Charting Progress 2

Feeder Report: Clean and Safe Seas

United Kingdom Marine Monitoring & Assessment Strategy

UKMMAS

Charting Progress 2

Feeder Report: Clean and Safe Seas

Preface

Charting Progress 2 seeks to show the extent to which the UK Government and the Devolved Administrations are making progress towards their vision of achieving clean, healthy, safe, productive and biologically diverse oceans and seas as set out in *Safeguarding our Seas*, in 2002. It builds on *Charting Progress*, the first assessment of the UK Seas, published in 2005, and its delivery is the responsibility of the United Kingdom Marine Monitoring and Assessment Strategy community (UKMMAS) community which was set up in response to a recommendation in *Charting Progress* to provide a more coordinated approach to the assessment and monitoring of the state of the UK marine environment. UKMMAS created four evidence groups (the Healthy and Biologically Diverse Seas Evidence Group – HBDSEG; the Clean and Safe Seas Evidence Group – CSSEG; the Productive Seas Evidence Group – PSEG; and the Ocean Processes Evidence Group – OPEG) to collect the evidence needed to assess progress towards achieving the vision. Each evidence group has a broad membership across the academic and research communities as well as experts in government agencies and non-governmental organisations, and was tasked to produce a 'Feeder Report' assessing all the evidence available under its remit which could be used as source material for the evidence chapters in the main *Charting Progress 2* report.

This Feeder Report forms the CSSEG contribution to *Charting Progress 2* and provides an assessment of the extent to which UK seas are clean and safe. Key contributors are listed at Page 327.

The authors of this report are responsible for all the information, including all data, technical information and graphic material, contained within this report, including the referencing, correct use and accuracy of such information.

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Executive Summary

The first of Defra's State of the Seas Reports, *Charting Progress*, was published in 2005. The general picture that emerged from the evidence was mixed. The UK seas were noted to be productive and to support a wide range of fish, marine mammals, seabirds and other marine life. The open seas were generally not affected by pollution and the levels of monitored contaminants had decreased significantly. The main contamination problems identified were, in part, due to the legacy of the past and seen at higher levels in industrialized estuaries or other areas local to the historic activity. However, human activity had resulted in adverse changes to marine life and was continuing to do so. For example, widespread commercial fishing practices threatened many fish stocks by overexploitation and damaged sea floor areas. Evidence was also noted which suggested that marine ecosystems were beginning to be affected by climate change.

This report, *Charting Progress 2*, updates our knowledge regarding the marine environment around UK coasts to 2007. Data are compiled and, wherever possible, assessed under eight broad topics: hazardous substances, radioactivity, eutrophication, microbiological contamination, oil and chemical spills, litter, algal toxins and underwater noise. The requirement for monitoring in UK waters (including rivers) has increased as a result of the implementation of the EU Water Framework Directive. This also introduces a new range of environmental quality standards, in many cases more stringent (i.e. requiring lower environmental concentrations of contaminants) which may increase the number of exceedances in the future. The implementation of the EU Marine Strategy Framework Directive will also require the assessment of UK seas in relation to 'Good Environmental Status' by 2012, and the methods and criteria by which this will be assessed against a suite of status descriptors are currently under development.

Key findings under each of the topics above can be found in Section 2. For hazardous substances, concentrations of contaminants which have been subject to earlier controls are falling, but in some cases (polychlorinated biphenyls, compounds previously widely used in electrical transformers, for example) relatively slowly. Inputs via rivers and the atmosphere and from produced formation water from oil production facilities are reducing. For new chemicals, such as the brominated flame retardants the polybrominated diphenyl ethers, data are limited and assessment criteria needed in order to establish their significance as environmental contaminants have not been developed as yet. Inputs and exposure to radioactivity are well controlled, and well below internationally agreed standards. Overall, the eight regions were classified as non-problem areas in respect to eutrophication. There is a considerable monitoring effort in relation to microbiology, encompassing both shellfish and bathing waters and shellfish themselves. In 2007, 96% of bathing waters met the EU imperative standard. Also in 2007, 40% of sampled shellfish waters met the guideline value, a significantly more stringent requirement than for bathing waters. In 2007, there were 654 accidental discharges

of oil from ships and offshore platforms. This represented increases of 29% in discharges from ships and 13% in discharges from platforms over 2006. The amount of litter in our seas and on our beaches continues to be a cause for concern, and there is little or no sign of reductions. Average litter densities on UK beaches remain high, at over 2000 litter items per kilometre surveyed. Comprehensive algal toxin monitoring programmes, compliant with EU legislation, are undertaken in all regions of the United Kingdom. These provide a satisfactory level of human health protection for consumers of shellfish. Underwater noise is a topic of concern, but currently there is no monitoring programme which would allow its effects to be assessed.

SECTION 1
INTRODUCTION

1.1 Introduction

One of the recommendations arising from the first UK-wide assessment of the state of the UK seas, *Charting Progress*, was to set up the United Kingdom Marine Monitoring and Assessment Strategy (UKMMAS) which would provide a more coordinated and integrated approach to monitoring. The overall aim of UKMMAS is to shape the UK's capability, within national and international waters, to provide and respond to, within a changing climate, the evidence required for sustainable development within a clean, safe, healthy, productive and biologically diverse marine ecosystem and, within one generation, to make a real difference.

UKMMAS consists of a high level, policy-lead Marine Assessment Policy Committee (MAPC). This is supported by a technical Marine Assessment and Reporting Group (MARG); which oversees the work of a number of initiatives. These include groups to investigate and report on objectives for the marine environment; preparation of integrated assessments; preparation of protocols and a monitoring manual; data archiving via the Marine Data and Information Partnership (MDIP) and the Marine Environment Data Action Group (MEDAG) now combined as the Marine Environmental Data Information Network (MEDIN); and three Evidence Groups to collate quality assured data on the themes of 'Clean and Safe', 'Healthy and Biologically Diverse' and 'Productive' seas.

What we mean by 'clean and safe' is that the results of monitoring studies, for example, within the UK Clean Seas Environmental Monitoring Programme (CSEMP) and other programmes, are assessed against thresholds and standards and found to be below levels at which harm or damage to marine life and human consumers will occur. These thresholds and standards are generally set at international level (e.g. in accordance with the OSPAR Convention or in EU Directives) or nationally. In some cases, such as for underwater noise and litter, these standards are not yet well developed.

The first of the State of the Seas Reports, *Charting Progress*, was published in 2005. The general picture that emerged from the evidence was mixed. UK seas were noted to be productive and to support a wide range of fish, marine mammals, seabirds and other marine life. The open seas were identified as generally not affected by pollution and the levels of monitored contaminants had decreased significantly. The main contamination problems identified were, in part, due to the legacy of the past and seen at higher levels in industrialized estuaries or other areas local to the historic activity. However, human activity had resulted in adverse changes to marine life and was continuing to do so. For example, widespread commercial fishing practices threatened many fish stocks by over-exploitation and damaged sea floor areas. Evidence was also noted which suggested that marine ecosystems were beginning to be affected by climate change.

The Clean and Safe Seas Evidence Group (CSSEG) focuses particularly on the direct effect on the marine environment of chemicals, and on pathogens, and their likely impact on human health. The present report, entitled the Assessment of the Clean and Safe Status of UK Seas, provides an updated assessment of the 'clean and safe' state of the seas across the whole of the UK Continental Shelf based on the status of eight biogeographical regions around the UK. The scope of the assessment includes hazardous substances, radioactivity, eutrophication, microbiological contamination, oil and chemical spills, litter, algal toxins and a

limited amount of information on noise in the marine environment. Findings from the various topic areas addressed by CSSEG are summarised in order to build up an overall assessment of the clean and safe status of UK seas up to 2008. As the quantity of evidence considered varies widely between these topics, the topic sections are similarly variable in length.

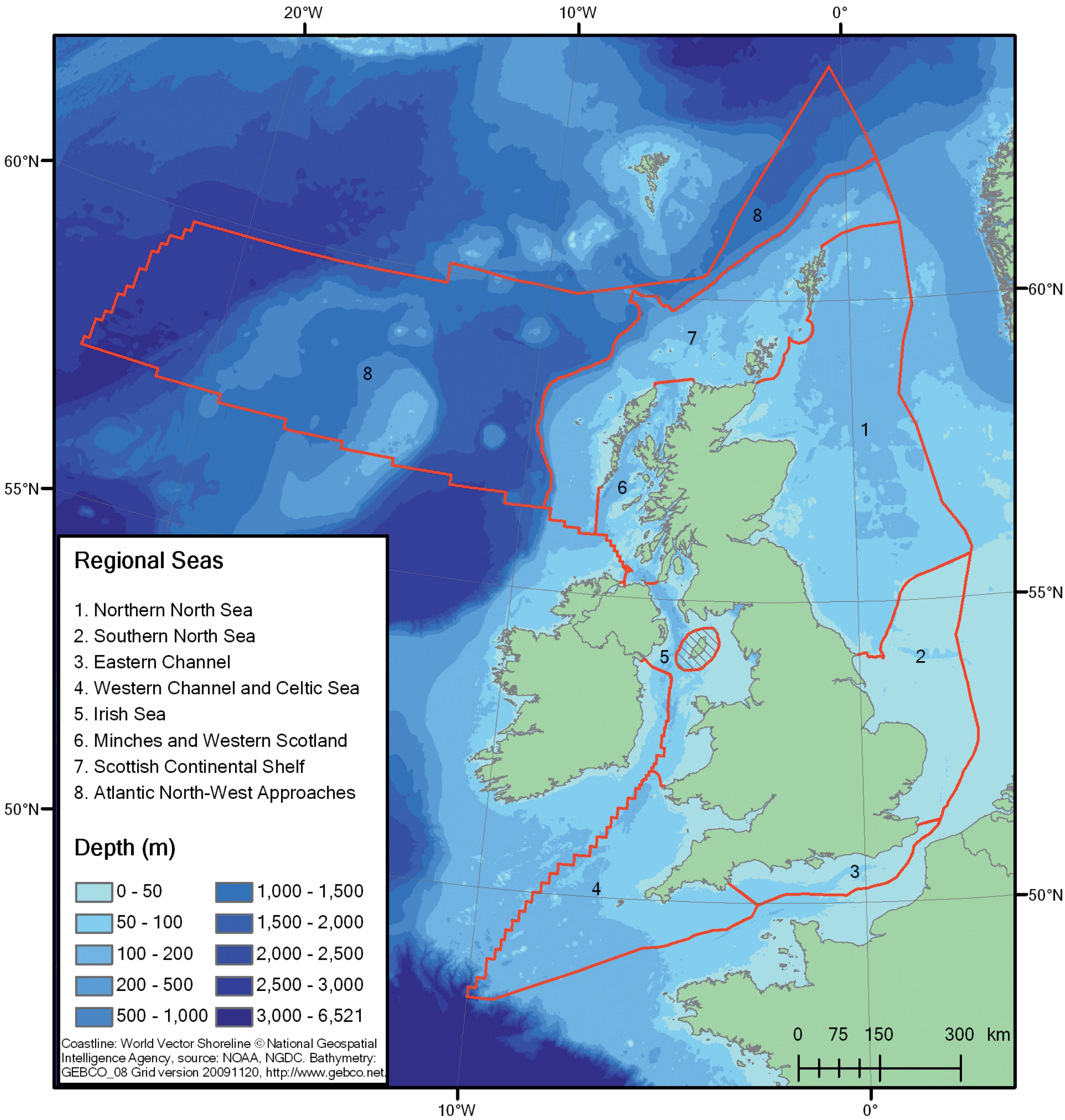
The assessments of current status within the report focus on the state of the topics (individually measured parameters, e.g. specific hazardous substances) in the eight biogeographical regions which make up the UK seas (see Figure 1.1). Regional assessments were made, where possible, for each of the eight biogeographical regions around the UK by using a 'traffic light' system. These were derived from regional waters classifications originally developed by the Joint Nature Conservation Committee (JNCC). The 'traffic light' system is based on three colours: red (many problems), amber (some problems) and green (few or no problems). The underlying criteria used to define the transition points between those three colours vary, and are explained in more detail within each topic section. The individual pieces of evidence obtained by monitoring in each Region are evaluated against standards or criteria which allow such an assessment. Some standards/criteria are well established (e.g. set in EU Directives or by OSPAR), some are newly developed (e.g. assessment criteria used to assess the status of various ecosystem components) and some are based on expert judgement. In some instances, blue has also been used to illustrate values close to background. For the EU Marine Strategy Framework Directive, member states are required to assess whether their waters attain 'Good Environmental Status' (GES) for a range of descriptors. A process has begun to identify methods and criteria which can be used for this purpose for each of the defined descriptors. However, it is important to note that, until that process is complete, there are still significant uncertainties associated with the quality levels that are to be established to determine GES under the Marine Strategy Framework Directive. It is also, therefore, unclear how the present assessments will relate to future assessment of GES under the Marine Strategy Framework Directive.

This type of 'traffic light' assessment could not be used for all parameters and therefore it was decided to assess those more generally. The parameters that could not be assessed in the regional/traffic light manner were:

- Oil and chemical spills, as the available data are based on the number of incidents rather than volumes lost and so it is not possible to assess impacts
- Noise, as there is not enough information available to assess the impact and amount of noise on a regional scale due to a lack of systematic monitoring
- Algal toxins, as these were assessed on the basis of monitoring undertaken to protect human health rather than the occurrence of harmful algal blooms. The toxic phytoplankton species that produce marine biotoxins occur naturally and are regularly reported to be present in phytoplankton assemblages
- Litter, as this was assessed on the basis of observed trends, as no objective criteria have been set for acceptability.

