used in quantities similar to those of agrochemicals and, consequently, this has led to their detection in STW. No published records could be found for the Essex Estuaries EMS; however, a number of compounds have been measured in ng l^{-1} quantities in water samples from the Thames Estuary (CEFAS, 2005a), including Mefenamic acid, Diclofenac, Clofibric acid, Clotrimazole and Ibuprophen (up to 928 ng l^{-1}), presumably discharged mainly from STW (e.g. Beckton and Crossness). Similar concentrations ranges can be found in other major urbanised/industrialised estuaries. As yet there is little understanding of impacts on aquatic life, or concentrations in sediments, and until this information is forthcoming risks from this group of contaminants are impossible to quantify.

6.1.6 Hydrocarbons (oil, petrochemicals, PAHs) and VOCs

Oil

Oil pollution is a continual threat to all inshore marine habitats, particularly estuaries due to their enclosed and sheltered nature. Risks include small leaks, spills and discharges, as well as the possibility of a major accident.

There are a number of ways in which oil could potentially impact on the interest features of the EMS. Inter-tidal habitats are under greatest threat from the physical effects of oil pollution: the most vulnerable of these are inter-tidal sand and mudflats and salt-marshes (see reviews of vulnerability of shores to oil damage by Gundlach and Hayes, 1978; Elliott and Griffiths, 1987).

Oil pollution may result in hydrocarbons becoming incorporated into sediments and buried. Heavily contaminated sediments are likely to have acute effects on populations of bottom-dwelling fish such as plaice and sole (in extreme cases, such as the Amoco Cadiz spill, whole year classes were wiped out over large areas of northern Brittany). Sub-lethal changes could be manifested as increased bioaccumulation and tainting, induction of components of the mixed function oxidase (MFO) enzyme system (e.g. ethoxyresorufin-O-deethylase – EROD - activity), and higher order changes in productivity, fecundity and behaviour. Olfactory responses in crustacea can affect their searching, feeding and grooming responses. Fish behaviour, including migration, is also affected by oil spills with most fish avoiding areas of heavy contamination. Eggs and planktonic larval stages of fish, molluscs and crustacea are also vulnerable to contact with oil in surface waters.

Sensitivity of *Zostera* beds to chronic exposure to oil, such as refinery effluent may not be very high (Hiscock, 1987). The likely impact of acute exposure (oil spillage) will be influenced by the type of oil, the degree of weathering and the nature of the habitat and in general, it is the associated faunal communities that are more sensitive to oil pollution than the *Zostera* plants themselves (Jacobs, 1980, Zieman 1984, Fonseca, 1992). Unfortunately, dispersants are likely to be more harmful to *Zostera* than oil, therefore it is generally considered that coated plants should be left untreated. The hydrocarbons present in crude oil can range from aliphatic (straight chain) compounds to more complex aromatic (containing a benzene ring) and polynuclear aromatic (containing two or more benzene rings) compounds. Processed products include petrol and diesel and a range of petrochemicals, e.g. propylene, acetylene, benzene, toluene and naphthalene. In addition to shipping, sources to the marine environment also include river-borne discharges, (including road runoff and licensed and unlicensed discharge to sewers) diffuse discharges from industrialised municipal areas, oil production sites and the atmosphere (PAH’s).

There are no EQS values for hydrocarbon oils in estuarine waters *per se*. Two directives list criteria which can be used as general guidance; the Bathing Waters Directive, under the heading organic substances: 300µg l^{-1} as the 90^{th} percentile (non-routine sampling prompted by visual or olfactory evidence of hydrocarbon presence); and the Shellfish Waters Directive listed under organic substances, which states that ‘hydrocarbons must not be present in such quantities as to produce a visible film on the surface of the water and/or a deposit on the shellfish, or to have harmful effects on the shellfish’. Also under the Shellfish Waters directive, hydrocarbon contamination is (presumably) included in ‘general physico-chemical parameters’ – tainting substances – where ‘the concentration of substances affecting the taste of shellfish must be lower than that liable to impair the taste of the shellfish’.

These EQS guidelines for Shellfish waters are obviously difficult to quantify, however tainting (an odour or flavour foreign to the product) can occur in commercial species contaminated with crude and refined oils. Species with a high body fat content such as salmon or herring are more easily tainted and retain the taint for longer than lean-muscle species. GESAMP (1993) report studies detecting taints in fish and macro-crustaceans resulting from exposure during acute incidents, chronic discharges and in experimental studies. There are no accepted permissible standards in organisms. In some instances, hydrocarbons may be present at well above background levels, even though no taint can be detected. Conversely fish can be tainted where analysis indicates that contamination is only at background levels. Experimental studies indicate that taints can be detected when fish are exposed to concentrations of oil in water in the range 0.01 to 1mg l^{-1}. Tainting can occur very rapidly on exposure (within a few hours at concentrations of oil above 1 mg l^{-1}), and fish have been shown to lose their taint within 1 to 4 days (experimental study on cod). However, field studies have indicated that fish may still be tainted days or weeks after a spill of fuel oil (GESAMP 1993). Because fine sediments absorb and retain oil, infaunal species such as clams and burrowing shrimps, and some demersal fish may be at risk of tainting on a more prolonged basis.

Surprisingly little published information on hydrocarbon oil concentrations in Essex Estuaries EMS could be found in the literature. Total hydrocarbons (THCs) in surface waters are generally low and there are very few recent measurements of THC levels in sediments. THCs in sea bed sediments from Barrow Deep in 1990 contained 63 µg g^{-1} dry wt as Ekofisk crude oil equivalents and would have exceeded OSPARs guideline 'no observable effects' concentration of 10 µg g^{-1} dry wt (MAFF, 1992). To put this in perspective, however, concentrations in the UKs most heavily contaminated estuarine sites such as the Tyne were and order of magnitude higher. Without more recent and more extensive data it is not possible to assess risk from oil to the site though, clearly, the Essex Estuaries may be vulnerable to accidental releases, given the large concentration of commercial shipping using nearby shipping lanes.

Median total petroleum hydrocarbon (THC) concentrations in unspecified invertebrates (mussels?) from Essex Estuaries are reported by Allen *et al* (2000) to be 12,500 µg kg^{-1} wet weight, and 11,450 µg kg^{-1} wet weight in sediments. On face value these concentrations are very high, though without any indication of species, sampling location, and in view of the small sample size (n=6 for 'invertebrates'; n=1 for sediment), little further comment can be made.

Polycyclic aromatic hydrocarbons (PAHs)

PAH's are ubiquitous environmental contaminants, estimated to constitute some 8% by weight of the total hydrocarbon composition (Kirby *et al*., 1998). Although they can be formed naturally (e.g. oil seeps, forest fires) the predominant source of PAHs is often anthropogenic emissions, and the highest concentrations are generally found around urban centres (Cole *et al*., 1999). In the aquatic environment, PAH levels are generally highest in sediments, followed by biota and lowest levels are found in the water column. They are of particular concern in the marine environment as the lower molecular weight PAHs are toxic to marine organisms, and metabolites of the higher molecular weight PAH's are carcinogenic (Law *et al*., 1997). PAH concentrations in sediments may be persistent (particularly where tidal action is restricted, and degradation limited by anoxia). Elevated levels have been linked to liver neoplasms and other abnormalities in demersal fish (Malins *et al.,* 1988). In addition, some PAHs have been identified as endocrine disruptors (Anderson *et al*., 1996a,b; Kocan *et al.,* 1996).

Based on observed environmental behaviour, physical and chemical properties, microbial degradation rates and statistical analyses, PAHs are divisible into two groups: Group 1, or low molecular weight (≤200) PAHs (including naphthalene, phenanthrene and anthracene) have a low affinity for particulates and are subject to microbial degradation. Their solubility and vapour pressure is higher than group 2 PAHs, and photo-oxidation and air-water exchange are important in estuaries. Consequently group 1 PAHs tend to have comparatively shorter residence times and often exhibit a complex distribution pattern. In contrast, group 2 or high molecular weight (≥200) homologues (including benzo(a)pyrene, fluoranthrene, pyrene and chrysene), are readily adsorbed onto particulates.

There are few published papers relating to PAHs in Essex Estuaries EMS. CEFAS (1998) carried out a pilot study of PAHs in seawater around the UK and found that concentrations were generally undetectable at intermediate or offshore stations, but significant concentrations were present in a number of estuaries. Although none of the component Essex Estuaries were included in this survey, data is given for two NMMP sites in the outer Thames Estuary (West Thurrock and Mucking). Figure 31 shows levels of PAHs in total (unfiltered) and dissolved (filtered) water (duplicated samples) from these two sites.

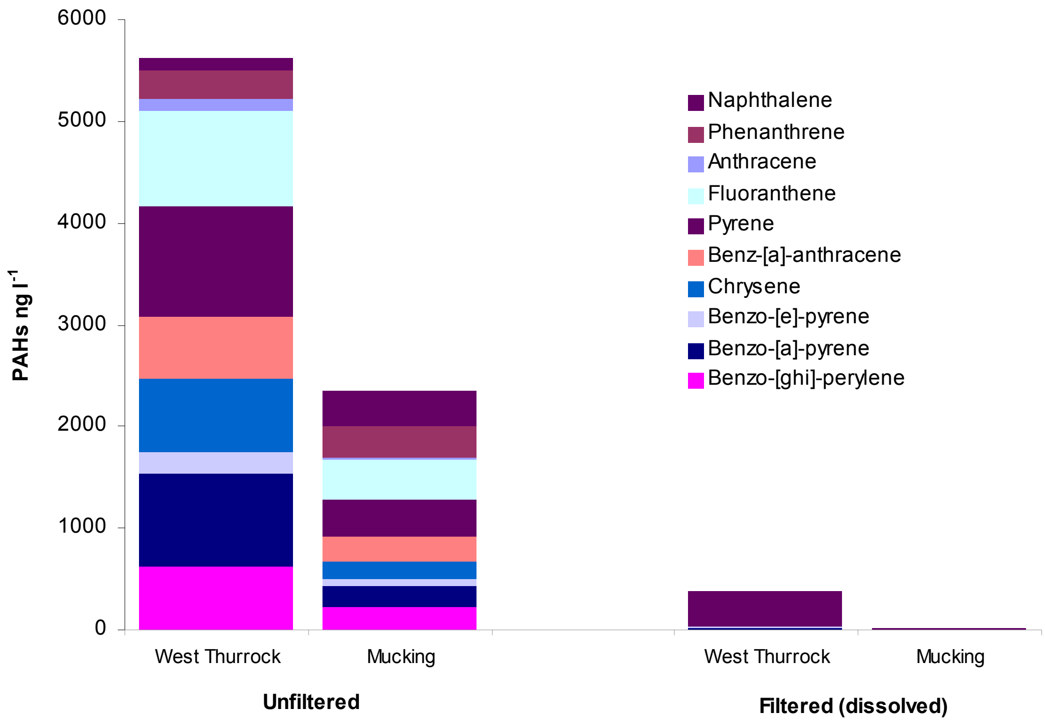


Figure 31. PAH concentrations (µg l^{-1}) in water samples collected in 1993 from West Thurrock and Mucking NMP sites in the outer Thames (data from CEFAS, 1998).

It is immediately obvious from figure 31 that the majority of PAHs are present in unfiltered water i.e. adsorbed to particulates. Furthermore, it is predominately the lower molecular weight PAHs (naphthalene, pyrene, anthracene) that are present in dissolved form whilst high molecular weight PAHs are adsorbed to particulates (CEFAS, 1998). This may have implications for the estuaries of the EMS, in that waterborne PAH-rich particulates may be transported into the site by tidal movement and remain due to sedimentation.

Perhaps since many PAHs have such an affinity for particulates, any existing information regarding concentrations of PAHs in the marine site are generally for concentrations in sediments, which tend to be much higher than those in water. Proposed guidelines for ΣPAH in sediments are shown in table 15.

Table 15. Proposed guidelines for ΣPAH in sediments µg kg^{-1} dry weight .

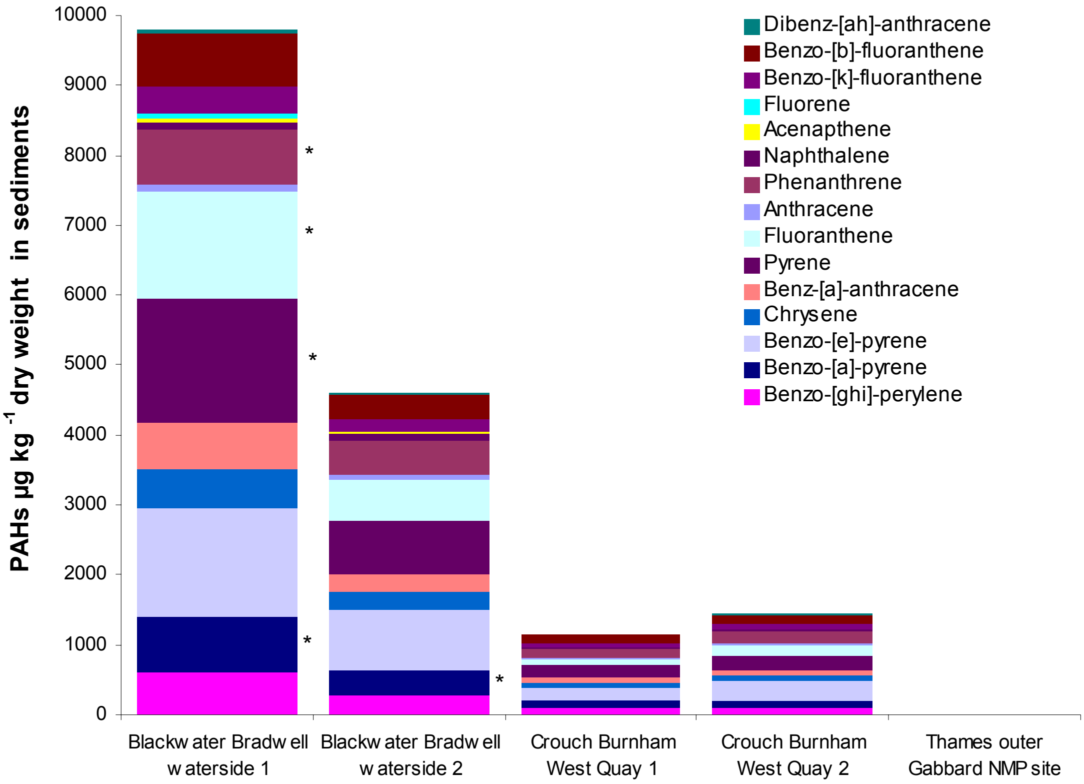
	TEL Threshold effect level*	PEL Probable effect level*
Naphthalene	34.6	391
Acenapthene	6.7	89
Fluorene	21.2	144
Phenanthrene	86.7	544
Anthracene	46.9	245
Fluoranthene	113	1494
Pyrene	153	1398
Benz-[a]-anthracene	74.8	693
Chrysene	108	846
Benzo-[e]-pyrene	-	-
Benzo-[b]-fluoranthene	-	-
Benzo-[k]-fluoranthene	-	-
Benzo-[a]-pyrene	88.8	763
Benzo-[ghi]-perylene	-	-
Dibenzo(a,h)anthracene	-	-
Indeno (1,2,3-cd)-pyrene -	-	
Total PAH	1684	16770

- Not set
*Thresholds from Macdonald *et al.,* (1996). Data for English Channel from Woodhead *et al.,* (1997).
N.B. TELs and PELs shown are the same as interim marine sediment quality guidelines (ISQGs) and probable effect levels (PELs) where they have been set by CCME (1999).

A CEFAS survey, carried out between 1993 and 1996, analysed PAHs in surface sediments at numerous UK coastal, estuarine (including the Blackwater and Crouch estuaries) and offshore sites (CEFAS, 1998;Woodhead *et al.,*1999). Results for EMS sites are exemplified in figure 32, which includes results for the (coastal) Thames outer Gabbard NMMP site for comparison. At the outer Thames site, PAH concentrations were below detection limits. Almost without exception the reported values for individual PAHs in both the Blackwater and Crouch estuaries are above TELs (where set). Phenanthrene, Fluoranthene, Pyrene and Benzo(a)pyrene exceed PELs at the Blackwater (Bradwell Waterside 1) site, and Benzo(a)pyrene exceeds PEL at the Blackwater (Bradwell Waterside 2) site. PEL for ΣPAHs is not exceeded at any of the sites, however, TEL for ΣPAHs is exceeded at both Blackwater Estuary sites.

With the exception of the Blackwater (Bradwell Waterside 1) site, ΣHMW PAHs were dominant, with benzo(e)pyrene present in highest concentrations. ΣLMW PAHs, notably pyrene, were dominant at the Blackwater (Bradwell Waterside 1) site, which might indicate a local source, as low molecular weight PAHs tend to have shorter residence times in the environment. However, benzo(e)pyrene and significant levels of other HMW PAHs were also present at the 2nd site. It is interesting that in both

Blackwater and Crouch Estuaries, the two samples were taken 1m apart, yet PAH concentrations differ (considerably for the Blackwater samples). Woodhead *et al.*, (1999) note that although both sediment samples were visually muddy, closer inspection showed that the Blackwater 2 sample also contained some coarser particulates, suggesting a grain size effect due to differences in adsorptive surface area.



* signifies concentrations above Probable Effects Levels (PEL)

Figure 32. PAH concentrations (µg kg^{-1} dry weight) in sediment samples collected September 1996 from The Blackwater and Crouch Estuaries, and June 1994 from the Thames outer Gabbard NMP site (data from Woodhead *et al.*,1999).

To put these results into some context, PAH data (from the same survey) for sediments from within some other UK SACs/SPAs, are shown in table 16, ordered by maximum concentration of ΣPAHs. Levels of ΣPAHs in the Blackwater Estuary are highest recorded of these sites, on the other hand they do not approach maximum levels found in some industrialised estuaries and ports: sediment collected from Milford Haven in 1996 contained concentrations of 102,471 µg kg^{-1} ΣPAHs (Woodhead *et al.,* 1999), although is probably exceptional following the grounding of the oil tanker Sea Empress in 1996.

Data from a pilot study of PAHs in sediments around the UK are reported as part of the National Marine Monitoring Programme (MPMMG, 1998). Concentrations range from undetectable at offshore sites (such as the central English Channel) to 43,470µg kg^{-1} dry weight on the River Tyne at Hebburn. Within this range, ΣPAH concentration in sediments from Essex Estuaries sites would probably classify as moderate.

However, carcinogenic and mutagenic intermediates, and the formation of PAH-DNA adducts may arise in fish through the actions of cytochrome P450. Parts of this enzyme system (e.g ethoxyresorufin-O-deethylase – EROD – activity) can be induced in fish exposed to sediments containing ΣPAH concentrations between 1000-3000 µg kg^{-1} dry weight, and possibly lower (Payne *et al* 1988; Woodhead *et al*., 1999) – i.e. within the range encountered in at least two of the component estuaries, and notably less than half of the reported concentration in the Blackwater Estuary. Similar levels have also been shown to reduce growth of larval fish, presumably weakening their resistance to disease and other stressors (Misitano *et al.,* 1994). Less is known of the toxicity to invertebrates, though sub-lethal effects to crustacea, perhaps exacerbated by UV, have been observed at 1000-6000 µg kg^{-1} dry weight (Alden and Butt, 1987; Ankley, *et al* 1994).

Table 16. PAH concentrations µg kg^{-1} dry weight in sediment samples from sites in UK SACs/SPAs collected between 1993 to 1996 (CEFAS 1998)

	Individual PAH range µg kg^{-1} dw			Total PAH range µg kg^{-1} dw	
	From	Up to	Sediment type	From	Up to/max
Blackwater	**<48**	**1760**	**mud/coarse mud**	**4603**	**9793**
Plymouth Sound & Est.	<15	1129	mud/ s,st,sh	3010	7410
Mersey	<4	1242	mud	6	6230
Exe	<1	876	mud/sand	nd	5889
Severn	37	843	mud	5425	5472
Humber	<1	981	mud/sand	132	7041
Poole Harbour	<1	270	mud	624	1694
Crouch	**8**	**260**	**mud**	**1153**	**1439**
Solent (east)	<1	105	mud	-	398

In a partial validation of the TEL/PEL procedure, Long *et al* (1998) showed that as the number of PEL excedences in sediments increases, so does the proportion of samples which are highly toxic to amphipods. Generally it is sediments of the highly industrialised estuaries on the north-east coast of England that are likely to contain acutely toxic concentrations of PAHs. Sediment bioassays using the amphipod *Corophium volutator* and polychaete *Arenicola* marina support this prediction (Thain *et al.,*1996; Matthiessen *et al.,*1998) although PAHs are not thought to be the single cause of this toxicity. However, Woodhead *et al*, (1999) consider that the more widespread excedences of TELs found in other UK estuaries during the 1993 – 1996 study suggested a more general problem of chronic PAH toxicity, and applied the EqP model developed by Schwartz, *et al.,*(1995) to predict total toxic units in sediments at some of the UK sites.

27 sites (27% of the total number of sites in the survey) were identified where the predicted toxic units were calculated to be 0.1 or greater (assuming 1.75% organic carbon content). In terms of biological significance, toxic units of >1 would be expected to elicit >67% amphipod mortality, >0.5 toxic units >30% mortality and >0.1 toxic units >9% mortality. Predicted PAH toxic units for the Blackwater (Bradwell waterside) 2 site were 0.22, therefore amphipod mortality was predicted to exceed 9%. Of the 27 sites, sediments of the Blackwater (Bradwell Waterside 2) site ranked as 10th most toxic (Woodhead *et al*, (1999).

PAHs are generated by two principal processes, combustion of organic matter at high temperature (pyrolitic) and the release of petroleum (petrogenic). Chromatograph profile of PAHs mixtures isolated from shellfish can be indicative of these sources, particularly when used in conjunction with ttr-aromatic steroid biomarker compounds present in crude and some refined oils (Bence *et al.,*1996; Boehm *et al.,*2001). For instance, pyrolitic PAHs are characterized by a dominance of unsubstituted PAHs with a wide range of molecular weights, whereas petrogenic PAHs are dominated by low molecular weight and alkylated PAHs.

CEFAS (2003a) reported on profiles of PAH isolated from bivalve molluscs from various shellfish harvesting areas in the UK and found that in the majority of cockle samples, PAH profiles indicated roughly equal quantities of both combustion and petroleum- derived PAHs. However, in cockles from the Thames Estuary area and off the East Essex coast (non-specified sites), profiles showed a higher proportion of petrogenic PAHs. This was considered to be partly due to lower concentrations of combustion-derived PAHs with distance from the source, rather than an increase in petrogenic PAHs. A list of possible sources in this area included shipping, several oil and coal-fired power stations along the Thames, oil refineries/terminals at Thurrock and Canvey Island and a variety of industrial sources (chemical and oil- and gas-related industries and metal works) (CEFAS, 2003a).

Irrespective of assimilation pathway, PAHs, because of their affinity for organic matter, have a tendency to be accumulated by organisms. Set against this, the MFO system of certain taxonomic groups is capable of metabolism and elimination of some of the smaller molecules (though as indicated above this does not make the hosts immune to effects). Fish and polychaetes are among the most efficient PAH metabolisers (tissue concentration factors generally less than 500) whilst in crustacea and bivalves metabolism is slow, particularly for larger PAHs (concentration factors 3000-6000).

Bivalves are therefore probably better indicators of PAH distributions (though not necessarily biological effects) compared with fish, and are of particular relevance in the Essex Estuaries marine site in view of the various designated shellfish waters within the site (see annex 6). To date EA monitoring of shellfish in the marine site has been limited to measurement of PAHs in mussels from the NMMP shellfish monitoring site at West Mersea Beach (see below).

Recent EA Monitoring Data

EA Monitoring data includes data for 'oil and grease visible' for the period 2000 to 2005. The results are given in terms of present or absent, presumably to comply with the criteria for shellfish waters regarding 'visible film' (see annex 4). Monitoring of water takes place at 15 sites within the EMS, including the NMP shellfish monitoring site at West Mersea Beach - opposite Fairhaven Ave, (for designated shellfish areas see annex 6). During this period visible oil and grease was only recorded on one occasion - in the Roach at the east end of Paglesham in 2000.

'Tars and floating material' were also monitored during 2000 – 2002 at two sites, the NMMP site and Brightlingsea Beach at Batemans Tower, with no positive results.

With the two exceptions above, there are no available EA data for hydrocarbons or PAHs in water or sediments of the EMS. Data has not been made available for concentrations in freshwater inputs, STW or industrial discharges therefore it is not possible to attribute any high levels directly to sources.

Table 17. PAH concentrations in mussels (µg kg^{-1} ww) from the NMMP shellfish monitoring site at West Mersea Beach (2000 – 2004). Data source: EA

			PAH concentration µg kg^{-1} ww		
	n	%less than values	Median	Minimum	Maximum
Indeno (1,2,3-cd) pyrene	30	100	0.50	0.50	1.00
Benzo (ghi) perylene	30	73	0.50	0.50	1.73
Fluoranthene	30	0	6.73	2.85	12.20
Phenanthrene	30	40	3.36	0.50	6.18
Naphthalene	12	67	2.75	0.50	18.00
Anthracene	30	100	0.50	0.50	1.00
Pyrene	30	0	6.42	2.25	11.70
Benzo (a) pyrene	28	86	0.5	0.5	0.8
Chrysene	30	20	3.3	0.5	4.6
Benzo (a) anthracene	30	7	1.2	0.5	3.1

EA data for selected PAHs in mussels from the NMMP shellfish monitoring site at West Mersea Beach are summarised in table 17. There are numerous 'less than' values in the data (some of which vary over the period) making it difficult to present an accurate assessment of any shellfish PAH contamination e.g. the 'less than' value of 1.0 µg kg^{-1} ww for anthracene is above the provisional ecotoxicological assessment criteria (EAC) value of 0.75 µg kg^{-1} ww for anthracene in mussels (OSPAR, 2000). These data do not suggest any significant PAH contamination of shellfish at this site. However, monitoring at other shellfish sites would appear to be an urgent requirement.

Provisional EAC for PAHs in mussels (from OSPAR, 2000) are shown in table 18.

Table 18 Provisional ecotoxicological assessment criteria (EAC) for PAHs in mussels (µg kg^{-1} dry weight and as a wet weight estimate (using a factor of 6.67). From OSPAR (2000).

PAH compound	Lower EAC µg kg^{-1}		Upper EAC µg kg^{-1}	
	Dry weight	Wet weight	Dry weight	Wet weight
Naphthalene	500	75	5000	750
Phenanthrene	5000	750	50,000	7500
Anthracene	5	0.75	50	7.5
Fluoranthene	1000	150	10,000	1500
Pyrene	1000	150	10,000	1500
Benzo(a)yrene	5000	750	50,000	7500

Volatile organic compounds (solvents, freons)

A number of volatile organic compounds (VOCs) are, potentially, endocrine disruptors, as well as being toxic, and may be discharged into the European marine site in small quantities. These include:

Trichloromethane (Chloroform)

Trichloromethane, a List I substance, is an industrial solvent used in the UK in the production of fumigants and anaesthetics. It is also a principal transformation product of chlorine-based biocide products used in cooling water systems: Trichloromethane (chloroform) is the major organohalogen formed at sites using freshwater sources, whereas bromoform predominates where water is taken from estuarine and marine sources (Cole *et al.,* 1999).

Trichloromethane has an EQS of 12µg l^{-1} (annual average) in all waters. EA data for tidal waters of the EMS (n=21) were invariably below detection limits of 0.1 µg l^{-1} (i.e. well below the EQS). However, as monitoring has only been done at one site (Stansgate, Blackwater) it is not possible to confirm that trichloromethane is not a concern for other sections of the EMS.

Tetrachloromethane (Carbon tetrachloride)

Tetrachloromethane is mostly produced for use in the manufacture of chlorofluorocarbons (CFCs) and is also used as a chemical intermediate in the manufacture of pharmaceutical and pesticide products. Tetrachloromethane production in the United Kingdom has recently ceased and its major use (production of CFC-11 and CFC-12) is now in decline. For UK marine waters, Willis *et al* (1994) reported levels to be between <0.1 – 44µg l^{-1}, with higher levels in source-dominated areas. Concentrations measured in the open ocean were generally much lower, at around 0.5 ng l^{-1}.

Tetrachloromethane is a List I compound, with an EQS of 12µg l^{-1} (annual average) in all waters. Of the data for EMS tidal waters (n=21, at Stansgate), all values were below detection limits (0.1 µg l^{-1}), i.e. below the EQS by two orders of magnitude, implying that tetrachloromethane is not likely to be a significant concern in tidal waters at this site. However, again it should be noted that sampling has not taken place elsewhere within boundaries of the EMS.

Chlorinated Ethylenes (e.g. trichloroethylene= trichloroethene)

These are List I substances produced in large quantities and widely used in industry in the production of food packaging, synthetic fibres and industrial solvents. MPMMG (1998) have indicated that concentrations of chloroethylenes in UK coastal and estuarine waters are unlikely to exceed relevant EQS (10µg l^{-1} annual average) derived for the protection of saltwater life.

Data for trichloroethylene (trichloroethene) in the EA database were examined here as being representative of this group of compounds. All 81 samples for EMS tidal waters were below detection limits (0.1 µg l^{-1}) - two orders of magnitude below the EQS -

but were restricted to the Stansgate site (Blackwater) and Rowhedge Ferry (Colne Estuary). At these two sites, trichloroethene is clearly not a concern.

Trichloroethane

Trichloroethane, a list II substance, is used as an industrial solvent. Potential sources of contamination include direct discharge of wastewaters, accidental spillages and deposition from the atmosphere. No data could be found for the Essex estuaries EMS

Brominated halogens

By-products formed during chlorination of power plant cooling water may have additional adverse effects on the growth of marine invertebrates during their larval stages (Stewart *et al.,* 1979). It has been known for some time that reproductive tissues, especially sperm, and the immature stages of organisms are sensitive to very low concentrations of organohalogens (Davis and Middaugh, 1978).

Chlorine (0.6 – 1mg l^{-1}) has been added to the cooling water of the Bradwell nuclear power station to prevent fouling by macrofauna (mussels, barnacles and hydroids) and biofilms. As this is a seawater system (high in bromine), potentially toxic bi-products, include volatile bromoform (derived from organic compounds such as fulvic acids) and, to a lesser extent, dibromoacetonitrile: concentrations of these compounds have been determined at 25 µg l^{-1} and 0.87µg l^{-1}, respectively in the Bradwell effluent stream (Jenner *et al*., 1997). Within the vicinity of the outfall this will be diluted (probably conservatively) with distance - as a function of cooling water mixing. Concentrations of bromoform were in the outfall were well below short-term toxicity values for marine algae, molluscs, crustacea and fish (12-140 mg l^{-1}) and therefore acute effects on estuarine biota are unlikely[20]. Again, the issue is probably whether or not chronic and additive impacts are possible. Since the power station was decommissioned in 2002 it may be that this procedure has now ceased.

It seems unlikely, on this evidence, that volatile organics represent a significant threat to the European marine site. However, since the number of sites sampled appears to be limited a rigorous assessment for VOCs is not possible.

6.1.7 Radioactivity

There is a consented discharge of radioactivity from a hospital in Chelmsford though the major licensed releases of radionuclides have been those from Bradwell nuclear power station. Information on monitoring of radioactivity in the region is principally documented in RIFE (Radioactivity in Food and the Environment) reports from which the following is largely derived (CEFAS, 2005b).

The Magnox power station at Bradwell at the mouth of the Blackwater Estuary has produced electricity for 40 years but ceased operating in March 2002 and is now undergoing defuelling prior to decommissioning. It is authorised to discharge gaseous

[20] No toxicity data is available for dibromoacetonitrile in marine species.

wastes to the local environment and liquid wastes to the estuary of the River Blackwater. Sampling of the aquatic environment has included sea water, and locally caught fish and shellfish (to monitor uptake and risk to humans via consumption), together with intertidal sediments (external exposure rates). The principal long-term surveillance of benthic organisms focused on the commercial oyster fishery in the northern part of the estuary. *Fucus vesiculosus* was also analysed as an environmental indicator material.

Measurements of selected radionuclides in native oysters (Tollesbury) and seaweed (Waterside Bradwell) are plotted in figure 33. and include results of the latest available survey conducted in 2004 (CEFAS, 2005b), together with earlier data from RIFE reports. Concentrations of artificial radionuclides were generally low in all aquatic sample types making source apportionment difficult. It is likely that, in addition to materials discharged from Bradwell, inputs from Sellafield and historical weapons testing contribute to loadings.

In the majority of biological samples, apart from spikes of ^{238}Pu and ^{241}Am in 2000, concentrations do not appear to have changed consistently in recent years (Figure 33). Exceptions are ^{65}Zn in oysters and also ^{137}Cs is indicated to have decreased in sediments over the last four years of monitoring (Cefas, 2005b). The most recent data for oysters also suggest a large reduction in ^{137}Cs in 2004. The ^{99}technetium detected in seaweeds at Waterside, Bradwell, which increased to a peak in 1999 and subsequently decreased is likely to be due to the long distance transfer of Sellafield-derived activity (bioaccumulation lagging behind discharge trends by perhaps two years), together with a small contribution from the reprocessing plant at Cap de la Hague.

The total beta activity in water from a coastal ditch (14 Bq L^{-1}) continued to be enhanced above background levels and was in excess of the WHO screening level of 1 Bq L^{-1} however the ditch is not known to be used as a drinking water source. Individual radionuclides were below detection limit in seawater. Gamma dose rates on beaches were difficult to distinguish from natural background. In 2004, the critical group of seafood consumers received 0.007 mSv, mostly due to the effects of external exposure, which was less than 0.5% of the dose limit for members of the public of 1 mSv. A more extensive survey of dose rates in intertidal areas in 1997 failed to find any evidence of localised hotspots.

In summary, the radiological impact of authorised disposals from Bradwell was considered to be low.

Monitoring of N Sea water off the Essex coastline (off Harwich) in 2002 indicated low (background) concentrations of ^{137}Cs and ^{3}H, at 0.004-0.006 Bq l^{-1} and 2-4 Bq l^{-1}, respectively.

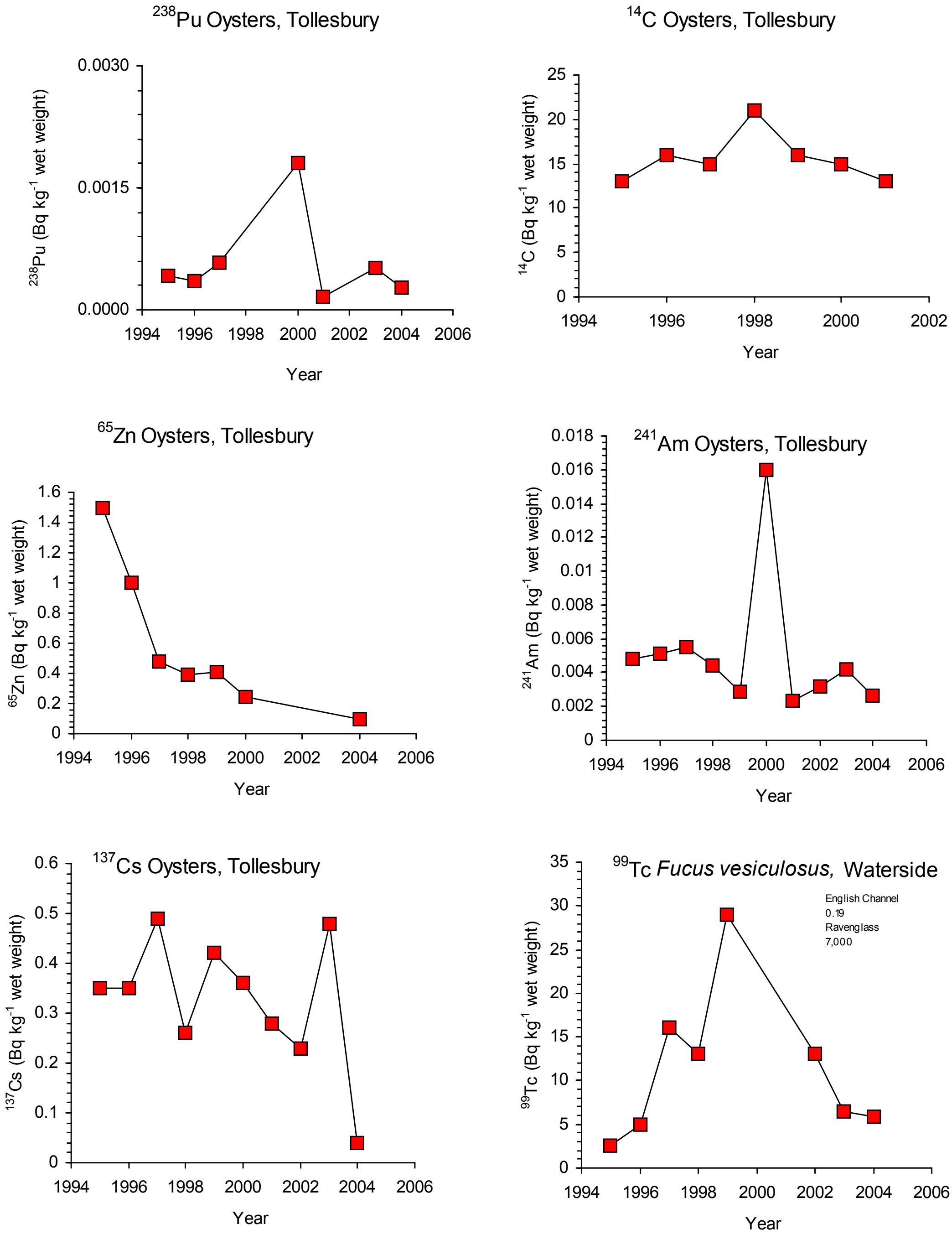


Figure 33. Trends in radionuclides in biota from the Blackwater Estuary. Oysters are native oysters *Ostrea edulis* from the Tollesbury fishery. *Fucus vesiculosus* from Waterside near Bradwell (data source RIFE reports, see CEFAS, 2005b).

6.2 Non-Toxic Contaminants

This section deals with non-toxic contamination in the Essex Estuaries EMS. Concentrations of non-toxic substances are often an important issue in marine sites. Areas of concern, identified by the nature conservation agencies include: nutrients (nitrogen, phosphorus and silicon), organic carbon, oxygen depleting substances (BOD and COD), pH, salinity, temperature (thermal discharges) and turbidity (Cole *et al.,* 1999)[21].

6.2.1 Nutrients

Water quality with regard to nutrients is primarily assessed in terms of the trophic status, or degree of nutrient enrichment of estuaries and near shore waters. 'Nutrient enrichment' generally refers to nitrogen and phosphorus species that are elevated beyond background levels as these are the two leading causes of poor water quality Nitrogen and phosphorus enter the estuarine environment via point or diffuse sources. Point sources are generally consented discharges and a direct result of human activities including; sewage effluent from sewage treatment works (STW), discharges from some industrial processes (including detergents) and cage fish farm installations. Diffuse inputs originate from both natural and anthropogenic sources. These comprise run-off/leaching from the land catchment (either directly into estuaries and coastal waters or via rivers and groundwater), atmospheric deposition, imports from off-shore waters and nitrogen fixation by plant life.

Table 19 shows estimated nutrient budgets for three Welsh estuaries (based on Parr *et al.,* 1999) and although many Welsh coast estuaries are oligotrophic, and different estuaries will vary according to the geology and urbanisation of the catchment area, these figures may be considered typical for many estuaries.

Table 19. Estimated source of estuarine nutrients (based on Parr *et al.*, 1999)

Source	Nitrogen %	Phosphorus %
Agricultural sources (livestock waste and inorganic fertiliser run-off)	25 - 49	3 - 49
Sewage Treatment Works	3 - 13	26 - 62
Atmospheric deposition	2 - 6	1.5 - 1.8
Nitrogen fixation	<5	-
Background	13 - 15	10 - 19

Ultimately, the potential for nutrient enrichment and localised effects will be determined by land use and the physico-chemical and biological characteristics of the site such as flow, seasonal variability, flushing, tidal regime, primary production and reactivity with sediments and rates of remineralisation.

The principal effect of extreme nutrient enrichment is eutrophication, defined as 'the enrichment of natural waters by inorganic plant nutrients, which results in the

[21] An updated version of Cole et al., 1999, prepared in 2003 by Power, B., Girling, A. and Fisk, B. (independent Environmental Consultants) is now available (ISBN: 1 85716 747 3), 206pp + Appendices.

stimulation of an array of symptomatic changes' (EA, 1998)[22]. These changes include an increase in phytoplankton growth - reflected by an increase in chlorophyll α concentrations. Dissolved oxygen levels in the water column fluctuate during the growth phase of a bloom and there is a potential for depletion of dissolved oxygen concentrations in the water column and sediments as a result of microbial activity following the die-off of phytoplankton blooms. pH may be affected. The bloom may contribute to increased turbidity in the water column, reducing light availability.

Some of these changes are quantifiable and, in addition to nitrogen, phosphorus and ammonia, a range of other parameters can be measured for determination of water quality in relation to nutrients. These include dissolved oxygen (DO), biological oxygen demand (BOD), chlorophyll *a*, suspended solids and turbidity. Nitrogen levels can be monitored as nitrate, nitrite and ammonium concentrations in tidal waters which, when added together, produce total inorganic nitrogen (TIN), an approximation of bioavailable nitrogen. Phosphorus is present in the aquatic environment in both inorganic and organic forms, although the principal inorganic form is orthophosphate and is measured as dissolved orthophosphate (soluble reactive phosphate SRP), or as total reactive phosphate (TRP) by measuring phosphate in unfiltered samples.

Parr *et al* (1999) report a wide range of nutrient levels in UK coastal waters and estuaries; concentrations of 0.07 – 1.85mg l^{-1} TIN and 0.007 – 0.165mg l^{-1} TRP are found in coastal waters, whilst the upper reaches of estuaries have nitrogen concentrations similar to those in river water, 0.1 - 15mg l^{-1} TIN. TRP in upper estuaries, as in rivers can also be variable, 0 – 11.4mg l^{-1}. Freshwater entering estuaries usually has a N:P ratio of >10, therefore the water column, particularly at the freshwater end of the estuary is more likely to be P- than N-limited, although saltmarshes are usually N-limited (Cole *et al* 1999).

Table 20. Designated sensitive areas within Essex Estuaries EMS and catchments

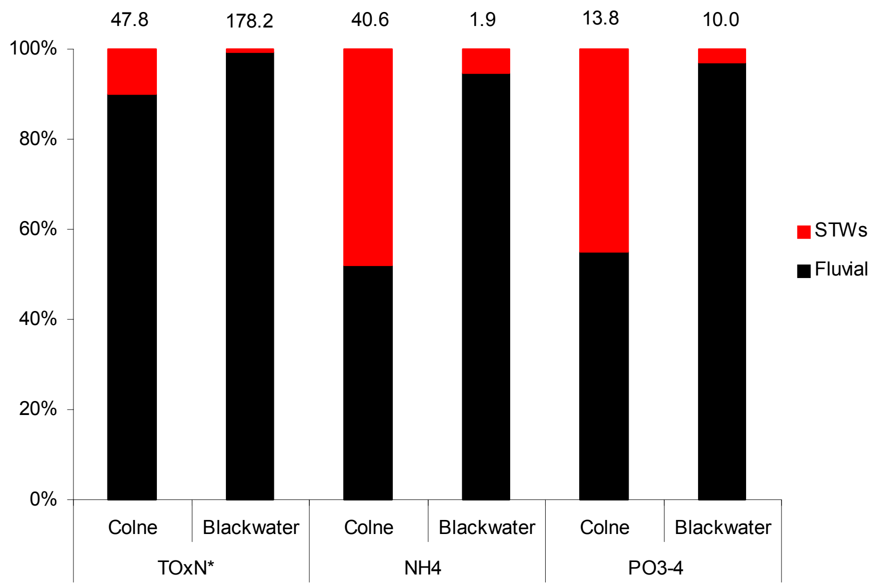
Site		Designation	Date identified
Blackwater River	- upper reaches of the Blackwater (near Braintree) to Langford (NW of Maldon)	SA(Eutrophic) SA(Nitrate)	July 1998 February 1997
River Colne	- Colne (near Halstead) to Colchester	SA(Eutrophic)	July 1998
West Mersea		SA (Bathing Waters)	March 2002
Strood Channel		SA(Shellfish Waters)	May 2003
Salcott Channel		SA(Shellfish Waters)	May 2003
Tollesbury Channel		SA(Shellfish Waters)	May 2003

Estuaries within Eastern Area of Anglian Region are generally considered to be nutrient rich systems compared to those in other parts of the UK (EA, 2002). Several areas within the Essex Estuaries EMS, and their freshwater catchments, are designated under the Urban Waste Water Treatment Directive (UWWTD) as Sensitive Areas - SA (Eutrophic, Nitrate, Bathing or Shellfish)[23]. These are listed in table 20.

[22] Also see Glossary
[23] (see Defra - http://www.defra.gov.uk for details of sensitive areas and designation criteria).

Published information on water quality data in the Essex Estuaries EMS indicates that excessive nutrients are problem (e.g. EN, 2003). In a recent discussion document, Defra (2003) state that Colne and Blackwater Estuaries are suffering from increased occurrences of algal mats and other symptoms of eutrophication, and that agriculture is the dominant nitrogen source. The SA designations of freshwater stretches of the Blackwater and Colne also suggest that freshwater inputs (including run-off from artificially nutrient-enriched land) are likely to be a substantial source of nutrients into the EMS. Supplementing these inputs are major sewage treatment work discharges at the head of some estuaries, nb Colchester on the Colne.

The Colne estuary is generally regarded as turbid and hypernutrified, with strong gradients of nitrate and ammonium inversely related to salinity (King & Nedwell, 1984, 1987; Ogilvie *et al.,* 1997). There are considered to be two major sources of N; freshwater inputs - run-off from the 500km^2 catchment area, much of which is rich arable land, and inputs from the large STW at Colchester Hythe, which dominates N inputs during summer when lower rainfall and abstraction reduces river flow (Robinson *et al.,* 1998). There are also two smaller STWs at Fingringhoe and Brightlingsea.



* TO_xN = Total Oxidised Nitrogen = nitrate + nitrate

Figure 34. Sources and loads of nutrients in STW effluents input directly (below the tidal limit) to the Colne and Blackwater Estuaries (average of loads for 1995 and 1996). Data for loads appear above bars and are in units of Mmol (moles x 10^6) nutrient y^{-1}. Sources (STW or fluvial) are displayed as % of total load. Data Source: Nedwell *et al.*, (2002).

Nedwell *et al.*, (2002) estimated nutrient loads in STW effluents input directly (below the tidal limit), and fluvial inputs to several UK estuaries including the Colne and Blackwater (figure 34). Results indicate that the fluvial contribution (largely fertilizer run-off) dominate nutrient inputs to both estuaries, although a high percentage of the total ammoniacal nitrogen and orthophosphate load to the Colne originates from STW effluent (Nedwell *et al.*, 2002).

Other estimates also indicate high nutrient inputs from STW to the Colne: Thornton *et al.,* (2002) report that the input of DIN to the Colne estuary from the Colchester STW during 1996, was 24.62 Mmol N, more than twice that of the river (10 Mmol N yr^{-1}). Kocum *et al*., (2002a) considered STW effluents to be the principal source of phosphate in the Colne, and interestingly, Kocum *et al*., (2002b) found the very high P levels in effluent from Colchester STW resulted in low N:P ratios at the head of the estuary. This is the opposite of what might be expected in a typical estuary (P limitation in freshwater and N limitation in seawater - Pennock & Sharp, 1994).

The result of these inputs is very high levels of P in the water column; MacGarvin (2004) noted that of 12 UK estuaries with the highest levels, the Colne ranked fourth in terms of high P concentrations.

In a study of phytoplanktonic primary production, nutrient and light availability in the Colne, Kocum *et al*., (2002a), reported high concentrations of nitrate (564µM), ammonium (291µM) and phosphate (52µM) at the head of the estuary. Despite these high levels, and the obvious potential for phytoplankton proliferation, chlorophyll-a concentrations did not reach levels typical of nuisance algal blooms, and were always below 50µg l^{-1} (annual average 5µg l^{-1}) - generally at the lower end of the range for estuarine habitats. Flagellates were the major component of the phytoplankton all year round, although in spring the relative abundance of diatoms increased. By summer, the species composition was again flagellate-rich (Kocum *et al*., 2002a). This is often a feature of coastal eutrophication and the development of nuisance algal blooms (Underwood & Kromkamp, 1999), although none developed during the study.

Paradoxically, phytoplankton were subjected to greater light-limitation in the clearer lower estuary than the turbid waters of the upper estuary, especially during winter. This situation arises because the water column is completely mixed and stratification does not occur. Thus, the shallow waters of the upper estuary offer less possibility of light limitation than deeper waters of the lower estuary. The greatest phytoplankton biomass in this study (37.5µg l^{-1}) therefore occurred at the head of the estuary (Kocum *et al*., 2002a).

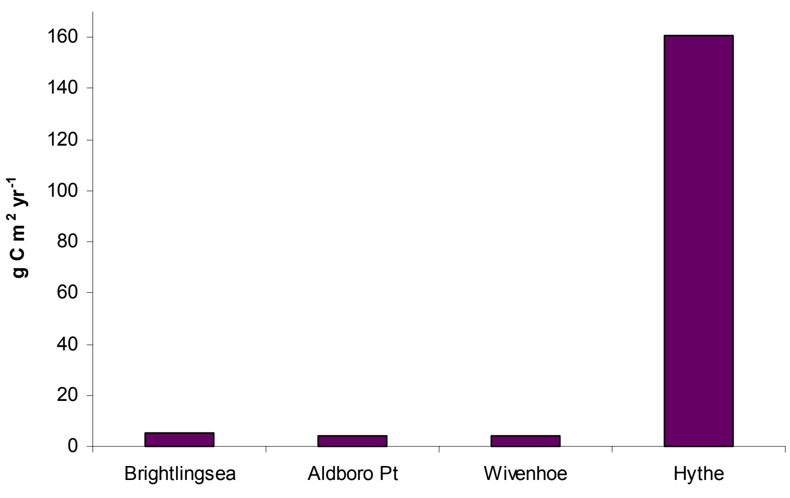


Figure 35. Estimated annual rate of productivity (g C m^{-2} yr^{-1}) in the Colne Estuary. Data source: Kocum *et al.,* (2002a)

Estimating the annual rate of productivity at four sites within the Colne, Kocum *et al.*, (2002a) found a much higher rate at the head of the estuary (Hythe) (figure 35). The authors concluded that planktonic algal photosynthesis throughout the estuary was always light-limited, although the more beneficial light regime in the upper reaches combined with an excess of nutrients, maintained higher levels of phytoplankton biomass. Underwood (2002) considers that the benthic microalgae are the major primary producers in the estuary. They form extensive biofilms on the sediment surface, notably between Colchester and Alresford, as they migrate to the sediment surface at low tide to photosynthesise, utilising nitrate, ammonium and phosphate. The community consists of euglenoid (green algae), diatoms and cyanobacteria. These algae are important for denitrification processes.

Dong *et al.* (2000) investigated denitrification in the Colne Estuary. Denitrification can remove significant quantities of nitrate from the water column to the sediments, and can play an important role in ameliorating the degree of eutrophication in estuaries with high N inputs (Ogilvie *et al.,* 1997; Nedwell *et al.,* 1999). Dong *et al.,* (2000) calculated that 32 – 44% of TOxN, and 20 – 25% of TIN inputs to the Colne Estuary were removed *en route* to the North Sea. Highest rates of removal occurred in the upper estuary (near the STW) and the lowest at the estuary mouth south of Brightlingsea. The higher rates corresponded with high nitrate concentrations in the water column and high organic content in sediment in the upper estuary.

Denitrification in the Colne represents significant attenuation of the flux of nitrogen through the estuary into the North Sea, which may be important for the coastal sea. NMMP (2004) report that median concentrations of TOxN exceed the relevant criteria in the coastal (Thames) area of the North Sea, into which the Colne discharges, and although the Thames itself is subject to very high nutrient loads (Nedwell *et al*., 2002) and undoubtedly contributes the major proportion of N inputs to this area, the contribution from the Essex Estuaries will only exacerbate the problem.

However, denitrification processes in hypernutrified estuaries such as the Colne may not always be environmentally beneficial as it can provide a significant source of nitrous oxide (N_2O) to the atmosphere (Robinson *et al.,* 1998). N_2O is the third most important greenhouse gas in terms of its overall effect, although current global budgets do not include estimates of estuarine and coastal sea sources. Robinson *et al.,* (1998) investigated emission fluxes of N_2O from tidally-exposed sediments and waters of the Colne. Estimates of the relative contributions of water-air and sediment-air N_2O fluxes in the estuary suggested that emissions from the water surface were about 4-fold that of the sediment, despite the large area of sediment exposed at low tide. This was attributed to the greater surface area of water, coupled with a 'tapering off' of N_2O flux from sediment after 1-2 hours of exposure. After this period of tidal exposure, the sediment nitrate pool becomes depleted in the absence of transport from the water column, and benthic microbial respiration may produce N_2 or ammonium (Kieskamp *et al.,* 1991).

Thus in hypernutrified estuaries such as the Colne, sediments act as a processing site for nitrate only when they are covered by the nitrate-rich seawater, when there is rapid recharge of the sedimentary nitrate pool from the overlying water and N_2O transport from sediment to water can occur (Robinson *et al.,* 1998). The total N_2O emission from the Colne Estuary was calculated as 1.2 x 10^5 mol N_2O-N yr^{-1}, of which 83% was from the water surface. This amounted to around 0.5% of the TON load to the estuary. Benthic N_2O production accounted for <2% of the nitrate denitrified in bottom sediments but is nonetheless considered significant when extrapolated globally (Robinson *et al.,* 1998).

Nutrient levels in the Blackwater Estuary are also considered to be elevated. Summarising data for 1993–1997, MacGarvin (2004) reported high ammoniacal nitrogen values off Osea Pier, in mid-estuary. TON values ranged between 3.64–141 µM and phosphorus between 1.06–5.26 µM. Outer estuary values, off West Mersea, were also said to be elevated. Phytoplankton chlorophyll concentrations were up to 108 µg l^{-1} off Osea Pier, and 14.1µg l^{-1} at West Mersea. There has also been concern about blooms of the macroalga *Enteromorpha* in the Blackwater (MacGarvin 2004; EN 2003; Defra, 2003, see also section 6.2.3).

Sources of nutrients to the Blackwater Estuary are considered to be principally fluvial (Nedwell *et al.,* 2002 – see figure 34) and include a significant amount of fertilizer run-off. With increasing 'coastal squeeze[24]', experiments are underway on the use of managed re-alignment[25] at several coastal around the Blackwater Estuary (Tollesbury, Northey Island Abbotts Hall and Orplands) to mitigate flood risk (Macleod *et al.,* 1999). Most of the sediments behind existing sea defences are/have been used for agricultural purposes and therefore have been subject to the addition of fertilizers. This can represent a significant source of nutrients to the estuary once the land is inundated with seawater. Nitrogen cycling in the soil environment is strongly affected by waterlogging and mineralisation of organic nitrogen can be restricted to conversion to ammonia, which accumulates in flooded soils, particularly those rich in organic matter (Ponnamperuna, 1984). Intermittent drying and flooding has also been shown to increase ammonium production (Reddy & Patrick, 1975).

Conversely, results from modelling exercises suggest that that managed realignment options in the Blackwater/Colne estuaries would enhance nutrient burial in the newly accreted sediments. Increased microbial metabolism of dissolved nitrate, attenuating flux to the southern North Sea, was also a modelled outcome (Shepherd *et al*., 2005 – see section 8).

Boorman *et al*., (1994) consider that generally, young, immature saltmarshes are flood dominated systems and net exporters of nutrients (and net importers of sediment and organic matter). In a study of saltmarsh organic and nutrient fluxes, the creek in Tollesbury saltmarsh was found to have very high levels of ammonium–N (up to 30 µmol l^{-1} at times) compared to systems in the Netherlands and Portugal. Even at 10µmol l^{-1} the levels in the Marsh Creek at Tollesbury were at least 100 times greater than in the adjoining areas of the North Sea and 10 times that of the adjoining inshore waters. Nonetheless, Boorman *et al*., (1994) reported a small import of ammonium-N to the Tollesbury Marsh over tidal cycles (figure 36a). Concentrations of nitrate were also comparatively high in the Tollesbury saltmarsh (up to ~170 µmol l^{-1}), over 100

[24] Coastal squeeze occurs as the salt marsh and mudflats become trapped between the sea walls and the rising sea. The seawall prevents these habitats moving upwards and landwards within the changing tidal frame and so become reduced in their extent.
[25] Managed realignment or retreat: The practice of allowing drained marshes that were once covered by the sea to flood again and be recolonised by saltmarsh vegetation. See also Saltmarshes in section 7.

times those found in the (offshore) North Sea, but similar to those of the adjoining waters. Tidal transport usually resulted in a net export of nitrate to the estuary (figure 36b). For phosphate, concentrations were in the range 0.1 to 0.8 µmol l^{-1}, and low in comparison to the French and Dutch saltmarsh systems. These levels were only marginally higher than those in adjoining waters, but ~ten times those of the North Sea. There was considerable variation in the direction of phosphate transport (figure 36c). The study found a clearly-defined boundary between the highly eutrophic saltmarsh waters and rather less eutrophic adjoining waters. Interestingly, Boorman *et al*., (1994) comment that the waters of the Blackwater are noted for their clarity (up to 2m in calm waters in summer) suggesting much less turbid conditions than in the neighbouring Colne Estuary.

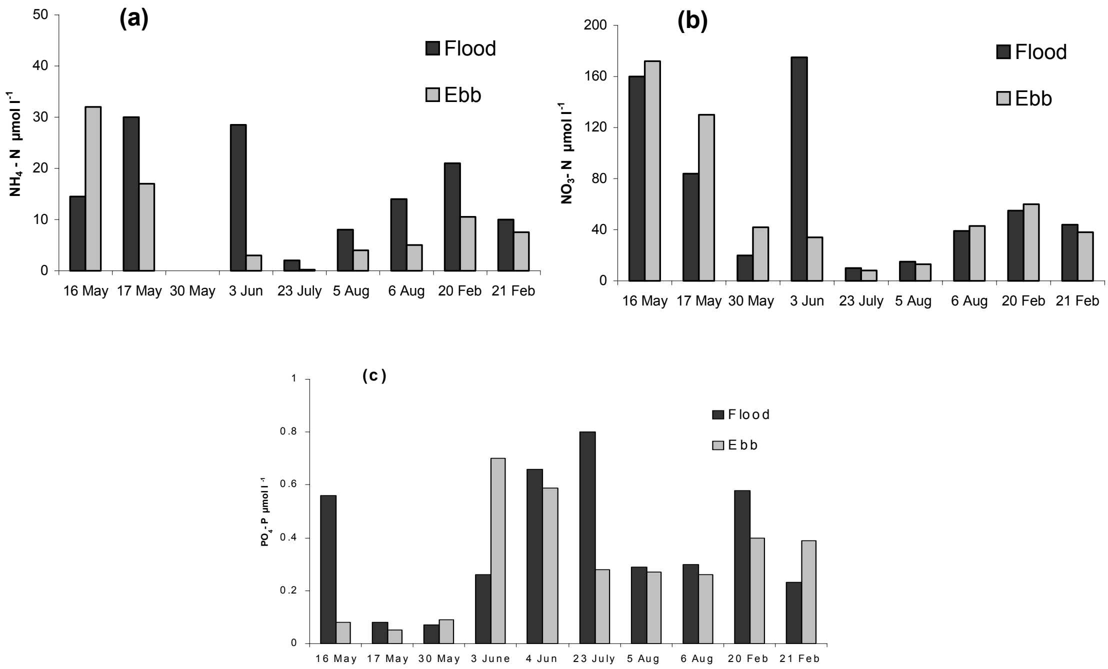


Figure 36. Concentrations of nutrients (µmol l^{-1}) during flood and ebb tide at Tollesbury saltmarsh on the Blackwater Estuary 1991-2. (a) ammonium-N, (b) Nitrate – N, (C) Phosphate- P, Redrawn from Boorman *et al.,* (1994).

This study was conducted before 1995, when the Tollesbury realignment site was created by breaching the sea wall thereby allowing 21 hectares of low-lying agricultural land to flood, extending the intertidal. Mudflats have now developed over much of the newly-created site, and there is an expanding area of saltmarsh[26]. As an immature marsh, this is likely to represent a more significant source of nutrients to the estuary than the older, previously established saltmarsh, particularly as the land-use prior to flooding was agricultural.

Point-sources of nutrients to the Blackwater Estuary include various STWs, i.e. West Mersea, Tiptree (via Salcott), Tollesbury, Bradwell On Sea and Maldon. The outfall from West Mersea STW is reported to have been a major cause of concern over recent years, because of the discharge of only primary treated sewage less than 1km from the designated bathing beach, and high bacterial counts in the upper reaches of the Salcott

[26] Defra.gov.uk

Channel close to shellfisheries. The beach, which was declared a bathing water beach under the EC Directive on Bathing Water Quality (1991), has regularly failed the required water quality standards (Blackwater Estuary Management Plan, 1996). However, a new sewage works opened in 1996 providing secondary and tertiary treatment prior to discharge, and anticipated improvements to water quality appear to have materialised; West Mersea bathing waters have complied with Bathing Water Directive (76/160/EEC) (passed mandatory standard for total and faecal coliforms and three physico-chemical parameters) in 2003-4, as have the other bathing waters in the EMS (Jaywick and Brightlingsea) (Defra, 2004).

There appears to be little published information regarding nutrient status of the Crouch and Roach estuaries. However, results of a modelling exercise carried out by Evans *et al.,*(1993) suggested that there are, or have been nutrient-related problems here too; STW loads from the inner Crouch were linked to elevated DO concentrations and high levels of chlorophyll-a near the tidal limit (see section 8).

Thus, it appears from the literature, that eutrophication as a result of nutrient enrichment is generally considered to be a significant problem in parts of the Essex Estuaries EMS. Macroalgal blooms are considered to be a concern although there is little available documented evidence of this. Despite hypernutrification, phytoplankton blooms do not generally occur because of light limitation in highly turbid conditions (especially in the Colne). However, Underwood (2002) voiced concerns that if the proposed tidal barrier to retain high water in the Colne was built at the Hythe[27], a reduction in tidal resuspension of sediments would allow the water to clear, and nutrient-rich water from the River Colne coupled with the huge source of nutrients buried in the sediments could provide a major stimulus for phytoplankton activity and a trigger for algal blooms in the Estuary. With the exception of one report noting that paralytic shellfish poisoning (PSP) (influenced by high nitrate concentrations in waters)[28], has recently resulted in the closure of Colne shellfish beds, significant nuisance blooms have not been reported.

Nutrient Quality Criteria.

Nutrient monitoring is required by the EC Urban Waste Water Treatment Directive (UWWTD) (91/271/EEC), the Nitrates Directive (91/676/EEC) and the OSPAR Strategy to Combat Eutrophication. The aim is to determine whether waters show signs of eutrophication: defined as 'an undesirable disturbance to the balance of organisms present and to the quality of the water concerned, which results from the accelerated growth of algae and higher plant forms caused by anthropogenic sources of nutrients' (91/271/EEC). Both the EC and OSPAR measures require action to reduce inputs when there are reasonable grounds for concern that eutrophication already affects waters or may occur in currently unaffected waters. The UWWTD requires treatment of point source discharges so as to reduce their nutrient content, the Nitrates Directive requires reduction of diffuse inputs of nutrients and OSPAR seeks to reduce all nutrient inputs to marine waters (MPMMG, 2004).

[27] The plans for the barrage are still under consideration
[28] http://www.defra.gov.uk/environment/water/quality/nitrate/ria.htm

Nutrient concentrations vary with salinity, therefore measurements collected simultaneously from different regions within the estuary, or from the same region but at different states of the tidal cycle, may show considerable differences and not be truly representative of water quality. To compound this difficulty, nutrient concentrations also vary throughout the year with freshwater flow. As yet there are no statutory water quality standards for nutrients in the UK and determination of the nutrient status of estuaries, and the ecological consequences, remain a notoriously contentious issue. The EA's Technical Guidance for Water Quality assessment framework (WQTAG089a) is divided into the assessment of a) the risk of eutrophication impacting on sites and b) the extent of ecological impact. Parameters including modelled predictions and temporal trends in nutrient loadings, nutrient concentrations, and impacts (phytoplankton biomass and species, DO status, evidence of species decline) are recommended for considering the risks and impacts of eutrophication in SAC and SPA estuaries for the purposes of reviewing new and existing permissions as required under the Conservation (Natural Habitats &c) Regulations, 1994. This "weight of evidence" based framework attempts to facilitate evaluations in a transparent way. However, judgement of nutrient status in the Essex Estuaries EMS, as elsewhere, still consists, to a certain extent, on subjective assessment of monitoring information concerning the primary variables, coupled with contextual information on the site characteristics and condition. The primary variables are generally considered to be nitrogen and phosphorous (though there is still great scientific debate as to which forms to measure). It is usually considered essential to monitor these parameters alongside initial biological response indicators such as chlorophyll-a (a measure of primary production), dissolved oxygen and, for example, Secchi depth (a measure of turbidity). These data may then be fed into models to develop criteria for the selection of numerical water quality objectives.

Although no statutory standards, Background Reference Concentrations (BRCs) or Ecotoxicological Assessment Criteria (EACs) exist for N and P in estuarine and marine SACs, a number of 'guideline values' have been established which could be of relevance for assessment of the status of nutrients in the catchment of the Essex Estuaries EMS, and for initiating management responses:

- EU nitrates directive 91/676/EEC, on the protection of all waters against pollution caused by nitrates from agricultural sources, calls for the identification of all waters that contain **50mg l^{-1} nitrate**.
- The USEPA is still in the process of arriving at their national nutrient strategy but has for many years proposed a limit of **10mg l^{-1} nitrate** for the protection of domestic water supplies (against overenrichment and impacts on human and animal health). A phosphorous criterion was reported some years ago in the EPA 'Red Book' as **0.1µg l^{-1} (as P)** to protect estuarine and marine organisms against the consequences of bioaccumulation (EPA, 1976). However, this was not established as threshold for eutrophication and is currently under review.

- The North Sea Status report stated that hypernutrification in sea water exists when winter (maximum) **TIN values exceed 0.144mg l^{-1}** (provided P>0.006mg l^{-1}), implying that nutrient concentrations need not be elevated by a large margin before algal proliferation commences (Parr, 1999). In estuaries however it seems likely that thresholds will be higher.

- Based on work in 2 eastern USA estuaries, Deegan *et al.,* (1997) have suggested that a DIN value of ~ **1mg l^{-1} DIN** or more might lead to poor habitat quality for fish populations, which may be due in part to cloaking effects of macroalgal mats on *Zostera* beds.

- There are suggested draft common assessment criteria for areas of UK waters subject to the OSPAR Common Assessment Procedure for eutrophication (table 21). The most recent NMMP report (MPMMG, 2004) uses these tentative values to evaluate present levels of contamination against pristine conditions, with the caveat that 'the results must be treated with a great deal of caution because BRC values do not yet reflect regional differences in geochemistry and EAC values have been extrapolated from limited datasets'.