This Feeder Report provides evidence from the Clean and Safe Seas Evidence Group (CSSEG) to Defra's state of the seas report, *Charting Progress 2*. In parallel, similar contributions are being prepared by the Healthy and Biologically Diverse Seas Evidence Group (HBDSEG) and the Productive Seas Evidence Group (PSEG)

Figure 1.1 Charting Progress 2 *Regional Sea boundaries.*



in line with Defra's vision for UK seas. These feeder reports were subject to peer review and presented at a stakeholder workshop, following which modifications were made to the text.

This report presents information on the current status of marine environmental quality in the waters around the UK. Within the peer review process for *Charting Progress*, it was suggested that a review of relevant R&D undertaken in relation to the topics covered in the Marine Environmental Quality feeder report should have been conducted so as to more fully utilise the academic studies lying within the timeline. For the present CSSEG feeder report, such a review was conducted under contract to Defra by staff at the University of East Anglia, and their

findings were passed to the topic leaders of each chapter for inclusion of relevant material. At the time of *Charting Progress*, it was recognised that there was evidence that the marine ecosystem was being altered by climate change. In preparing each topic section of this report, potential impacts of future climate change within the topic area have been considered, to the extent that these are currently understood. These are not included within the feeder report, but will be compiled and included as a separate section within *Charting Progress 2*.

The scale at which and the criteria by which the assessments have been undertaken varies by topic.

1.2 Significant developments since *Charting Progress*

The increasing need for information on the UK marine environment is being in driven in part by a number of significant policy developments that have arisen since *Charting Progress* including the Marine and Coastal Access Act 2009, the Marine (Scotland) Act 2010, and similar legislation is being developed for Northern Ireland. All of these bills include provisions for a marine planning system and for a network of Marine Conservation Zones and Marine Protected Areas: information on marine ecosystems will be vital in order to support these initiatives. Another hugely significant development since *Charting Progress* has been the adoption of the EU Water Framework Directive and the implementation of new monitoring programmes to provide a holistic assessment of water body status. Monitoring was started in December 2006 with the first assessments delivered in spring 2009. Further offshore, the EU Marine Strategy Framework Directive adopted in 2008 requires that the UK takes *...the necessary measures to achieve or maintain Good Environmental Status in the marine environment by the year 2020 at the latest*, although an initial assessment of the current environmental status is required by 2012 and will rely heavily on the information gathered for this report. Along with the need for information to support assessments due as part of other obligations (e.g. the OSPAR Quality Status Report 2010), it becomes clear that the rapid expansion in the number of policy drivers even since *Charting Progress* has increased the need for a comprehensive assessment such as this one.

Another major change has been the general improvement in water quality in most UK estuaries over the past five years. These changes have largely been the result of major legislative obligations to clean-up catchments and water courses under various EU Directives, including the Dangerous Substances Directive, the Urban Waste Water Treatment Directive, the Nitrates Directive, the Shellfish Waters Directive and the Bathing Waters Directive.

Key legislation and other drivers in this area include:

National

- The Food and Environment Protection Act, 1985

International

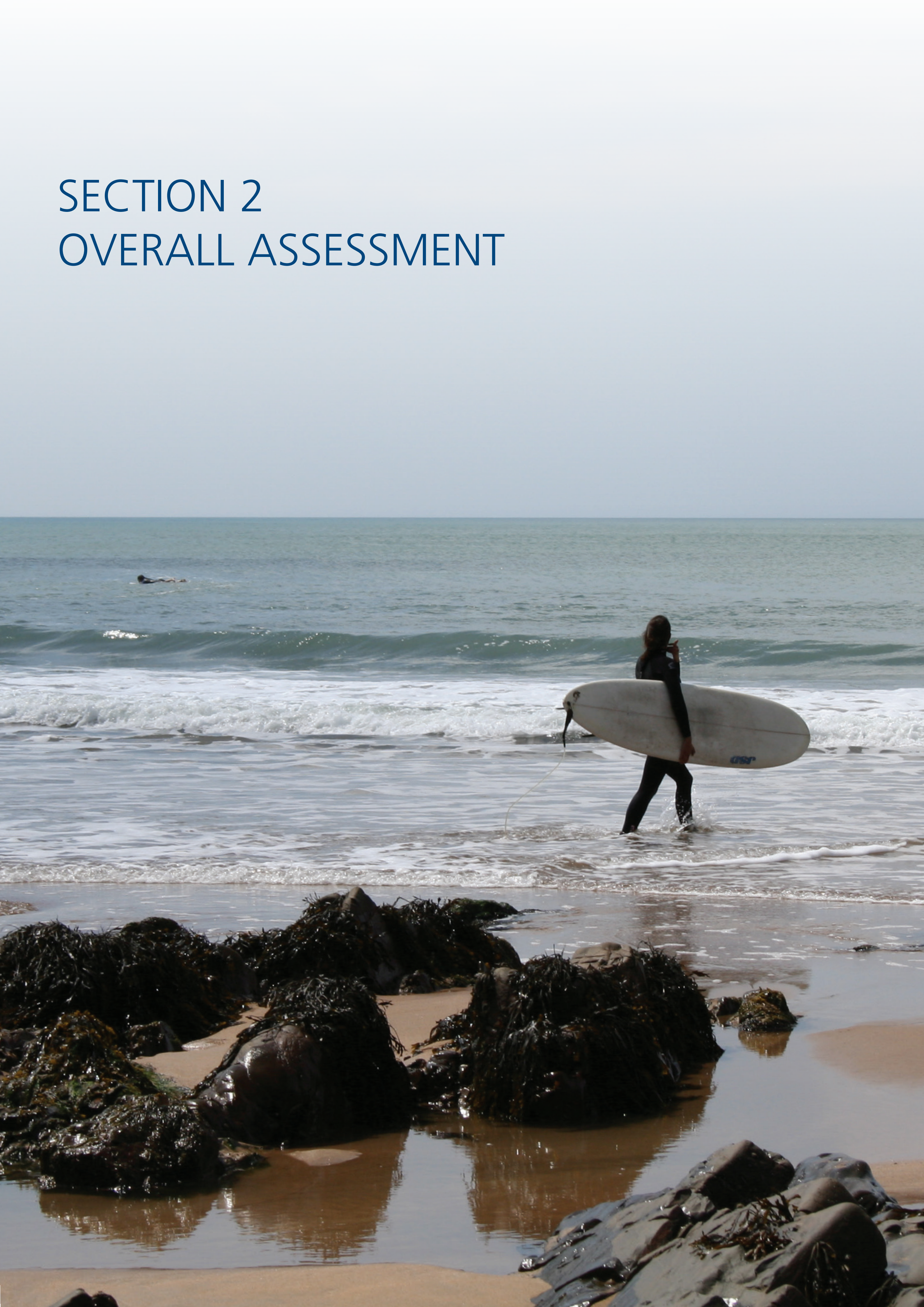
- EU Water Framework Directive
- EU Marine Strategy Framework Directive
- EU Shellfish Waters Directive
- EU Shellfish Hygiene Directive
- EU Bathing Waters Directive
- EU Dangerous Substances Directive
- EU Habitats Directive

- Restriction of Hazardous Substances in Electrical and Electronic Equipment Directive (2002/95/EC)
- The Nitrates Directive (1991/676/EEC)
- The Integrated Pollution Prevention and Control Directive (2008/1/EC)
- The Urban Waste Water Treatment Directive (1991/271/EEC)
- EC Commission Regulation (1907/2006/EC) on the registration, evaluation, authorisation, restriction of chemicals
- EC Commission Regulation (2006/1881/EC) which sets maximum toxic equivalent (TEQ) concentrations for the sum of dioxins, furans and dioxin-like CBs in the muscle meat of fish and fishery products
- EC Commission Regulation (2005/208/EC) which sets maximum levels for benzo[*a*]pyrene as a PAH marker compound in seafood
- OSPAR Hazardous Substances Strategy
- OSPAR Strategy for Radioactive Substances
- International Convention for the Prevention of Pollution from Ships, 1973
- Stockholm Convention on Persistent Organic Pollutants
- UNECE Convention on Long-range Transboundary Air Pollution
- Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas

The main aim of CSSEG is to oversee the UK national Clean Seas Environmental Monitoring Programme (CSEMP) and to ensure that it is conducted in an effective and co-ordinated manner. It includes representatives from the following organisations:

- Agri-Food and Biosciences Institute, Northern Ireland (afbini)
- British Oceanographic Data Centre (BODC)
- Centre for Environment, Fisheries and Aquaculture Science (Cefas)
- Department for Environment, Food and Rural Affairs (Defra)
- Environment Agency (EA)
- Marine Scotland Science
- Food Standards Agency (FSA)
- Marine Conservation Society (MCS)
- Northern Ireland Environment Agency (NIEA)
- Scottish Environment Protection Agency (SEPA)

SECTION 2
OVERALL ASSESSMENT



2.1 Introduction

The UK Government's first report on marine stewardship, *Safeguarding Our Seas,* outlined a vision of clean, healthy, safe, productive and biologically diverse oceans and seas. In 2005, the first full assessment of the state of UK seas, *Charting Progress*, showed that recent regulations and recommendations had led to a marked fall in the amounts of measured contaminants reaching the sea. It also found that levels of contaminants in water, sediments and biota were generally low, apart from in a few estuaries that had been heavily contaminated by historical industrial and domestic discharges.

This chapter builds upon *Charting Progress*, showing what progress we have made towards the vision of clean and safe seas. Based on the Feeder Report produced by the Clean and Safe Seas Evidence Group, it assesses the impacts of six major components associated with the cleanliness of the sea (hazardous substances, radioactivity, eutrophication, oil and chemical spills, litter, and underwater noise). It also assesses the safety of seafood to consumers and bathing waters to swimmers through reporting on algal toxins and microbiological contamination.

For this assessment, we used criteria developed in accordance with the OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. We highlight where relevant assessment criteria are lacking.

Assessments of the potential impact of individual chemicals use an environmental quality standard (EQS), derived from the concentration at which laboratory data suggest that the chemical could be harmful. Most of these standards are for water. For sediments and biota, we have data on background assessment concentrations (BACs) and environmental assessment criteria (EACs), the latter based on toxicological information and which often also add a margin of safety to give a very conservative estimate of potential harm. Alternative approaches involve combining modelling, laboratory and field toxicity data to generate a range of concentrations within which potential effects may occur. For example Effects Range-Low (ERL) and Effects Range-Median (ERM) concentrations. This approach, first developed for the United States Environmental Protection Agency, is now widely used around the world.

We have outlined the specific approach taken within each section, and made individual assessments for each of the eight biogeographical areas (see Figure 1.1 on Page 4). The chapter concludes with a table summarising the outcome of the assessments by topic and region.

Sampling off Stonehaven, North-East Scotland

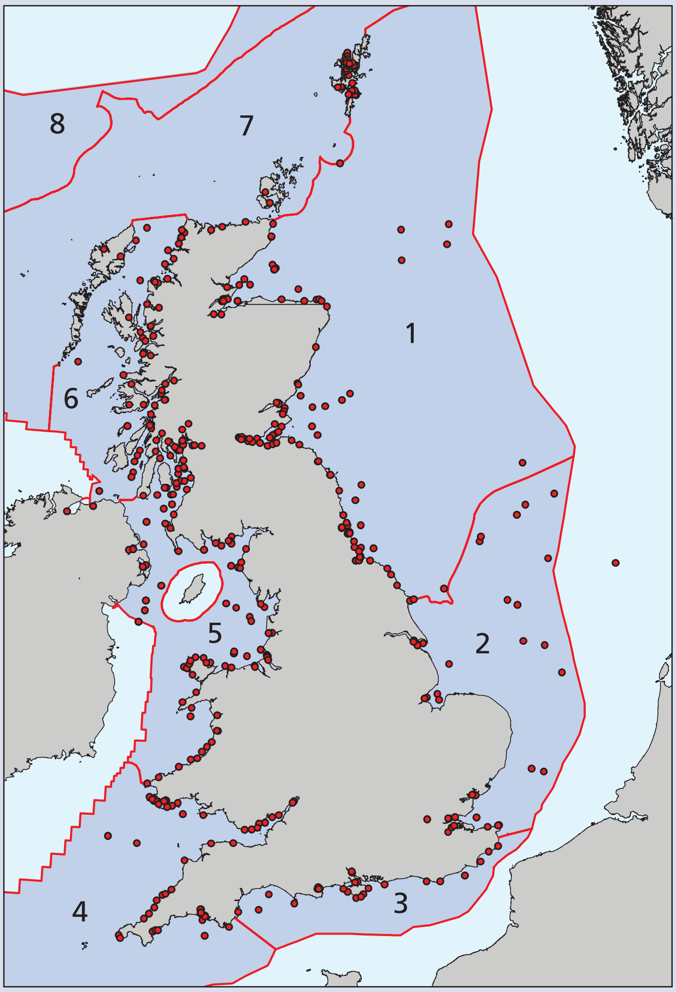
National Monitoring for Environmental Quality

The main programme for monitoring the status of contaminants in UK waters is the *Clean Seas Environmental Monitoring Programme (CSEMP)*. CSEMP measures the concentrations of specific chemicals which are persistent, toxic and have the ability to accumulate in food chains, at almost 500 sites around the UK (see map). Concentrations of nutrients are also measured at some locations. The CSEMP measures the levels of biological effects in animals which can reflect the influence of many of the potentially hazardous chemicals in the environment. Finally, it assesses the ecology of the benthic communities at some of these sites to separate out the effects of hazardous substances from other causes of change. The results are used to report on progress made in delivering the vision of clean and safe seas, and are stored in the MERMAN database and sent annually to the International Council for the Exploration of the Sea (ICES) to fulfil the UK commitment under the OSPAR Convention.