Table 21. Draft Common Assessment Criteria for the OSPAR Comprehensive procedure (from MPMMG, 2004)

	Dissolved Inorganic Nitrogen (salinity related and/or region specific) background concentration	Elevated winter Dissolved Inorganic Nitrogen levels (roughly set at >50% above salinity related and/or region specific background concentration)	Dissolved Inorganic Phosphorus (salinity related and/or region specific) background concentration	Elevated winter Dissolved Inorganic Phosphorus levels (roughly set at >50% above salinity related and/or region specific background concentration)
Offshore North Sea	10 µmol l^{-1}	>15 µmol l^{-1}	0.6 µmol l^{-1}	>0.8 µmol l^{-1}
Channel	9 µmol l^{-1}	>15 µmol l^{-1}	0.4 µmol l^{-1}	>0.8 µmol l^{-1}
Irish Sea (saline waters)	12 µmol l^{-1}	>18 µmol l^{-1}	0.8 µmol l^{-1}	>1.25 µmol l^{-1}

- There is a proposed EQS of **0.021mg l^{-1} un-ionised ammonia (NH_3-N) AA (Annual Average)** for the protection of saltwater fish and shellfish, although due to the technical difficulties in measuring the unionised form, total ammonium is usually monitored and NH_3 calculated. However, even calculations can be difficult as the relative proportion of ionised and un-ionised ammonia depends on salinity, temperature and pH.

- The proposed EQS of **0.021mg l^{-1} un-ionised ammonia** (NH_3 N) also applies to EC designated salmonid and cyprinid freshwaters. In addition there is an EQS of **0.78mg l^{-1} total ammonia** for these waters (Seager *et al.,*1988).

- In recognition of the impact that the ammonium ion may have at higher salinities, a total ammonia limit of **1.1 mg l^{-1} -N AA (annual average)** and **8.0 mg l^{-1} –N MAC (maximum allowable concentration)** is also proposed for UK waters based on USEPA, 1989, 1999 recommendations and a review carried out by Eddy (2004) for the Environment Agency.

Recognising the dilemma in arriving at standards, there have been other attempts in recent years to develop and test General Quality Assessment (GQA) schemes for nutrients in estuaries and coastal waters, which may be adopted nationally and internationally. One such scheme is proposed for the EA by the WRc as part of their

General Quality Assessment (GQA) scheme (Gunby *et al.,* 1995). For nitrogen, this method uses the combined concentrations of nitrate, nitrite and ammonium concentrations in tidal waters (total inorganic nitrogen, TIN), as an approximation of bioavailable nitrogen. Assuming conservative behaviour for TIN and a standard concentration in marine waters, allows the TIN concentration in the freshwater input to be calculated, provided salinity data are available. For phosphorus, Total Reactive Phosphate (TRP - phosphate in unfiltered samples) is measured and, as for nitrogen, the concentration in freshwater calculated. Estuaries are then be grouped according to the following class boundaries (table 22):

Table 22. TIN and TRP classification criteria for estuaries (based on Gunby *et al.*, 1995)

Class	Median projected TIN (mg l^{-1})	Class	Median projected TRP (mg l^{-1})
A/B	5.3	A/B	0.087
B/C	8.1	B/C	0.35
C/D	11.1	C/D	1.00

In view of the hydrodynamic differences between estuaries, together with seasonal and other site-specific factors, it is not known how these thresholds would apply to the Essex Estuaries or how valid they may be. Nevertheless, in the absence of site-specific guidelines they at least represent benchmarks as to the potential threats, against which to draw comparisons. Based on these criteria, and published data from other estuaries, it is possible to attempt a brief analysis of nutrient monitoring observations supplied by the Agency including;

- determination of background (reference) values and 'hotspots' for the area
- examination of historical data and trends in the Essex Estuary
- comparisons with other areas
- validity of guideline values and classification schemes

GQA scheme: TIN and TRP

Cole *et al*., (1999) made a comparison of the nutrient status of UK estuaries, having extrapolated freshwater values (from seawater values) on the basis of conservative mixing. Using these criteria for the Essex Estuaries, the projected GQA classification for TIN and TRP are below average (both grade D, table 23), which indicates that the Essex Estuaries are suffering from nutrient enrichment problems.

There are other schemes, which estimate the nutrient status from freshwater load inputs, thus encompassing point source discharges. Dong *et al.,* (2000) calculate estuarine nutrient loads by multiplying annual average of all nutrient concentration measurements for contributing rivers, by the annual freshwater flow, however there is scope for error in that diffuse freshwater sources entering directly into the estuary will not be accounted for, likewise estuarine sources, if they occur, would make this type of estimate unreliable.

Table 23. Classification nutrient status of selected estuaries in England according to GQA TIN/TRP projection methodology (Cole *et al.,* 1999)

Estuary	Projected median TIN concentration (mg l^{-1}) in freshwater	GQA TIN class	Projected median TRP concentration (mg l^{-1}) in freshwater	GQA TRP class
Blackwater	**14.3**	**D**	**6.8**	**D**
Camel	5.9	B	0.4	C
Carrick	5.4	B	4.6	D
Colne	**12.7**	**D**	**4.2**	**D**
Crouch	**11.3**	**D**	**5.3**	**D**
Dart	4.3	A	0.2	B
Deben	11.5	D	6.2	D
Exe	5.4	B	0.3	B
Fal	9.4	C	5.1	D
Fowey	4	A	0.1	A
Hamford Water	10	C	6.8	D
Helford	7.3	B	3.2	D
Humber	8.8	C	0.1	B
Itchen	5.6	B	0.3	B
Lynher	5.5	B	0.1	A
Medway	5.1	A	0.4	C
Mersey	7.1	B	0.4	C
Nene	15.1	D	0.9	C
Ore/Alde	9.5	C	-1.0	A
Orwell	14	D	3.2	D
Ouse	12.2	D	0.8	C
Roach	**11.9**	**D**	**11.4**	**D**
Severn	7.6	B	0.5	C
Stour	13.3	D	2.5	D
Tamar	4.6	A	0.2	B
Test	6.3	B	0.3	B
Thames	12	D	2.4	D
Wash	13	D	1.5	D
Welland	13.1	D	0.4	C
Witham	21.9	D	0.5	C
Wyre	9	C	7.9	D
Yare	9.7	C	0.6	C
Yealm	5.9	B	4.2	D

The issue of whether or not to focus on nutrient concentrations in the tidal waters or loading criteria has been a contentious one among both scientists and managers. As noted above, the characteristics of estuaries differ significantly, and therefore nutrient sources, their fate and effects in the estuarine environment are not easily predicted. Rather than relying on a classification scheme for the estuary as a whole it may be more beneficial to investigate the distribution of key determinands in finer detail:

{Note: It is generally assumed that an N:P ratio of 10:1 is ideal for plant growth. At N:P ratios>10:1 (mainly in FW) P is thought to be limiting and at N:P ratios < 10:1 (mainly in SW) N is thought to be limiting (though there are 3 coastal areas in the UK where P may be limiting –from the Solent to Dartmouth; around the Severn from Padstow to Oxwich and from the Humber to Essex). In many estuaries however enrichment may be such that nutrients are more likely to promote algal growth; turbidity may be the limiting factor instead (Parr *et al.,* 1999).

Recent nutrient monitoring data

The majority of EA data reviewed here for nutrients in the EMS are for tidal waters, and for the period 2000-2005, therefore it is not possible to determine longer-term trends. Data has not been made available for concentrations in STW or industrial discharges and there are few data for concentrations in freshwater, therefore distributions cannot be directly attributed to sources.

6.2.1.1 Phosphate

Phosphorus is present in the aquatic environment in both inorganic and organic forms. The principal inorganic form is orthophosphate which can be measured as dissolved orthophosphate (or soluble reactive phosphate SRP - phosphate in samples that have been filtered through a 0.45 μm mesh), or as total reactive phosphate (TRP - phosphate in unfiltered samples). Much of the monitoring undertaken by the Environment Agency in England and Wales involves the measurement of TRP, while the National Monitoring Programme uses SRP for sites in estuaries and coastal waters of the UK (MPMMG 1998).

There are very few data for concentrations of phosphate in freshwaters entering the Estuaries, however the three freshwater sites for which data are available, are in the Colne and Blackwater SPA's, close to, or in the receiving waters for major STW discharge outfalls (see figures 2-4). The highest individual concentrations of phosphate in these freshwater sources occurred in Washlands outlet point 'C' (up to 5.54 mg l^{-1} - figure 37). Median levels of phosphate range from 0.21 mg l^{-1} (West Dyke) to 1.45 mg l^{-1} (Tenpenny Brook). Thus the very limited data indicate that freshwater inputs may constitute a significant source of phosphate, notably for the Colne, and no doubt contribute to the high levels observed in parts of the Colne Estuary.

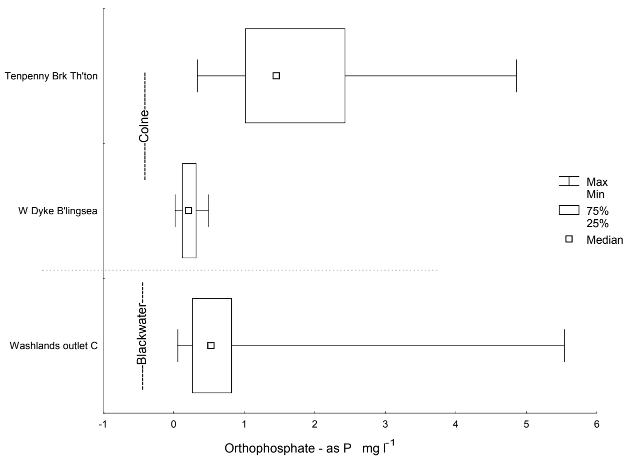


Figure 37. Concentrations of Orthophosphate - as P (mg l^{-1}) in fresh water sources to the Colne and Blackwater Estuaries. Data source: EA . Data are for 01/2000 – 07/2005.

EA phosphate data for tidal waters of the Essex Estuaries EMS are principally as orthophosphate as P (unfiltered - TRP). For some sites the data are for all years 2000 – 2005, whilst for others the data are for 2003-2005 or 2000-2003. The more extreme concentrations (up to 7.9mg l^{-1}) have occurred at inner estuarine sites towards the freshwater inputs (figure 38): notably in the Crouch SPA - Fen Creek - south of Emotes Farm, and the Colne SPA - Rowhedge Ferry, and Ivanhoe.

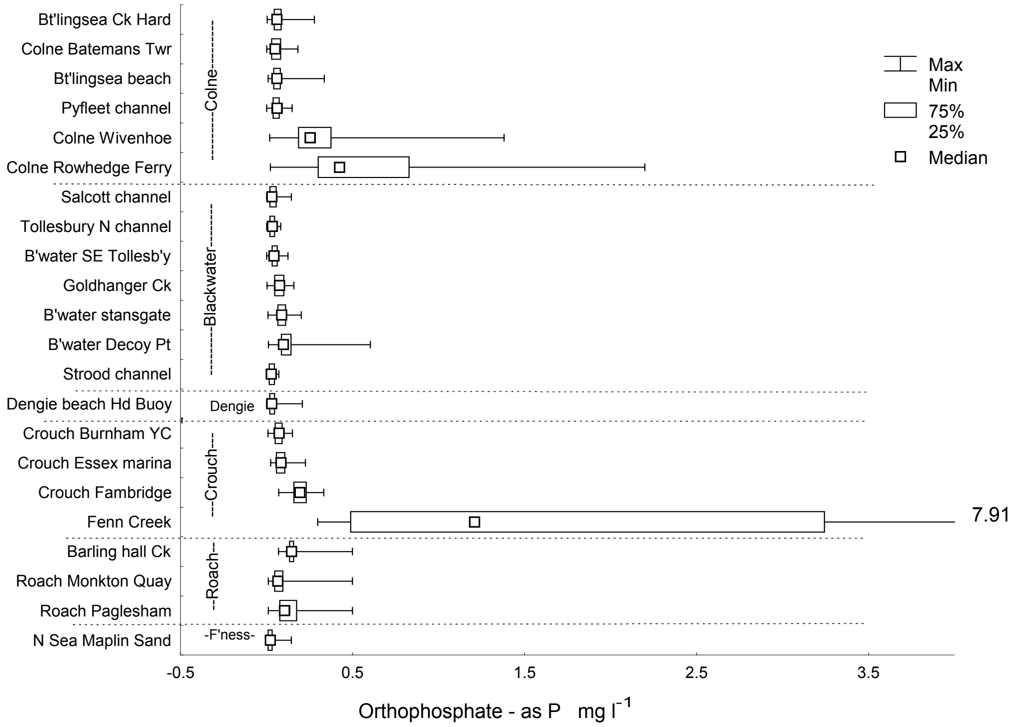


Figure 38. Concentrations of Orthophosphate - as P (mg l^{-1}) in tidal waters of the Essex Estuaries. Data source: EA . Data are for 02/2000 – 07/2005.

Median concentrations of phosphate for the period are also generally more elevated toward the freshwater end of the Estuaries - up to 1.21 mg l^{-1} at Fen Creek, again suggesting that the origins of much of the phosphate load are upstream of the Estuary. The approximate background for the tidal waters sites (25th percentile) in the estuary is in the range 0.01 – 0.49mg l^{-1}, invariably above the 0.1μg l^{-1} (0.0001 mg l^{-1}) criteria set by the EPA(US) to protect estuarine and marine organisms, and generally in the upper range reported by Parr *et al* (1999) for coastal waters (0.007 – 0.165mg P l^{-1}).

During the period 2000 to 2005 there has been a general reduction in phosphate levels at almost all of the sampling sites throughout the Essex Estuaries EMS. Some of these reductions are statistically significant; notably Rowhedge Ferry in the Colne SPA where the annual average was 0.93 mg l^{-1} in 2000, 0.39 mg l^{-1} in 2004, and 0.35 mg l^{-1} up until July 2005 (R^2 = 0.19 p<0.001). Fen Creek in the Crouch & Roach SPA has also seen significant reductions (R^2 = 0.13, p<0.01); the annual average has fallen from 3.23 mg l^{-1} in 2000 to 1.25 mg l^{-1} in 2004, with indications that 2005 may see further reduction (1.16 mg l^{-1} up until July). However, levels still remain high in relation to phosphate criteria. This general trend is exemplified in figure 39.

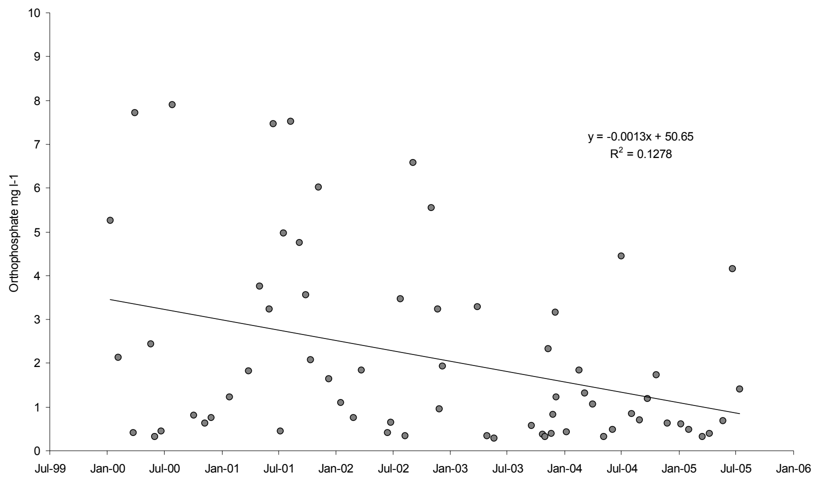


Figure 39. Temporal trend for concentrations of Orthophosphate - as P (mg l^{-1}) in tidal waters of Fen Creek, South Of Emotes Farm, in the Crouch & Roach SPA. Data source: EA .

In contrast to this, there have been increases in phosphate levels at Baling Hall Creek in the Roach Estuary and Decoy Point in the Blackwater Estuary, although monitoring has been lees frequent (n=7 and n=19 respectively) at these two sites making annual averages unreliable.

6.2.1.2 Nitrogen Species

There are very few data for concentrations of N in freshwaters entering the Estuaries, however the three sites for which data are available are close to, or in the receiving waters for major STW discharge outfalls. The highest concentrations of TOxN (total oxidised nitrogen - nitrate + nitrite) in these freshwater sources occurred in Tenpenny Brook at Thorrington Mill Sluice (up to 24.6 mg l^{-1} - figure 40). Median levels of TOxN range from 1.43 mg l^{-1} at Washlands outlet point 'C' to 12.8 mg l^{-1} in Tenpenny Brook. Thus the very limited data indicate that freshwater inputs (with concentrations of N probably enhanced by STW discharges) are a significant source of TOxN, notably for the Colne, and contribute to the high levels observed in the Colne Estuary

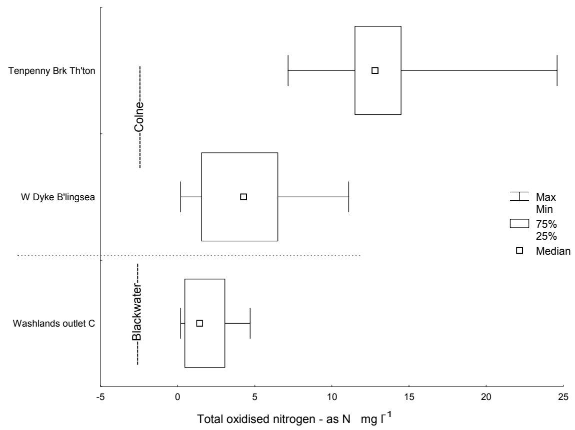


Figure 40. Nitrogen –total oxidised – as N (mg l^{-1}) in freshwaters entering the Colne and Blackwater Estuaries 2000 – 2005. Data source: EA

The available EA monitoring data for nitrogen species in tidal waters of the Essex Estuaries consists principally of TOxN (total oxidized nitrogen - nitrate + nitrite) and ammonia values. For some sites the data are for all years 2000 to 2005 inclusive, whilst for others the data are for 2003-2005 or 2000-2003. There are also limited data for nitrate, nitrite and TIN. All are expressed as N. Analysis of the data for these determinands allows a relatively good representation of the nutrient status of the EMS with regard to nutrients, both spatially and for the five-year period.

Toxin values for the period are shown in figure 41. As is the case with phosphate, the more extreme concentrations (up to 24.6 mg l^{-1}) have occurred at inner estuarine sites towards the freshwater inputs: notably in the Crouch SPA - Fen Creek - south of Emotes Farm, and the Colne SPA - Rowhedge Ferry, and Wivenhoe. Again, data for concentrations in freshwater inputs, STW or industrial discharges are too limited to attribute high levels directly to sources.

Median concentrations for the period are also more elevated toward the freshwater end of the estuary, up to 3.85 mg l^{-1} at Rowhedge Ferry, with peaks at Fenn Creek, Wivenhoe and Fambridge. Maximum values are all below the EC nitrate Directive's 50 mg l^{-1} threshold for any waters. The 25th percentile values for TOxN are calculated to be between 0.3 and 2.21 mg l^{-1}, which arguably approximates to a background reference for the area.

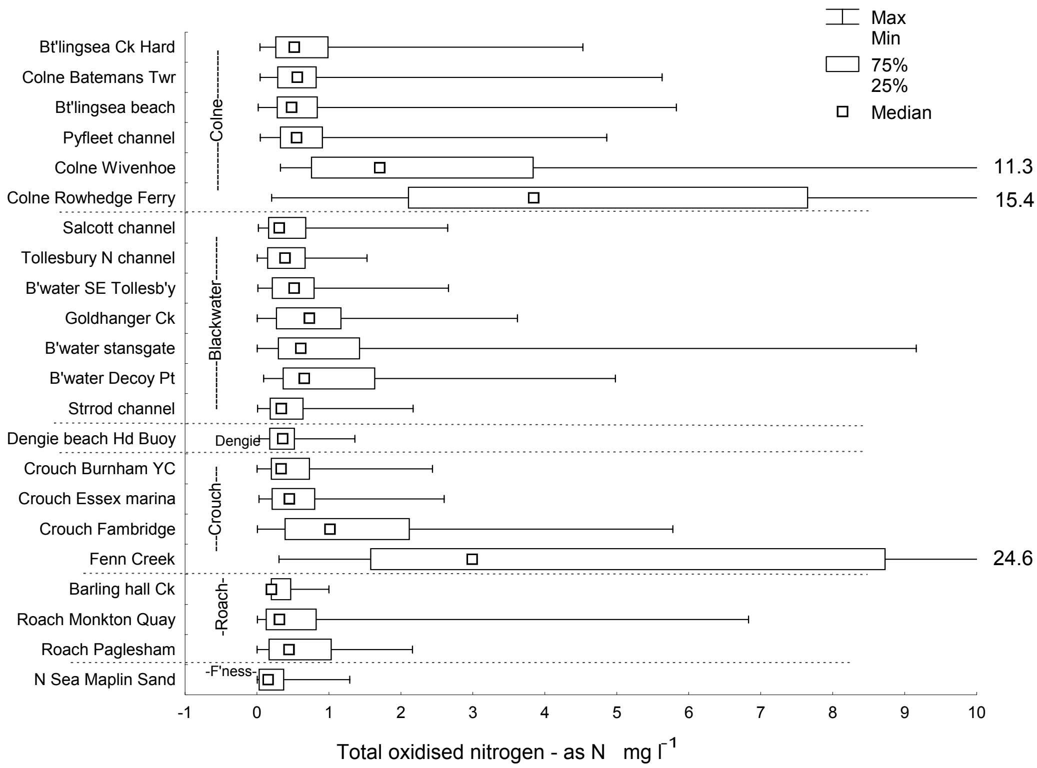


Figure 41. Nitrogen –total oxidised – as N (mg l^{-1}) in tidal waters of the Essex Estuaries 2000 – 2005 and periods therein. Data source: EA

On a temporal scale, there have been general decreases in TOxN throughout the EMS over the 5-year period (figure 42), although mean values for 2005 sampling (up until July) remain relatively high at some sites, notably Fenn Creek in the Crouch & Roach SPA (2.64 mg l^{-1}), Rowhedge Ferry and Wivenhoe Barrier in the Colne SPA (1.95 and 1.8 mg l^{-1} respectively).

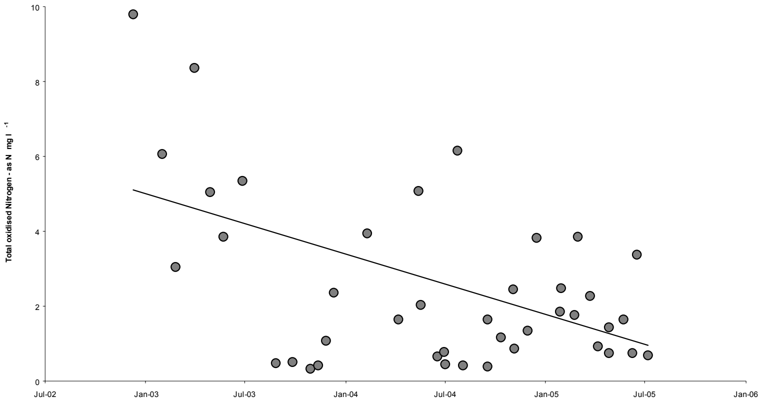


Figure 42. Temporal trend for concentrations of Nitrate - as Total Oxidisable Nitrogen (mg l^{-1}) in tidal waters of Fenn Creek, South Of Eyotts Farm, in the Crouch & Roach SPA. Data source: EA .

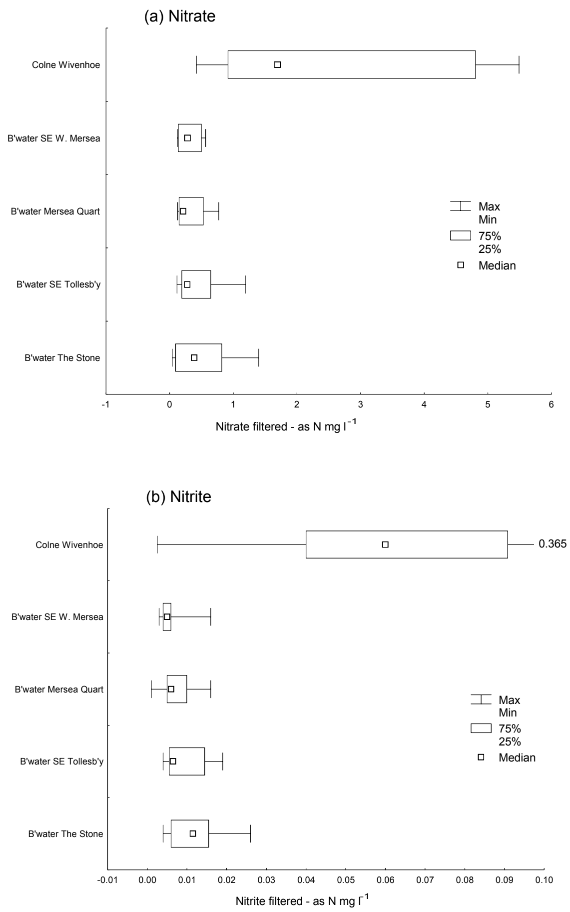


Figure 43. Nitrate (a) and Nitrite (b) concentrations - filtered as N – (mg l^{-1}) in tidal waters of the Colne and Blackwater Estuaries. Nitrate values are for 2004 – 2005 (Wivenhoe, 2003) ; Nitrite for 2004-5 (Wivenhoe, 2003 – 2005). Data source: EA . NB Note the different scales

EA monitoring data for Nitrate and Nitrite are limited to one site in the Colne and four sites in the Blackwater Estuaries, and are for 2003-5 only (figure 43). Not unexpectedly, nitrate values are higher than those for nitrite (up to 5.49 mg l^{-1} at Fenn Creek), indicating that nitrate comprises the greater proportion of TOxN (nitrate typically makes up the largest proportion of nitrogen inputs to estuaries, with nitrite and ammonia usually accounting for < 10%).

Both nitrate and nitrite levels are generally higher in the Colne where the sampling site is at Wivenhoe Barrier, toward the freshwater reaches of the estuary. In the Blackwater Estuary, values increase from the mouth of the estuary SE of West Mersea toward 'the Stone' further up the channel.

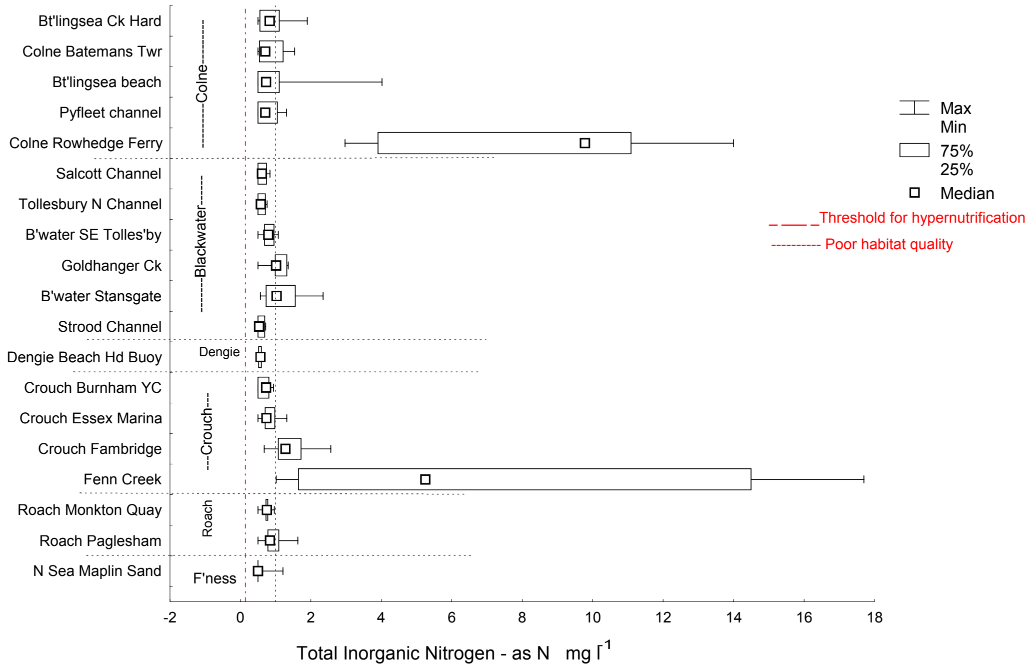


Figure 44. Nitrogen –total inorganic – as N (TIN) (mg l^{-1}) in tidal waters of the Essex Estuaries 2000. Data source: EA. See text for explanation of thresholds.

The available monitoring data for TIN is for the year 2000 only, and is represented in figure 44. Values range from 0.5 – 17.7 mg l^{-1}, generally in the upper range reported by Parr *et al* (1999) for UK coastal waters and estuaries (0.07 – 1.85mg l^{-1}). Spatially, TIN distribution is similar to that of TOxN, with 'hotspots' at inner estuarine sites; the Crouch - Fen Creek - south of Eyotts Farm, and the Colne - Rowhedge Ferry. All TIN values, (n = 181) for tidal waters sites during 2000 are higher than the TIN value (0.144mg l^{-1}) considered to represent the threshold for hypernutrification in coastal waters (North Sea Quality Status Report), and 28% are above the (1mg l^{-1}) effects level suggested by Deegan *et al* (1997) as responsible for poor habitat quality for estuarine fish populations, (due in part to cloaking effects of macroalgal mats).

6.2.1.3 Ammonia

Whereas the effects of nutrient enrichment tend to be indirect, some forms of ammonia can be toxic to marine life. A review of the effects of ammonium on estuarine and marine benthic organisms is given in Nixon *et al* (1995). Toxicity data are presented for shrimps, mysids and lobsters (in which ammonia appears to interfere with the ability of lobsters to adjust to different salinities). Estimated 96-hour LC50s for juvenile school prawns *Metapenaeus macleayi* and leader prawns *Penaeus monodon* are 1.39 and 1.69 mg un-ionised ammonia NH_3 (N) l^{-1} (26.3 and 37.4mg l^{-1} total ammonia (N)) respectively (Allan *et al*., 1990). For the nauplius of the marine copepod *Tisbe battagliai*, Williams and Brown (1992) estimated a 96-hour LC50 of 0.787 mg NH_3 (N) l^{-1} (24.6mg NH_4 (H) l^{-1}), and tests on several life stages showed a No Observed Effect Concentration (NOEC) of 0.106mg NH_3 (N) l^{-1} (3.34mg NH_4 (N) l^{-1}). For invertebrates, toxicity appears to increase as salinity decreases (Miller *et al.,* 1990, Chen and Lin 1991), although more work is needed to establish whether this pattern is typical for all, or most, invertebrates (Nixon *et al.,* 1995). Several studies indicate that ammonia toxicity is greatest to early life stages of invertebrates.

The majority of ammonium toxicity data relates to fish, although most of the species tested are freshwater species, with many coarse fish appearing to be as sensitive to ammonia as salmonids (Mallet *et al.,* 1992). Acute toxicity of ammonia to fish increases with low dissolved oxygen concentrations in both fresh and marine water environments (Seager *et al.,* 1988, Nixon *et al.,* 1995). For this reason, the proposed GQA scheme for ammonia in estuaries was combined in a proposed joint scheme for dissolved oxygen and ammonia (Nixon *et al.,* 1995).

Ammonium toxicity to fish is also related to salinity, and appears to be lowest at intermediate salinities (~10psu), but below this may increase as salinity reduces towards freshwater (Seager *et al*., 1988, Miller *et al*., 1990). This may be of relevance, in terms of synergistic effects with low oxygen especially in estuaries where DO sags can occur at low salinities.

Ammonia does not accumulate in the sediments, although ammonifying microbial activity in sediments can result in ammonia release. This activity is greatest when large quantities of macroalgal biomass decline (Owens and Stewart, 1983) and is potentially toxic to sediment dwelling organisms and those organisms that use water in the boundary layer between the sediment and the water column for feeding or respiration (molluscs, crustacea and most annelids).

Ammonia is present in all natural waters, even if only at very low concentrations. It is derived either from the breakdown of organic nitrogen (mineralisation) or by the reduction of nitrate (a process known as denitrification). Ammonia as an intermediate stage in nitrogen fixation (conversion of atmospheric N_2 to fixed nitrogen and subsequent incorporation into microbial proteins, etc) is a relatively unimportant source in comparison to mineralisation (Cole *et al.,* 1999). However, anthropogenic sources are generally more important in estuaries, notably sewage treatment effluent and, in some situations, run-off from agricultural land (Seager *et al* 1988). The toxicity of ammonia can therefore be a cause for concern in estuarine European marine sites close to STW outfalls.

The un-ionised form of the ammonium ion (NH_3) is the most toxic although ammonia as N is more commonly monitored. The toxicity of ammonia to aquatic life is affected by temperature, pH, dissolved oxygen and salinity. In general, ammonia toxicity is greater, the higher the temperature and pH and the lower the levels of dissolved oxygen and salinity. Of these three factors, salinity is the least important .

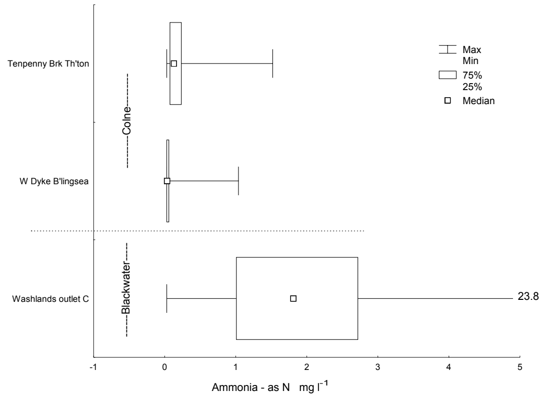


Figure 45. Ammonia (filtered as N) in fresh water entering the Colne and Blackwater Estuaries. Data are for 2000 – 2005 (Colne) and 2000 - 2004 (Blackwater). Data source EA

Again, the few data for concentrations of ammonia in freshwaters entering the Estuaries, are for the three sites close to, or in the receiving waters of major STW discharge outfalls. The highest concentrations of ammonia - as N in these freshwater sources occurred in Washlands outlet 'C' to the Blackwater Estuary (up to 23.8 mg l^{-1} - figure 45). Median levels of ammonia ranged from 0.034 mg l^{-1} in West Dyke Brightlingsea to 1.81 mg l^{-1} in Washlands outlet 'C'. Loadings information is needed however, since although ammonia concentrations entering the Colne from discharges are no higher than the input at Washlands, levels in the upper tidal sections of the Colne are sometimes higher.

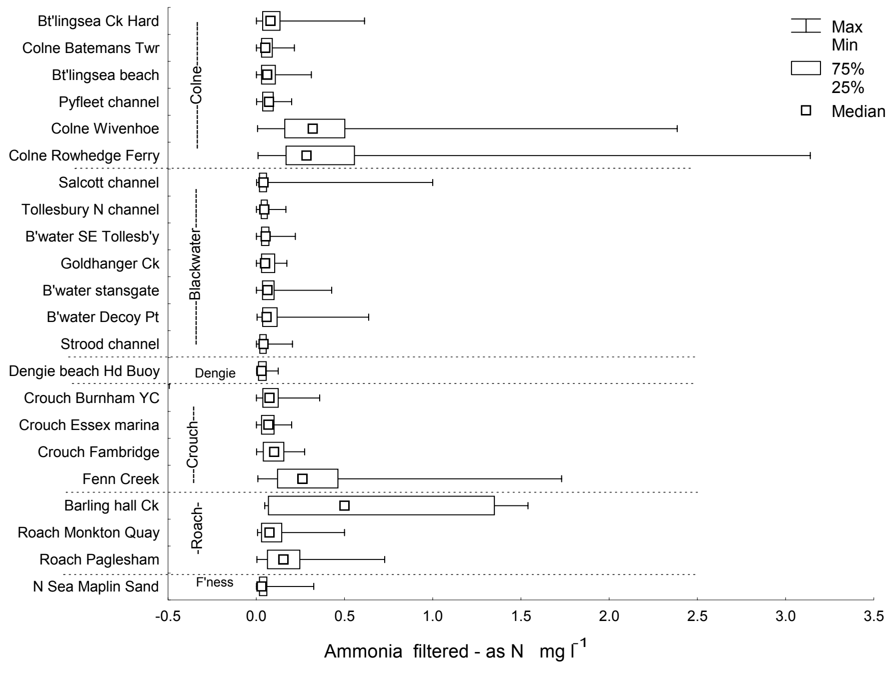


Figure 46. Ammonia (filtered as N) in tidal waters of the Essex Estuaries EMS. Data are for 2000 – 2005 and periods therein. Data source EA

EA monitoring data for ammonia (filtered as N) in tidal waters of the EMS are summarised in figure 46.). For some sites the data are for all years 2000 – 2005, whilst for others the data are for 2003-2005 or 2000-2003. Values ranged from 0.01 – 3.14 mg l^{-1}, and did not exceed the 8.0 mg l^{-1} –N MAC (maximum allowable concentration) proposed for UK waters. Highest individual concentrations were recorded towards the freshwater end of the Colne Estuary (Rowhedge Ferry).

In recognition of the impact that the ammonium ion may have at higher salinities, a total ammonia limit of **1.1 mg l^{-1} -N AA (annual average)** and **8.0 mg l^{-1} –N MAC (maximum allowable concentration)** is also proposed for UK waters based on USEPA, 1989, 1999 recommendations and a review carried out by Eddy (2004) for the Environment Agency.

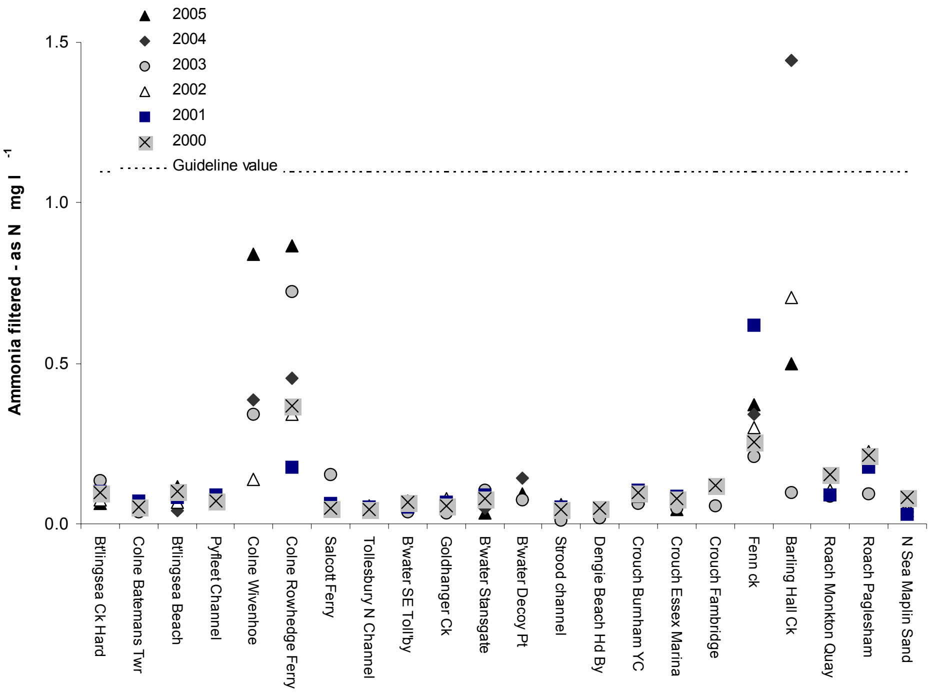


Figure 47. Mean annual levels of ammonia (filtered as N) in tidal waters of the Essex Estuaries. Data are for 2000 – 2005 and periods therein. Guideline value shown. Data source EA.

In order to compare values with standards and guidelines for ammonia, EA data have been also plotted as annual averages (figure 47). The guideline value of 1.1 mg l^{-1} -N (AA) was exceeded in 2004 at Barling Hall Creek, a tributary of the Roach Estuary.

Over the five-year period, mean ammonia levels have varied, generally decreasing or remaining similar at most sites throughout the estuary. Notable exceptions are Rowhedge Ferry and Wivenhoe Barrier in the inner Colne Estuary, where there have been significant increases in ammonia concentrations (Rowhedge r^2 = 0.10, p <0.01; Wivenhoe r^2 = 0.13, p<0.05, respectively) and annual averages for 2005 (up until July) approached the guideline value. Temporal trends in annual averages for this period are shown in figure 47.

Note that the ammonia data in figures 46 and 47 are for ammonia as N, and values for the more toxic unionised ammonia, NH_3 (N), have to be calculated from the these values, taking account of pH, temperature, and salinity. As a rough guide; for a pH of 8.2, a temperature of 20°C, and a salinity of about 30, 0.44 mg l^{-1} total ammonia (N) relates to about 0.021mg l^{-1} NH_3 (N), which is the proposed EQS (as 95th percentile).

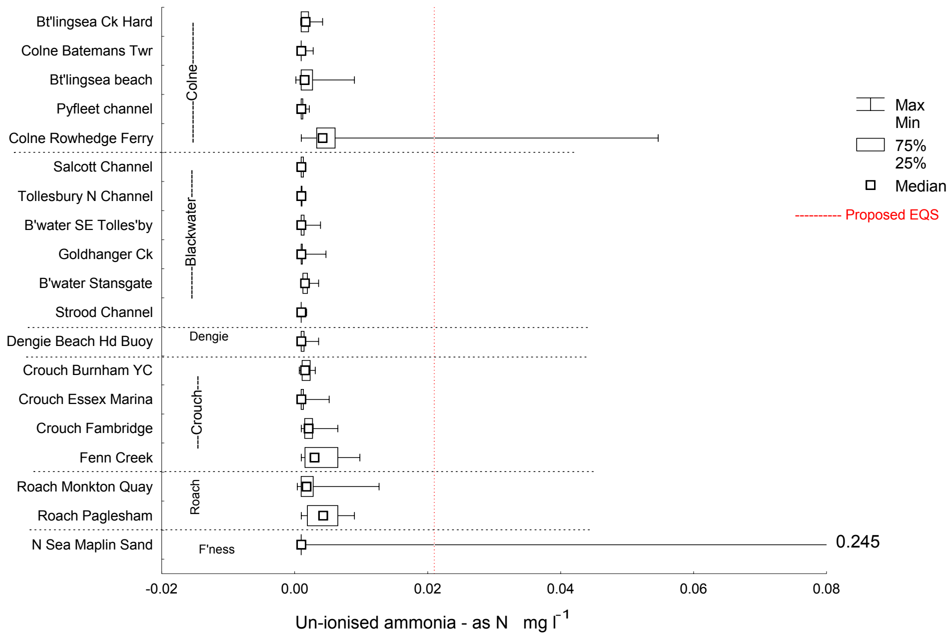


Figure 48. Unionised ammonia, NH_3 (N), in tidal waters of Essex Estuaries. Data are for 2000 - 2001. Data source EA. NB **The proposed EQS of 0.021mg l^{-1} NH_3 (N) is shown for information only as it relates to an annual average.**

Using monitoring data, EA have calculated the equivalent unionised ammonia, NH_3 (N) from some of this data and the results are shown in figure 48. The highest individual concentration (0.245 mg l^{-1}) appears to have been recorded at the Maplin Sands sampling site, although this seems questionable and may be an error in the data. Elsewhere, the more elevated concentrations occurred at Rowhedge Ferry in the Colne SPA - 0.056 mg l^{-1}, and Monkton Quay, near the mouth of the Roach in Foulness SPA - 0.013 mg l^{-1}. Though high, these values appear to be compliant with the EQS.

As data for unionised ammonia are limited to 2000 and 2001, temporal trend analysis is not possible. Again data is not available for concentrations in freshwater inputs, STW or industrial discharges; therefore the 'hotspots' cannot be directly attributed to sources.

6.2.2 Microalgae -Chlorophyll

It is important to distinguish between natural blooms and those induced by "artificial" causes. Levels of chlorophyll would be expected to increase in spring due to the natural spring bloom. It is pronounced or persistent blooms which cause concern. Elevated and prolonged spring and summer levels of chlorophyll *a* are one of the primary symptoms of increased nutrient inputs to estuarine waters and, as such, are another response variable measurement. Chlorophyll *a* is the molecule mediating photosynthesis in almost all green plants including phytoplankton. Rapid proliferation or blooms of phytoplankton, as reflected in elevated chlorophyll *a* levels, can occur throughout the ocean but are typically associated with temperate coastal and estuarine waters such as the Essex Estuaries EMS. During winter months, growth of phytoplankton populations are at a minimum because of reduced temperature, light availability, and water column stability, and chlorophyll-*a* levels generally remain low. Monitoring of chlorophyll *a* is more often restricted to spring and summer months when estuarine concentrations in optimum growing conditions may exceed 50-80µg l^{-1} (Monbet 1992).

In the UK, an indicator (mean) value for suspected eutrophic conditions is set at 10µg l^{-1} chlorophyll *a* (Dong *et al.,* 2000). Criteria used to inform UWWTD and WFD risk assessment suggests a concern when chlorophyll-a concentrations reach >100 µg l^{-1} (max) and 25µg l^{-1} (annual average) in fresh water or 10 µg l^{-1} in fully saline waters. In addition for its OSPAR assessments of marine waters, the UK has adopted an annual average value of 15µg l^{-1} to indicate a "problem area". This is based on waters having 50% higher chlorophyll levels than regional background (Wither, 2003a).

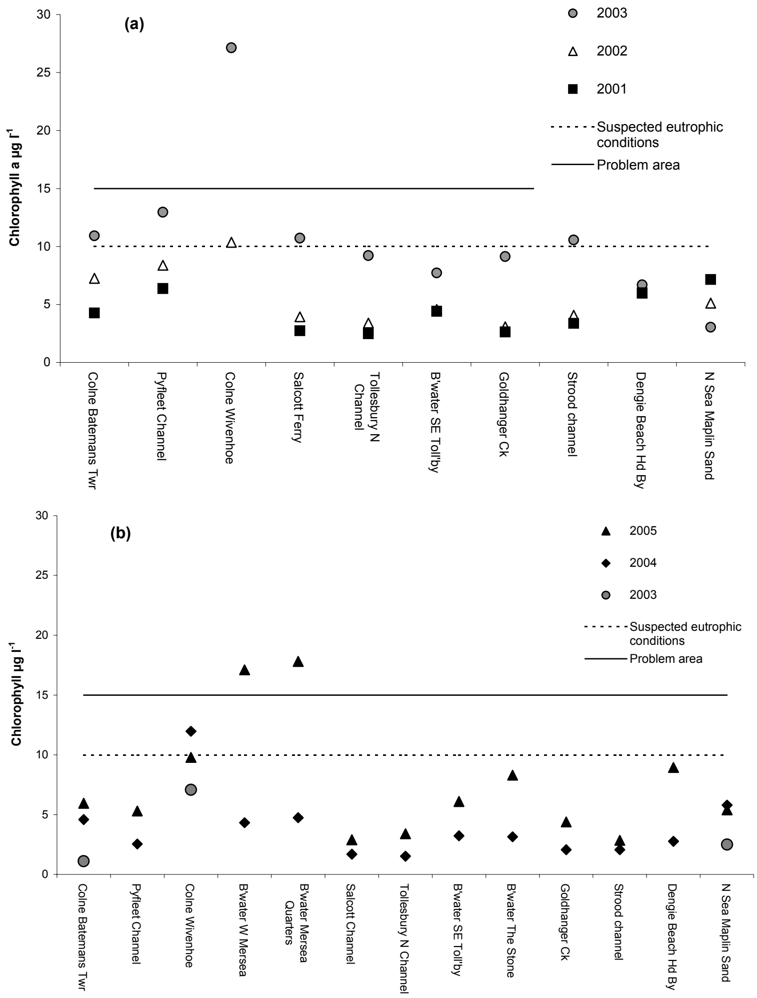


Figure 52. Mean annual chlorophyll concentrations (µg l^{-1}) for tidal waters of the EMS. Data are for (a) Chlorophyll-a, 2001 - 2003 (b) Chlorophyll, 2003 – July 2005. Data source:EA.

The available EA monitoring data for chlorophyll in the EMS are very patchy, and are for ‘chlorophyll a - methanol extract’ (2001-3), and ‘chlorophyll - by cold acetone extraction and spectrophotometry’ (2003-5). These data are summarised in figure 52. For most sites, values represent samples taken 5 to 12 times per year, mostly between March and October, although measurements taken in November to January are included at some sites. Nevertheless figure 52 gives a general indication of levels in the estuary.

Individual values for chlorophyll-a range from 1.0 to 87 µg l^{-1}; and 1.0 to 46 µg l^{-1} for ‘chlorophyll’; generally within the normal range during spring and autumn blooms (up to 80µg l^{-1}) and below the 100 µg l^{-1} (max) criteria used by UWWTD and WFD to suggest problems. Highest values were recorded between May and October. Mean annual concentrations appear to be most elevated (exceeding the criteria that indicates a ‘problem area’), at Wivenhoe in the Colne (2003) and near West Mersea (up to July 2005), although with such patchy data, it is not possible to assess the situation with any confidence. Elsewhere, mean annual concentrations of chlorophyll for recent years scarcely exceeded the lower threshold for eutrophic conditions, of 10µg l^{-1}.

6.2.3 Macroalgae

Macroalgae are a common feature of most estuaries. Green macroalgae are the natural flora of inner estuaries as they are generally more tolerant of salinity fluctuations than are red and brown seaweeds. Thus, the mere presence of macroalgae does not necessarily indicate nutrient enrichment or degraded conditions. It is the excessive growth of opportunist species, which blanket the substratum that may be a consequence of high nutrient levels.

Because of their simple structure and wide physiological tolerances, green seaweeds such as *Enteromorpha, Ulva, Chaetomorpha*, *Cladophora* and simple brown seaweeds such as *Pilayella* and *Ectocarpus* can utilize elevated levels of nitrogenous and phosphorus nutrients to form blooms. The resultant mats are often monospecific, although different species can dominate at different shore heights and at different times in the season (Fernandes *et al.,* 2004). Such blooms of opportunistic macroalgae, principally very fast-growing species of simple green seaweeds, are increasingly becoming a problem in many shallow marine areas. Morand & Briand (1996) report that the thickness of macroalgal mats can vary between 2 to 100 cm and weigh between 0.2 to 400 kg m^{-2}. These algae can outcompete the slower growing, longer-lived seaweeds, such as fucoids, which they often replace at high density. Macroalgal mats may be attached or free-living, and can occur intertidally or sublittorally. Intertidally, they tend to blanket sand and mud flats causing reducing conditions in the underlying sediment. Anoxia and ammonia release in sediment beneath the mat, and within lower layers of the mat, can inhibit benthic invertebrate populations and interfere with feeding by wading birds. Sublittorally, they can also outcompete and replace seagrasses, blanketing whole beds and causing problems for the diverse fauna usually associated with seagrass. Indirectly, therefore, the secondary productivity of benthos will almost certainly be linked to nutrient status through effects on sediment and epibenthic flora, including phytoplankton.

As blooms subside, only part of the biomass of algae is consumed by grazers, and a pool of organic nutrients can build up in sediments. This can persist after nutrient sources are reduced. Thus blooms may not be immediately cured by changes in effluent treatment (Fernandes *et al.,* 2004).

Setting an EQS for macroalgal cover at a site is obviously a difficult task, as quantifying macroalgal cover and assessing the potential threat to the integrity of a site can be largely subjective. A number of complex guidance criteria are currently used by the EA (from Wither, 2003b):

1) Greater than 25% of the inter-tidal area is impacted with green seaweed, each sub area is defined as impacted if there is at least 25% cover.
2) A DETR workshop held in 2001 made the following recommendations:

   a The preliminary reference level = 5% cover*
   b A problem area is one with > 15% cover of intertidal area on soft sediments.
   c Some rocky shore areas can have ~100% cover with no adverse impact.

   Although (1) implies 6.25% (25% of 25%) cover can represent a problem area, in reality impacted zones may typically have ca. 60% cover. This is consistent with (b) (25% of 60% =15%)

   d reference level for mass of weed = $100 g/m^2$ wet wt.
   e up to $500\ g/m^2$ wet wt. is not a problem
   f $1\ 000\ g/m^2$ wet wt. is a problem

   **The term cover does not mean the % of the estuary with some weed cover, it means the % of the total area covered by weed. For example if surveys find the following:*

   50 hectares with 0 cover

   10 hectares with 1-25% cover

   10 hectares with 26-50% cover

   5 hectares with 100% cover

The % cover would be 13.4% not 33%.
i.e. ((1+25)/2x10/100 + (26+50)/2 x10/100 + 100x5/100) / 75

The guidance goes on to say that the above criteria should not be used in isolation. Consideration needs to be focussed on the consequences of the excess algal coverage for the functioning of the ecosystem and the consequent effects on the interest features. Ideally, to demonstrate a problem there should be some supporting evidence of adverse effects, for example

Invertebrate fauna reduced
1) Wading bird feeding distribution modified
2) Cockle numbers reduced
3) Deposited weed smothers other saltmarsh vegetation
4) Public complaints about odour
5) Floating rafts of weed impacting on boating activity
6) Anoxia in surface sediment layer (e.g. top 2 cm)

However, evidence of adverse biological effects, though preferred, may not always be available, requiring expert judgment for the appropriate assessment.

In common with many affected sites, there is a dearth of good data on the extent and regularity of macroalgal blooms in Essex Estuaries EMS and reports of problems are largely anecdotal. At Tollesbury saltmarsh adjacent to the Old Hall Marsh on the northern shore of the Blackwater Estuary, growth mats of the alga *Enteromorpha* have been linked to poorer establishment of *Salicornia* spp (Reading *et al.,* 2000). The Environment Agency have recently conducted a survey of intertidal areas of Essex Estuaries by helicopter using digital photography (figures 53A and B).

(A)

(B)

Figure 53. Aerial photographs taken July, 2005. (A) the Blackwater Estuary showing 50% macroalgal cover. (B) the Roach Estuary showing <5% algal cover. Source:EA

With the exception of Foulness SPA, all of the component estuaries in the EMS were affected by macroalgal cover to some extent. The Blackwater Estuary appeared to be the most impacted with approximately 50% algal cover (figure 53A). The Colne, Crouch and Roach had <5% and the Dengie <1%. These EA results are summarised in figure 54.

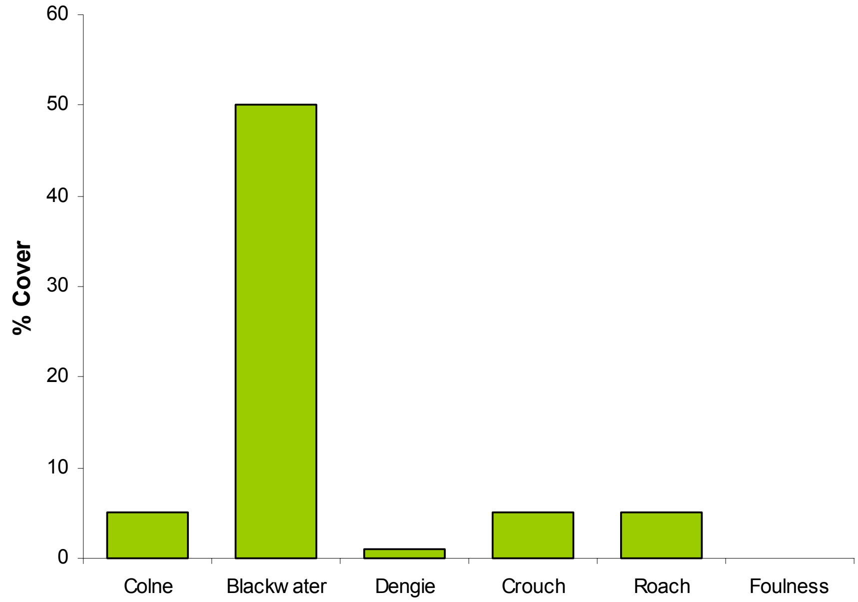


Figure 54. Macroagal cover in Essex Estuaries EMS assessed from digital photographs taken on helicopter flights over the site on 25th July 2005. Data source: EA

6.2.4 Dissolved Oxygen

The principal sources of DO in the marine environment are the atmosphere, via O_2 gaseous exchange across the air-sea surface, and *in situ* production by algae and aquatic plants during photosynthesis. DO levels vary with temperature, with lowest levels in estuaries occurring during the summer months. MPMMG (1998) reported summer and winter concentrations of DO at National Monitoring Programme sites in the UK in the range 4 to 11 mg l^{-1} expressed as a median, with lowest concentrations occurring in estuaries during the summer.

Increased levels of nutrients in estuarine waters can stimulate growth of both macro algae and phytoplankton (algal bloom), resulting in an intensification of both seasonal and diurnal variation in DO. Daytime photosynthetic activity may result in O_2 supersaturation of the water column; whilst at night severe depletion can occur due to respiration. These fluctuations can cause problems for fish and invertebrate communities. During bloom die-offs, microbial decomposition of algal cells leads to an increase in oxygen demand and acute DO depletion, which again can result in lethal and sub-lethal effects to fish and invertebrate communities.

Parts of the Essex Estuaries may be particularly vulnerable, and DO problems could be anticipated in some areas of the EMS such as the inner estuaries, where nutrient levels are highest.

DO is measured in estuaries and coastal waters in terms of either a concentration (mg l^{-1}) or as a percent saturation (%). Table 24 shows recommended EQS values for saline waters derived from the review of Nixon *et al.,* (1995).

Table 24. Recommended EQSs for dissolved oxygen in saline waters (from Nixon *et al.,* 1995)

Saltwater use	EQS	Compliance statistic	Notes
Designated shellfishery	70% saturation 60% saturation 80% saturation	50%ile, mandatory standard Minimum, mandatory standard 95%ile, guideline value	EC Shellfish Water Directive
Saltwater life	5 mg l^{-1} 2 mg l^{-1}	50%ile 95%ile	
Sensitive saltwater life (e.g. fish nursery grounds)	9 mg l^{-1} 5 mg l^{-1}	50%ile 95%ile	
Migratory fish	5 mg l^{-1} , 3 mg l^{-1}	50%ile, 95%ile	Higher values may be required where fish have to traverse distances >10 km, or where high quality migratory fisheries are to be maintained

In addition, various class thresholds for estuaries in England and Wales, based on DO over a continuous period of >1 hour were proposed by Nixon *et al.,* (1995) (see table 25) and although this scheme has not been implemented, the class thresholds are a useful indication of the levels of DO that are likely to cause effects if organisms are exposed for a continuous period of greater than one hour.

Table 25. Proposed GQA class thresholds for dissolved oxygen in estuaries in England and Wales (from Nixon *et al.,* 1995)

GQA class boundary	Threshold value of DO (mg l^{-1})
A/B	8 mg l^{-1}
B/C	4 mg l^{-1}
C/D	2 mg l^{-}

More recently, the Water Quality Technical Advisory Group (WQTAG) have recommended that a salinity-related standard is the most practical approach, as the solubility of oxygen declines as salinity increases; e.g. the solubility at 10°C declines from 11.3 mg l^{-1} in fresh water to 9.1 mg l^{-1} in sea water (Wither, 2004). The threshold determined for triggering likely significant effects for dissolved oxygen levels in estuaries is based on a saline level of 5 mg l^{-1} (annual 5%ile) and is consistent with the EQS recommended in Nixon *et al.,* (1995) to protect sensitive saltwater life (table 24). In freshwater the salinity adjusted level is 6 mg l^{-1} This is consistent with the Freshwater Fish Directive which sets an annual 5%ile of 6 mg l^{-1} as the trigger level in salmonid waters. The salinity related threshold is represented by the line in figure 49.

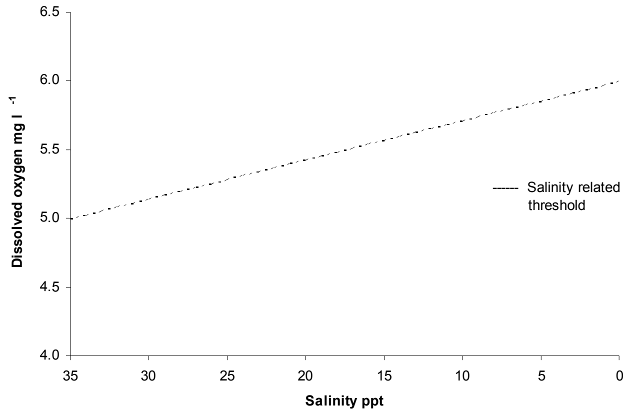


Figure 49. Dissolved oxygen threshold vs salinity.

As the appropriate threshold value changes with salinity, calculating the 5%ile from field data can be difficult. Suggested approaches to calculation of the 5%ile include:

- Plotting field data against salinity and check whether 5% or more of the points lie below the threshold line
- Converting all data to the 'zero salinity equivalent value' using the following simplified equation: DO (adjusted to salinity zero) = DO (at salinity x) + x/35 then calculate the 5%-ile and compare with a value of 6mg/l.[29]

The EA monitoring data for dissolved oxygen in waters of the Essex Estuaries EMS are for the period 2000 to 2005 and are generally for tidal waters. The data are a mixture of mg l^{-1} and % saturation records, depending on date, enabling some comparison with EQSs. The former values are summarised in figure 50.

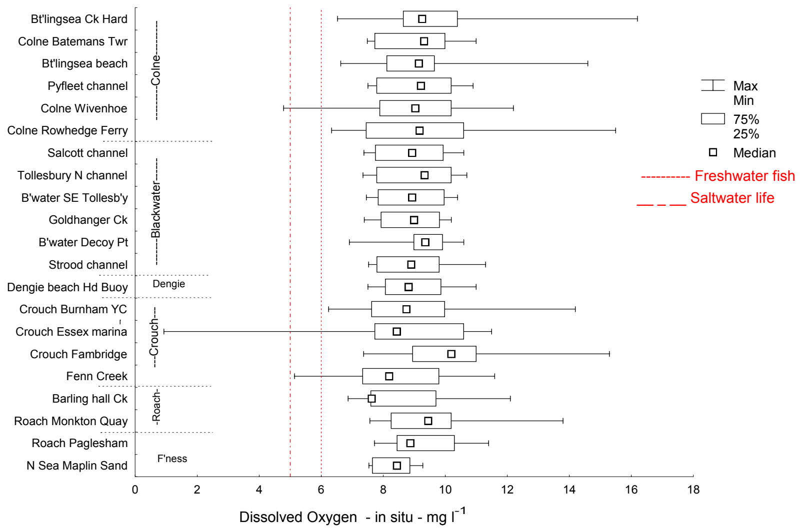


Figure 50. Dissolved oxygen (mg l^{-1}) in tidal waters of the Essex Estuaries 2003 - 2005. Data source:EA

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[29] NB. It is noted in the guidance notes that estimations of the 5%ile are not necessarily required for each individual sample point, and that it might be inappropriate to represent all the data for one estuary on one plot. The estuary may therefore be divided into a number of 'representative' areas and the 5%ile estimated for each of those areas. Subjective judgement will be needed to define these areas (Wither, 2004).

Data for DO (mg l^{-1}) in waters of the EMS ranged from 0.926 to 16.2 mg l^{-1}, with values generally in the range for UK estuaries reported by MMPMG (1998) (4 - 11mg l^{-1}). Lowest individual values occurred at Essex Marina in the Crouch and Wivenhoe in the Colne (0.926 and 4.79mg l^{-1} respectively). Unsurprisingly, summer values reflect the greatest oxygen depletion. Median DO for the period did not fall below the 5 mg l^{-1} recommended EQS for sensitive saltwater life (or 6 mg l^{-1} for freshwater) between 2003-5 (figure 50).

The available salinity data do not tie up with sampling date and times for DO samples, therefore the data cannot be plotted against salinity as recommended by Wither (2004) (figure 49), However as very few individual values were below the threshold levels (0.5% <5 mg l^{-1}; 1.6% <6 mg l^{-1}), this would seem unnecessary.

Data for dissolved oxygen (% saturation) give the greatest temporal coverage for the period 2000 to 2005. 50%ile and 95%ile comply with recommended EQS values for dissolved oxygen (70% and 80% respectively) (table 24) and values fell below the minimum mandatory standard (60% saturation) on two occasions/sites only. These %ile standards apply to designated shellfish waters only and are included to serve as a rough guide to water quality in relation to DO % saturation (designated shellfish waters within the EMS are listed in annex 6).

Figure 51 shows DO as % saturation, presented as mean annual values for each sampling site, which did not fall below 80% saturation during this period. Thus, DO levels appear to be good in the EMS, however, it is worth noting that 35% of all individual values for the period have exceeded 100% DO saturation indicating possible O_2 supersaturation. This is sometimes symptomatic of algal blooms (Cole *et al.,,* 1999).

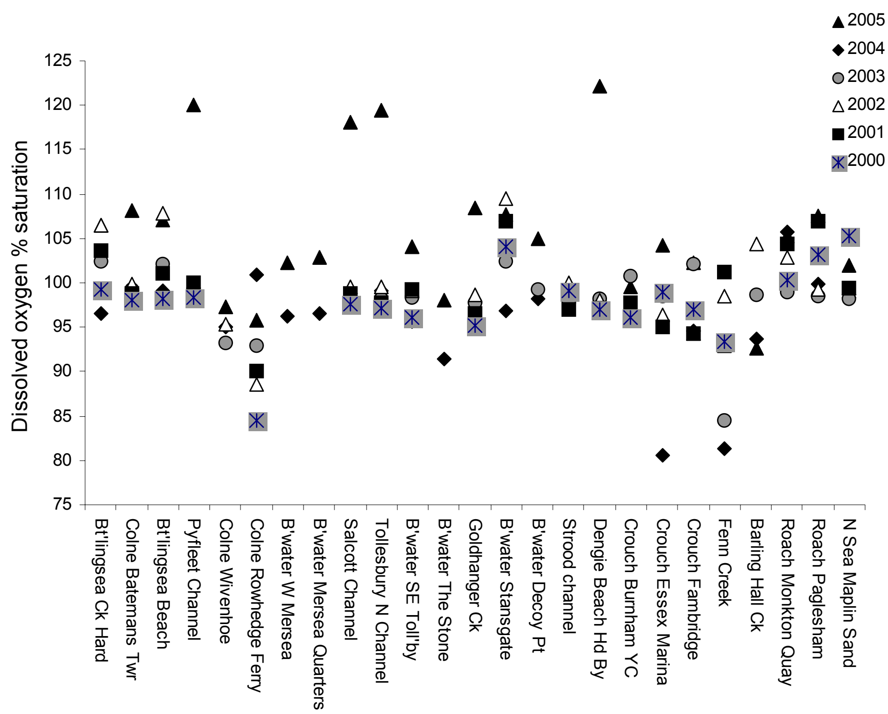

Figure 51. Mean annual levels of dissolved oxygen (% saturation) in tidal waters of the Essex Estuaries EMS. Data are for 2000 – 2005 (up until July for 2005) and periods therein. Data source EA.

6.2.5 Turbidity

Turbidity is a measure of the attenuation of light in the water column and may be defined as the properties of water that cause light to be scattered and absorbed. Turbidity is caused by particles and dissolved substances in water, including organic and inorganic particulate suspended matter, and dissolved substances that contribute to the colour of water. During blooms, the organic component can include significant amounts of algae.

The composition of particulate matter varies but is derived from: directly eroded material, sediments that have settled to the substratum and become resuspended during periods of high flow, dredging, suspended solids in discharges, chemical flocculation (at the salt/freshwater interface) and plankton.

Methods for measuring turbidity vary, utilising different combinations of light transmission and scattering, water transparency (secchi disc) and suspended solids (sample filtered and dried at 105^{o}C or 500^{o}C) or remote sensing. The results of these methods are not readily inter-convertible, making comparisons problematic.