Data on the loads of contaminants entering our seas from river basins are submitted to OSPAR under their *Comprehensive Study on Riverine Inputs and Direct Discharges (RID)* programme. Data on contaminants in air are also submitted to OSPAR under their *Comprehensive Atmospheric Monitoring Programme (CAMP)*.

To dispose of dredged material, to remove seabed sediments for use as aggregate or to build structures at sea all require a licence or consent. Several organisations check regularly for compliance at key disposal sites, particularly those receiving large quantities of dredged material or those close to sensitive or conservation areas. This monitoring, conducted mainly by government agencies, assesses the concentration of contaminants and the presence of fauna of various species in the sediments, as well as the physical distribution and transport of material. The findings are published regularly in journal articles and reports.

CSEMP monitoring sites (●) around the UK in 2007. Crown copyright 2010: permission granted by Cefas



The *Cetacean Strandings Investigation Programme* undertakes post-mortems of stranded and accidently caught whales and dolphins in order to establish causes of death. It also provides a bank of tissue samples, some of which are analysed to determine contaminant concentrations. These data provide information on variations in the levels of contamination between species, and on spatial and temporal differences in the levels of contaminants in harbour porpoises.

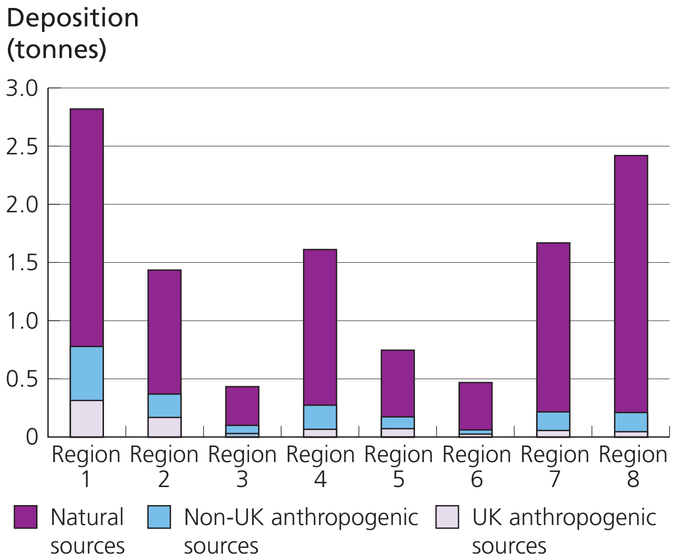
Clean Seas

2.2 Hazardous substances

Hazardous substances enter the marine environment from natural sources (e.g. polycyclic aromatic hydrocarbons (PAHs) from oil seeps, volcanoes and forest fires) and as a result of human activities, and reach the sea via direct discharges, through rivers and estuaries or via the atmosphere. The potential hazard associated with the different substances depends on their individual properties and behaviour following release. As explained on Page 8, we have identified concentration thresholds above which these substances could be toxic both to marine organisms and to human consumers of seafood. The Clean Seas Environmental Monitoring Programme (see box on previous page) directly monitors a limited number of hazardous chemicals, selected on the basis of a risk assessment and subject to there being agreed methodological guidelines, quality assurance procedures and assessment criteria available. For other contaminants, such as tributyltin (TBT), we can assess impacts by studying the biological effects they cause.

Data are still sparse at the regional scale; we may have too few sampling sites to characterise a region with high confidence. However, a major development since *Charting Progress* has been a redesign of the hazardous substances monitoring programme to make it more effective at detecting changes over time. For the first few years, the old and new programmes have run side to by side to provide continuity.

Figure 2.1 Deposition of cadmium to sea areas surrounding the UK in 2006. © AEA Technology.



2.2.1 Inputs of hazardous substances

The downward trend in inputs of contaminants over time reported in *Charting Progress* for rivers, sewage works and industrial discharges has continued for mercury, cadmium and lindane to both the Celtic Sea and the North Sea, but for polychlorinated biphenyls (PCBs) concentrations have stabilised.

Between 1990 and 2007, anthropogenic emissions of cadmium to the atmosphere decreased by 84%, of copper by 57%, of lead by 96%, of zinc by 55% and of mercury by 80%. Figure 2.1 shows deposition of cadmium to sea areas surrounding the UK in 2006.

While industrial change has caused some of these decreases, improved abatement at the remaining sources has also contributed.

Produced water from offshore oil and gas platforms contains natural toxic aromatic hydrocarbons which are a component of oil as well as treatment chemicals. Discharges of oil in produced water fell by about 25% during the

period 2002 to 2006, largely due to reductions in the volume of water discharged. The oil content of the discharged water remained constant at about 20 ppm until 2006, but reduced to about 15 ppm in 2007 thanks to improved produced water management. This brought the overall reduction in the amount of oil discharged from 2002-2008 to close to 50% (see Figure 2.2). This is a good example of effective regulation, in which the UK exceeded the reduction required by OSPAR.

Emissions of PAHs to the atmosphere have decreased by 84% since 1990. In 2007, the largest source of PAHs was road transport combustion, followed by domestic combustion (Figure 2.3). Twelve years earlier, the major source was the aluminium smelting industry, which contributed around 50%. Since then, thanks to improved practices, this industry is now responsible for only 1% of total PAH emissions.

2.2.2 Seawater

The evidence from seawater measurements is very encouraging. Inputs and concentrations of the most commonly monitored contaminants in seawater have fallen since *Charting Progress* as a result of earlier controls placed upon their use and are generally below UK EQS limits (for more information on UK EQSs, see www.environment-agency.gov.uk/research/planning/40295.aspx).

In addition, we found virtually no toxicological hazard from metals in water samples analysed for the EU Directives on Dangerous Substances (mainly in estuarine waters) and Shellfish Waters (mainly in coastal waters); nearly 99% of metal concentrations were below the UK EQS values in 2007, although 6% of copper concentrations exceeded the EQS (e.g. see Figure 2.4). Biological water quality (assessed using the percentage of

Figure 2.2 Oil discharged in produced water by the offshore oil and gas industry, 2001–2008. © Crown copyright 2010: permission granted by Cefas.

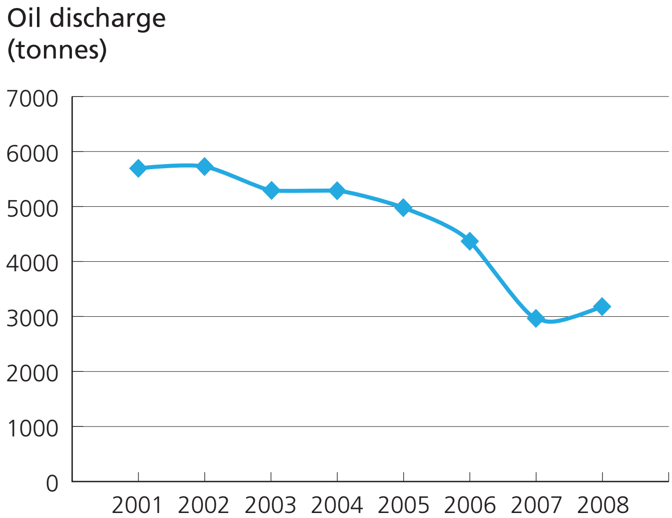
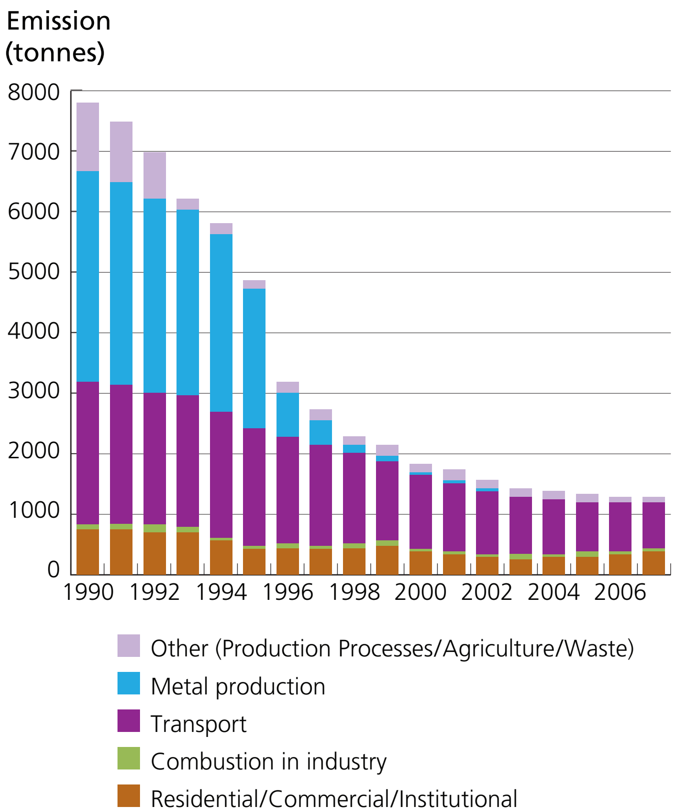


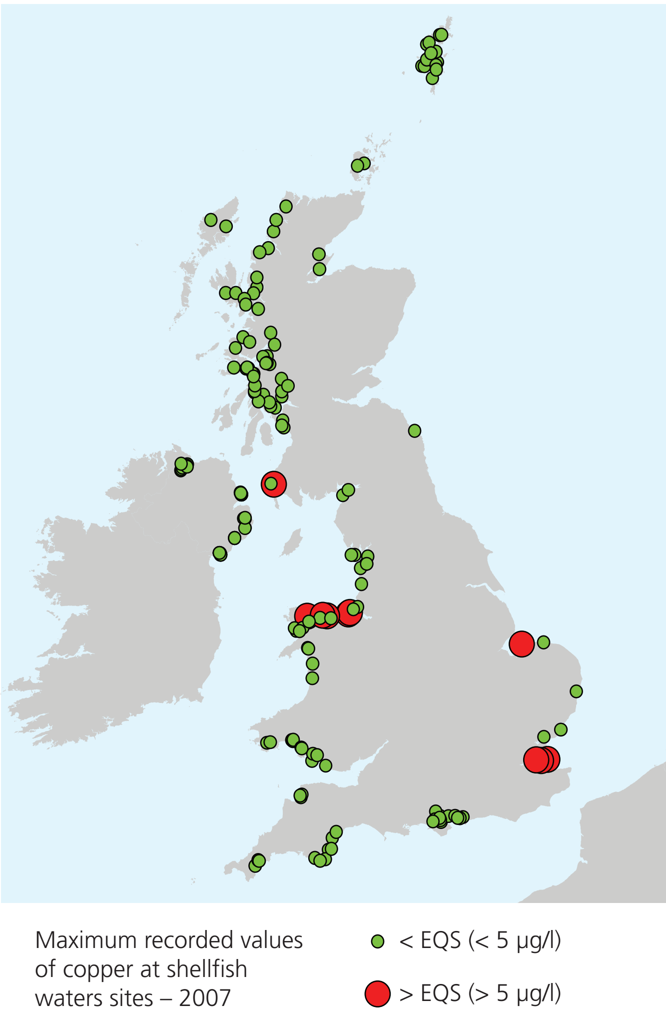
Figure 2.3 Emissions of PAH to the UK atmosphere, 1990–2007. © AEA Technology.



oyster embryos that develop successfully in the water samples) is good or very good everywhere studied, except in the Tees estuary close to industrial discharges.

However, because concentrations of many contaminants in water are both low and very variable, the UK marine monitoring programme under CSEMP focuses on measuring contaminant levels in biota and sediments, where accumulation means that concentrations are generally higher and less variable. This increases the power of the monitoring programme to observe changes in concentrations over time.

Figure 2.4 Copper concentrations in filtered water samples relative to the UK EQS. © Crown copyright, Marine Scotland, 2010.



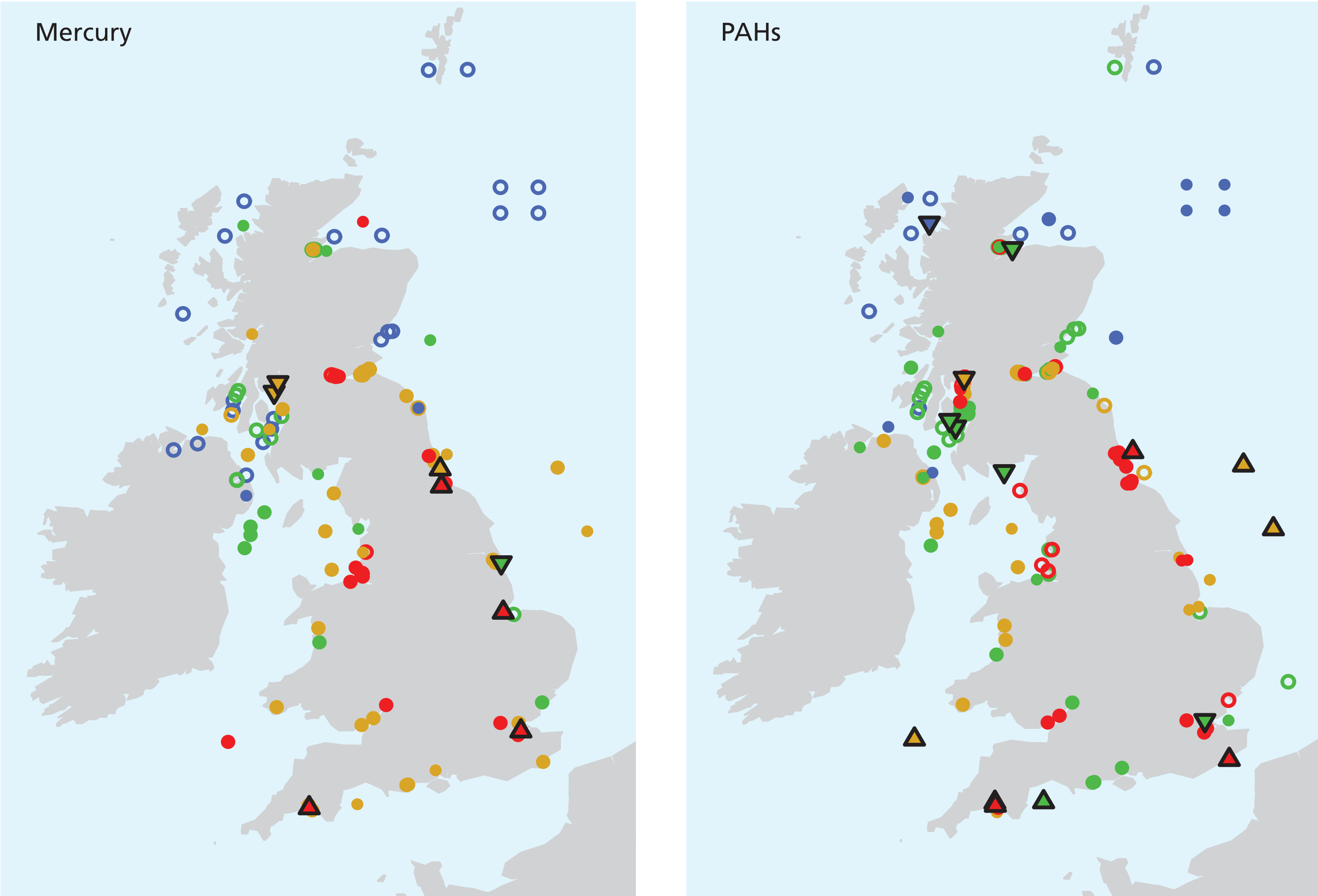
2.2.3 Sediments

Analysis of contaminants in sediments reveals more clearly where there are problems, particularly in estuaries that have been heavily industrialised over time. While metal concentrations in sediments are generally lower in Scotland and the western Irish Sea, they are higher in England and Wales, with a number of industrialised estuaries, such as the Tees, Tyne, Thames, Severn and Mersey (in the case of mercury), showing levels that are high enough to have potential toxicological effects (Figure 2.5). Similarly, sediment PAH concentrations were high in the Tees, Tyne and Wear estuaries, for example, and hence potentially toxic to sediment-dwelling organisms (Figure 2.5). The capacity of the sediment to support biota is generally good at all locations studied, except in parts of the Tees, Wear and Thames estuaries where the presence of adverse biological effects may be linked to the high PAH concentrations.

There has been no significant overall trend in the concentrations of metals in sediments since *Charting Progress*, although both upward and downward trends can be seen at specific locations for all eight metals (cadmium, mercury, lead, arsenic, chromium, copper, nickel, zinc) determined. However, if metal inputs from rivers, sewage discharges and industry continue to decline, we would expect future assessments to find decreasing concentrations in the sediments where anthropogenic inputs have exceeded natural sources.

Concentrations of PCBs, were also determined in surface sediments. They are present in the environment as a result of widespread historical

Figure 2.5 Normalised mercury and PAH concentrations in sediment. © Crown copyright, Marine Scotland, 2010.



Blue, green and amber symbols indicate concentrations below the BAC, ERL and ERM concentrations respectively; red symbols indicate concentrations above the ERM.

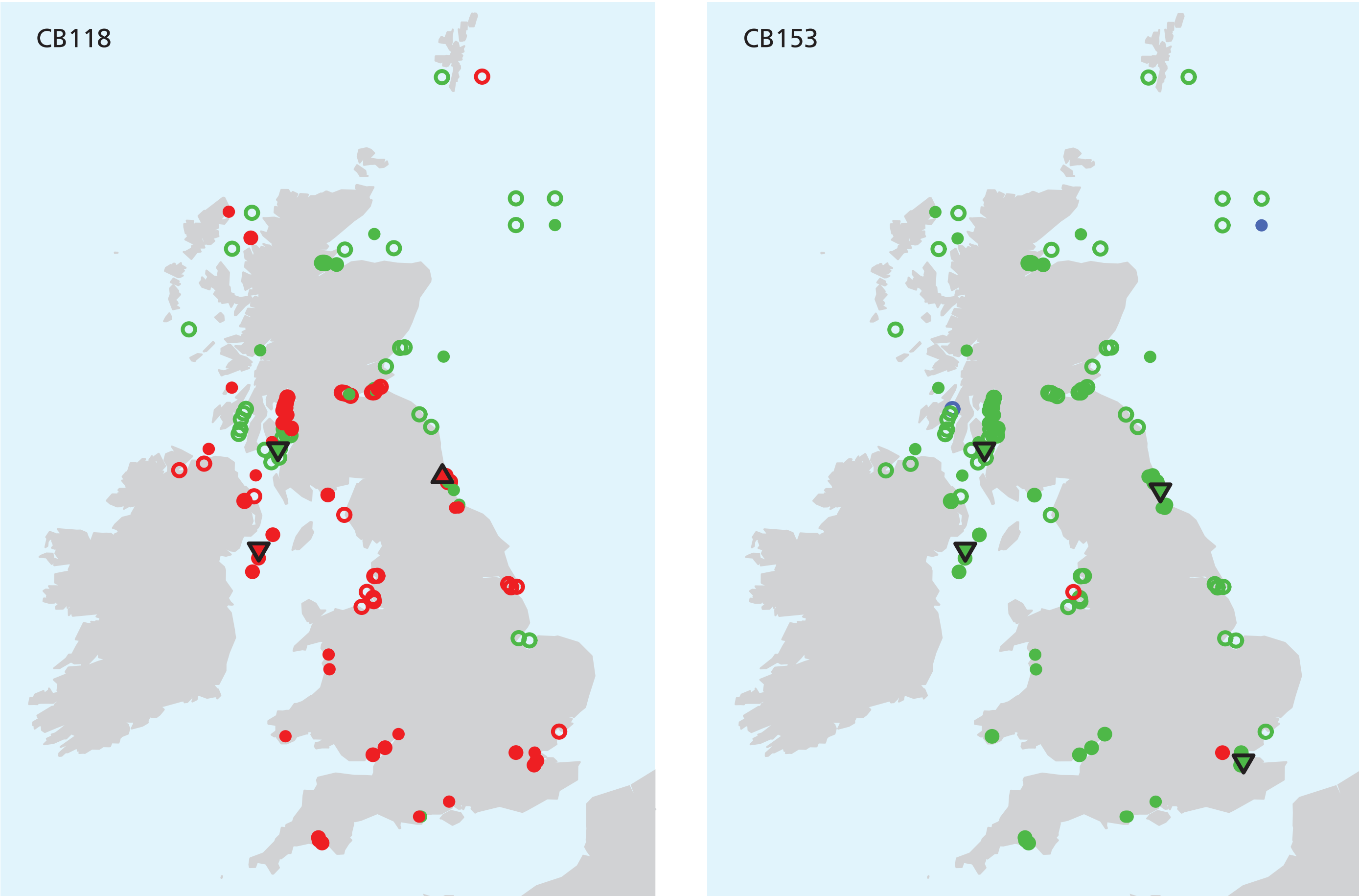
Key to Figures 4.5, 4.6 and 4.7

- ● *5+ years of data, no trend*
- ▲ *5+ years of data, upward trend*
- ▼ *5+ years of data, downward trend*
- • *3–4 years of data, trend not investigated*
- ○ *1–2 years of data, insufficient data to assess trend*

use of these products, mainly in electrical transformers. In particular, we found that concentrations of the most toxic congener included in the analyses are above the EAC in most areas (CB118: Figure 2.6). This is significant because CB118 can affect neurological, immunological and reproductive processes in marine biota and humans. Generally speaking, we found the lowest concentrations of CBs at Scottish offshore sites, and the highest around south-eastern England (CB153: Figure 2.6).

The few temporal trends we saw were mostly downward, although in many instances there was no apparent trend. Although the ban on new uses of PCBs was put in place in 1981, these compounds are very persistent in the environment and significant falls in environmental concentrations may take decades.

Figure 2.6 Normalised CB118 and CB153 concentrations in sediment. © Crown copyright, Marine Scotland, 2010.



Blue and green symbols indicate concentrations below the BAC or EAC respectively; red symbols indicate concentrations above the EAC. Key to symbols on Page 13.

CB congeners

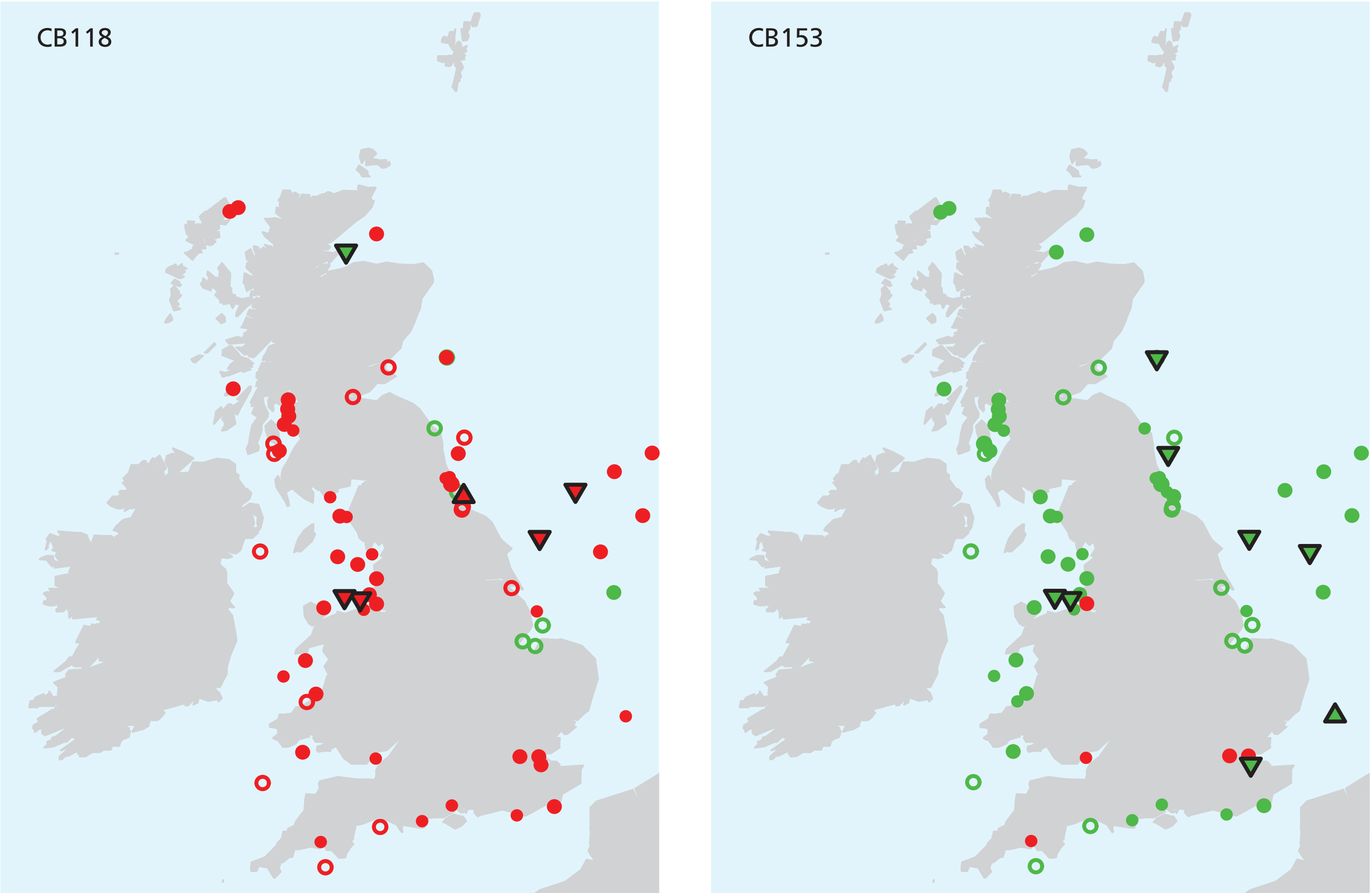
For this assessment we used the ICES7 CBs, selected by the ICES Marine Chemistry Working Group so as to facilitate comparison between different studies in which different sets of CB congeners are determined. If this minimum set of congeners is included in all studies, as is generally the case, then their sum can be used for comparative purposes.

The seven CB congeners are: CB28, CB52, CB101, CB118, CB138, CB153 and CB180.

2.2.4 Biota

In fish and shellfish, we found the highest concentrations of contaminants in industrialised estuaries. Mercury levels in fish flesh are high in the Mersey and Thames estuaries, and lead concentrations in fish liver are similarly high in many estuaries including the Forth, Tyne, Tees, Wear, Severn, Mersey and Bann as well as in a few other coastal areas. Cadmium, mercury and lead concentrations were slightly elevated in mussels from some industrialized estuaries: Dee, Humber and Thames for cadmium; Thames and Mersey for mercury; Tyne, Tees and Forth for lead. Silver concentrations were higher in mussels from the Severn and Thames estuaries than

Figure 2.7 Normalised CB118 and CB153 concentrations in fish liver. © Crown copyright, Marine Scotland, 2010.