Two principal methods are used by the EA for quantifying turbidity in waters of the Essex Estuaries EMS: light scattering, measured using a turbidimeter calibrated with Formazin (units Formazin Turbidity Units, FTU), and transparency. There are also data for suspended solids @105°C although spatial coverage of these data is not as widespread as for FTU. For the purposes of this report FTU units and suspended solids have been used to assess turbidity, and transparency for comparison with EQS.

Turbidity data for tidal waters of the EMS are for the period 2000 – July 2005. Figure 55 exemplifies the data expressed as FTU. The more elevated median values for the period are for Monkton Quay in the Roach and Fenn Creek in the Crouch (34.1 and 31.5 FTU, respectively). Individual values are in the range 1.2 – 1130 FTU and are highest for the Colne - Brightlingsea Beach (at Batemans Tower) and the Roach at Paglesham (Foulness).

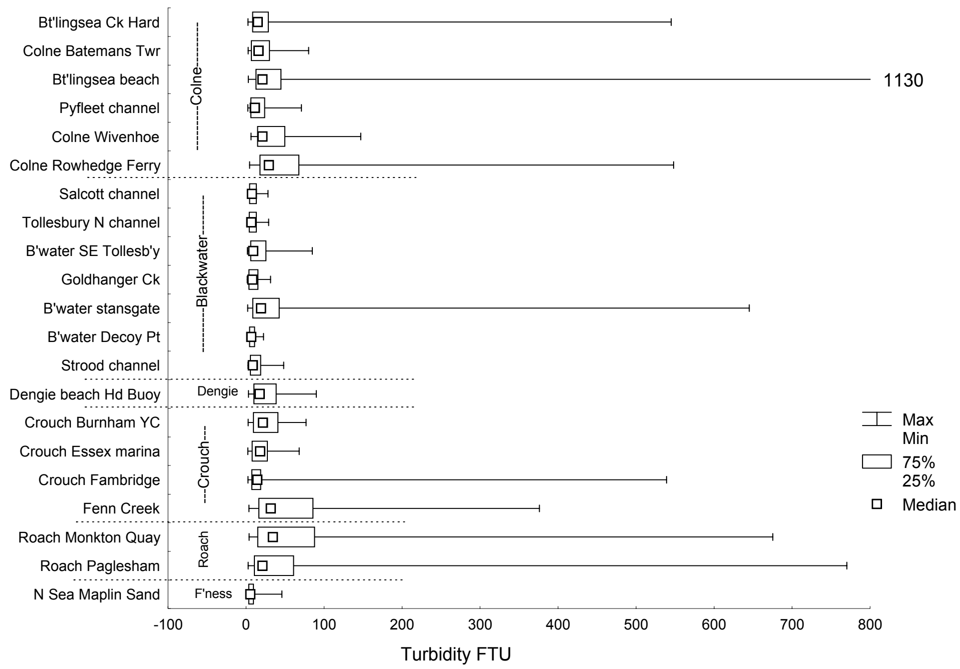

Figure 55. Turbidity (FTU) in waters of Essex Estuaries EMS. Data are for 2000 – 2005 and periods therein. FTU = Formazin Turbidity Units and refers to the standard used to calibrate the turbidimeter. (Data source: EA)

Data for suspended solids (105oC) in the water column of the EMS are summarised in figure 56 as annual means. During the period 2000 - 2005, values for annual means ranged from 5.7 - 289mg l-1, with highest values, again, in the Blackwater (Stansgate) and Foulness SPA in the Roach.

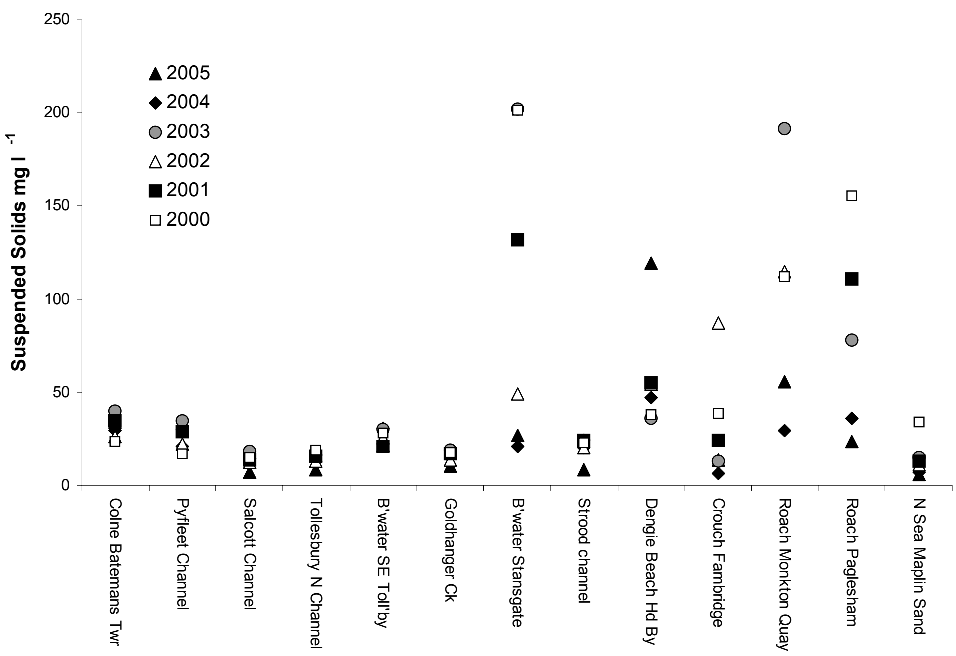

Figure 56. Mean annual values for suspended solids (mg l^{-1} @105°C) in waters of the EMS. Data source; EA

To put these turbidity levels into some perspective, Cole *et al* (1999) cited typical annual values for mean suspended solids (105^{o}C) around the English and Welsh coast as 1-110mg l^{-1}, and suggested that anything >100 mg l^{-1} could be considered high. Using this criterion, some areas of the EMS therefore appear to suffer from relatively high turbidity.

Increased or sustained turbidity in the water column can result in a reduction in algal (macroalgae and phytoplankton) growth rates due to reduced light availability. Subsequent adverse effects to zooplankton, benthic communities and fish populations (a general reduction in biodiversity) would be anticipated as particulates are suspended and re-deposited. An accompanying reduction in food availability may have secondary effects to higher trophic levels. However, the principal source of turbidity is often quoted as being sediment resuspension (Parr *et al.,* 1998) and peak levels are generally confined to a discrete area in the mid-upper reaches of the system, which moves up and down with the tide (Cole *et al*., 1999). The level of suspended solids depends on a variety of factors, including: substrate type, river flow, tidal height, water velocity, wind reach/speed and depth of water mixing (Parr *et al*., 1998). Turbidity measurements could therefore be significantly influenced by these natural characteristics as opposed to anthropogenic impacts.

The only EQS for turbidity appears under the Bathing Waters Directive (76/160/EEC) and relates to transparency using a secchi disc (guide value 90th percentile >2m, imperative 95th percentile >1m). Designated bathing beaches within the EMS are Brightlingsea, West Mersea and Jaywick, (east of Colne Point). Guideline and imperative values are only applicable during the bathing season and may be waived in the event of 'exceptional weather or geographical conditions'. Nevertheless, for comparison with these quality standards, and as a further guide to the water quality in the EMS, we have calculated the appropriate values from the transparency data and summarised them in figure 57.

For the majority of sampling sites within the EMS, 90%ile values fall below the 2m guideline. Lowest 90%ile values occur in the Colne at Wivenhoe and Blackwater at West Mersea, and 95%ile values for these two sites also fall below the imperative standard of 1m. However, these sites are not bathing beaches. At Brightlingsea Beach (at Batemans Tower) and West Mersea Beach (opposite Fairhaven Ave- also an NMMP Shellfish Site) the majority of data for these sites are recorded as 'less than, (or, less frequently, 'greater than' 1), hence both 90th and 95th percentile values in figure 57 approximate to the EQS. Table 26 shows the data for these sites, broken down into total number (n), number of actual measurements, number of 'less than 'and 'greater than' values recorded during 2000-2005. On this analysis, it appears that transparency failed to comply with the EQS.

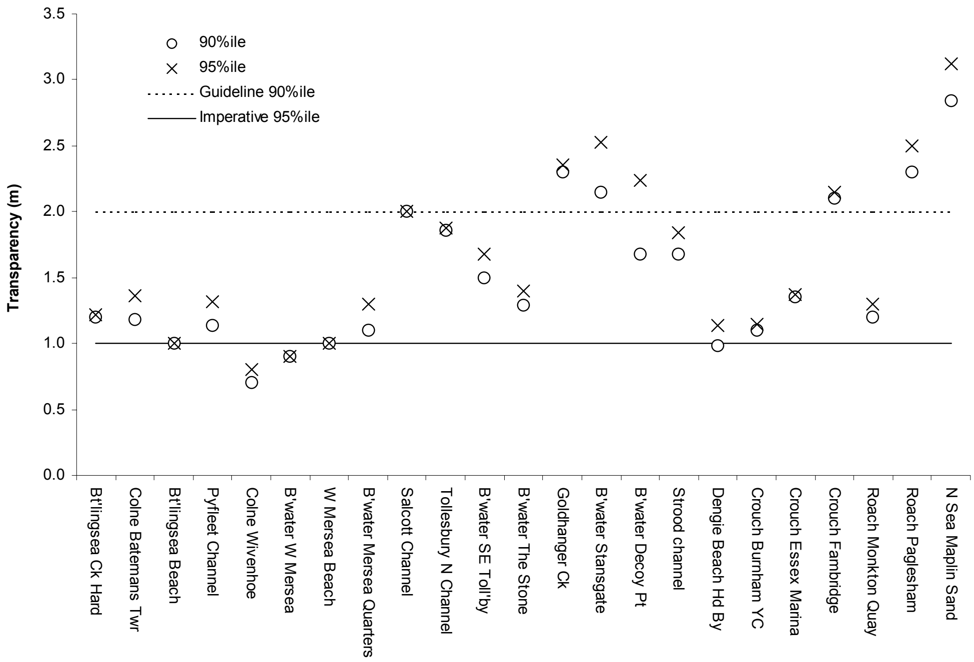

Figure 57. 90th and 95th percentile values for transparency (m) in waters of Essex Estuaries EMS. Data are for 2000 – 2005 and periods therein. (Data source: EA). Guideline and Imperative standards shown as stipulated in the Bathing Waters Directive 76/160/EEC for fresh and saline waters (note that the standards are minimum values).

Table 26. Breakdown of EA monitoring data for Bathing Water sites

	n	Actual	<1	>1
Brightlingsea Beach	230	12	186	32
West Mersea Beach	226	24	148	54

7. BIOLOGICAL IMPACTS

These are reviewed here under two sections. The first deals with bioaccumulation (an indication of exposure to contaminants) whilst the second lists evidence for effects at various levels (biochemical and molecular to community structure and function).

7.1 Bioaccumulation

Metals

Measurements of contaminants in sediments and water are a useful guide to environmental contamination, but ultimately it is the impact on biota which is of most concern. One of the arguments against the use of EQSs as a measure to achieve environmental protection is that they do not take account of bioavailability. For example, it may not be cost-effective, or protective, to upgrade a discharge which is shown to exceed the EQS if bioaccumulation studies indicate the substance(s) is not biologically available. Conversely, enhanced bioavailability can sometimes occur as a result of changing environmental conditions at a site, even though concentrations of contaminant in water and sediment appear unremarkable. Clearly this is a complex issue which is further complicated by the fact that there is no single universal bioindicator. Different organisms react differently to pollutants; usually it is preferable to examine a variety of different ecological/feeding types to appreciate the range of responses (Bryan *et al*., 1985).

For this reason we have examined a range of available information to provide a broad impression of bioaccumulation, and to assess whether or not body burdens are influenced by anthropogenic activities. EA biomonitoring data focus on mussels from the NMMP shellfish site at West Mersea. Cefas have conducted a number of long-term surveys, with a variety of species, notably in relation to recovery from TBT. MBA also has data on metals (and TBT) in a range of species, for surveys conducted in the 1980s in the Essex estuaries (together with Foulness and Shoeburyness, between 1997-2001), to coincide with sediment data, described above. Metals results are discussed firstly, focusing on the deposit-feeding clam *Scrobicularia plana* and the omnivorous ragworm *Nereis diversicolor*.

Scrobicularia is a valuable indicator species, particular in terms of understanding trends in sediment metal bioavailability, though it is sometimes less widespread than *Nereis diversicolor,* and in places much less abundant.

Table 27 lists the mean concentrations of metals in *Scrobicularia plana* from the Essex Estuaries and Foulness. For comparison, equivalent data for the average of the lowest ten sites in our UK data set (UK min) - which encompasses most of the estuaries in England and Wales - are shown.

Scrobicularia samples in the Crouch estuary contained slightly higher levels of Cu, Pb and Sn than other Essex estuaries. The average degree of enrichment for Crouch clams, relative to UK baselines, decreases in the order Ag>Sn>Hg>Cd=Pb>Cu=Ni>Mn>Cr=As=Zn>Fe (table 27), indicating that it is the enrichment of the pollutant-type metals which is most significant. Enrichment in these

Essex Estuary clams is much less marked if compared to the regional background at Foulness, and for some metals (notably Ag, Cu) levels are higher at the latter site.

Table 27. *Scrobicularia plana*. Average metal concentrations (µg g^{-1} dry weight) in the Essex Estuaries, Foulness and UK baselines (data source: MBA).

	Ag	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Hg	As	Sn
Colne	8.52	0.87	1.20	19.7	1329	16.8	6.80	17	946	0.48	24.7	0.77
Blackwater	6.86	1.13	1.20	30.8	603	27.5	7.33	15	705	0.54	28.1	0.86
Crouch	14.1	0.96	1.10	55.2	567	23.0	5.45	25.6	851	0.50	21.7	1.87
Foulness	24.8	0.35	2.56	83.5	706	61.4	10.35	18.5	534	0.51	21.6	0.35
UK min	0.05	0.13	0.26	9.5	226	4.8	0.89	3.79	193	0.04	5.7	0.08
	Enrichment Factor											
Crouch ÷ UK min	283	7	4	6	3	5	6	7	4	13	4	23

Nereis diversicolor also accumulates most metals in amounts which reflect bioavailability in their sedimentary environment (Bryan *et al.,* 1980; 1985; Langston, 1980, 1982). Tolerance to a wide range of salinity makes *Nereis* extremely useful for monitoring in estuaries and *N. diversicolor* is relatively abundant throughout most of the EMS. As demonstrated in clams, *Nereis* in the Crouch estuary tended to display higher levels of bioaccumulation than other Essex estuaries, for a number of metals. The average degree of enrichment for Crouch worms, relative to UK baselines, decreased in the order Cd>Ag>Pb>Sn>Hg>Cr>Ni>Cu=As>Mn>Zn>Fe. This pattern is slightly different to that in *Scrobicularia* but confirms that enrichment is most significant for the pollutant-type metals, whilst the relatively low enrichment of Zn and Fe is almost certainly a result of the ability of *Nereis* to regulate these essential elements.

Thus, body burdens in different species may reflect subtle differences in bioavailability. Variability may be caused by both abiotic (e.g. redox, speciation, complexing ligands) and biotic factors (e.g. growth rates, feeding strategy, food quality and quantity) (Langston *et al*., 1998). Nevertheless, *Nereis* and *Scrobicularia* demonstrate overall similarity in terms of enrichment patterns and highlight, consistently, the bioaccumulation of 'pollutant' metals in Essex samples. As none of these metals is naturally enriched in sediments of the area, they are presumed to reflect anthropogenic origin. The high ranking of Ag is considered to reflect the significance of sewage related inputs, of which the largest source in the area is likely to be the Thames Estuary[30] (and, in the past, dumping grounds in the Barrow and Black Deeps). Several other metals may also have a sewage origin, whereas Sn, Cu and Zn could arise from a variety of sources in the area, including antifouling.

[30] Descriptions of trends in the bioavailability of metals along the Thames Estuary have been described in detail elsewhere (Langston *et al*., 2004)

Published surveys with winkles, *Littorina littorea*, confirm that Cu and Ag bioavailability was above background in the Colne, Crouch and Blackwater and that Ag may be a tracer of sewage influence albeit from sources which are principally outside the EMS (Bryan *et al*, 1983).

Elevation of body burdens above background could be taken as another indication that the Essex Estuaries are not in optimum condition. However, it is important to keep in perspective the fact that, as with sediments, levels of certain metals (such as Cu, Zn, As, Pb and Mn) can reach much higher levels in biota from heavily mineralized regions of the UK; likewise Ag, Cd and Hg concentrations in *Scrobicularia* and *Nereis* from highly industrialized estuarine regions exceed those in the Essex estuaries by a considerable margin. Body burdens in the EMS samples are therefore unlikely to be acutely toxic to the organisms themselves, though contributions to chronic effects cannot be ruled out and potential for food chain transfer exists. EN is currently engaged in a project to assess direct toxicity to birds through ingestion of prey (M. Coyle, *pers comm*.)

It is stressed that the Essex data represent a past perspective of contamination. Nevertheless, they illustrate that the use of a validated suite of indicator species is a useful way of estimating bioavailability around the system. The current status of the EMS needs to be more extensively defined in similar fashion, and at intervals in the future, to track changes in bioavailability.

Summaries of UK data on cockles, mussels and oysters collected in the mid 1990s at designated shellfish beds around the UK for statutory purposes have indicated that samples from the region were sometimes at the upper end of the UK range (Cefas, 2000). This included: Ag in cockles from the outer Thames and mussels from the Blackwater; As in Colne mussels (> 3 mg kg^{-1} ww); Cr in Blackwater mussels (>5 mg kg^{-1} ww); Cu in Thames oysters and cockles; Fe in Blackwater cockles and Thames mussels, Ni (Thames); and Se in Blackwater oysters. However; these concentrations do not appear to pose a human health risk. Even for high level shellfish consumers, estimated dietary intake of metals based on 1995-96 data, appears to represent less than half of the Provisional Tolerable Daily Intakes (PTDI) (CEFAS, 2000).

Current statutory 'bioaccumulation' monitoring by EA is focused on mussels from the designated shellfish area at the mouth of the Blackwater, collected under the NMMP programme as part of the commitment to OSPAR, who have set background/reference concentrations for Hg, Cd, Pb, Cu and Zn.[31] Based on comparisons with these reference criteria, the summary statistics for 2000-2004, for mussels from the NMMP shellfish site at West Mersea, indicate elevated values for Cd, Hg, Cu, Pb and, to a lesser extent, Zn (Table 28). The site was also singled out as being relatively enriched by UK standards in a recent review of results from the marine monitoring programme (NMMP, 2004). Using NOAA 'mussel watch' criteria however, the above group of metals would fall below the 'high category' (below the top 15% of US samples) by a small margin, though Se and As would classify as such (also by a relatively small margin).

[31] Note however that partial regulation of Cu and Zn may mean that mussels underestimate contamination of these essential metals.

Table 28. Metals and TBT in mussels *Mytilus edulis* from the West Mersea NMMP site in comparison with OSPAR and NOAA guidelines (summary statistics, 2000-2004:data source EA).

Concentrations in *Mytilus edulis* mg kg^{-1}wet weight

						guideline values		
	Median	Minimum	Maximum	25th percentile	75th percentile	lower BRC*	upper BRC*	†NOAA 'High'
Arsenic	3.12	2.05	3.69	2.6925	3.25			2.55
Cadmium	0.24	0.13	0.31	0.23	0.28	0.07	0.11	0.93
Chromium	0.18	0.12	0.27	0.17	0.22			‡0.63
Copper	1.74	1.31	2.13	1.45	1.88	0.76	1.1	1.8
Lead	0.32	0.27	0.56	0.30	0.35	0.01	0.19	0.126
Mercury	0.034	0.028	0.056	0.032	0.051	0.005	0.01	0.035
Nickel	0.39	0.31	0.51	0.36	0.45			0.495
Selenium	1.09	0.77	1.20	1.00	1.10			0.525
Silver	0.05	0.01	0.11	0.05	0.05			
Zinc	16.0	12.1	26.4	14.1	17.6	11.6	30	30
TBT	0.44	0.36	0.84	0.38	0.74	0.001	0.01	0.11

median exceeds the upper background reference concentration (OSPAR)

median exceeds the lower background reference concentration (OSPAR)

*OSPAR background/reference concentration
†NOAA 'high' category for mussels, converted to wet weight guidelines, assuming a solids content of 15%
(‡Cr guideline is US EPA screening value - EPA # 823-R-95-007)

Again it is important to put these results in context, since they are not necessarily synonymous with deleterious effects. Some of the earliest biomonitoring in the UK showed that young oysters (*Crassostrea gigas*) from the Colne accumulated Cu, but were relatively uncontaminated in comparison with those from estuaries in Cornwall and N Wales where the catchment was heavily influenced by previous mining and smelting activity (Thornton *et al*., 1975). A similar conclusion may be drawn from the data for clams, mussels, polychaetes and winkles. Metal bioaccumulation above background is evident in Essex estuaries, and in some cases may be related to anthropogenic sources; however, body burdens fall well below thresholds which are associated with deleterious effects at highly contaminated sites. Nevertheless, sublethal impacts and the potential for food chain transfer of some contaminant metals has yet to be addressed.

Because long-term data are relatively scare, temporal trends cannot be depicted with any certainty. There were no consistent changes in metal concentrations in mussels from West Mersea between 2000 and 2004 for the majority of metals, with the exception of Ag, whose annual median concentration has fallen steadily throughout this period (Figure 58). As Ag tends to be a good marker for the influence of sewage (Bryan and Langston, 1992), these results may be an indication of reduced inputs via STW, and in particular a possible result of recent improvements to the West Mersea works.

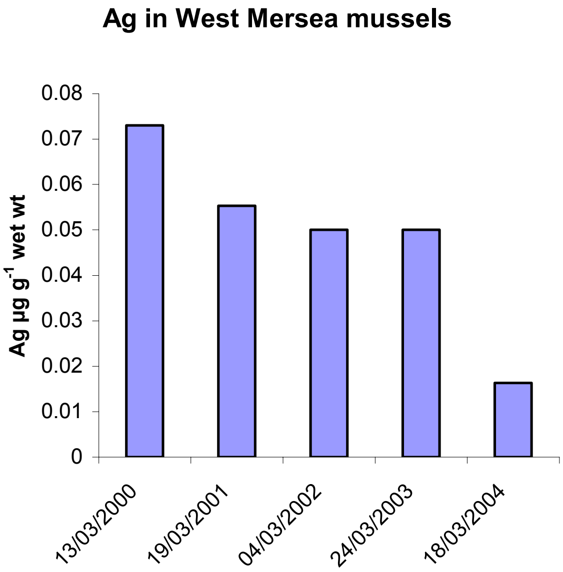

Figure 58. Ag in mussels at the West Mersea NMMP shellfish site, annual means 2000-2004 (data source EA).

Metal bioaccumulation trends in salt-marsh plants follow distributional patterns seen in sediments and invertebrates. Comparisons of metals in halophytes at two Essex Marsh Sites, Two Tree Island and Salcott, in the mouth of the Thames and Blackwater, respectively, suggest that plants from the former site accumulated more metal, particularly Cd, Hg and Pb, as anticipated from the higher levels of contamination present in the Thames Estuary (Williams *et al*., 1994). However, as concentrations in the plants (*Salicornia spp, Spartina spp, Aster tripolium, Atriplex portulacoides*) did not accurately reflect particulate loadings, it is unlikely they are useful bioindicators of sediment contamination. Metals were predominantly found in the roots (with some transport of mobile elements Cd, Mn and Zn to leaves), though at levels which did not appear to compromise health of these saltmarsh plants (Table 29). Broader investigations, over a range of sites and conditions, may be needed to confirm these conclusions.

Table29. Concentrations of metals in *Spartina spp* in the Blackwater and Thames (from Williams et al., 1994).

	Blackwater (Salcott)		Thames (Two Tree Island)		
	roots	leaves	roots	leaves	Toxic body burden
Cd	0.37	0.06	1.0	0.18	
Cr	3.3	1.1	1.4	0.73	
Cu	13	4.6	11.0	4.8	20[a]
Ni	3.4	1.0	2.1	0.98	20[b]
Pb	7.1	0.95	4.4	2.9	
Zn	102	51	108	68	400[a]
Mn	52	78	22	58	
Fe	5400	966	1490	590	

[a] Lepp (1981); [b] Hutchinson (1981)

There are no recent results for metal residues in higher organisms within the EMS. Results for Hg in muscle of fish caught in the Thames Estuary indicate this remains an important source in the area. The median values for flatfish (dab, flounder, plaice) sampled between 1999 and 2001 were in the range 0.21 μg g^{-1} – 0.23 μg g^{-1} wet weight – higher than OSPAR Background/Reference Concentrations (0.03-0.07 μg g^{-1}), within the JMP 'medium' category for contamination (0.1- 0.3 μg g^{-1} wet weight) but below the directive (93/351/EEC –mercury in fish products) limit of 0.5 μg g^{-1} wet weight for maximum limits in individual fish species (NMMP, 2004). It is also less than the mercury EQS, required in areas receiving significant mercury inputs, 'that the concentration in a representative sample of fish flesh chosen as an indicator should not exceed 0.3μg g^{-1} wet weight', set by the Paris Commission in 1980.

Cd concentrations are elevated in liver of flatfish from the Thames Estuary (median 0.166 μg g^{-1} – above the value of 0.150 μg g^{-1} for other UK sites) (NMMP, 2004), but are likely to be lower at offshore sites (Cefas, 1998). Records for dab from Thames Warp, in 1993, indicated concentrations of 0.12 μg g^{-1} Cd (liver) and 0.06 μg g^{-1} Hg (muscle).

Bioaccumulation of organic contaminants

The most comprehensive data set for bioaccumulation of organic contaminants in the EMS is for tributyltin. TBT concentrations in tissues of shellfish (oysters and mussels) have been measured regularly by Cefas as part of the program to monitor the effectiveness of TBT legislation, and, as with water, indicate significant and rapid reductions in body burdens in the Crouch and Blackwater (Figures 59 and 60). In comparison, responses in the Blackwater were less rapid immediately after the 1987 restrictions (and possibly reversed in 1991) - perhaps as a result of illegal usage (Maff, 1993). More surprisingly, data for mussels from the West Mersea site in 2000 suggest this upward trend may still be continuing (Figure 60), and requires further investigation.

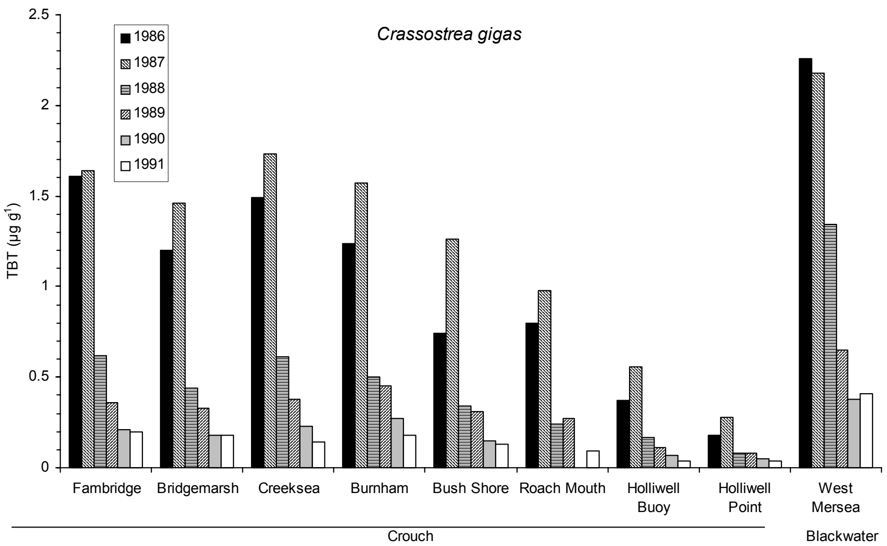

Figure 59. TBT concentrations (μg g^{-1} wet wt as cation) in oysters *Crassostrea gigas*, Crouch and Blackwater Estuaries, following TBT legislation in 1987 (plotted from Cefas data, MAFF 1993).

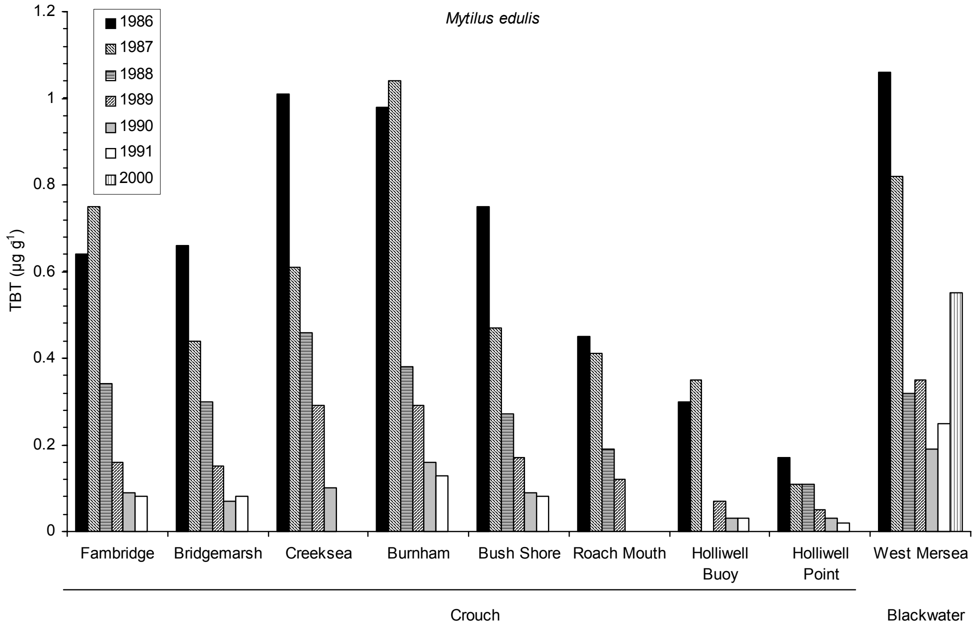

Figure 60. TBT concentrations (µg g^{-1} wet wt as cation) in mussels *Mytilus edulis* (bottom), Crouch and Blackwater Estuaries, following TBT legislation in 1987 (plotted from Cefas data, MAFF 1993, data for W. Mersea mussels in 2000 are from EA database).

Based on OSPAR reference criteria, the summary statistics (2000-2004) for TBT in mussels from the NMMP shellfish site at West Mersea indicate body burdens were substantially above background/reference concentrations. Using NOAA 'mussel watch' criteria, TBT in West Mersea mussels would also be classified as belonging to the 'high category' (top 15% of US samples).

The summary statistics for pesticides in mussels from the NMMP shellfish site at West Mersea (2000-2004) are shown in table 30. The majority of values are below detection limits and, based on OSPAR ecotoxicological assessment criteria (EAC) and NOAA 'mussel watch' categories, there appears to be little risk at this particular site. No recent data could be found for other locations within the EMS. Earlier results from 1990 imply low body burdens (1-4 µg kg^{-1} ww) of γHCH, dieldrin and ppDDE at Creeksea (Crouch) and Southend – marginally above the lower EAC threshold (MAFF 1993). However, summaries of 1995-96 UK shellfish data collected by CEFAS (2001) indicate that samples from the region were sometimes at the upper end of the UK range (dieldrin in mussels and oysters up to 7 µg kg^{-1}ww in the Thames Estuary; pp'DDE up to 6µg kg^{-1}ww in the Roach at Blackledge).

It was not possible to assess spatial or temporal trends rigorously on the basis of available data, but it seems unlikely that residues are increasing, given that most OC pesticides are now banned and many others are subjected to either national legislation or voluntary control. In fact comparisons with 'mussel watch' results from 1978 (Murray 1982) suggest that, at West Mersea, concentrations of dieldrin and DDE are now lower by perhaps four- and two-fold respectively.

Table 30. Pesticides in mussels *Mytilus edulis* from the West Mersea NMMP site in comparison with OSPAR, NOAA and EPA guidelines (summary statistics, 2000-2004:data source EA).

Concentrations in *Mytilus edulis* µg kg^{-1} wet wt

						guideline values			
	n	Median	Maximum	Minimum	% less than	lower EAC*	upper EAC*	†NOAA 'High'	‡EPA
Aldrin	30	0.2	1.4	0.1	100				
Dieldrin	28	0.86	1.43	0.1	36	0.75	7.5	1.36	7.34
Endrin	30	0.1	2	0.1	100				
Isodrin	30	0.1	2.8	0.1	100				
HCHα	30	0.2	1	0.1	100				
HCHβ	30	0.2	1	0.1	100				
HCHδ	30	0.2	1	0.1	100				
HCHγ	30	0.2	1	0.1	100				84
DDT (op)	30	0.1	1	0.1	80			(ΣDDT) 21	(ΣDDT) 314
DDT (pp)	24	0.2	1	0.1	83				
TDE (pp)	30	0.59	1	0.28	20				
DDE (pp)	30	1.74	2.66	1.25	0	0.74	7.4		
HCB	0.2	1	0.1						73
HCBD	0.2	1	0.1						

*OSPAR ecotoxicological assessment criteria

†NOAA 'high' category for mussels, converted to wet weight guidelines, assuming a solids content of 15%

‡ US EPA screening value - EPA # 823-R-95-007

PCB body burden data are also limited spatially and temporally. In 1990 concentrations of 37 µg kg^{-1} ww were found in mussels at Creeksea (Crouch) and Southend (MAFF 1993). This is above lower and higher ecotoxicological assessment criteria (EAC) of 0.75 and 7.5 µg kg^{-1} wet weight, respectively (for ICES7 CBs) set by OSPAR (NMMP 2004) and also exceeds guidelines of 10 µg kg^{-1} ww proposed by EPA (but not the 60 µg g^{-1} ww 'High' category indicated by NOAA).