Green symbols indicate concentrations below the EAC[passive]; red symbols indicate concentrations above the EAC[passive]. There were no concentrations below the BAC. EACs for CBs in sediment are expressed for sediment of 2.5% organic carbon. It is possible to calculate lipid-normalised concentrations of CBs in fish liver in equilibrium with sediment containing CB concentrations equal to the EACs in sediment. These so-called EAC[passive] values are used as the green/red boundary for CBs in biota. Key to symbols on Page 13.

elsewhere. Historically, the major inputs of silver were due to its widespread use in photography, which should be declining with the growth in use of digital cameras. However, the increasing use of silver nanoparticles as an antimicrobial agent may result in increased inputs. Although these metal concentrations are higher than background values, none pose a risk to human health because, in general, the shellfish tested were not from commercially harvested beds.

Although concentrations of CB138, CB153 and CB180 in fish liver are below the respective EACs, those of the more toxic CB118 are above the EAC and thus potentially toxic to the fish (see Figure 2.7). Fish liver is not eaten in the UK, and fish liver oil for use as a dietary supplement is cleaned-up during processing to reduce CB levels. We found the lowest concentrations in Scotland (Region 6) and highest in eastern England (Region 2). We found high levels of ethoxyresorufin-*O*-deethylase (EROD) enzyme activity in fish liver, which reflects exposure to contaminants such as dioxins, furans, planar CBs and PAHs, at sites in the North Sea and Liverpool Bay and at two historic sewage disposal sites close to the Scottish east coast.

Figure 2.8 *Site classifications for mussels, assessed for individual PAH compounds. © Crown copyright 2010: permission granted by Cefas.*

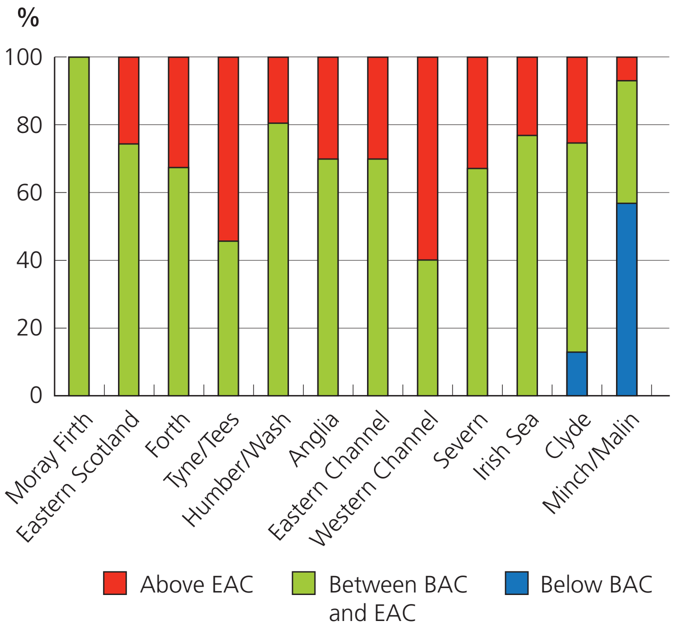
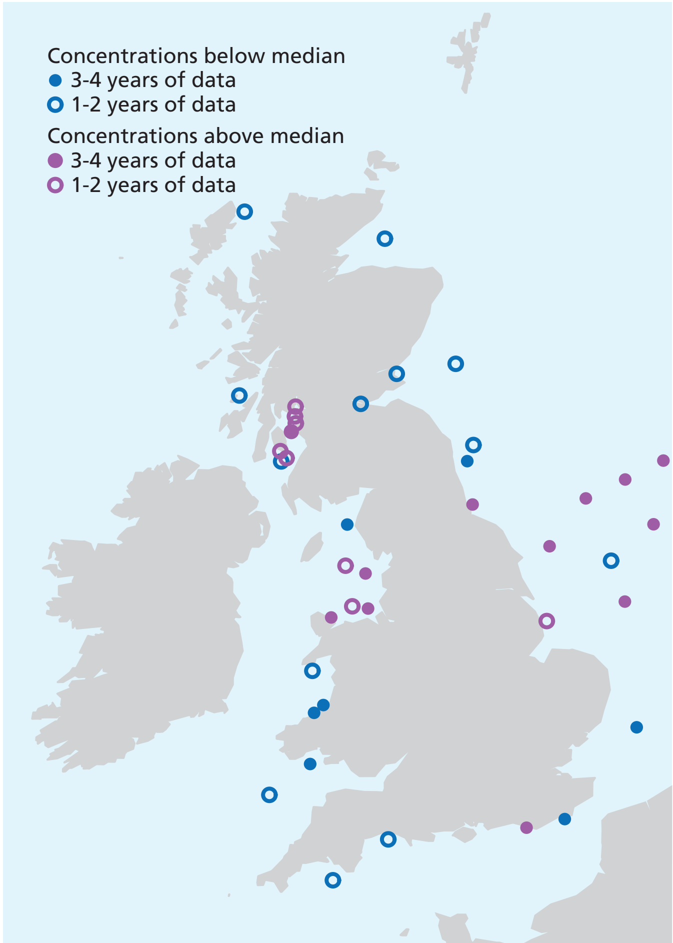


Figure 2.9 *BDE47 concentrations in fish liver. © Crown copyright, Marine Scotland, 2010.*



PAHs are potentially dangerous to fish (and humans) as, when metabolised, some PAH compounds can form potentially carcinogenic compounds that can bind to DNA. We found little change in the levels of these DNA adducts in the fish from some industrialised estuaries since *Charting Progress*, when high concentrations were reported, suggesting that marine organisms are still at risk due to PAH contamination at these locations. PAH concentrations in mussels are illustrated in Figure 2.8.

With their introduction into the formal monitoring programmes, more data are now available for the polybrominated diphenyl ethers (PBDEs), than was the case for *Charting Progress* (see Figure 2.9). However, we do not have enough information to identify trends over time other than in harbour porpoises, nor have assessment criteria yet been developed within OSPAR. We found the highest concentrations in fish in industrialised estuaries, including the Clyde, Tees and Humber, and the lowest off the Scottish coast, in the Western Channel and off eastern England. In harbour porpoises from UK waters, a rapidly rising trend in blubber concentrations of the brominated flame retardant hexabromocyclododecane (HBCD) after 2001, has been reversed since 2003. This is probably because of the closure of two UK plants, one manufacturing HBCD and the other using HBCD in the manufacture of expanded polystyrene.

Concentrations of CBs in harbour porpoise blubber are reacting more slowly to controls on the use of PCBs, although these have been in place since the 1980s, and levels are declining only slowly. Concentrations of BDEs in harbour

BDE congeners

Brominated diphenyl ethers (BDEs) are flame retardant compounds. For this assessment we determined the nine OSPAR congeners.

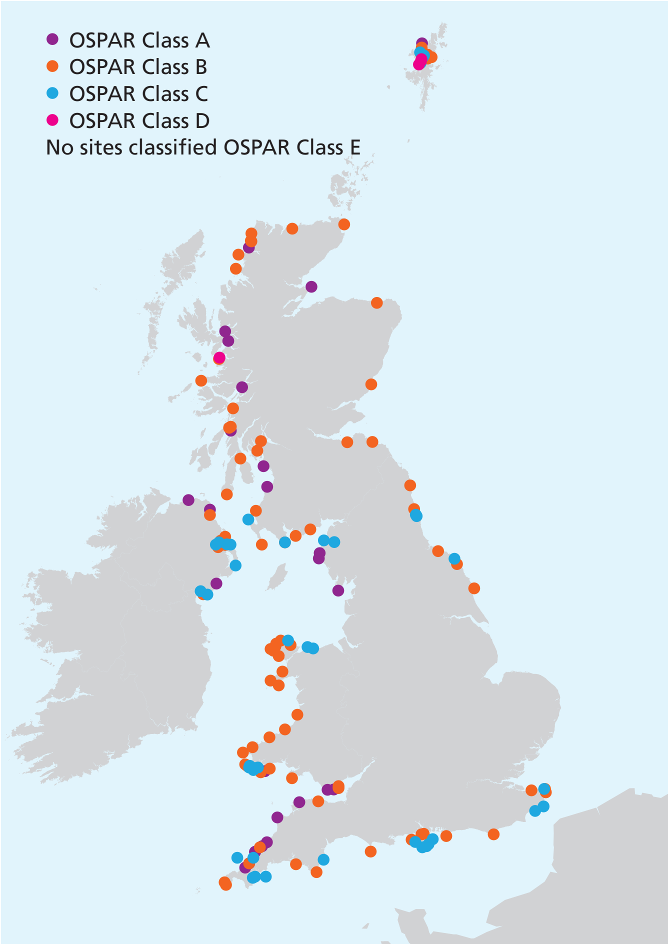
The nine BDE congeners are: BDE28, BDE47, BDE66, BDE85, BDE99, BDE100, BDE153, BDE154 and BDE183.

porpoise blubber have also been declining over the period 1998 to 2008, following EU risk assessment and regulation. The tissues in deep-sea fish collected from the Rockall Trough to the west of the UK contained both CBs and BDEs, but not HBCD or tetrabromobisphenol-A (TBBP-A).

Fish liver pathologies, including cancers, are higher and potentially increasing at certain Irish Sea sites, higher but static at some North Sea sites, and low and static (approaching or at background levels) at Inner North Sea and English Channel sites. The causes of the higher levels are unknown, but cancers do not result solely from exposure to hazardous substances.

Another indicator of poor health in marine biota is imposex – the imposition of male characteristics on female organisms caused by exposure to tributyltin (TBT; Figure 2.10) or hybrid male/female conditions caused by a wider range of chemicals. The extent of imposex in marine snails as a result of exposure to TBT has declined since 1998, showing that the bans on the use of TBT in antifouling paints for ships have been very effective, with evidence of recovering populations and wider improvements in the range of bottom-dwelling organisms in previously impacted areas.

Figure 2.10 OSPAR classification of dogwhelks from UK sampling sites in relation to imposex, 2007. OSPAR classifications go from A (no incidence of imposex) to E (populations unable to reproduce). © Crown copyright 2010: permission granted by Cefas.



Marine snail (dogwhelk)

Charting Progress reported evidence of endocrine disruption resulting from exposure to oestrogenic chemicals in flounder from a number of UK estuaries (Tyne, Tees, Mersey, Clyde and Forth). It assessed endocrine disruption, in this case feminisation, using vitellogenin (VTG). This is a protein normally only found in the blood of female fish. Thus, finding it in male fish indicates exposure to oestrogenic chemicals. There has been no further work since then, so we cannot say what the current status is, or assess trends since *Charting Progress*. Although the concentrations of VTG in males of offshore species of fish, cod and dab, are generally close to background levels, in cod from the North Sea and around Shetland, we found a marked increase in the amount of VTG at a body mass of 5 kg. This is about the size at which cod switch their diet from eating benthic invertebrates to eating other fish, both benthic and pelagic. We saw similar results in dab from UK offshore waters, suggesting that the affected fish are gradually accumulating persistent oestrogenic compounds through their diet. One report also showed the presence of egg cells in the testes of male peppery furrow shells, a filter-feeding bivalve sampled in a number of estuaries in south-west England during 2004 to 2005, including the Avon estuary previously considered to be a reference site due to the low population level and lack of industry. These findings suggest continuing impacts from oestrogenic compounds, although we cannot assess their scale. Similar studies in cod and bivalves were not reported in *Charting Progress*.

Our assessment shows that reductions in emissions, discharges and losses are having an impact, since we find downward trends for certain contaminants in specific contexts such as the BDEs in harbour porpoise blubber. However, it is also clear that, for some legacy chemicals, concentrations in sediments are reducing only very slowly and contaminated sediments will act as a source of persistent organic pollutants for years to come. However, even their concentrations are generally below those likely to cause effects except for historically contaminated estuaries and very coastal locations.

During 2009, there were initial assessments of the status of the UK seas under the EU Water Framework Directive (WFD). Extensive data collection within monitoring implemented for the EU Dangerous Substances and Shellfish Waters Directives, etc, informs these WFD chemical status assessments. All Scottish transitional and coastal waterbodies achieved good status for contaminants. In England and Wales, 69% of transitional waters and 91% of coastal waters assessed were at good chemical status. Less than good chemical status was, in the majority of cases, related to TBT contamination. There were few breaches of the contaminant standards at sites in Northern Ireland, with the exception of ammonia. Programmes of measures will be developed where necessary.

The WFD monitoring uses EQS limit values for water developed by the EU which are in many cases lower than those used in this assessment, with the aim of achieving improved environmental protection. For a list see Common Position (EC) No 3/2008, at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:C:2008:071:0001:01:EN:HTML.

Within the EU Marine Strategy Framework Directive, monitoring will be undertaken with a view to assessing Good Environmental Status against 11 attributes, including whether concentrations of contaminants are at levels not giving rise to pollution effects; and whether concentrations of contaminants in seafood are below the regulatory limits set to protect human consumers.

2.2.5 Future work

We need to understand more about the effects of mixtures of chemicals. Classical toxicology has provided a huge amount of information on the hazards of individual compounds but, in their environment, animals are exposed to more than one compound at a time. For endocrine disruptors, the effects of different chemicals might just be additive, but they could also cancel each other out or exacerbate each other's effects. We also need more data to determine the significance of pharmaceuticals in the marine environment.

OSPAR and ICES are currently developing guidelines for monitoring and assessing integrated chemical and biological effects, initially for use within OSPAR. These will also be adopted within the CSEMP, where appropriate, and so integrated data will be available for the next in this series of UK status reports.

2.3 Radioactivity

Radioactivity in the marine environment arises from both naturally occurring and man-made sources, and can be harmful to humans and non-human species. The major sources of discharges are shown in Figure 2.11. For this assessment we have used data on changes in radioactivity concentrations in the environment from national monitoring programmes and OSPAR periodic reports.

Generally we have found that radioactive discharges are strictly controlled, discharge levels have reduced and a strategy is in place to further reduce discharge levels in the future.

Figure 2.11 Licensed nuclear sites discharging radioactive material into the marine environment. © Crown copyright 2010.

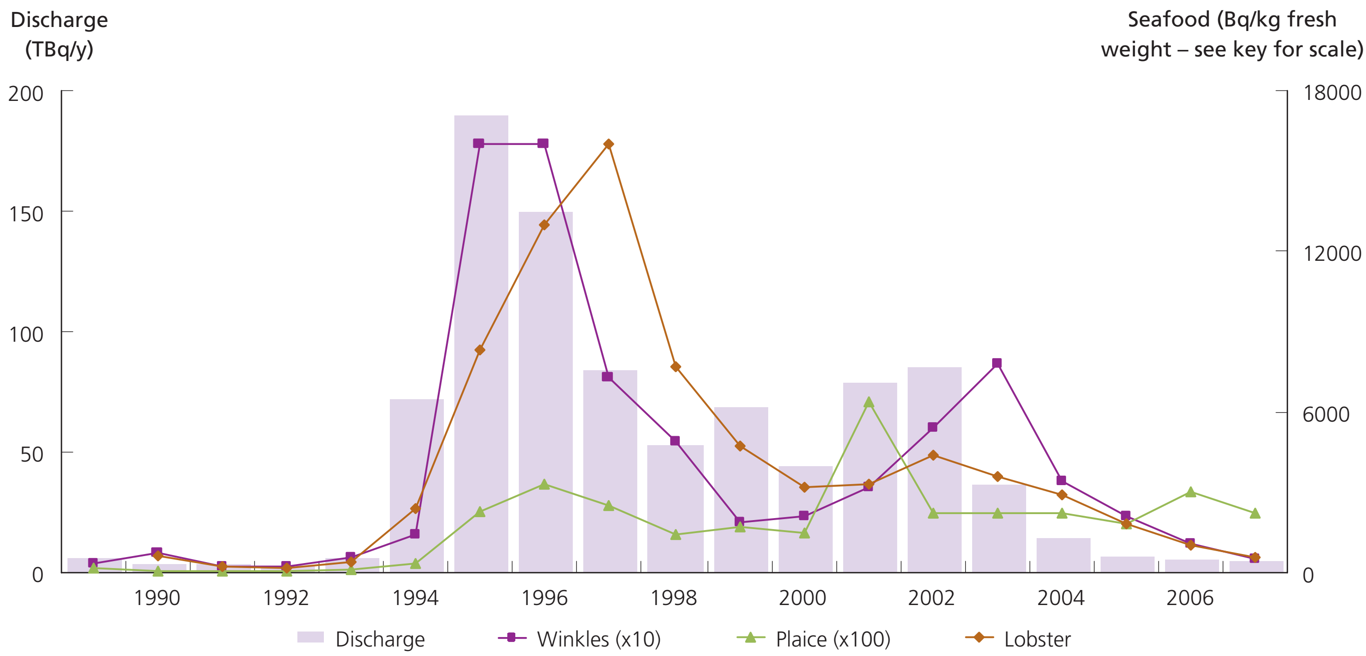
The aims of the OSPAR Radioactive Substances Strategy are to reduce radioactive discharges, emissions and losses, so that concentrations in the marine environment will eventually be near background values for naturally occurring radioactive substances and close to zero for artificially produced radionuclides. The revised UK Strategy for radioactive discharges, published in July 2009, sets out how the UK intends to achieve OSPAR's interim objective that additional concentrations above historic levels are close to zero by 2020. This builds on the UK Strategy for radioactive discharges published in 2002, widening its scope to include aerial as well as liquid discharges from the decommissioning and operational activities of the nuclear and non-nuclear sectors. The Strategy reports on the progress that has been made on reducing discharges and concentrations to the marine environment since 2002 on a sectoral basis and sets projections and expected outcomes for radioactive discharges up to 2030, based on a set of environmental principles. Forecasts indicate the UK's consistent progress with meeting the OSPAR commitments.

This strategy is having a noticeable effect, and inputs have fallen further since *Charting Progress*. The annual reports in the Radioactivity in Food and the Environment (RIFE) series confirm that radioactivity levels in UK waters currently pose no risk of harm to humans or wildlife.

With regard to specific radionuclides:

- Since 2005, technetium-99 (^{99}Tc) discharges from processes at Sellafield have fallen below 10 TBq per annum, and have met the end of 2006 target set in the UK Strategy for radioactive discharges (2002). Environmental concentrations of this radionuclide have also decreased significantly overall since 1995 (see Figure 2.12 for biota). Figure 2.13 shows the current distribution of ^{99}Tc in subtidal sediments of the Irish Sea. Note that the highest concentrations are in a patch of muddy sediments off Sellafield.
- Remobilisation of radionuclides from deeper sediment layers into surface sediments and overlying waters is now the principal source of caesium-137 (^{137}Cs) and plutonium in the Irish Sea. Increased concentrations of plutonium-239/240 (^{239,240}Pu) in certain areas of the Irish Sea and Solway Firth suggest redistribution of historically contaminated sediments is an emerging factor for ^{239,240}Pu.
- Polonium-210 (^{210}Po) was historically discharged by a phosphate processing plant near Whitehaven. The levels of ^{210}Po in

Figure 2.12 Annual ^{99}Tc liquid discharge from Sellafield and concentrations in winkles, plaice and lobster collected near Sellafield. © Crown copyright 2010.

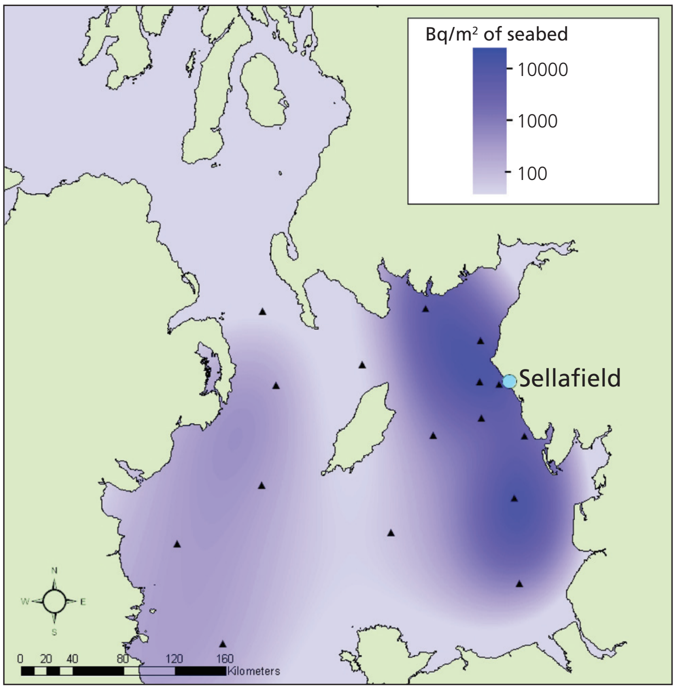


seafood around Whitehaven have fallen to within the range of natural variability. ^{210}Po is responsible for ~50% of the radiation dose to seafood consumers around Sellafield, which remains well within the UK and EU annual dose limit of 1 mSv set to protect human health. Most of this dose is due to the legacy of earlier discharges. Current discharges from Sellafield are very low relative to their 1970s peak and continue to fall.

- Concentrations of tritium (^{3}H) and carbon-14 (^{14}C) in fish and molluscs near the radiopharmaceutical plant in Cardiff are decreasing, although tritium levels remain higher than elsewhere in coastal waters.
- The offshore oil and gas industry is responsible for a large proportion of the total alpha-emitting radioactivity entering UK waters, as a result of discharges of the 'produced water', which contains elevated levels of the naturally occurring radionuclides radium-226 (^{226}Ra), radium-228 (^{228}Ra) and lead-210 (^{210}Pb). However, discharges fell by about 25% between 2000 and 2006, and will continue to reduce in line with declining production of oil and gas.

There is evidence of radioactive particles on beaches around Sellafield and Dounreay. However, the beaches remain open at both locations as there is no risk to users of these beaches. For the protection of consumers, the harvesting of seafood in the vicinity of Dounreay was banned under FEPA in 1997. Monitoring programmes are in place at both Sellafield and Dounreay to locate and retrieve contaminated particles from the foreshores. At Dounreay, an offshore programme of particle recovery is also underway.

Figure 2.13 Distribution of ^{99}Tc in Irish Sea subtidal sediments based on 2005 and 2006 survey data (small black triangles denote sampling locations). © Crown copyright 2010



In a screening assessment, modelled dose rates in aquatic systems were below the threshold of 40 µGy/hr in all cases except near the Springfields nuclear fuel manufacturing site in Lancashire, where new discharge limits should ensure that, in the future, the dose rates do not exceed the threshold.

40 µGy/hr threshold

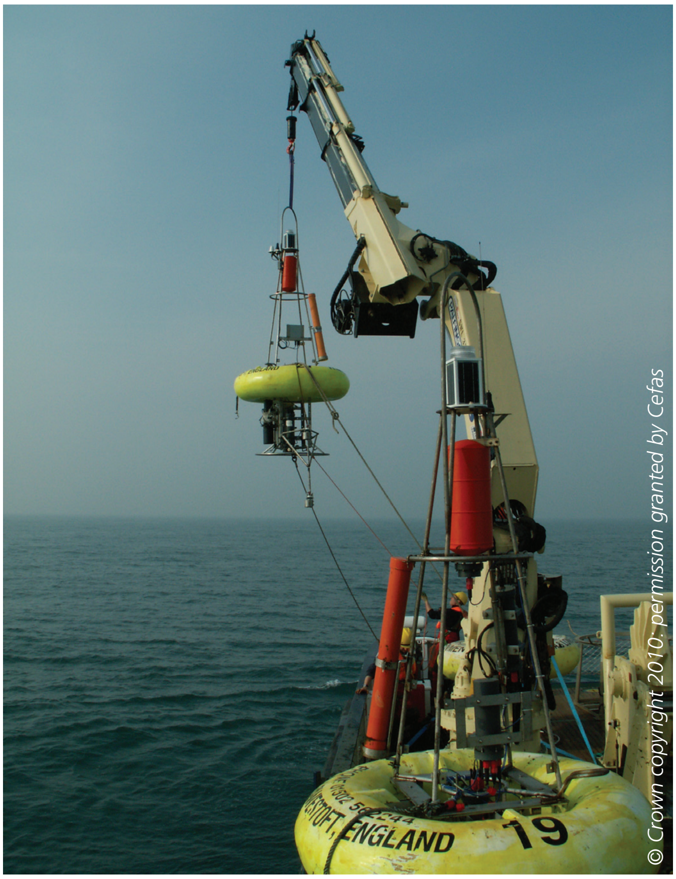
The 40 µGy/hr guideline is the dose rate below which there will be no harm to the species at the population level (Radioactive Substances Regulation – Environmental Principles, Environment Agency, 2009).

2.4 Eutrophication

Eutrophication is one of the major threats to the health of estuarine, coastal and shelf sea ecosystems around the world. It occurs when the enrichment of water by nutrients (often from fertilizer run-off from agricultural land or sewage discharges) causes an accelerated growth of algae and higher forms of plant life. This in turn leads to an undesirable disturbance to the balance of organisms present in the water and to the quality of water concerned. (This definition is based on the Urban Waste Water Treatment Directive, UWWTD; 91/271/EEC.) For this assessment, undertaken in 2007, we focused on the risk posed by nutrient enrichment in the period 2001 to 2005 and in the near future, and the extent of eutrophication problems in UK waters.