More recent summary statistics for PCBs in mussels from the NMMP shellfish site at West Mersea (2000-2004) are shown in table 31. The majority of values for congeners 28, 52 and 180 are below detection, whilst congeners 101, 118, 138 and 153 were sometimes accumulated to levels above OSPAR EACs. Concentrations at the West Mersea site appear to have declined since 2001 (illustrated in figure 61 for congener 138) and in the most recent sample (2004) all congeners were below the detection limit of 0.1µg kg^{-1} ww. Risk, if any, appears to be decreasing, but is difficult to assess with any certainty as detection limits are above EAC.

Table 31. PCBs in mussels *Mytilus edulis* from the West Mersea NMMP site (summary statistics 2000-2004:data source EA).

			PCB concentration µg kg^{-1} ww		
	n	%less than	Median	Minimum	Maximum
PCB NO.28	30	100	0.2	0.1	0.2
PCB NO.52	30	100	0.2	0.1	0.2
PCB NO.101	30	20	0.3	0.1	0.54
PCB NO.118	30	20	0.25	0.1	0.48
PCB NO.138	30	20	0.35	0.1	1.06
PCB NO.153	30	20	0.58	0.1	1.06
PCB NO.180	30	93	0.1	0.1	0.11

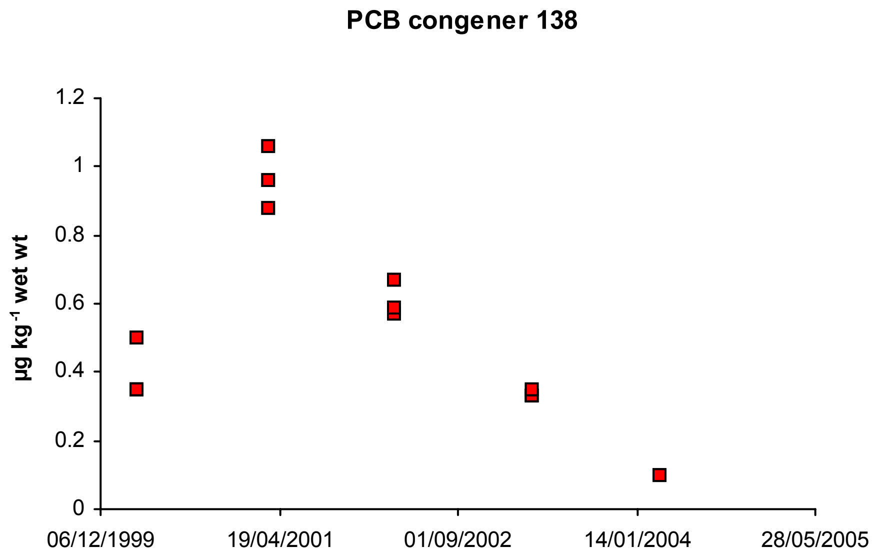

Figure 61. PCB congener 138 in mussels *Mytilus edulis* from the West Mersea shellfish site (data source EA). Trends for congeners 101, 118 and 153 were similar.

CEFAS monitoring of cockles, mussels and oysters from commercial beds around the UK suggests PCBs do not pose a human health risk. Even for high level shellfish consumers, estimated dietary intake of residues, based on 1995-96 data, represent a small (<2%) proportion of the Acceptable Daily Intakes (ADI)/Provisional Tolerable Daily Intakes (PTDI) (CEFAS, 2001). Polychlorinated biphenyls were elevated in the outer Thames mussels and oysters, though even these body burdens (<0.1 mg kg^{-1} wet weight) were more than an order of magnitude below guidelines set by Norway/Sweden (at 2 mg kg^{-1} wet weight).

Nevertheless, surveys of PCBs in flatfish (dab, flounder and plaice) confirm that the Thames Estuary is still one of the most contaminated areas monitored in the UK (along with Liverpool Bay and the Mersey Estuary). Between 1999 and 2001 the median concentration in Thames fish was 4100 µg kg^{-1}, normalised to lipid weight. The lowest value of the 78 UK sites sampled was 219 µg kg^{-1} from the Minches (NMMP, 2004).

Offshore, at Thames Warp, records for dab in 1993 indicate concentrations of 190 µg kg^{-1} ww (for ICES7 CBs in liver) which is above the OSPAR EAC range of 1-10 µg kg^{-1} ww for fish tissue (Cefas, 1998). Pesticides detected in this same sample included dieldrin (49 µg kg^{-1} ww; EAC range 5-50 µg kg^{-1}), pp DDE (45µg kg^{-1} ww; EAC range 5-50 µg kg^{-1}) and ΣDDT (63 µg kg^{-1} ww). These clearly suggest the presence of such compounds in elevated concentrations. As in other examples, however, EAC values could be over-protective and do not necessarily imply effects, nor do they seem likely to result in problems for human consumption.

We are not aware of any recent data on PCBs or pesticides in fish from within the Essex Estuaries.

The lack of thorough, recent information on bioaccumulation across the site appears to be a substantial knowledge gap. Extended sampling would help to establish trends and risks in the EMS. In terms of the Essex Estuaries EMS, monitoring of detritivores and other infaunal types should be incorporated as a priority in long-term surveillance programmes. This is essential to address the issue of bioavailability of sediment contaminants. Soft-substrate habitats occupy most of EMS and are becoming increasingly important, relatively, as diffuse sources. Many infaunal species are also important prey items for predators including the birds for which the site is designated.

The issue of food chain transfer (and sublethal effects) of key contaminants such as Ag, Hg, PCBs and other persistent chemicals in higher predators is difficult to assess because of the extremely small number of samples. This topic might be suitable for a national R&D project, rather than as purely regional objective.

7.2. Biological effects studies

To date, there have been few investigations of the effects of water quality on biological and ecological quality which focus specifically on the Essex Estuaries EMS. A number of studies conducted by Cefas on biological effects indicators (biomarkers, bioassays and ecological impacts) include sites in the Crouch, or in the Thames Estuary (including offshore stations) as part of UK-wide comparisons. These form the basis of the synthesis given below

7.2.1 Biomarkers

Metallothionein

The physiological significance of metal burdens, can be examined specifically by determination of metallothionein (MT) induction and intra-cellular metal-binding patterns. The MT assay may act as an 'early-warning' indicator of metal exposure. There is also scope to deploy the assay to show temporal trends; i.e. to demonstrate improvements arising from planned schemes to reduce metals, standstill provisions of the Dangerous Substances Directive, or as may be required under the Habitats Directive to achieve Favourable Condition.

So far however only preliminary studies (in 2001) have been undertaken, measuring the protein in dab livers at 16 sites around the UK. The closest site to the EMS was the outer Gabbard. Metallothionein levels in these fish did not appear to be elevated and variability at each site was relatively high. Concentrations were most consistently correlated with Cu in both sexes and to a lesser extent with Pb (males) and Zn, Hg and Cd (females) (NMMP, 2004). Metallothionein studies within the EMS would be useful to indicate whether organisms are attempting to respond, or are adapted, to metal stress.

EROD

PAHs, PCBs (planar), dibenzo-b-dioxins and dibenzofurans can, when absorbed by benthic flatfish, induce the synthesis of the cytochrome p450 group of enzymes, typified by CYP1A1-dependent EROD activity. The close association of these fish with sediments (which tend to scavenge hydrophobic contaminants) may make them a particularly useful biomarker of this type of contamination.

Surveys of EROD levels in dab *Limanda limanda* conducted at NMMP sites in 2001 and 2002 indicated that fish from the North Sea tend to have rather low levels compared with contaminated estuaries and coastal areas (CEFAS 2003a;2005a). Dab at the outer Gabbard displayed mean activities of 122 pM/min/mg protein which were five times lower than fish caught at the Burbo Bight and Liverpool Bay suggesting comparatively low –level exposure to organic contaminants off the Essex coast.

Analysis of major PAH bile metabolites (e.g. 1-OH Pyrene) a possible surrogate measure of exposure tended to confirm this. Samples from the outer Gabbard contained 280 $\mu g\ l^{-1}$ compared with 402 $\mu g\ l^{-1}$ at the Burbo Bight near the mouth of the Mersey Estuary (CEFAS, 2005a). The outer Gabbard is of course a considerable distance offshore in the southern North Sea: PAH sediment concentrations increase from the outer Gabbard (0.8 $\mu g\ kg^{-1}$ in 2003) sharply towards the mouth of the Thames (123 $\mu g\ kg^{-1}$). EROD samples have not been measured at estuarine sites within the Essex estuaries EMS, though this would be a useful addition towards future assessment of site condition.

Fish Pathology And Disease Biomarkers.

CEFAS have established prevalence of external disease and pathological changes in liver of dab populations around UK coasts in recent years, as part of the NMMP programme (CEFAS, 2005a). Disease symptoms observed included lymphosystis, epidermal ulceration, epidermal papilloma, hyperpigmentation, macroscopic liver lesions and number of parasites. At the outer Gabbard, the incidence and severity of disease and histopathological lesions appear relatively low compared with the Dogger Bank region of the North Sea, which has had a high incidence in recent years (CEFAS, 2001, 2005a) Apart from occasional observations from the outer Thames estuary however, little is known of areas close to and within the Essex Estuaries EMS. Further attempts to establish disease status here would be useful in future assessments (CEFAS, 2005a).

7.2.2 Bioassays

An oyster larval development assay has been used in the assessment of water quality around the UK coastline by NMMP and, in 1999-2001, incorporated sites in the Thames Estuary (on a number of occasions). The nearest site to the Essex Estuaries EMS, however, was at Mucking, where short-term tests indicated significant toxicity (>10%) compared with controls (artificial sea water) particularly in 1999 and 2000 (NMMP, 2004).

Earlier oyster larval assays of water quality at offshore sites identified poor quality at the Thames Warp site off Maplin in 1994 (47%); however, this was not seen in earlier years. It is not known if water quality within the EMS is influenced to the same extent. At the Gabbard site, responses were considered as near background (Cefas 1998).

Toxicity in sediment samples has been examined at a number of sites around the UK using whole animal bioassays with infaunal species such as *Arenicola marina* and *Corophium volutator*. In addition to lethal endpoints in these short-term tests, sublethal indices included cast production in *Arenicola*, a sign of reduced feeding activity, (CEFAS, 2001). Mortalities of *Arenicola* and *Corophium* held in sand from Shoebury in the 10 day assays were consistently low (7-13% and 0-7%, respectively) and these have been used by CEFAS as 'reference tests' with which to compare sediment quality elsewhere (acute mortality varied from 0-100% over a suite of 25 sediment sites). The cast production by *Arenicola* in Shoebury sediment varied from 3.1-3.6 casts d^{-1} compared to a range varying from 0- 4.1 casts d^{-1} at other sites. On this basis the sands at Shoebury (and presumably in adjacent parts of the Foulness SPA) appear to be low risk.

At offshore sites, earlier sediment tests with the same assays gave rather equivocal results: thus, at the Thames Warp site, responses were consistently near background but at the Gabbard site up to 82% suppression of feeding was observed in *Arenicola* in one year (1994 only). Sediment elutriate tests with oyster larva also suggested toxicity in Gabbard sediments at that time (but not Thames Warp). Proximity to the South falls disposal site was put forward as a possible explanation (Cefas, 1998).

It may be useful to consider similar tests over a wider range of sites within the EMS as an indication of the effects of sediment condition.

An in vitro bio-analytical assay, DR-CALUX (a coupled reporter-reporter gene system responsive to compounds which interact with the aryl hydrocarbon receptor), has been used to assess the presence of dioxins and dioxin-like compounds, including furans, in sediments from the Thames Estuary (CEFAS, 2005a). Expressed as toxic equivalents (TEQ) to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD, the most active dioxin), responses in Thames samples varied from 1 ng TEQ kg^{-1} to 2.3 ng TEQ kg^{-1} dry weight. These are much lower than the highest value found (106 ng TEQ kg^{-1} in the Forth) and close to guideline thresholds recommended in Canada, the USA and Holland (0.85, 2.5 and 13 ng kg^{-1}, respectively). Adverse effects arising from inputs within the Thames (a major source of contaminants to the southern N.Sea) seem unlikely, though there are no data for the Essex EMS itself.

7.2.3 Ecological responses

TBT –Impact And Recovery In The Crouch Estuary

CEFAS long-term monitoring of TBT in the Crouch Estuary (reviewed in section 6) has been conducted in parallel with benthic surveys to assess faunistic responses.

Improved TBT water quality in the Crouch has coincided with improvements in biological quality, both in infaunal and epifaunal communities (sampled from 1987 to 1997). Among the most significant changes were a marked and sustained increase in numbers of some sedentary **epifaunal taxa** at inner estuary sites, including a range of ascidians (nb *Ascidiella aspersa*, *Ascidia conchilega, Ciona intestinalis*) and a large population of the native oyster *Ostrea edulis* (Rees *et al.,* 2001). The most obvious of these changes, in trawl samples, occurred prior to 1992, i.e. in the immediate aftermath of the ban. Slower change has taken place since then.

Other epifaunal taxa demonstrating net increases in density (up to 1997) included Polynoidae (scale worms), spider crabs *Macropodia rostra*, *Gibbula* spp and *Buccinum undatum*. Distributions of slipper limpets, shore crab, common starfish and sand goby have changed little in recent years whilst a number of species (particularly migrating shrimps and their predators) have declined in numbers, though this is attributed more to factors in the open sea than water quality in the estuary (Rees *et al*, 2001).

Thus, it seems that, at the community level, some indication of recovery may occur over timescales of ~5 years. At the population level, improvements are indicated by re-colonization of TBT-sensitive ascidians and oysters. Nevertheless, some continuing low-level impacts from previous TBT inputs may still be occurring in the region and may take much longer to eradicate. Species which are extremely sensitive to TBT such as *Hinia reticulata* and *Nucella lapillus* - reported to be present in surveys of the Crouch prior to the use of TBT-based antifouling paint - were still absent from samples taken more than a decade after legislation (Rees *et al* 2001).

Another casualty of TBT has been the American oyster drill *Urosalpinx cinerea* which was probably introduced to the UK in consignments of *Crassostrea virginica* at the turn of the 19^{th}/ 20^{th} centuries and thrived on the Essex oyster beds (Colne, Blackwater, Crouch and Roach) up until the 1970s. Observations between 1987-1990 in Goldhanger Creek in the Blackwater Estuary indicated that populations of *Urosalpinx* had become scarce, with little recruitment of young individuals (Gibbs *et al*., 1991). Masculinisation symptoms in females were similar to those observed in TBT-effected *Nucella*, and included penis and vas deferens development, oviduct and egg-capsule malformation, and sterility. Little viable spawn was observed during this period. It remains to be determined whether recovery has occurred since then, and whether other populations along the Essex coast have been similarly effected. Though this species is far from being considered an interest feature (and has inflicted considerable destruction of large numbers of native oysters) it nevertheless represents a useful surrogate measure of biological impact for other sensitive species. Given the limited dispersion of *Urosalpinx* (no free-swimming larval stage), recolonisation might be expected to be slow.

A marked increase in the biodiversity of sub-littoral **infaunal benthic communities** of the Crouch, between 1987 and 1992, was also linked to a decline in TBT loadings at the most contaminated upstream sites, with molluscs and amphipods displaying the most significant increase in abundance and diversity (Waldock *et al* 1999). Typical sediment-TBT concentrations at the head of the estuary declined from 0.16 µg g^{-1} dw to 0.04 µg g^{-1} over this five year period (coinciding with an increase in species numbers from 8 to 37 at Stow Creek). TBT reductions in coarser sediments at the mouth were less obvious (0.02 to <0.01 μg g^{-1} DW) with species number remaining fairly constant. Diversity of polychaetes has increased subsequently at upstream sites since 1987 (Stow Creek was then dominated principally by *Aphelochaeta marioni* and the oligochaete *Tubificoides benedii*) but has remained relatively stable at the mouth - a similar trend to that observed in molluscs and crustaceans. The most successful recruitment noted for molluscs was in *Mysella bidentata, Macoma balthica, Abra alba* and *Thracia* spp, together with smaller numbers of *Cerastoderma edule* and *Nucula turgida.* Other 'recovering' taxa of note were ascidians, anemones and pycnogonids.

Overall, therefore, the change has been from a gradient of low diversity upstream and high diversity at the mouth in 1987, to a more even pattern along the estuary in recent samples. Although the evidence is circumstantial to an extent, rather than unambiguous proof of cause and effect, these changes are almost certainly the result of declining TBT levels. During the same period reductions in the shell thickening response of oyster, and improvements in *Littorina* populations, were further indications of the success of legislation (Waite *et al* 1991; Waldock *et al* 1999; Matthiessen *et al*., 1995). The numbers of eggs and veliger larvae of *Littorina littorea* found in plankton surveys of the Crouch, together with frequency of 0-group individuals on intertidal flats at Creeksea increased markedly in the years following the ban, coinciding with a simultaneous decrease in TBT residues in tissues (from 0.17μg g^{-1} ww in May 1988 to ~0.05 μg g^{-1} in July 1991). TBT-induced impacts on egg production and development have been demonstrated in the laboratory at environmentally realistic TBT concentrations, illustrating the likely cause-effect nature of this relationship (Matthiessen *et al*., 1995)

Evidence for biological improvements in other Essex estuaries arising from TBT restrictions includes the re-opening of the oyster fishery in the Blackwater Estuary: oysters had reached near-marketable size by 1988, following the closure in the 1970s when TBT contamination was beginning to rise (Waite *et al* 1991).

Endocrine disruption (oestrogenic responses)

The masculinising influences of TBT are discussed above. In recent years, there have also been concerns that chemicals that mimic female hormones (including both natural and xenobiotic substances) may be causing feminisation in some organisms, thus influencing the condition of aquatic biota and habitats. Initial observations of endocrine disruption in the aquatic environment were mainly in freshwater fish populations, and suggested that oestrogenic effects decreased rapidly within relatively short distances from sources (usually STW). It was therefore anticipated that dilution in estuaries and the open sea would render the threat harmless in these environments. Since then, the discovery that some benthic estuarine fish, notably flounder (which have close contact with sediment and feed on benthic infauna), also exhibit symptoms of impact from oestrogens, has led to the recognition that adsorption to particulates may be responsible for impeding dispersal and dilution, and that sediments could be acting as a secondary source (Allen *et al*., 1999a,b).

Habitat and feeding preferences make species such as flounder vulnerable to the type of sediment-associated pollution which occurs in estuaries. Thus, bioaccumulation and biomagnification of endocrine disrupting compounds, via infaunal organisms, could contribute to the oestrogenic responses (raised vitellogenin - VTG - and ovotestis) seen in flatfish (Lye *et al.*, 1999; Matthiessen *et al.*, 2002). Causes of these responses are still unclear: no direct link has been established between volumes of domestic sewage and ED, though some link with trade wastes may be discerned (Matthiessen *et al*. 1998). As there are likely to be complex mixtures of a number of endocrine disrupting chemicals in the environment - including natural and synthetic hormones (e.g. E2 and EE2), alkylphenols, PCBs and OC pesticides, phthalates and other xenoestrogens - the causative agents may well be acting in combination, and the relative contributions of each remains uncertain (Matthiessen *et al*. 1998; Lye *et al*., 1999).

Flounder from the Crouch Estuary (Fambridge and Roach) were sampled in 1996, as part of a broader survey of UK populations, focusing on VTG induction and the incidence of ovotestis in males (Matthiessen *et al*., 1998). Compared with the heavily industrialized and urbanized Tees and Mersey Estuaries - where plasma VTG concentrations were highest ($>10^7$ ng ml^{-1}) - the Crouch population displayed low levels of male vitellogenesis; the Roach sample was, however, statistically higher than the reference estuary, the Alde (16-25 ng ml^{-1}). The ovotestis condition, where oocytes develop in the testis of male fish, was absent in the Crouch (compared with a maximum incidence of 17% in the Mersey), apparently signifying little cause for concern in terms of reproductive output. Arguably, however, the presence of vitellogenesis in male fish, albeit at low levels, implies that the Crouch is not in optimum condition (Allen *et al*., 2000). There are no data on endocrine disruption in other sections of the EMS, or other species, including rare fish populations and invertebrates; this is seen as a gap which justifies further investigation. Flounder breed offshore and may be less susceptible to ED than organisms breeding in estuaries. Based on these arguments, the Essex Estuaries EMS has been identified as a medium priority site for further research (Allen *et al*., 2000).

Flounder have been sampled from the Thames Estuary at Nore Sand (off Southend, but outside the EMS) and further upstream at Mucking and Galleons Reach (Beckton). Although VTG levels were slightly elevated, only in the latter sample was VTG in plasma (14×10^3 ng ml^{-1}) significantly higher than the Alde reference population (Matthiessen *et al*., 1998).

Ecological indices and general site condition

Extensive intertidal habitats (mudflats and sandflats), together with their infauna, are important ecological components of the European Marine Site, providing feeding areas to sustain the qualifying bird populations. The Essex saltmarshes are also important roosting and feeding areas for waterfowl. Unfortunately biotic indices have seldom been scored within the EMS: most information is for NMMP sites and adjacent offshore areas.

Benthic Macrofauna

Quantitative temporal comparisons of benthic species abundance at offshore sites around the UK, including the outer Thames Estuary, have indicated that physical characteristics (particle size, salinity, depth), rather than contamination, probably dominate community distributions. Off the mouth of the Thames the sediment is dominated by medium sands, with an associated fauna that is characteristically fairly sparse (Cefas 1998).

Benthic community analysis was undertaken at Thames Gabbard, in 2000, as part of the NMMP programme (table 32), and attempts were made to link indices such as diversity, abundance, feeding groups (infaunal trophic index – ITI) and AZTI marine biotic index (AMBI score- classification according to pollution sensitivity), with environmental parameters such as sediment contamination and type and water depth (NMMP, 2004). The ITI index – 63% - was just above the borderline between normal (> 60%) and 'changed' (30-60%). AMBI scores (2.45) suggested a low-moderate degree of pollution stress (0 = unstressed; 7 = highly polluted). Multivariate analysis of the biotic data set indicated that this site hosted a relatively diverse community, with a fairly heterogeneous range of substrates.

Inside the mouth of the Thames Estuary, at Mucking, the lower ITI index (30.6%) implied a degree of alteration, and the AMBI index indicated a higher pollution stress (3.37 – see table 32). However, physical characteristics of sediment (%silt/clay), salinity and water depth, were likely to be most influential in determining the differences in macrofaunal community patterns observed at these sites. Biological and substrate diversity at Mucking were considered moderate for the type of estuarine habitat. Further upstream, at Woolwich, the benthic community was more impoverished (dominated by two species of oligochaete). It is possible that the trend towards poorer water quality in the upper Thames Estuary were sufficient to contribute to less favourable ecological condition (NMMP, 2004). Unfortunately there are no comparable data for sites within the EMS.

Table 32. Benthic community scores at coastal and estuarine NMMP sites (data source, NMMP, 2004)

Diversity and other biotic indices

	species	abundance	Diversity(H')	ITI*	AMBI**
		Coastal sites			
Gabbard	21	114	1.88	63	2.45
UK median (n=31)	20	102	2.02	61.1	1.67
		Estuarine sites			
Mucking	14	7322	1.48	30.59	3.37
Woolwich	13	1442	1.03	1.31	1.33
UK median (n=28)	17	5671	1.44	40.81	3.29

*ITI- Infaunal trophic index (Codling and Ashley, 1992)
**AMBI – AZTI marine biotic index (Borja *et al*, 2000)

Impacts on biota arising from the disposal of sewage and dredge spoils off the Essex coast, have been assessed by CEFAS, though these have focused on the immediate disposal area rather than the EMS. Monitoring of benthos at the former sewage disposal site at Barrow Deep, in 1992, suggested that the main driver of community composition and distributions was the physical nature of the sediment, with cluster analysis revealing two benthic groupings (Cefas, 1997). The well sorted fine sands/muddy sands of the main channel systems tended to support impoverished communities typified by *Bathyporeia elegans, Nephtys spp., Abra alba* and *Scoloplos armiger.* Enriched fauna was found at the disposal ground itself and in adjacent areas of coarser, but more stable, sediments which typically supported serpulid worms, anemones and some taxa, such as cirratulid polychaetes, which tend to thrive on a degree of organic enrichment. Thus, this latter grouping may have been modified on the basis of a subsidiary influence of sewage sludge, although there was no evidence for elimination of susceptible suspension feeders or the introduction of typical pollution indicators such as *Capitella*. There was an indication of some metal enrichment in sediments along the Barrow and Middle Deeps but concentrations are not thought to be high enough to have influenced biota at the site. It would seem unlikely that the sewage dumping operations (now ceased) would have had any deleterious impact on the EMS.

A major marine disposal operation in recent years was the dumping of 27 million tonnes of dredgings at the Roughs Tower Ground, some 20 m NE of the Thames Estuary, arising from the expansion of the port of Harwich, in 1999. This project carried the risk of exceeding the dispersive capacity of the site. Subsequent monitoring has confirmed that benthic fauna were reduced at the grounds, relative to adjacent control areas, and though some recovery commenced within a few months, a return to equilibrium is likely to take longer (CEFAS, 2003b). Likewise, catches in some crustacean fisheries were reduced, relative to adjacent reference sites, but were anticipated to achieve parity in the near future. Fortunately, there were no indications of increased contaminant concentrations in sediments and it seems that any impact of sludge disposal is likely to be localized, largely due to physical smothering, and is not anticipated to impact significantly on adjacent Essex coastline

Meiofauna

Meiofauna play an important role in ecosystem function. Assessing their diversity and abundance may be a useful addition to more conventional biological screening based on benthic macrofauna.

Nematode and harpacticoid copepod assemblages have been assessed in relation to environmental and sedimentary variables at 12 NMMP sites around the coastline, including ‘off the Thames’ in the region of the Essex Estuaries EMS (Cefas, 2003c; NMMP, 2004). The assemblages from different locations around the UK were found to co-vary with physical attributes of the sediment, and with metal concentrations (though the latter may be coincidental and is not unambiguous evidence of cause and effect). Nevertheless, ‘Off the Thames’ assemblages were typical of low metal sediments, and differed from those of more polluted sites such as the Tyne and Mersey. This supports the notion that the ecological characteristics of coastal sections of the EMS are unlikely to be at high risk from general water quality.

Saltmarshes

Saltmarshes are important wildlife habitats, suppliers of nutrients for fisheries, and natural barriers against tidal incursion, notably so along the Essex coastline where they provide a substantial contribution to coastal stability. This function is under threat from a variety of natural causes related to sea level rise and it is thought that some Essex marshes have been suffering extensive erosion since the 1930s (Allen and Pye, 1992). The Dengie peninsula, for example, suffered a 16% loss in saltmarsh area between 1970 and 1985 (Harmsworth and Long, 1986). This may be exacerbated to an extent by human activities such as land reclamation and there are suggestions that contaminants could play a role in the process, either by stimulating or perpetuating the loss of saltmarsh plants. Detailed descriptions of five Essex marsh systems and the impacts (in 1991) of local contaminant inputs on their 'health' (defined by general appearance - whether actively accreting with dense vegetation, or exhibiting degradation) are given in Fletcher *et al*. (1994a, b). Three of the five study sites fall within the EMS; South Woodham Ferrers (Crouch, mixture of agricultural and urban hinterland), Tollesbury Wall and Salcott (both Blackwater, mainly agricultural influence, but also some STW contribution). The general classification of these saltmarsh systems is indicated in table 33.

Table 33. Vegetation type and superficial health 'designation' of three Essex salt marsh systems Metal enrichments in sediments refer to comparisons with 'standard shale'(Fletcher et al 1994a,b)

	vegetation	Designation	Metal enrichment
South Woodham Ferrers	Limited species diversity. Main species *Atriplex* and *Spartina*, minor species *Salicornia* and *Aster tripolium* (plus filamentous green algae and *Porphyra*)	'unhealthy'	As (1.6) Pb (1.7) Hg (1.5) Zn (1.1)
Tollesbury Wall	Lower levels dominated by *Salicornia* with *Atriplex* and abundant *Aster* at higher levels. Other species *Spartina anglica, Festuca, Puccinellia, Limonium, Sarcocornia*	Both healthy and unhealthy areas	Fe (1.6) Pb (1.5)
Salcott	Isolated patches *Spartina* near sea wall, more diverse elsewhere with *Festuca, Suaeda, Armeria, Spergularia, Inula*, but dominated by *Atriplex*	'unhealthy' near sea wall due to bare mud mounds	Fe (1.2) Pb (1.1) Se (6.1) Zn (1)

It is difficult to attribute health of the marsh to the presence or absence of specific contaminants because of the modifying effect of a complex array of environmental factors: this includes tidal height and strength - which in turn influences the degree of inundation, waterlogging, redox, oxygen diffusion, pH, grain size, and the balance between accretion rate and contaminant deposition. A number of these factors are also closely linked with the diagenetic processes which can control contaminant behaviour.

Other natural variables which confound simple prediction of the links between contaminants and marsh condition are the presence of mussel and oyster beds, algal mats, bacterial exudates and other sediment stabilisers.

One consistency may be the progression towards retention of finer, less scoured, organic-rich (and contaminant rich) sediment as the marsh develops. Vegetated marsh areas might therefore be expected to trap more contaminants than adjacent well-flushed intertidal flats and channels, though the consequences of this mechanism for the Essex sites still requires elucidation. In attempting to do so, and to investigate the links between contaminants and marsh 'health' and stability, Fletcher *et al* (1994b) determined a range of metals and metalloids (As, Ca, Cd, Cu, Cr, Fe, Hg, Mg, Mn, Ni, Pb, Se and Zn) at the above Essex sites. Results confirmed that pollutant type metals (as opposed to geogenic elements) were higher in vegetated saltmarsh regions, relative to mudflat, possibly reflecting differences in granulometry (see also O'Reilly Weise *et al* 1995). However, based on comparisons with a standardised shale (and a more contaminated marsh site near Southend), seldom was enrichment greater than two-fold (Table 33). The validity of such comparisons might be questioned, as the shale is not a 'local background'; nevertheless, results tend to confirm the conclusions drawn for metals in inter-tidal muds in the previous section (6), that Essex estuary sediments are generally only mildly contaminated with metals, probably from a range of anthropogenic sources.

Complementary studies at the same saltmarsh sites (and Dengie) in the early 1990s also concluded that contamination in these Essex systems was relatively low-level (compared with the Thames) and widespread, with distributions reflecting a variety of processes and inputs, both contemporary and historical (Leggett *et al.*, 1995). For Dengie sediments the main elements with an enriched component were again the anthropogenic metals (O'Reilly Weise *et al* 1995, Leggett *et al.*, 1995). Even then, the degree of enrichment in Dengie mudflats, relative to average shale, was generally small (As 1.4, Hg 1.8, Pb 1.5, Se 7; other metals <1) and could possibly be explained by granulometry, as much as inputs (see also 'pesticides', section 6, for examples of organic micropollutant enrichment).

Thus, although it has been suggested that contaminants may be a contributory factor in the decline of Essex marshes (NRA 1993a; Leggett *et al.*, 1995), the evidence for this is at best circumstantial and it seems likely that physical processes will be dominant except perhaps close to effluent streams.

Impacts of Managed Retreat

Because of the subsidence of the southern North Sea Basin, of all UK regions, the threat of erosion and flooding induced by sea-level rise is greatest along coastal margins of SE England (up to 2 mm y^{-1}), encompassing the Essex Estuaries EMS. Unless accretion rates match this, the marsh cannot survive[32]. In the Blackwater Estuary in particular, there is a need to recreate accreting saltmarsh due to the inability of some existing erosion-threatened saltmarshes to retreat landward because of sea

[32] On the Dengie peninsula accretion rates are of the order of 11-22 mm y^{-1} (Reed 1988). At other actively accreting sites in Essex this may range from 3.5-25mm y-1 (NRA, 1993b) depending upon topography and local hydrodynamic regime

walls. As a result, some 23% out of a total of 880 ha of marsh has been lost through erosion since 1973. The scale and rate of loss has led to a number of pilot investigations into the practice of managed retreat (MR).

Whilst successful colonisation and development of salt marsh plants (following anthropogenically-induced tidal incursion) is largely dependent on physical conditions, sediment movement, and topography, there could be consequences, from MR, for contaminant behaviour, with possible repercussions for biota. Studies of geochemical changes at Orplands Farm near the mouth of the Blackwater Estuary have indicated that, in addition to the expected rise in Na, Mg and Ca, concentrations of particle-reactive metals such as Pb, Cr and Cu may increase in newly formed salt-marsh sediments following tidal inundation. In contrast, Fe, Mn and Cd could be lost as a result of pH and redox changes, biogenic decomposition, and changing ionic strength (Macleod *et al*., 1999). After breaching, soils tend to be become covered with an accreting layer of estuarine-derived sediment, below which there may be some variability in metal profiles, dependent on previous land-use patterns: partitioning of several metals can shift from a predominantly refractory phase to more labile (and, potentially, bioavailable) forms during the development of anoxic horizons, as may occur when grass and other vegetation decays after flooding (Emmerson *et al*., 2000). The issue of increased metal bioavailability at MR sites does not appear to have been addressed.

A similar study at Tollesbury MR site, on the northern shore of the Blackwater Estuary, investigated the relationship between sediment transport, metal mobility and their impacts on saltmarsh development (Chang *et al*., 2001). Following breaching of the sea wall in 1995, topography and tidal flow dominated the distributions of accreting sediment particles at the site. After two years, the development of *Salicornia* spp, the dominant saltmarsh plant, was observed in accreted finer deposits at the MHWN level. Metal behaviour was broadly similar to that in the Orplands study, above, with small increases in Cu and Cr deposition in sediments attributed to transport into the site from the estuary (along with Na, Ca, and Mg), and a decrease in Cd - attributed to post depositional remobilisation following the saline incursion. The distribution of metals was again shown to be influenced by granulometry, with highest concentrations determined by the fine-sediment transport pattern.

The high affinity of some contaminants with sediments that are accreting (through the implementation of MR) has been perceived favourably by some - in terms of providing a sink for pollutants scavenged from overlying water (notably those contaminants with high sediment-water partition coefficients). Whilst metal loadings in MR sediments may not appear exceptionally high (acute toxicity not expected), the consequences of altered geochemistry on colonising saltmarsh plants (e.g. resulting from the creation of an anoxic zone), and for bioaccumulation in dietary organisms and predators (including designated birds), are not known and should be investigated. Particular caution may be warranted where MR sites which promote accretion are located near to localized sources of contaminants. Erosional aspects may also be need to be considered if there is a likelihood that MR could expose deeper, more-contaminated, sediment layers. On the evidence presented here however, it is unlikely that either metals or organic contaminants are present at acutely toxic levels in sediments, should they be remobilised.

Implications of MR for nutrient cycling are discussed elsewhere in this report.

Another scheme under investigation to reverse erosional losses is that of recharge - which attempts to shore up eroded areas by deposition of sediments from elsewhere. At Westwick Marina, in the Crouch Estuary, this has allowed experimentation on recolonization. In initial trials it was found that, provided sediments are not deposited in too thick a layer (>15cm will hinder vertical migration), and are not excessively rich in 0_2–depleting organic matter, recolonization can be initiated very quickly, though the timescales of complete recovery have yet to be determined (Cefas, 2003c)

Birds

A synopsis of BTO trends and alerts in designated bird populations for SPAs within the Essex Estuaries EMS can be found in section 5 of this report.

8. MODELS

Evans *et al.,*(1993) developed a Combined Distribution model (consisting of a suite of programs) for simulating the long-term behaviour of the water quality of estuaries and applied it to the Crouch and Roach Estuary. The underlying reason for the (then) NRA commissioning the work was to acquire a method that would assist in devising sampling programmes to determine compliance with EQS's. Thus, the model was designed to incorporate discharges from STWs, storm overflows, agriculture and industry and predict the impact that they will have on receiving waters. Output forecast data were defined in statistical terms, such as percentiles, over periods of time (e.g. a year) to enable comparison with relevant EQS criteria.

Combined Distribution Estuary Modelling (CDEM) is broken down into four key stages:

- The construction of a mathematical model of an estuary
- A focus of effort on a description of the long-term behaviour of the inputs to an estuary
- Driving the model with synthetic inputs
- Analysing the synthetic outputs of the model

The model was validated with previously calibrated and validated models of the Crouch and Roach (Clark, 1990, 1991 and 1992 – unreferenced).

The purpose of the simulation runs for the Crouch and Roach was principally to examine the effects of different loading regimes. These were found to be unpronounced except at the tidal limits. The authors postulated that this was because of relatively small tidal flow rates. Results included the identification of a stretch of high mean DO concentrations at stations near the tidal limit in the Crouch, which existed despite STW loads from the inner Crouch. The explanation for this was that high levels of chlorophyll-a were also present at the same stations (and presumably resulted in super-saturation of DO in the water column due to daytime photosynthetic activity).

Results from the studies of sensitivity showed, not surprisingly, that synchronisation of STW effluent discharge with the ebb tide resulted in generally lower BOD levels. There was a general shortage of available data for this exercise, such that some 'template files' or estimated data were used in some places and the study was not exhaustive. However the authors considered that the results were promising and that despite drawbacks, the method was feasible with the amount of data typically available for UK estuaries.

It is not known whether CDEM has been widely used, although in a report on the use of water quality predictive models in coastal and estuarine areas, Bastreri, (2004) does not indicate that CDEM is a favoured modelling package within the EA. However, several of the component programs developed and used by Evans, *et al.,* (1993) as part of CDEM are listed as currently in use by various EA regions.

In an effort to improve technologies for assessing the costs and benefits of managed re-alignment of coastal defences, Shepherd *et al.,*(2005) built a coupled hydrodynamic and biogeochemical model of the Blackwater Estuary and used it with plausible scenarios to estimate changes in carbon and nutrient fluxes. The project involved several steps:

- Developing hydrodynamic (PRISM) and biogeochemical (COSE) models and merging them
- Developing an algorithm for denitrification within the models
- Initialising and validating the models for the Blackwater estuary
- Developing one demonstration managed realignment strategy for the Blackwater
- Estimating total nitrogen fluxes in the Blackwater, and changes in the fluxes associated with denitrification from managed realignment strategy
- Off-lining of the modelled estimate, changes in carbon, nitrogen and sediment burial in the estuary associated with managed realignment
- Comparing sediment burial changes to offshore supply
- Conducting a cost benefit analysis of managed realignment in the estuary for the selected scenario using existing information from other projects
- Incorporating effects of changes in C and N fluxes from managed realignment into cost benefit analysis.

The Blackwater/Colne estuary was chosen for this study primarily because of the hydrodynamic, water quality, and biogeochemical data available, as well as the existence of highly monitored experimental realignment schemes. This enabled the authors to initialise and validate numerical models. Shepherd *et al*., (2005) noted that the loss to reclamation, and hence the potential for realignment in the Blackwater, was less than some other estuaries (e.g. the Crouch), and considered, therefore, that any estimates of potential increased nutrient removal will have been conservative.

Results from modelling demonstrated clearly that managed realignment options in the Blackwater/Colne estuary would enhance C, N and P burial in the newly accreted sediments. Increased microbial metabolism of dissolved nitrate, attenuating flux to the southern North Sea was also a modelled outcome.

Environmental benefits associated with managed realignment included improved water quality in the estuary, as well as landscape, amenity, recreation and habitat augmentation (however, possible adverse environmental effects of N_2O production [33] were not taken into account in this exercise). Cost-benefit analysis (CBA) demonstrated that managed realignment would provide positive economic benefits, regardless of whether the sedimentation rate was assumed to be low (1.5 $mmyr^{-1}$) or high (6 $mmyr^{-1}$). The different scenarios considered over a range of time-spans produced a range of benefits. Notable points include:

- The longer the time-span the greater the benefits, with all scenarios producing positive benefits over the 100 year period. These were deemed economically efficient.

[33] See section 6.2

- The 'hold the line' (HTL) or 'do nothing' scenario represented an overall cost to society

Overall, the project was deemed to be successful and was considered by the authors to make a valuable contribution to research into the sustainable management of coastal ecosystems in general.

Other Water Quality Modelling. There have been a number of attempts to model key parameters for antifouling compounds; notably that by Harris and Cleary (1987) for TBT in the Crouch, and generic information on the partitioning of booster biocides by Thomas *et al*. (2004).

9. CONCLUDING REMARKS AND RECOMMENDATIONS

Biological Status

Quantitative diversity indices are frequently used to assess the effects of environmental degradation on the biodiversity of natural assemblages of benthic organisms. Several indices, which help to put biological status into perspective, have been applied to a number of other cSAC/SPAs, but not to the Essex Estuaries. To help in comparisons between sites, and in long-term assessment of condition, it may be useful to apply a selection of these quantitative techniques as part of a future monitoring strategy (see Annex 8).

Regarding bird populations, a review (level 1) of statistics by BTO has shown that numbers of the internationally important Dark-bellied Brent Goose, Knot, and Ringed Plover have decreased in various component estuaries of the site, and High Alerts were been triggered for the Dark-bellied Brent Goose in the Colne SPA, and the Knot in Foulness SPA; Medium Alerts were triggered for Dark-bellied Brent Goose and Ringed Plover in the Blackwater Estuary SPA, and the Knot in Foulness SPA. The BTO review suggested that further investigation into these declines in the Blackwater and Colne SPAs was warranted (Armitage *et al*., 2002). Findings of the recent level 2 review indicated that factors including increased water abstraction and reduced freshwater inputs, changes in waste treatment and consequent reduction in organic loadings, disturbance, changes in surrounding land-use patterns, saltmarsh loss and erosion, and sediment changes may have influenced some of the bird population declines. Of these, reduced freshwater input and changes to waste water treatment may be directly related to EA consents.

Some of the declines mirror national and regional trends: i.e. decline of Ringed Plover populations in the Blackwater over the 25-year period, and Shelduck numbers on the Colne Estuary over the 5-year period, were related to changes in the species' national and regional population respectively (as recorded by WeBS).

In view of the suspected sensitivity of the site to changes in nutrient status, and the largely unknown status of many toxic contaminants, the acquisition of any further information which addresses possible links between environmental quality and biological consequences seems desirable.

Chemical Status

At present there is little unequivocal evidence from chemical data indicating that modifications to biota of the European Marine Site have occurred, or would be expected to occur, due to toxic contaminants. The potential *combined* threat from multiple inputs of nutrients, selected metals (Cu, Zn, Hg, Ag), residual TBT and episodic pesticide inputs is probably of most concern. However, the available evidence on biological effects from both toxic and non-toxic contaminants is very patchy. To address this issue in the future a more subtle, targeted assessment of impact will be required in order to establish cause and effect (discussed in greater detail later in this section). Until more appropriate, integrated monitoring is put in place it is only possible to assess, subjectively, individual contaminants, or groups of contaminants that could, in theory, impinge on sensitive species and benthic communities: a much abbreviated summary is also provided in table 34.

1) Organotins (TBT)

Legislation in 1987 was extremely effective in reducing inputs of TBT, as reflected in concentrations in water in the Crouch Estuary. Concentrations of 40-50 ng l^{-1} were not uncommon (sometimes more than an order of magnitude higher in marinas). By 1998, concentrations in the estuary were generally around the EQS of 2ng l^{-1} but in marinas concentrations could exceed this by three-fold (some illegal usage during the peak season?). TBT concentrations have sometimes also exceeded the EQS in the Blackwater Estuary in recent years, probably as a result of illegal usage and perhaps sediment re-suspension. There are no rigorous TBT time series for other parts of the EMS though it is expected that reductions seen in the Crouch have been mirrored in other small-boat dominated sections of the site.

Long-term monitoring in the Crouch Estuary by Cefas has linked the initial improvements in environmental quality with biological recovery. The threat from TBT is therefore receding, at least in terms of acutely toxic levels in water, but has not entirely disappeared. The persistence of sediment-bound TBT is also, potentially, a source of long-term chronic impact in poorly flushed areas, and could be exacerbated by dredging activities.

There were some substantial reductions in sediment TBT loadings in the immediate aftermath of the legislation (although there are indications of a much slower decline in the long-term). Sediments are probably now low in TBT throughout most of the EMS. However, as with water, concentrations near boatyards and marinas can exceed provisional (though perhaps overcautious) ecotoxicological guidelines set by OSPAR (0.005-0.05μg kg^{-1}). A precautionary approach may be needed if such areas are to be dredged. Processes including physical resuspension and bioturbation could remobilise these sinks. Furthermore, TBT in such contaminated sediments is likely to be bioavailable, and potentially harmful, to deposit-feeders and infauna (many of which are important dietary items for birds).

These data suggest TBT is no longer likely to be a *major* risk to the biota of the EMS though localised effects could be expected for sensitive species, particularly gastropods. In view of the moderate risks, a more detailed survey is needed to establish current status and trends for TBT in water, sediments and biota, and also to monitor imposex and population status of neogastropods, given the toxicological significance of this compound.

Table 34. A Summary of Water and Sediment Quality issues in Essex Estuaries EMS (Findings for each of the numbered 'contaminant categories' are explained in more detail in the accompanying text).

'contaminant'	Area	Potential Sources	Most vulnerable features/biota
1) Organotins (TBT, TPT?)	Current status unclear but likely to only be of significance near sources and in upper estuaries: persistent in sediments	Mostly historic but possibly some continuing input from shipping (docks), illegal use on small boats, STW and sediments.	Molluscs, particularly neogastropods
2) Metals (e.g. Cu, Zn, Ag, Hg)	No acute problems identified but sediment sometimes exceed ISQG/TEL guidelines in some upper Estuary sites. Potential for bioaccumulation in Managed Retreat Sites?	Possibly STWs, urban run-off, boats and ships. Thames Estuary an important source to the area, notably for Ag and Hg. Sediments principal sink for most metals. Modified conditions at MR sites could increase bioavailability?	Invertebrates (primarily molluscs and crustaceans), larval fish. Bioaccumulation in birds not evaluated
3) Nutrients (Macroalgal blooms, Hypernutrification)	Especially, Blackwater, Colne, inner Crouch	Agricultural sources, notably N (livestock, fertiliser application) STW discharges also important, notably P in Colne Possibly sediments – (phosphate, ammonia)	*Zostera spp.* Invertebrates, fish (esp. early life stages), seabirds, General diversity
4) Hydrocarbons, PAHs	Poorly defined, due to lack of sampling but some evidence of enrichment in sediments, notably Blackwater	Incomplete combustion of fossil fuels, run-off, discharges, aircraft, boats and ships: sediments main reservoir (PAHs probably of pyrogenic origin)	Benthic invertebrates and fish (NB those in contact with sediment). Very few bioaccumulation data.
5) Pesticides, herbicides and other synthetic organics	Estuarine Sediments probably main reservoir but little information on spatial distributions within the EMS. Direct toxicity improbable; extent of endocrine disruption not fully tested.	Not quantified due to insufficient data but probably includes a significant component from agricultural run-off, and sewage discharges. Thames estuary probably a major source in the area of PCB and some other POPs, but outside EMS	Invertebrates (esp. crustacea), fish Very few bioaccumulation data in indicator species other than mussels Extent of endocrine disruption not fully tested.

…(cont.)

Table 34. continued

6) Radionuclides	Blackwater Estuary principal monitoring site for radioactivity. Areas nearest major discharge at mouth of Blackwater potentially most at risk.	Major consented discharge is Bradwell Nuclear Power Station (ceased operation in 2002). Smaller consented discharge at head of Blackwater Estuary (from Chelmsford Hospital)	No evidence of any threat to aquatic species monitored, or to human consumers

2) Metals

There were no data available for metal concentrations in rivers or consented discharges entering the EMS; nevertheless the balance of evidence suggests such inputs would be unlikely to cause acute problems. During the last five years of tidal waters monitoring occasional samples in the Crouch/Roach (Cu, Cr, Hg, Pb, Zn), Blackwater (Cd, Cr, Cu, Zn) and Colne (Cr, Cu, Hg, Zn) have been above the guidance value, though there have been no cases of EQS excedences (based on annual averages, mainly). For Cu this represents an improvement on the situation a decade ago when there was a 22% failure rate. At the time, dominant sources of Cu were considered to be boats and ships (including dockyards and other maintenance facilities). There are indications from monitoring data that Zn inputs (from multiple sources) may have also decreased, though there are no clear links between concentrations in tidal waters and sources (probably multiple).

Metal concentrations in sediments are considered fairly typical for estuaries in the region (low level but widespread enrichment above background). None of the sediment metals was above sediment ‘probable effects levels’, but Ag, As, Cu, Hg and Ni exceeded the ISQG/TEL at a least one site. On this basis, according to the guideline criteria, chronic effects cannot be excluded.

Industrial and urban sources (run-off and STW) within the Thames probably influence distributions of pollutant metals such as Ag and Hg within the Essex Estuaries. In the past there may also have been a contribution from sewage sludge dumping off the outer Thames Estuary. However, it is also likely that the relative abundance of organic and oxyhydroxide coatings – characteristics which govern the ability of sediments to sequester metals and act as sinks – are a factor: upper estuary (and saltmarsh) sediments tend to be enriched naturally in these coatings compared to those of marine origin.

It is stressed that the data on sediments may not be representative of conditions now (though there are indications at one or two sites that levels have changed little during the past two decades). Re-survey is seen as a particularly important issue in terms of meeting standstill requirements for sediments under the Dangerous Substances Directive and attainment of Favourable Conservation Status under the Habitats Directive.

Bioaccumulation surveys conducted in the Essex Estuaries in the 1980s indicated some elevation of body burdens for pollutant metals including Ag, Sn (from TBT), Hg, Cd and Pb. Updated studies with infaunal species such as worms and clams are recommended, as there is little recent spatial information on metal bioavailability or bioaccumulation within the EMS, other than occasional samples from the NMMP shellfish sites at West Mersea and commercial beds in the outer Thames. Statutory monitoring at these designated beds in the mid 1990s indicated that samples from the region were sometimes at the upper end of the UK range. These included: Ag and Cr in Blackwater mussels, Ag and Cu in cockles from the outer Thames. Current 'bioaccumulation' monitoring by EA is focused on mussels from the designated shellfish area at West Mersea. Based on comparisons with OSPAR reference criteria, values (2002-2004) for Cd, Hg, Cu, Pb and, to a lesser extent, Zn were relatively enriched by UK standards. However, these concentrations do not pose a human health risk.

Acute ecotoxicological impact due to metal contamination is not anticipated throughout the site as a whole, but it is possible that a degree of anthropogenic enrichment, coupled with conditions in sediments (particularly anoxia), could increase the bioavailability of several metals. This may be pertinent for Managed Retreat Sites, where tidal inundation sometimes creates anoxic conditions in sediment horizons, enhancing the lability of several metals. There is scope for more research to establish the physiological and ecological significance of metal body burdens. Sub-lethal biomarkers, notably metallothionein induction, provide sensitive and selective measures of metal stress and can help map affected areas, as well as monitoring temporal trends.

3) Nutrients

Much of the marine site (notably the Blackwater, Colne and Crouch & Roach) is hypernutrified and is subject to eutrophication. Nutrient-associated water quality problems have been apparent for a number of years. Although the majority of nutrient inputs in the system may be due to diffuse sources such as agricultural run-off, sewage discharges constitute additional loading and result in chronic contamination of inner estuarine areas. The larger STWs (Colchester, Chelmsford and Maldon) which enter the most sensitive inner estuarine areas, and provide bio-available nutrients, are likely to be the major point source inputs on the Colne and Blackwater, respectively, but smaller STW discharges (e.g. Fingringhoe, Brightlingsea, West Mersea, Tiptree (via Salcott), Tollesbury, Bradwell On Sea) will also contribute to the general hypernutrification. Without analysis of data for concentrations and loadings arising from these discharges, however, it is impossible to assess particular sources.

Nutrient-associated water quality problems have been recorded for some time and include macroalgal proliferation, notably on the Blackwater Estuary, and much less extensively on the open coasts of Dengie and Foulness SPAs. High nutrient levels in several of the estuaries are in the range and ratios liable to trigger nuisance microalgal blooms, although, the very high levels of turbidity (especially in the Colne) limit light penetration and inhibit photosynthesis, and with the exception of one report of paralytic shellfish poisoning (PSP) causing the closure of Colne shellfish beds, significant nuisance blooms have not been reported.

The question of possible reductions in infaunal diversity and abundance, or sublethal effects to biota, arising from raised nutrient levels, are largely unresearched. The problem of hypernutrification has lead to various Sensitive Area designations within the EMS and its catchment area, which, it is hoped, will herald reductions in nutrient loadings from both point, and diffuse sources.

Some investment has been made to upgrade the wastewater treatment facilities at a number of STWs affecting the EMS, which is expected to lead to localised improvements in water quality.

Continuing seasonal surveys of nutrient concentrations in tidal waters (<u>and major effluent flows</u>) is recommended to establish the long-term trends in point source influences on water quality, and to determine whether further P (or N) reduction from point sources would be of benefit.

There is a complex interaction of nutrients between sediment and overlying water, which in the case of N, involves a range of processes including nitrification, denitrification, mineralisation, assimilation and fixation, which may all vary spatially and temporally. It has been shown that ammonifying microbial activity is greatest when macroalgal biomass declines, resulting in the release of ammonia from sediment. It is likely that this occurs in parts of the EMS, due to the presence of significant biomass of green macroalgal species (*Enteromorpha spp.*) which die back annually. Such conditions, are liable to increase the toxicity of ammonia, posing a threat to sensitive estuarine communities. The complexity of the nitrogen (and phosphorus cycle), and the significance of sediments, has been long appreciated, nevertheless monitoring still largely involves measurements of nutrients in water. Until more data becomes available for sediments, the significance of related microbial activity, and attempts at evaluating the sediment as source or sink for nitrogen, ammonia and phosphorus is difficult.

Changes to consents (quantities and location) should therefore be considered carefully to avoid the risk of further enrichment.

4) Hydrocarbons, PAHs

PAH enrichment in Essex Estuaries sediment is probably consistent with an anthropogenic origin. The source of these PAHs may include airborne particulates derived from combustion products (i.e. several major airports), urban run-off, various trade and domestic inputs, and marina activity. The Thames Estuary may also contribute to the PAH load in the marine site, with tidal movements transporting PAHs derived from similar aforementioned sources in the Thames catchment area.

From the somewhat limited amount of literature, PAH concentrations in the EMS (Blackwater and Crouch) are not as high as those recorded in highly industrialised estuaries, although the reported levels represent a threat of harmful effects to benthic organisms, notably in the Blackwater Estuary. Exposure pathways to most biota probably consist of both water and dietary sources including sediments. Feeding habit and lifestyle will be significant factors in modifying the bioavailability of PAHs, as with most contaminants. Because of their stronger affinity for sediments, high molecular weight PAHs are thought to be accumulated more strongly by deposit-feeders, rather than by those types which process overlying waters.

Reported PAH concentrations in sediments of the Blackwater Estuary are above the proposed Threshold Effect Levels for ΣPAHs, but below the Probable Effects Level, and almost all individual PAHs in sediments of both the Blackwater and Crouch are above proposed TELs. The concentrations in sediments of the Blackwater Estuary are sufficiently high to generate sub-lethal effects in fish and fish larvae, i.e. EROD induction, although EROD has not been measured in fish from estuarine sites within the Essex estuaries EMS. This should be remedied in order to assesss the biological condition of the marine site with more confidence.

PAH concentrations in much of the marine site are unknown, as there appears to be no EA monitoring data, a situation that should be remedied as a matter of urgency. Similarly, EA monitoring of PAHs in shellfish appears to have been limited to one site, and data contains numerous 'less than' values (some of which vary over the period) making it difficult to present an accurate assessment of any shellfish PAH. In view of the high levels of PAHs recorded in sediments of the Blackwater Estuary, where there are a number of designated shellfish areas, extending regular monitoring to include other shellfish sites would also appear to be an urgent requirement. Any effects, if they occur, are likely to be chronic, although acute effects to some infaunal species cannot be ruled out. In view of the uncertainties over EQS guidelines, further work will be needed to assess the actual biological consequences of PAHs for the marine site: the presence of elevated levels of other contaminants may be a complicating feature and demonstrates the need to integrate biological and chemical monitoring.

5) Pesticides, Herbicides, PCBs and other persistent organic contaminants.

There is a lack of qualitative evidence on the distribution, bioavailability and toxic effects of organic micropollutants on the flora and fauna of the Essex Estuaries EMS. There are concerns, nevertheless, that they may contribute, along with inorganic contaminants, towards loss of condition of features at the site. In particular, a number of herbicides may be considered as a *potential* threat to sensitive salt marsh plants (as well as algae and *Zostera*) in the region, though as yet these threats remain largely unquantified.

Most synthetic organic compounds analysed by the Agency in tidal waters (2000-2005) were below detection limits and appear to comply with EQS standards. On this basis risks to biota within the European Marine Site, from aqueous sources might be assumed to be low. However, spatial and temporal coverage are not adequate to give a detailed assessment.

For the majority of POPs reviewed there are no data to evaluate sources. In general terms, STW are likely to act as an important source of such contaminants, along with run-off from agricultural land, weed control in urbanized locations, and landfill sites. The agricultural nature of the catchments feeding into the EMS implies that application of agrochemicals may have been widespread. Episodic pulses into intertidal saltmarsh systems, arising from herbicide and pesticide application in adjacent agricultural areas, has been demonstrated in a small number of studies. Extrapolations from toxicity data suggest resulting impacts are possible. Further investigations need to be undertaken to establish whether or not such events represents a real risk to the Essex marshes.

Published sediment data from the 1990s include a number of examples which exceeded the PEL or TEL by a small margin, again indicating that impacts are possible (for example MCPP , 2,4-D and other members of the CPH herbicide group at S. Woodham Ferrers (Crouch) and Salcott (Blackwater), probably of agricultural origin). Elevated concentrations of DDT at Dengie are surprising given its open coastal location: as PCB is also elevated here, possible sources are unlikely to be purely agricultural and may include a component from offshore sludge dumping grounds and the Thames Estuary. Apportioning sources of POPs is in most cases difficult since they tend to be fairly ubiquitous, low level contaminants and have been used for urban as well as agricultural application.

Results of EA bivalve monitoring in shellfish waters contain a predominance of <DL values suggesting that most POPs pose little bioaccumulative threat in the region, though only the West Mersea site has been sampled in recent years (2000-2005). Earlier results from 1990 in the Crouch and outer Thames Estuary indicated γHCH, dieldrin, ppDDE and PCBs were marginally above the OSPAR threshold and in the mid 1990s were sometimes at the upper end of the UK range. More recent fish and shellfish data for PCBs suggest the Thames Estuary is still one of the most contaminated areas monitored in the UK and for a number of other POPs concentrations are above OSPAR guidelines.

The sparse data on POPs are indicative of low level but extensive contamination throughout the area. However, further targeted sampling in the EMS is needed to characterise current sources and to assess the threat of bioaccumulation thoroughly; this should incorporate sediments and sediment-dwelling species such as clams and worms.

The possibility of endocrine disruption has not been investigated extensively in the European marine site and should be addressed in future in view of the observations of feminization in male flounder from the Crouch Estuary.

6) Radioactivity

The major licensed releases of radionuclides to the EMS are those from Bradwell nuclear power station at the mouth of the Blackwater Estuary, which is currently being decommissioned. Monitoring of radioactivity by CEFAS in nearby water, sediment, fish and shellfish is focused on monitoring uptake and risk to humans via consumption and external exposure. Concentrations of artificial radionuclides are generally low in all aquatic sample types (below detection limits in seawater) making source apportionment difficult. It is likely that, in addition to materials discharged from Bradwell, inputs from Sellafield, Cap de la Hague and historical weapons testing contribute to loadings. The principal long-term surveillance of benthic organisms is focused on the commercial oyster fishery near West Mersea and indicates significant reductions in ^{65}Zn and ^{137}Cs in recent years. In summary, the radiological impact of authorised disposals from Bradwell is considered to be low and is anticipated to continue to decline.

Future Research Requirements

Better, more integrated information on environmental chemistry, 'health' and biodiversity are obvious, generalised top-level needs to address the 'quality' of the Essex Estuaries European Marine Site, just as rigorous monitoring of habitats will be needed to provide estimates of 'quantity' and extent.

With regard to the latter, an appropriate surveillance strategy should be adopted to quantify the nature and extent of biotopes in order to fulfil Favourable Conservation Status[34] as encompassed in the Habitats Directive. Monitoring techniques are described in detail by Davison and Hughes (1998) in relation to surveillance of *Zostera* beds but are equally applicable to biotope monitoring in a broader context. These fall into four categories: *aerial remote sensing*; *sublittoral remote sensing*; *underwater video*; *conventional benthic sampling.* (e.g. grabs and cores and observations arising from inter-tidal surveys and diving). The first three options have potential for large-scale generalised mapping of distribution and extent, whilst the fourth provides a more detailed synthesis of biotope structure and condition. Ideally, the preferred monitoring scheme would probably consist of a blend of these approaches taking in to account questions of scientific objectives, practicality and cost. (Davison and Hughes, 1998).

In view of the sensitivity of the site to the possible effects of eutrophication it is recommended that monitoring of both phytoplankton and benthic macroalgae is undertaken, alongside nutrient determination. More work is needed to construct spatial and seasonal models of nutrient sources and fluxes and should also include the role of sediments. Further hydrodynamic modelling and simulation of nutrient distributions would be a useful supplement.

The requirement to fulfil FCS and other drivers relating to toxic contaminants and water and sediment quality (such as the 'standstill' provision under the Dangerous Substances Directive) may be difficult to monitor. The fragmented nature of much of the available environmental quality data prevents all but a first approximation of the status of the site for many contaminants. This needs to be addressed if we are to

[34] Favourable Conservation Status (FCS) for a given habitat/species requires that it's condition has to be characterised and, if considered necessary, brought up to a level where the habitat/species is sustainable in the long term. Under the Habitats Directive, Member States must report on the progress towards FCS for all nominated sites. English Nature has produced Favourable Condition tables to aid this process, which encompasses a number of attributes, including the extent and biological quality of the interest features (summarised in Annex 1 and 2 for the Essex Estuaries EMS).

progress our understanding of how environmental quality, and in particular anthropogenic inputs, are affecting the status of the European marine site.

A major issue central to the current project is how to monitor the health of the environment within the site i.e. to ensure that conditions are favourable for the survival of biota and, if they are not, to establish any cause and effect relationships.

Some of the classes of contaminants which, in our opinion, should be prioritised in future surveys have been discussed above. Traditionally, surveillance of chemical parameters in water has often been carried out by the EA for the purposes of compliance monitoring and is not necessarily intended for survey purposes (or for the type of characterisation being undertaken here). This should perhaps be reviewed in future to maximise the value of the information.

Thus, in addition to nutrients, accurate, up-to-date chemical data on e.g. metals, PAHs, organotins, PCBs and some pesticides and herbicides, are needed to give better impressions of fluxes from rivers and discharges, and to provide details of their current distribution, sources and sinks within the Essex Estuaries EMS. It is particularly important that future sampling programmes incorporate more information on sediment contaminants and their role as diffuse sources. Metal concentrations in sediments presumably consist of both anthropogenic and geogenic components which have yet to be distinguished. Methods are becoming available which address this problem and the consequences for bioavailability and re-release. It is recommended that these be applied. Bioindicator organisms should be collected, concurrently, to try to link sediment loadings and 'speciation' of contaminants with their biological consequences. This should incorporate infaunal organisms, particularly bivalves and polychaetes, which are capable of reflecting sediment-bound contamination. Surveys should be repeated at intervals in the future, to ensure that bioavailability does not increase. Furthermore, many sediment-dwelling organisms are essential food items for the important bird species and assemblages for which the site is designated. The threat posed by bioaccumulation and food chain transfer of priority contaminants needs to be qualified in these species, requiring a much more extensive data set.

Biotope mapping and biodiversity indices are important components of the site assessments but may not be sufficiently sensitive to quantify subtle threats, or predict future change. In recent years, a range of biomarker and bioassay techniques have been developed to assess sub-lethal biological impact in greater detail, and would allow targeted biological-effects screening of the Essex estuaries, including possible problem discharges. Also, they can be used to assess the risk from diffuse sources. Current sediment quality criteria are useful initial guidelines to scale threats, but may not be entirely appropriate for all species or sediment types. Individual site conditions are likely to modify threats considerably, therefore accurate assessments need to be customised to the habitat in question. By selection of an appropriate suite of indicators/biomarkers, a sampling strategy could be tailor-made[35] to establish the causes and extent of damage (if any), or improvement, in the EMS with greater certainty. This would ideally include; conventional quantitative ecological survey (for

[35] Examples of biological effects techniques from which a selection could be chosen to assess condition are shown in annex 8. Note also that a trial project on the value of such techniques in assessing the condition of EMS is currently being undertaken by PMSP (University of Plymouth and MBA); EN project FST 20-18-028.

identifying changes in the abundance and diversity of species), targeted chemical and biomonitoring procedures as outlined above (for determining the concentrations and bioavailability of anthropogenic contaminants), and biochemical, physiological and behavioural biomarkers which signal exposure to, and in some cases, adverse effects of, pollution. Some examples of biological effects techniques are summarised in Annex 8.

When such procedures are used in combination in well-designed survey programmes, they can provide insights into which pollutants are responsible for environmental degradation. They may also be useful in addressing the long-standing problem of additivity/ synergism. A major criticism of many current statutory monitoring assessments, whether using comparisons with EQS values, sediment quality guidelines, or some other marker, is that they address only single contaminants at a time. Even if individual chemicals do comply with limit values (as most appear to do in the Essex Estuaries) it does not necessarily mean the environment is healthy. Biological effects may occur if several contaminants act together. The majority of outfalls and sediments contain a particular cocktail of chemicals whose true impact can usually only be assessed through a site-specific evaluation, taking into consideration the interactions that occur between different components and also the local environment. Sediments in some areas for example, may become highly anoxic as a result of overlying macroalgal mats, or deliberate tidal inundation - a feature that can modify the impact of a variety of contaminants as well as inducing mortality directly. By incorporating biological-effects monitoring, alongside chemical surveillance, it may be possible to make substantial progress towards understanding and managing these complex environmental issues and would provide more reliable and objective site characterisations in the future. If such an integrated approach were put in place at an early stage, to provide baselines, it would clearly be amenable for measuring long-term trends.

In the absence of specific information on discharges it is not possible to comment on the arguments for and against more rigorous regulatory action on individual consents. Presumably EA will be assessing loadings from these sources as part of the Review of Consents process. Review of the available chemical data for tidal sections of the Essex Estuaries rarely raises concerns over acute toxicity within the EMS, though the possibility that combined pressures might impair performance of sensitive species and communities cannot be ruled out. At present, the (limited) evidence probably does not justify expensive remedial action on sources. However, there is sufficient uncertainty to justify a more targeted and detailed programme of research and surveillance to measure actual biological impacts at a variety of levels (e.g. biochemistry, bioaccumulation, biomarkers, community structure). If results indictate deleterious effects, which can be attributed to known causes, then the case for remedial action against key sources (which may include multiple inputs) would be placed on a stronger, scientifically-sound basis. At the very least, such a program would provide a benchmark for assessing future changes in the condition of the site, and likely contributions from water quality.

Hypernutrification – any measurable increase in the concentration of a dissolved nutrients (ICES, 1984*). Hypernutrification is deemed to exist where winter nutrient concentrations “significantly” exceed 12 mM DAIN m^{-3} [>12 μM l^{-1} or 168 μg l^{-1} N] in the presence of at least 0.2 mM DAIP m -3 [≥0.2 μM l^{-1} or 6.2 μg l^{-1} P] (EC, 1999**).