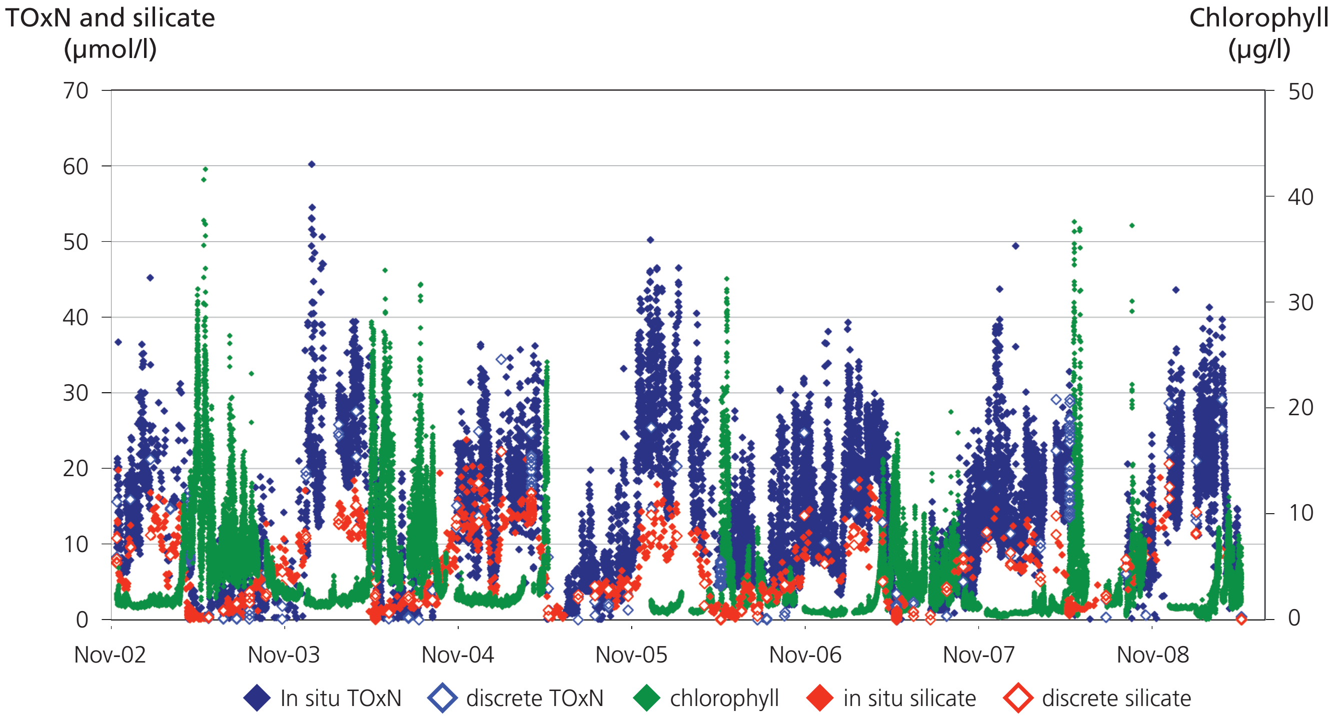
We used OSPAR's Comprehensive Procedure for identifying the eutrophication status of coastal and offshore waters. This uses a 'weight of evidence' approach to identify 'non-problem areas', 'potential problem areas' at risk of eutrophication, and 'problem areas' that are already experiencing undesirable disturbance to the balance of organisms. For estuarine water we used the results of similar assessments carried out in support of the EU UWWTD and Nitrates Directive. We assessed field measurements against a checklist of parameters including concentrations of nutrients, chlorophyll and dissolved oxygen, phytoplankton indicator species, macrophytes and toxin-producing algae. Figure 2.14 shows data collected using a SmartBuoy. Higher concentrations of nitrogen and silicate are seen during the winter and of chlorophyll in the spring and summer, in response to blooms of algae which consume nutrients and photosynthesise. Each dot represents a single, automatic, measurement. Traditionally, nutrients levels and trends were monitored using a relatively small number of discrete samples analysed on-board ship or in the laboratory. Newer developments allow the gathering of much more frequent data (as shown in Figure 2.14).

Deployment of a SmartBuoy to provide detection of environmental change in UK waters



We found that UK coastal and offshore waters in each of the eight regions are currently non-problem areas with respect to eutrophication. The coastal waters include five areas that had caused concern in an earlier assessment undertaken in 2002 and reported in *Charting Progress* – these were East England, East Anglia, Liverpool Bay, the Solent and the Firth of Clyde. Although these areas are still nutrient enriched, and some showed evidence of accelerated growth of algae, there was no evidence for undesirable disturbance, and the risk is not increasing.

Figure 2.14 A time series of nitrate + nitrite (TOxN), silicate and chlorophyll concentrations (in situ data) from the Cefas SmartBuoy in Liverpool Bay, together with the results from periodic ship-based surveys (discrete). © Crown copyright 2010: permission granted by Cefas.



Opportunistic green algae growing in an estuary

However, in 2007 we identified 17 small estuaries and harbours as problem areas and 5 as potential problem areas. These water bodies are also designated as Sensitive Areas under the UWWTD and as Nitrate Polluted Waters (eutrophic) under the Nitrates Directive, and hence are already subject to nutrient reduction programmes. But there is such a large reservoir of nutrients in soils and sediments that the environmental response to the reduction in nutrient inputs is likely to be slow. Moreover, it is not clear to what extent these protective measures will lead to ecological recovery, because the eutrophication process is complex and may not be easily reversible.

The biggest pressures on eutrophication status occur in the east, south and north-west of England where nutrients of human origin (notably nitrate and phosphate from agriculture and urban waste water sources) have enriched

coastal waters. We found no changes in eutrophication status over the period 1996 to 2005, and re-assessment of the five areas cited above with additional data confirmed them to be non-problem areas. The designation of Nitrate Vulnerable Zones covering 69% of the land in England, 14% of Scotland, 4% of Wales and the whole of Northern Ireland is likely to lead to a reduction in nutrient inputs from agriculture, as is the effective implementation of the UWWTD which will reduce nutrient inputs from waste water. Since 1998, total inputs of phosphate have declined by around 6% to 9% per year in all regions, while total inputs of dissolved inorganic nitrogen have decreased by 2% per year in the North Sea (Region 1) and the Irish Sea (Region 5).

We have high confidence in the assessments of eutrophication in most areas due to the availability of extensive datasets, and enhanced monitoring which was put in place in areas that were previously reported to be vulnerable.

In conclusion, we have reached a situation where eutrophication problems are apparent in some small estuaries, which occupy only a small percentage of our seas (< 0.2% of the area in each region, and < 0.03% overall). However, we should continue to reduce nutrient pressures through appropriate actions under EU Directives to address areas where there are still problems, even though recovery may take many years. A further application of the OSPAR Comprehensive Procedure to assess the current eutrophication status will begin shortly.

2.5 Oil and chemical spills

Although oil and chemical spills are generally short-term and localised, their effects can be significant. We could not assess the regional impact of accidental spillages of oil and chemicals, because in general they are logged as the number of incidents reported rather than as volumes lost. Most happen in major shipping lanes or where the offshore oil and gas industry operates. We have high confidence in the estimates of oil lost from offshore platforms as the UK Government has a mandatory reporting requirement, but our confidence is lower in relation to spills from ships, because these are usually detected using aerial surveillance and satellite data rather than being reported by those responsible.

In 2007, the most recent year for which data are available, there were 654 accidental discharges of oil from ships and offshore platforms into UK waters, an increase of 29% in discharges from ships and 13% in discharges from platforms compared to 2006. However, most were small, with only 47 incidents involving the loss of oil or chemicals in excess of 2 tonnes. We could not assess compliance with the OSPAR Ecological Quality Objective (EcoQO) for the proportion of oiled common guillemots found on beaches around the North Sea due to a lack of monitoring data; however, in Orkney and Shetland, the EcoQO has been met and the proportion of oiled guillemots is decreasing. The only incident of note since *Charting Progress* involved the container ship MSC *Napoli* which was beached in January 2007, spilling a total of 302 tonnes of oil, of which 150 tonnes affected Lyme Bay on the Devon/Dorset coast. The incident was effectively dealt with by the Secretary of State's Representative for Maritime Salvage and Intervention and the Maritime and Coastguard Agency, and only had a small local impact on seabirds.

The growing traffic in heavy fuel oils from the former Soviet Union past UK coasts is raising the risk of accidental spillages of oil and chemicals, as is the increasing size of container vessels.

MSC Napoli *grounded in Lyme Bay in January 2007*

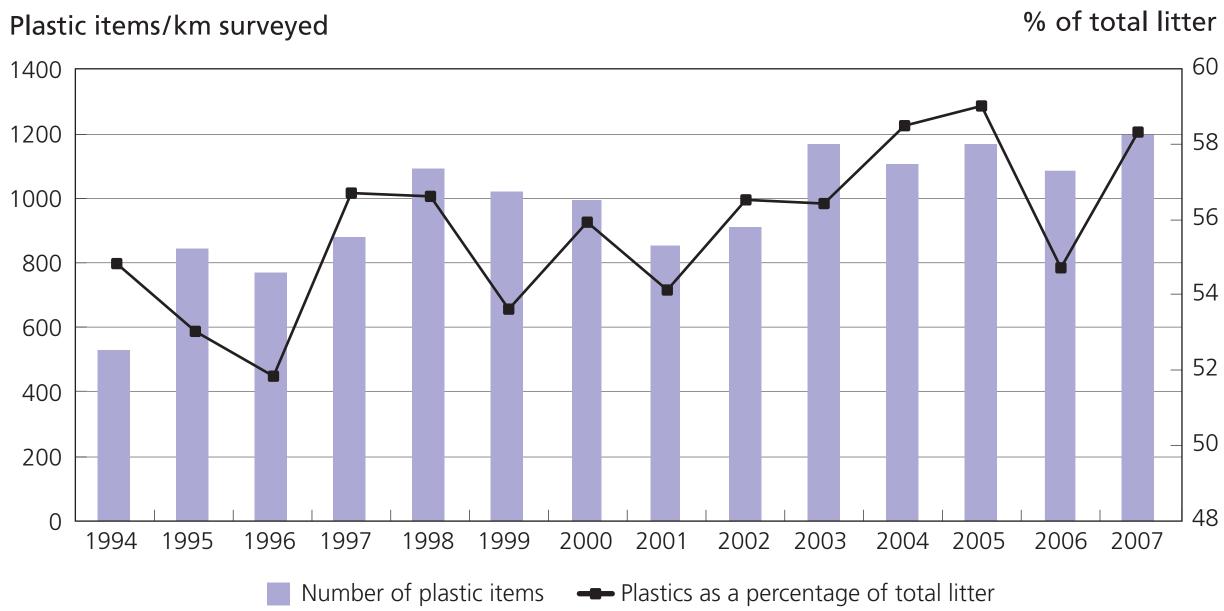
Prevention is better than cure. The best hope of reducing incidents comes from international efforts to ensure that the best modern ships and well-trained and efficient crews are passing through our waters. Other mitigation measures, such as traffic separation schemes and the provision of emergency towing vessels, are in place. The UK also has a good response capability, under the umbrella of the National Contingency Plan for Dealing with Pollution from Ships and Offshore Installations.

In the absence of a major spill, oil and chemical spills are generally of relatively minor significance in preventing progress towards the UK vision of clean and safe seas.

2.6 Litter

Significant amounts of litter appear in our seas and on our beaches. It is unsightly and can cause harm to marine wildlife through entanglement and ingestion, and through smothering of the seabed. Litter has both environmental and economic effects through harm to wildlife, costs to local communities in terms of clean-up costs and lost tourism, and costs to fishermen through lost catch and snagged nets. It can also pose a hazard to seafarers through fouling of ship propellers. Plastics are the main type of litter found both on beaches (Figure 2.15) and offshore, including increasing quantities of microscopic pieces of plastics resulting from degradation of larger plastic products in the sea. These may act as a vector for transferring

Figure 2.15 Plastic litter items per kilometre of UK beaches surveyed and as a percentage of total beach litter. © Marine Conservation Society.



Beachwatch litter survey

toxic chemicals to the food chain. Plastic litter can take hundreds if not thousands of years to break down, and it may never truly biodegrade. International and UK legislation prohibits the disposal of all plastics into the sea.

To assess beach litter, we used the annual series of surveys undertaken by volunteers for the Marine Conservation Society over one weekend each year. Offshore litter data come from CSEMP and other research cruises. Seabed litter has been surveyed at only a few sites and data are sparse, making assessment difficult. The KIMO 'Fishing for Litter' initiative enables litter trawled up from the seabed by fishing boats to be landed ashore and disposed of responsibly. The quantities of litter landed by region are reported, but the source locations are not logged.

2.6.1 Beach litter

Among the beach litter that can be identified, the main sources are the general public, fishing, sewage discharges and shipping. In general, there has been no appreciable fall in the

quantities of litter on UK beaches since *Charting Progress*. In fact, if we consider data collected since the start of monitoring, there has been a considerable increase. In 1994, an average of around 1000 items per kilometre was recorded but, by 2007, this had almost doubled. The majority of this increase occurred between 1994 and 2003; since then litter levels have been relatively steady although still high (Figure 2.16).

The methodology presently used by the Marine Conservation Society is comparable to that used by OSPAR and with the recently published UNEP/IOC guidelines on survey and monitoring of marine litter. But, more frequent sampling would increase confidence in the assessment of trends.

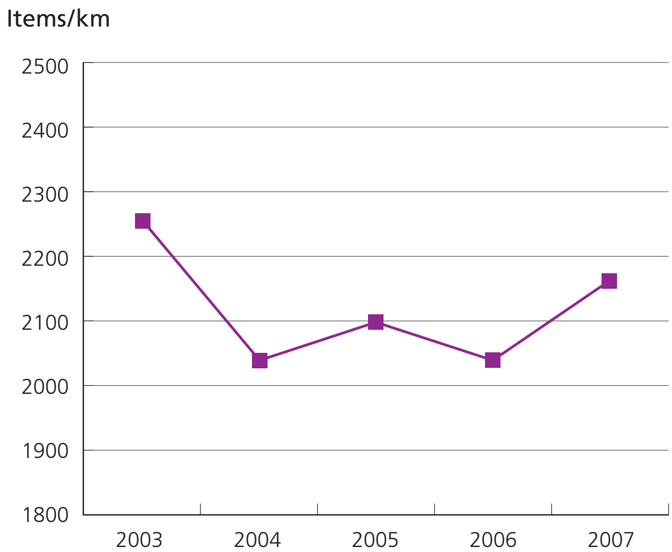
Some beaches are not surveyed every year making a comparison of these sites more difficult and some areas have sparse data sets. Up to 40% of litter items remain unassigned each year, either because they are too small or too weathered to identify a source, or because they could have come from a number of sources. Although it was assigned a ‘red’ status (unacceptable) in some areas in *Charting Progress*, the overall ‘traffic light’ status assigned to beach litter is amber (some problems) in Regions 1 to 5. However, with the exception of Region 3 which has improved, the status has not changed significantly since *Charting Progress*.