Eutrophication –Any measurable increase in primary production resulting from hypernutrification (ICES, 1984*). “The enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance[36] to the balance of organisms present in the water and to the quality of the water concerned” (EC, 1999**).

*International Council for the Exploration of the Sea. I.C.E.S., 1984. (E:12). General Secretary, I.C.E.S., Palaegade 2-4, DK 1261, Copenhagen K, Denmark.

**European Commission Directorate General XI (1999).Verification of Vulnerable Zones Identified under the Nitrate Directive and Sensitive Areas Identified under the Urban Waste Water Treatment Directive. Environmental Resources Management London United Kingdom

[36] The “undesirable disturbance” aspect is used more and more by EA to decide on whether action should be taken

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11 ANNEXES

Annex 1. Summary of the interest features and conservation objectives Essex Estuaries European Marine Site: (Adapted From English Nature, 2000)

Interest features under the EU Habitats Directive. NB. A number of sub-features have been identified relating to each of the interest features and further information can be found in the relevant Regulation 33 advice (English Nature 2000).

Essex Estuaries SAC Annex I habitat features:

- ***Salicornia* and other annuals colonising mud and sand (pioneer saltmarsh)**

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

Glasswort (*Salicornia* agg.) / annual sea-blite (*Suaeda maritima*) community and sea aster (*Aster tripolium var. discoides)* community

- ***Spartina* swards (*Spartinion*) (Cordgrass)**

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

Small cordgrass (*Spartina maritima*) and smooth cordgrass (*S. alterniflora* community

- **Atlantic salt meadows (*Glauco-Puccinellietalia*)**

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

- Low/mid-marsh communities
- Upper marsh communities
- Upper marsh transitional communities
- Drift-line community

- **Mediterranean and thermo-Atlantic halophilous scrubs (*Arthrocnemetalia fruticosae*) (Mediterranean saltmarsh scrubs)**

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

- Shrubby sea-blite community
- Rock sea lavender/sea heath community

Annex 1. (cont.)

Essex Estuaries SAC Annex I habitat features:

- **Estuaries**

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

- Saltmarsh communities
- Intertidal mudflat and sandflat communities
- Rock communities
- Subtidal mud communities
- Subtidal muddy sand communities
- Subtidal mixed sediment communities

- **Mudflats and sandflats not covered by seawater at low tide** (intertidal mudflats and sandflats)

Conservation objectives focus on maintaining in favourable condition subject to natural change. In particular:-

- Mud communities
- Muddy sand communities
- Sand and gravel communities

- **Subtidal sandbanks***

- Sandbanks which are slightly covered by sea water all the time

* This feature was added after Reg 33

Annex 2. Interest features under the EU Birds Directive:

Each SPA within the Essex Estuaries European marine qualifies for a number of different internationally and nationally important bird species. Conservation objectives for bird populations focus on maintaining habitats in favourable condition subject to natural change.

The Colne Estuary SPA supports

- Internationally important breeding populations of the regularly occurring Annex 1 species: little tern (*Sterna albifrons*).
- Internationally important wintering population of the Annex 1 species hen harrier (*Circus cyaneus*) [1]
- Internationally important assemblage of waterfowl (wildfowl and waders)
- Internationally important populations of regularly occurring migratory species
- Nationally important breeding populations of the regularly occurring migratory species: ringed plover (*Charadrius hiaticula*)
- Nationally important breeding population of the migratory species pochard (*Aythya farina*)[2]

[1] The habitat required for this species to feed - grassland/grazing marsh - does not occur within the European marine site, as it occurs above Highest Astronomical Tide. Objectives to maintain this aspect of bird interest in favourable condition are found within English Nature's conservation objectives for the relevant SSSI within the SPA boundary.

[2] The freshwater habitat required for this species to breed also does not occur within the European marine site, as it occurs above Highest Astronomical Tide. Objectives to maintain this aspect of bird interest in favourable condition are found within English Nature's conservation objectives for the relevant SSSI within the SPA boundary.

The Blackwater Estuary SPA supports:

- Internationally important breeding populations of the regularly occurring Annex 1 species: little tern (*Sterna albifrons*).
- Internationally important wintering population of the Annex 1 species hen harrier (*Circus cyaneus)*[1]
- Internationally important assemblage of waterfowl (wildfowl and waders)
- Internationally important populations of regularly occurring migratory species.

[1] The habitat required for this species to feed - grassland/grazing marsh - does not occur within the European marine site, as it occurs above Highest Astronomical Tide. Objectives to maintain this aspect of bird interest in favourable condition are found within English Nature's conservation objectives for the relevant SSSI within the SPA boundary.

Annex2. (cont.) Interest features under the EU Birds Directive:

The Dengie SPA supports:

- An internationally important wintering population of the Annex 1 species hen harrier (*Circus cyaneus*)[1]
- An internationally important assemblage of waterfowl (wildfowl and waders)
- Internationally important populations of regularly occurring migratory species.

[1] The habitat required for this species to feed - grassland/grazing marsh - does not occur within the European marine site, as it occurs above Highest Astronomical Tide. Objectives to maintain this aspect of bird interest in favourable condition are found within English Nature's conservation objectives for the relevant SSSI within the SPA boundary.

The Crouch and Roach Estuaries SPA supports:

- An internationally important assemblage of waterfowl (wildfowl and waders)
- Internationally important populations of regularly occurring migratory species.

The Foulness SPA supports:

- Internationally important breeding populations of regularly occurring Annex 1 species: sandwich tern (*Sterna sandvicensis*), common tern (*Sterna hirundo*), little tern (*Sterna albifrons*) and avocet (*Recurvirostra avosetta*)
- Internationally important wintering population of the Annex 1 species hen harrier (*Circus cyaneus*) [1]
- Internationally important assemblage of waterfowl (wildfowl and waders)
- Internationally important populations of regularly occurring migratory species
- Nationally important breeding populations of a regularly occurring migratory species: ringed plover (*Charadrius hiaticula*).

[1] The habitat required for this species to feed - grassland/grazing marsh - does not occur within the European marine site, as it occurs above Highest Astronomical Tide. Objectives to maintain this aspect of bird interest in favourable condition are found within English Nature's conservation objectives for the relevant SSSI within the SPA boundary.

Annex 3. Water Quality Standards

List I (EC Dangerous Substances Directive) and list II substances (from Cole *et al.,* 1999, derived by WRc according to the methodology described in Grimwood and Dixon 1997)

List I substances

Parameter	Unit	Water quality standard		Standstill Provision [a]
		Estuary[b]	Marine	
Mercury	µg Hg/l	0.5 DAA	0.3 DAA	yes[c]
Cadmium	µg Cd/l	5 DAA	2.5 DAA	yes
Hexachlorocyclohexane[d]	µg HCH/l	0.02 TAA	0.02 TAA	yes
Carbon tetrachloride	µg CCl_4/l	12 TAA	12 TAA	no
Dichlorodiphenyltrichloroethane				
(all 4 isomers, total DDT)	µg DDT/l	0.025 TAA	0.025 TAA	yes
(para, para-DDT)	µg ppDDT/l	0.01 TAA	0.01 TAA	yes
Pentachlorophenol	µg PCP/l	2 TAA	2 TAA	yes
Total drins	µg /l	0.03 TAA	0.03 TAA	yes
Aldrin	µg /l	0.01 TAA	0.01 TAA	yes
Dieldrin	µg/l	0.01 TAA	0.01 TAA	yes
Endrin	µg /l	0.005 TAA	0.005 TAA	yes
Isodrin	µg/ l	0.005TAA	0.005 TAA	yes
Hexachlorobenzene	µg HCB/l	0.03 TAA	0.03 TAA	yes
Hexachlorobutadiene	µg HCBD/l	0.1 TAA	0.1 TAA	yes
Chloroform	µg $CHCl_3$/l	12 TAA	12 TAA	no
1,2-Dichloroethane (ethylenedichloride)	µg EDC/l	10 TAA	10 TAA	no
Perchloroethylene (tetrachloroethylene)	µg PER/l	10 TAA	10 TAA	no
Trichlorobenzene (all isomers)	µg TCB/l	0.4 TAA	0.4 TAA	yes
Trichloroethylene	µg TRI/l	10 TAA	10 TAA	no

Notes: Substances are listed in order of publication of Directives.

D Dissolved concentration, ie usually involving filtration through a 0.45-µm membrane filter before analysis
T Total concentration (ie without filtration).
AA standard defined as annual average
[a] Most directives include, in addition to the standards for inland, estuary and marine waters, a provision that the total concentration of the substance in question in sediments and/or shellfish and/or fish must not increase significantly with time (the "standstill" provision).
[b] In the UK the standards for estuaries are the same as for marine waters - The Surface Waters (Dangerous Substances) (Classification) Regulations 1989
[c] In addition to a standstill provision applying to sediments or shellfish there is a further environmental quality standard of 0.3 mg Hg/kg wet flesh "in a representative sample of fish flesh chosen as an indicator".
[d] All isomers, including lindane

Annex 3 (cont.) Water quality standards for the protection of saltwater life.

List II substances

Parameter	Unit	WQS (see footnotes)	Uncertainties in the derivation : Details obtained from the relevant EQS derivation reports
Lead	µg Pb/l	25 AD[1,5]	The preliminary EQS was multiplied by a factor of 2 to account for overestimation of Pb toxicity in laboratory studies compared to the field environment. The EQS was considered tentative as a result of the paucity of reliable data, in particular for sub-lethal chronic studies with invertebrates and fish, and for field studies.
Chromium	µg Cr/l	15 AD[1,5]	There were limited data on the sub-lethal effect of Cr and long-term exposure to freshwater and saltwater life. Separate standards for different Chromium valences (Cr(VI) and Cr(III)) were not recommended as a consequence of the lack of data for Cr(III). In addition, a comparison of the toxicities of each oxidation state was not possible. Some data were available that indicated higher sensitivity of some saltwater organisms to low salinities. The EQS was based on data generated at salinities typical of normal seawater. Therefore, further research on the effect of Cr at lower salinities was recommended.
Zinc	µg Zn/l	40 AD[1,5]	The dataset available for the toxicity of Zn to saltwater life illustrated that at the EQS, adverse effects on algal growth had been reported. However, it was considered that there was currently insufficient evidence to suggest that the EQS would not adequately protect saltwater communities.
Copper	µg Cu/l	5 AD[1]	Further data were considered necessary on the sensitivity of early life stages and life-cycle tests to confirm the sensitivity of saltwater life.
Nickel	µg Ni/l	30AD[1]	Marine algae were reported to be adversely affected by Ni at concentrations as low as 0.6 _g l^{-1} which is below the EQS to protect saltwater life However, it was considered that there was insufficient evidence to justify a lower EQS based solely on results with algae and that further research into this area was desirable. There was also limited evidence to suggest that invertebrates in estuarine systems may be more susceptible to the effects of Ni than invertebrates in marine systems. Thus, an EQS to protect estuarine life may be needed in future when further data become available.
Arsenic	µg As/l	25AD[2]	Based on crab 96 hour LC50, and an extrapolation factor of 10 applied. Standards may need to be more stringent where sensitive algal species are important features of the ecosystem
Boron	µg B/l	7000 AT[1]	Few data available. However the standard was based on Dab 96 hour LC50, with an extrapolation factor of 10 applied
Iron	µg Fe/l	1000AD[1,5]	The EQS for the protection of saltwater life was based on observed concentrations and general assessments of water quality. It was recommended, therefore, that the standard should be reviewed as soon as direct observations of water concentrations and biological status become available. Limited data did not allow an assessment of the importance of Fe species.
Vanadium	µg V/l	100 AT[1]	Data on the toxicity of vanadium on saltwater life were limited. As there were limited data for vanadium, it was not possible to recommend standards based on dissolved concentrations or separate standards for migratory fish. With regard to the latter, it may be necessary to base judgement of any risk in applying the EQS on knowledge of local risks and circumstances.
Tributyltin	µg/l	0.002 MT[2]	The standards for TBT were tentative to reflect a combination

			of the lack of environmental data, toxicity data or data relating to the behaviour of organotins in the environment.
Triphenyltin (and its derivatives)	µg /l	0.008 MT[2]	The standards for TPT were tentative to reflect a combination of the lack of environmental and toxicity data or data relating to the behaviour of organotins in the environment.
PCSDs	µg /l	0.05 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Cyfluthrin	µg /l	0.001 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values
Sulcofuron	µg /l	25 PT[1]	As a consequence of the general paucity of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values. The data for sulcofuron suggested that embryonic stages for saltwater invertebrates could be more sensitive than freshwater species and, therefore, the EQS for the protection of marine life, derived from the freshwater value, may need to be lower.
Flucofuron	µg /l	1.0 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were based on freshwater values.
Permethrin	µg /l	0.01 PT[1]	In view of the lack of data for the mothproofing agents, both from laboratory and field studies, the EQSs were reported as tentative values.
Atrazine and Simazine	µg /l	2 AA[2] 10 MAC[4]	The EQSs for the protection of saltwater life were proposed as combined atrazine/simazine to take account of the likely additive effects when present together in the environment.
Azinphos-methyl	µg /l	0.01AA[2] 0.04 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Dichlorvos	µg /l	0.04 AA 0.6 MAC[2]	Based on data for sensitive crustaceans
Endosulphan	µg /l	0.003 AA[2]	There is little evidence on the ultimate fate of endosulfan and its metabolites or degradation products in sediments and on any effects on freshwater benthic organisms. Consequently, it is possible that some sediment-dwelling organisms, such as crustaceans, may be at risk.
Fenitrothion	µg /l	0.01 AA[2] 0.25 MAC[4]	As there were limited data with which to derive EQSs to protect saltwater life, the freshwater values were adopted. However, the annual average for the protection of freshwater life may be unnecessarily stringent in view of the uncertainties associated with the acute toxicity data used in its derivation. The uncertainties exist because the original sources were unavailable for certain studies. Lack of confirmatory data existed in the published literature and data for warm water species were considered in the derivation.
Malathion	µg /l	0.02AA[2] ;0.5MAC[4]	It was recommended that further investigation for both field and laboratory conditions into the effects of malathion on crustaceans and insects and on UK *Gammarus* species, in particular, should be carried out.
Trifluralin	µg /l	0.1AA[2]	None mentioned with regard to the annual mean.

		20 MAC[4]	
4-chloro-3-methyl phenol	µg /l	40 AA[3] 200 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2-chlorophenol	µg /l	50 AA[3] 250 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4-dichlorophenol	µg /l	20 AA[3] 140 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
2,4D (ester)	µg /l	1 AA[3] 10 MAC[4]	For the EQS proposed for 2,4-D esters, comparison of the data and derivation of standards were complicated by the number of esters and organisms for which studies were available. In addition, the toxicity of the esters may have been underestimated in some of the studies due to their hydrolysis. There were limited data on the toxicity of 2,4-D ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
2,4D	µg /l	40 AA[3] 200 MAC[4]	There were limited data on the toxicity of 2,4-D non-ester to saltwater life. Consequently, the freshwater value was adopted until further data become available.
1,1,1-trichloroethane	µg /l	100 AA[3] 1000 MAC[4]	The 1,1,1-TCA dataset available for freshwater species contained comparatively few studies where test concentrations were measured and, consequently, comparison of studies using measured concentrations vs. those using nominal values indicated that data from the latter type of study could be misleading.
1,1,2-trichloroethane	µg /l	300 AA[3] 3000 MAC[4]	For 1,1,2-TCA, few data were available on chronic toxicity to freshwater fish. There were limited data on the toxicity of 1,1,2-TCA to saltwater life and, consequently, the EQS to protect freshwater life was adopted.
Bentazone	µg /l	500 AA[3] 5000 MAC[4]	In view of the relatively high soil organic carbon sorption coefficient, it is likely that a significant fraction of the pesticide present in the aquatic environment will be adsorbed onto sediments or suspended solids. However, it is likely that this form will be less bioavailable to most aquatic organisms. As the adsorbed pesticide is more persistent than the dissolved fraction, it is possible that levels may build up that are harmful to benthic organisms. Insufficient information on saltwater organisms was available to propose a standard. In view of the paucity of data, the standards to protect freshwater life were adopted to protect saltwater life.
Benzene	µg /l	30 AA[3] 300 MAC[4]	Limited and uncertain chronic data available.
Biphenyl	µg /l	25 AA[3]	The data available for marine organisms were considered inadequate to derive an EQS for the protection of marine life. However, the reported studies for saltwater organisms indicate that the EQS for freshwater life will provide adequate protection.
Chloronitrotoluenes (CNTs)	µg /l	10 AA[3] 100 MAC[4]	The dataset used to derive the EQS to protect freshwater life was limited. Toxicity data were available for comparatively few species and there was limited information on the bioaccumulation potential of the isomers. There were few chronic studies available to allow the assessment of the long term impact of CNTs. There were no reliable data for the toxicity to or bioaccumulation of CNTs by saltwater species and, therefore, the EQSs proposed for freshwater life were adopted.
Demeton	µg /l	0.5 AA[3] 5 MAC[4]	Insufficient saltwater data were available to propose a standard. Therefore, the standard was based on freshwater value.
Dimethoate	µg /l	1 AA[3]	The available data for marine organisms were considered

			inadequate to derive an EQS for the protection of marine life. Crustaceans were considered to be the most sensitive organisms, but more data are required to confirm this. In view of the uncertainties associated with the marine toxicity dataset, the freshwater EQS was adopted. This was based on the toxicity of dimethoate to insects. Although there are no marine insects, there is some evidence that marine organisms are more sensitive than their freshwater counterparts.
Linuron	µg /l	2 AA[3]	In view of the lack of data for saltwater life, the EQS proposed for the protection of freshwater life was adopted until further data become available.
Mecoprop	µg /l	20 AA[3] 200 MAC[4]	There were limited data relating to the toxicity of mecoprop to aquatic life. The dataset for saltwater life comprised data for one marine alga, a brackish invertebrate and a brackish fish. Consequently, the freshwater values were adopted until further data become available.
Naphthalene	µg /l	5 AA[3] 80 MAC[4]	Limited and uncertain chronic data available.
Toluene	µg /l	40 AA[3] 400 MAC[4]	The dataset used to derive the EQS to protect saltwater life relied on static tests without analysis of exposure concentrations. Consequently, the derived values are considered tentative until further data from flow-though tests with analysed concentrations become available.
Triazophos	µg /l	0.005 AA[3] 0.5 MAC[4]	The dataset available for freshwater life was limited to a few studies on algae, crustaceans and fish. No information was available for the target organisms (insects), on different life-stages or on its bioaccumulation in aquatic organisms. There were no data on the toxicity or bioaccumulation of triazophos in saltwater organisms. Consequently, the EQSs to protect freshwater life were adopted until further data become available.
Xylene	µg /l	30 AA[3] 300 MAC[4]	Limited information available. Freshwater data used to § back up§ the standards.

Notes

Substances are listed in the order of publication of Directives.

A annual mean
D dissolved concentration, ie usually involving filtration through a 0.45-µm membrane filter before analysis
T total concentration (ie without filtration)
µg/ l micrograms per litre
AA standard defined as annual average
MAC maximum concentration
[1] DoE Circular in 1989 (Statutory standard)
[2] Statutory Instrument 1997 (Statutory standard)
[3] Statutory Instrument 1998 (Statutory standard)
[4] Non- statutory standard
[5] revised standards have been proposed but are not statutory

Annex 4. Quality Standards Stipulated In The Shellfish Waters Directive

(from Cole et al., 1999)

Parameter	Unit	G	I
A. GENERAL PHYSIO-CHEMICAL PARAMETERS			
Colour			(a)
Dissolved oxygen	% sat	>80 T95	>70 TAA(b)
pH			7-9 T75
Salinity	g/kg	12-38 T95	40 T95(c)
Suspended solids			(d)
Tainting substances			(e)
Temperature		(f)	
B. METALS AND INORGANIC ANIONS			
Arsenic		(g)	(h)
Cadmium		(g)	(h)
Chromium		(g)	(h)
Copper		(g)	(h)
Lead		(g)	(h)
Mercury		(g)	(h)
Nickel		(g)	(h)
Silver		(g)	(h)
Zinc		(g)	(h)
C. ORGANIC SUBSTANCES			
Hydrocarbons			(i)
Organohalogens		(g)	(h)
D. MICROBIOLOGICAL PARAMETER			
Faecal coliforms	per 100 ml	300 T75(j)	

Notes:

G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)
D dissolved concentration ie usually involving filtration through a 0.45-μm membrane filter before analysis
AA standard defined as annual average
75 standard defined as 75-percentile
95 standard defined as 95-percentile
MA maximum allowable concentration

Pt/l concentration of platinum (Pt) determined photometrically on the Platinum/Cobalt scale as a measure of colour in water

[a]A discharge affecting shellfish waters must not cause an increase in colouration of more than 10 mg Pt/l compared to the waters not so affected (waters filtered in both cases). This standard is expressed as a 75-percentile.

[b]If an individual result indicates a value lower than 70% of saturation, the measurement must be repeated. Concentrations below 60% of saturation are not allowed, unless there are no harmful consequences for the development of shellfish colonies.

[c]A discharge affecting shellfish waters must not cause an increase in salinity of more than 10% compared to the water not so affected. This standard is expressed as a 75-percentile.

[d]A discharge affecting shellfish waters must not cause an increase in the concentration of suspended solids by more than 30% compared to the water not so affected. This standard is expressed as a 75-percentile.

[e]The concentration of substances affecting the taste of shellfish must be lower than that liable to impair the taste of the shellfish.

[f]A discharge affecting shellfish waters must not cause an increase in temperature of more than 2 °C compared to the water not so affected. This standard is expressed as a 75-percentile.

[g]The concentration of this substance or group of substances in shellfish flesh must be so limited that it contributes to the high quality of shellfish products.

[h]The concentration of this substance or group of substances in water or in shellfish flesh must not exceed a level which gives rise to harmful effects in the shellfish or their larvae. Synergistic effects must also be taken into account in the case of metal ions.

[i]Hydrocarbons must not be present in water in such quantities as to produce a visible film on the surface of the water and/or a deposit on the shellfish, or to have harmful effects on the shellfish.

[j]In shellfish flesh and intervalvular fluid. However, pending the adoption of a directive on the protection of consumers of shellfish products, it is essential that this value be observed in waters from which shellfish are taken for direct human consumption.

Annex 5. Bathing Waters Quality Standards

Quality standards for fresh and saline waters stipulated in the Bathing Waters Directive (from Cole *et al.,* 1999)

Parameter	Unit	G	I
A. INORGANIC SUBSTANCES AND GENERAL PHYSICO-CHEMICAL PARAMETERS			
Colour			(a, b)
Copper	mg Cu/l		
Dissolved oxygen	% saturation	80-120 T90	
pH			6-9 T95(b)
Turbidity	Secchi depth m	>2 T90	>1 T95(b)
B. ORGANIC SUBSTANCES			
Floating waste(c)		(d)	
Hydrocarbons	µg l-1	300 T90(e)	(f)
Phenols	µgC_6H_5OH	5 T90(e)	50 T95(e)
Surfactants(g)	µg l-1 as lauryl sulphate	300 T90(e)	(k)
Tarry residues		(d)	
C. MICROBIOLOGICAL PARAMETERS			
Faecal coliforms	per 100 ml	100 T80	2 000 T95
Total coliforms	per 100 ml	500 T80	10 000 T95
Faecal streptococci	per 100 ml	100 T90	
Salmonella	per 1 l		0 T95
Entero viruses	PFU/10 l		0 T95

Notes
G guide value
I imperative (mandatory) value
T total concentration (ie without filtration)80 standard defined as 80-percentile*
90 standard defined as 90-percentile*
95 standard defined as 95-percentile*
It is further stipulated that of the 20, 10 or 5% of samples from designated waters which exceed the standard, none should do so by more than 50% (except for microbiological parameters, pH and dissolved oxygen) and that "consecutive water samples taken at statistically suitable intervals do not deviate from the relevant parametric values" (Article 5 of CEC 1976).
[a]No abnormal change in colour
[b]May be waived in the event of exceptional weather or geographical conditions
[c]Defined as wood, plastic articles, bottles, containers of glass, plastic, rubber or any other substance
[d]Should be absent.
[e]Applies to non-routine sampling prompted by visual or olfactory evidence of the presence of he substance
[f]There should be no film visible on the surface and no odour
[g]Reacting with methylene blue
[k]There should be no lasting foam

Annex 6. Designated Shellfish areas within the EMS (from FSA, 2005)

Production Area	Bed Name	Species	Class	Notes
Colne	Pyefleet Spit	*Crassostrea gigas*	B	
	Pyefleet Channel	*Cardium edule*	C	
	Brightlingsea	*Mytilus edulis*	B	1
	Brightlingsea	*C. gigas*	B	
West Mersea	Strood Channel	*Ostrea edulis*	B	
	Strood Channel	*C. gigas*	B	1
	The Nothe	*O. edulis*	B	
	Freeground	*O. edulis*	B	
	Tollesbury South	*O. edulis*	B	
	Tollesbury North	*O. edulis*	A	Seasonal classification 1 December to 30 June (reverts to B at other times)[c]
	Salcott	*O. edulis* *C. gigas*	B B	
	Little Ditch*	*C. gigas*	B	
Blackwater	Goldhanger	*C. gigas*	B	
	Thirslet Creek	*M. edulis*	B	
	Bench Head	*O. edulis*	B	1
	St Peters Flats	*O. edulis*	B	1
	Batchelor Spit	*O. edulis*	B	1
	Buxey Sands	*C. edule*	B	
	Dengie Flats	*C. edule*	A	1
	Ray Sands	*C. edule*	A	1
	The Nass	*O. edulis*	B	
Crouch	All beds	*O. edulis*	B	
	Althorne Creek	*M. edulis*	B	2
Roach	Paglesham Pool	*C. gigas*	A	Seasonal classification – 1 June to 31 August inclusive (reverts to B at other times)[c]
	Paglesham Pool	*Tapes philippinarum*	B	
	Paglesham Reach	*Ostrea edulis* *C. gigas*	B B	
	Quay Reach	*O. edulis*	B	
	Dunhopes	*O. edulis* *M. edulis*	B B	
	Pond Lays	*O. edulis* *Mercenaria mercenaria*	B B	
	Middleway	*C. gigas*	B	
	Devils Reach	*C. gigas*	B	
	Blackledge	*M. edulis*	B	
Thames Estuary Mid and NE	Maplin Sands	*C. edule*	B	3
	Foulness Sands	*C. edule*	B	3

1 Area classified at higher level, but showing marginal compliance.
2 Classification is provisional due to insufficient sample results, either in number or period of time covered
3 Area classified at lower level due to enforcement issues.
[c]. A 'Seasonal classification' may be considered when sample results indicate a clear and consistent period when the shellfish are of a quality to be harvested compared to the rest of the year. The period for which the seasonal classification applies is indicated.

*Also a designated relaying area for bivalve molluscs (Class B) - effective from 26 September 2003

Annex 7. Sediment Quality Guidelines

Interim marine sediment quality guidelines (ISQGs) and probable effect levels (PELs; dry weight) [1]: metals and organics (from Cole *et al.,* 1999)

Substance	ISQG	PEL
Inorganic ($mgkg^{-1}$)		
Arsenic	7.24	41.6
Cadmium	0.7	4.2
Chromium	52.3	160
Copper	18.7	108
Lead	30.2	112
Mercury	0.13	0.70
Nickel	15.9	42.8
Silver	0.73	1.77
Zinc	124	271
Organic (μgkg^{-1})		
Acenaphthene	6.71	88.9
Acenaphthylene	5.87	128
Anthracene	46.9	245
Aroclor 1254	63.3	709
Benz(a)anthracene	74.8	693
Benzo(a)pyrene	88.8	763
Chlordane	2.26	4.79
Chrysene	108	846
DDD[2]	1.22	7.81
DDE[2]	2.07	374
DDT[2]	1.19	4.77
Dibenz(a,h)anthracene	6.22	135
Dieldrin	0.71	4.30
Endrin	2.673	62.4[4]
Fluoranthene	113	1 494
Fluorene	21.2	144
Heptachlor epoxide	0.60[3]	2.74[4]
Lindane	0.32	0.99
2-Methylnaphthalene	20.2	201
Naphthalene	34.6	391
PCBs, Total	21.5	189
Phenanthrene	86.7	544
Pyrene	153	1 398
Toxaphene	1.5[3]	nd[5]

[1] from CCME, (1999)
[2] Sum of *p,p´* and *o,p´* isomers.
[3] Provisional; adoption of freshwater ISQG.
[4] Provisional; adoption of freshwater PEL.
[5] No PEL derived.

Annex 8. Examples Of Biological Effects Monitoring Techniques

Immunotoxicity Assays – these assay measures the immunocompetence of haemocytes from invertebrates, reflecting both the extent of exposure to immunotoxins and the general well-being of the test organism. Various immunological parameters (e.g. cell counts, generation and release of superoxide anions, phagocytosis, lysosomal enzyme activity) have proved useful in monitoring the status of shellfish in response to oil pollution and PAHs (Pipe *et al*, 2000; Raftos and Hutchinson, 1995; Dyrynda *et al*., 1998).

EROD (ethoxyresorufin-O-deethylase) is a marker for the activity of the mixed function oxidase (MFO) system, whose induction is usually associated with exposure to, and the detoxification of xenobiotics such as PAHs and PCBs. Occasionally these transformations may produce deleterious side effects due to the formation of carcinogenic or genotoxic compounds (e.g. formation of benzo(a) pyrene diol epoxide from the benzo(a) pyrene). Genotoxicicity assays (see below) may help to establish this possibility.

Metallothionein (MT) induction and associated changes in metal metabolism are specifically induced by metals and are sufficiently sensitive to be used to detect elevated levels of bioavailable metal in the field or arising from metals in discharges (e.g. Langston *et al*., 2002). The induction of MT protein, and associated metal-binding patterns can therefore be used to map spatial and temporal trends in biological responses to metals.

Genotoxicity-The Comet Assay - The single cell gel-electrophoresis (comet) assay is ideal for screening for possible genotoxicity associated with point-source and diffuse inputs to the system.

Respiratory physiology - Cardiac activity in bivalve molluscs and decapod crustaceans – Heart rate provides a general indication of the metabolic status of mussels and crabs. The CAPMON technique (Depledge and Anderson, 1990) permits the non-invasive, continuous monitoring of cardiac activity using infra-red sensors attached to the shell.

Tolerance Studies - More widespread investigations of community tolerance to establish their adaptation to contamination levels. Mapping the genetic composition of tolerant populations of individual species (*Hediste, Littorina* and others) in relation to induction of detoxification systems (such as EROD and metallothionein) should also be considered. This could add an interesting temporal dimension to biological monitoring – e.g. in determining the consequences of anticipated improvements in environmental quality (arising from planned schemes, standstill provisions of the Dangerous Substances Directive, or as required under the Habitats Directive to achieve Favourable Conservation Status).

Toxicity Studies on sensitive species - Toxicity has been studied in a relatively small number of species to date. It would be useful to examine subtle sublethal-effects in some of the less well represented and, perhaps, sensitive species. Also to include

Annex 8 (cont.)

sediment bioassays to look at growth and survival of juvenile bivalves. Compare responses in Essex Estuaries EMS biota with those elsewhere to look for signs of adaptation.

Biodiversity indices: quantitative techniques to assess community-level response to environmental degradation - *Species richness* - an indication of the number of taxa per unit area; -*Shannon-Weiner diversity index* (H^1) - expresses the relationships between the occurrence of species and the apportioning of individuals among those species (relative dominance); -*Pielou's evenness index (J)* compares the diversity of the data with its theoretical maximum, where all species would be equally abundant. Lower values would be associated with samples from sites numerically dominated by only one or two species, which is generally indicative of stressed communities; -*The Simpson index* relates the contribution made by each species to the total population. Although observing similar aspects of the dataset, this index is unrelated to the Shannon-Weiner diversity index. Higher values in the Simpson index equate to the presence of a few dominant species in the assemblage; -*Taxonomic distinctness* This index, which captures phylogenetic diversity rather than simple species richness, is more linked to functional diversity. Clarke and Warwick (2001) have refined and developed this index to pick out degraded locations; -*Abundance Biomass Comparison* - can be used to establish whether observed patterns resulted from the effects of natural environmental variables, or whether they were affected by some unnatural disturbance such as chemical pollution, organic enrichment from sewage, frequent bait digging etc. This method depends on the fact that the distributions of biomass among species in marine macrobenthic communities show a differential response to disturbance, which can be demonstrated by the comparison of k-dominance curves for abundance and biomass.

Various forms of multivariate statistical analysis of benthic communities and associated environmental variables are useful in examining spatial and temporal trends in communities in relation to contaminants (Warwick *et al*., 1998).

It is stressed that the above procedures have been selected primarily with regard to their ease of use, low cost and relevance to known environmental problems. Ideally, all components to the scheme need to be synchronised and run in tandem to achieve best value and to provide the most useful information on causal links and mechanisms. The results will assist environmental managers in identifying those consents and activities which most require attention and hopefully may help to decide on the best options for action.

Titles in the current series of Site Characterisations

Characterisation of the South West European Marine Sites: **The Fal and Helford cSAC**. Marine Biological Association of the United Kingdom occasional publication No. 8. pp 160. (2003)

Characterisation of the South West European Marine Sites: **Plymouth Sound and Estuaries cSAC, SPA**. Marine Biological Association of the United Kingdom occasional publication No. 9. pp 202. (2003)

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Characterisation of the South West European Marine Sites: **Summary Report**. Marine Biological Association of the United Kingdom occasional publication No.14. pp 112 (2003).

Characterisation of European Marine Sites. **Mersey Estuary SPA**. Marine Biological Association of the United Kingdom. Occasional Publications No. 18, 185pp. (2006).

Back cover: Abbotts Hall Managed Retreat Site: before (top) and after (bottom) breaching of sea defences
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