2.6.2 Offshore litter

We found a wide variability in offshore litter between sites and sometimes in successive years for locations sampled. There is generally not much litter on the seabed, but seabed smothering could be an issue in particular locations.

The presence of significantly higher densities of litter at Carmarthen Bay, North Cardigan Bay, in the Celtic Deep and in Rye Bay suggests that these are areas of accumulation, where litter

Figure 2.16 Beach litter items (all types) per kilometre surveyed in all UK regions, 2003–2007. © Marine Conservation Society.



gathers because of the effects of winds and currents. The frequency of litter ranged from 0 to 17 items per hectare. Rope, polypropylene twine and hard plastics are the most common forms of offshore litter. However, data are too sparse to allow a meaningful assessment of changes in quantities of litter either regionally or over time, and we also know too little about the impacts of litter in the sea to draw any reliable conclusions about the effects.

Drivers for change include the Merchant Shipping (Prevention of Pollution by Sewage and Garbage from Ships) Regulations, and for Port Waste Reception Facilities, and the Updated Code of Practice on Litter and Refuse. Increased participation in recycling schemes by the general public and implementation of relatively new legislation may take time to show effect.

At present, responsibility for marine litter is spread across a number of UK agencies. Further co-operation between all organisations responsible for litter will help coordinate efforts to control marine litter.

While marine litter would appear to be largely preventable, the wide range of sources of litter, the number of pathways by which it enters the marine environment, and the fact that litter can be easily transported by winds and currents, all make managing the problem highly complex.

Litter is one of the 11 qualitative descriptors that will be used for assessment of Good Environmental Status under the EU Marine Strategy Framework Directive, so gathering data to allow a robust assessment will be essential. The UK Marine Monitoring and Assessment Strategy Assessment (UKMMAS) community will need to develop a more comprehensive programme to do this.

Marine litter has economic, environmental and aesthetic impacts. What is not yet clear is the full extent of these impacts in the UK.

2.7 Noise

For most marine mammals, many marine fish, and possibly some shellfish, sound is important for communication, locating mates, searching for prey, avoiding predators and hazards, and for short- and long-range navigation. Noise at inappropriate volume and frequency can mask biologically relevant signals; it can lead to a variety of behavioural reactions; hearing organs can be adversely affected, and at very high levels, sound can injure or even kill marine life. Man-made sound sources of primary concern are explosions, shipping, seismic surveys, offshore construction and offshore industrial activities and sonars of various types, including military sonar, which has previously been implicated in deaths of beaked whales.

There is currently not enough evidence to provide a quantitative assessment of underwater noise in UK waters, but increasing activity in constructing, for example, offshore wind farms, is likely to have raised local noise levels while the developments were underway. Further large-scale developments of offshore wind farms are likely in the future. The management of subsurface noise emitted from shipping is currently the subject of international debate within the International Maritime Organization and we expect further guidance on this issue in the future.

Meanwhile, this area requires considerably more research. Future studies should focus on mapping and modelling ambient noise, observational and experimental studies, and developing frameworks for assessing noise related risks. The UK Marine Monitoring and Assessment Strategy (UKMMAS) community will eventually need to develop a monitoring programme regarding underwater noise to meet the requirements of the EU Marine Strategy Framework Directive.

Offshore construction

Safe Seas

2.8 Microbiological contamination

Microbiological monitoring of the marine environment is currently focussed on identifying faecal pollution of bathing waters and shellfish harvesting areas. There are three national programmes covering bathing waters, shellfish waters and shellfish hygiene. Management regimes are in place to protect public health in relation to shellfish, ensuring that contaminated shellfish does not reach the market.

We have assessed microbiological data against standards set within the EU Bathing Waters Directive, the Shellfish Waters Directive and the Shellfish Hygiene Standards within the EU Food Hygiene Regulations. Current standards assess bacterial contamination as indicative of levels of faecal pollution. This serves as a proxy for other agents such as viruses. Limited ability to measure viral loads in the environment and a lack of understanding of the dose-response relationship in humans means that viral standards have yet to be established. Such issues continue to be investigated with a view to developing a viable approach to the management of viruses.

Bathing Waters: In 2007, 96% of bathing waters met at least the 'imperative' (compulsory) standard and 76% met the 'guideline' (desirable) standard under the EU Bathing Waters Directive. This is similar to the findings in *Charting Progress*. The 2003 assessment showed that 98% of designated bathing waters met the imperative standard and 74% met the guideline standard. See Figure 2.17 for a regional summary of compliance with the imperative bathing waters standard.

Bathers on a Cornish beach

Figure 2.17 Location of UK identified bathing waters in 2007 (●) and regional compliance with the imperative bathing waters standard in UK bathing waters, 1998–2007. Sampling areas include a small number of inland bathing waters. © Crown copyright 2010: permission granted by Cefas.

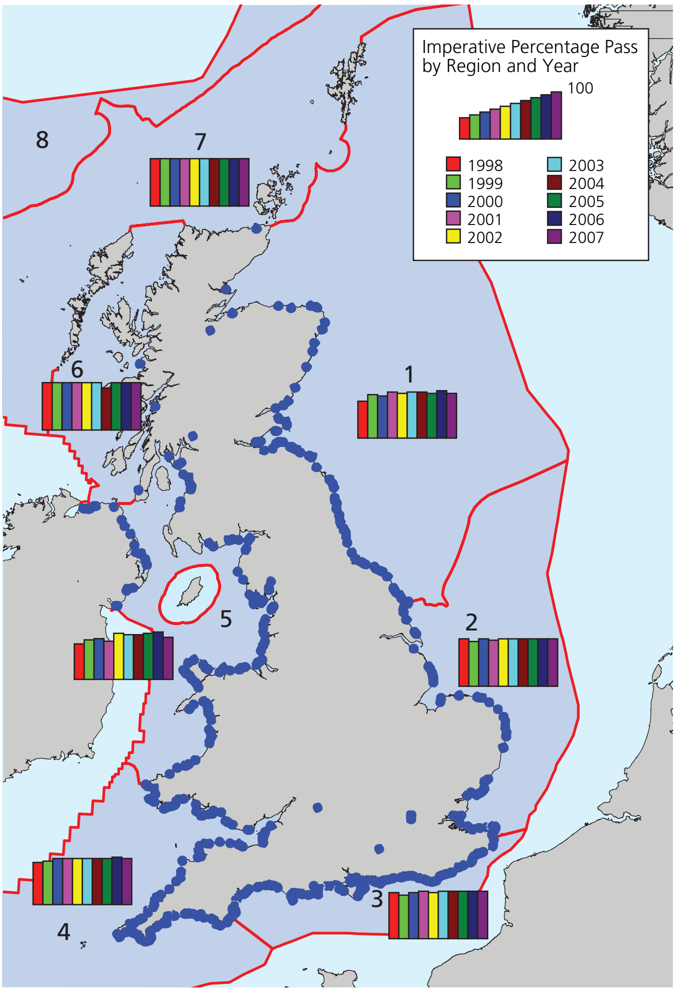
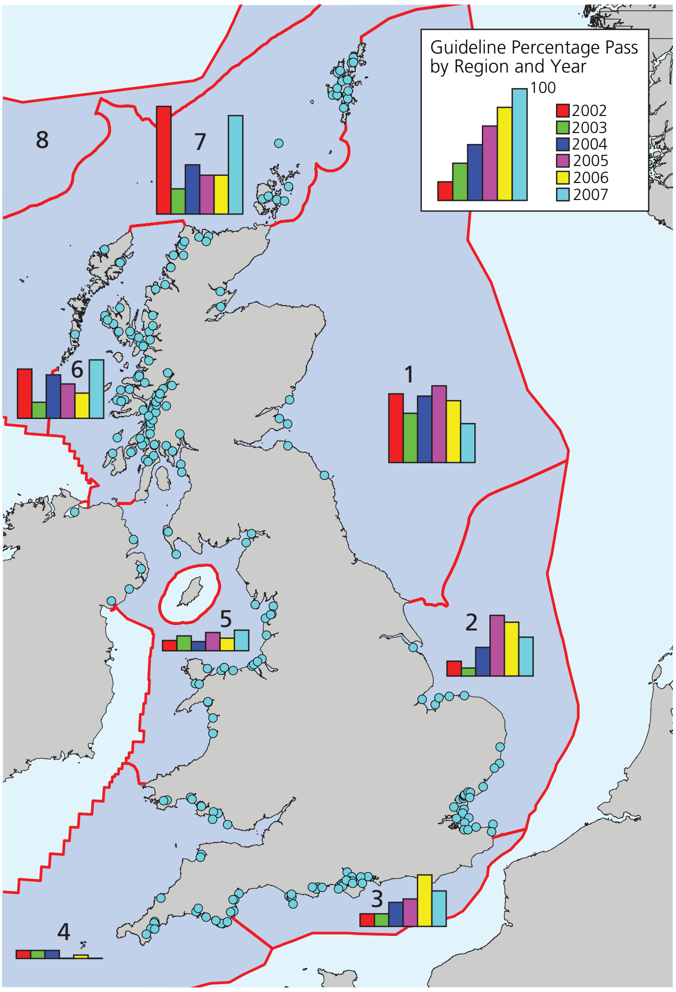


Figure 2.18 Location of UK designated shellfish waters in 2007 (●) and regional compliance with the guideline shellfish waters value, 2002–2007. There is year-to-year fluctuation; overall, the percentage compliance was greatest in Regions 6 and 7 and least in Region 4. © Crown copyright 2010: permission granted by Cefas.



All bacteria concentrations per 100 ml of seawater; Bathing Waters Directive: Imperative standard 10 000 total coliforms, 2000 faecal coliforms in 95% of samples; guideline standard 500 total coliforms and 100 faecal coliforms, in 80% of the samples.

Shellfish Waters: In 2007, 40% of sampled shellfish waters met the guideline value under the EU Shellfish Waters Directive (see Figure 2.18 for locations and time series). This value is significantly more stringent than the guideline standard in the Bathing Waters Directive. (Shellfish taken from the more contaminated waters are cleansed prior to sale for human consumption, to reduce bacterial contamination to a safe level.) *Charting Progress* undertook only a limited assessment so we cannot determine whether any significant change has taken place since then.

Shellfish Hygiene: In 2007, shellfish from 21% of areas could be consumed without treatment, while 78% required some treatment. Less than 1% was prohibited from harvest on the grounds of microbiological contamination. Comparable figures in *Charting Progress* were 17%, 82% and 1%, respectively.

The levels of compliance reflect significant investment in sewage treatment and infrastructure driven by the Bathing Waters and Shellfish Waters Directives. Water companies plan to spend over £300 million on additional improvements under these Directives over the next five years. Further improvements in microbiological quality will also require measures to reduce the impact of land run-off. This includes reducing misconnections in piping, sustainable drainage systems, and in changes to land management, such as establishing buffer zones excluding grazing animals from the vicinity of water courses. Viruses are also of concern and further work is needed to measure them and establish suitable standards.

2.9 Algal toxins

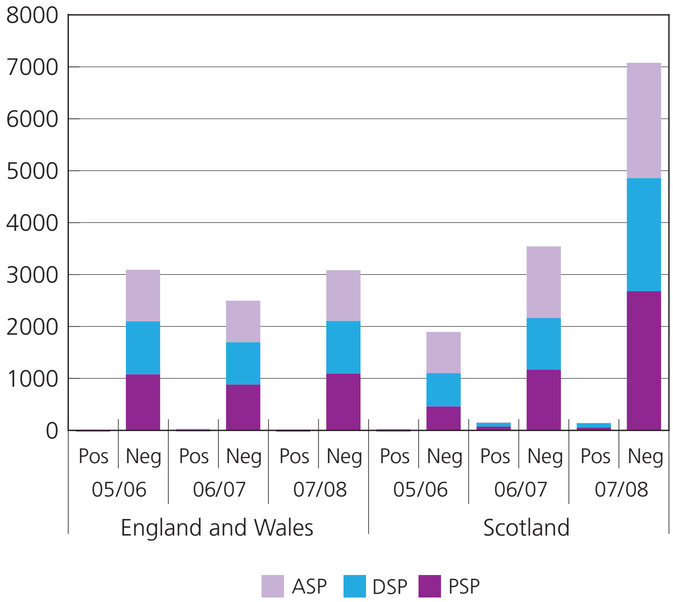
Algal toxins (also known as biotoxins) are natural compounds produced by certain species of marine algae. They are of concern because shellfish such as mussels, cockles and oysters can accumulate them when they feed, and this has the potential to affect human consumers. However, when toxins are detected in shellfish by the statutory monitoring programme (Figure 2.19), appropriate management measures are taken to protect shellfish consumers. Despite some concern that high levels of nutrients could increase the occurrence of harmful algal blooms in which the toxins are produced, a recent study concluded that the abundance of the harmful algal bloom species in UK and Irish coastal waters is not related to nutrient enrichment from human sources. Essentially, harmful algal blooms are a natural phenomenon beyond our control and, as such, will occur. However, we will need to continue with monitoring programmes to protect public health and shellfish production, particularly as climate change may alter the frequency of toxic blooms.

Figure 2.19 The occurrence of positive samples for diarrhetic shellfish poisoning (DSP) between April 2005 and March 2006. Exceedance of the limit value results in temporary closure of the fishery until levels fall. © Crown copyright 2010: permission granted by Cefas.

For this assessment we used data from comprehensive biotoxin monitoring programmes covering the period 2005 to 2008. These include monitoring for relevant phytoplankton species in all regions of the UK to comply with EU legislation. (The testing methods employed by all official monitoring laboratories on behalf of the competent authority follow United Kingdom National Reference Laboratory [UKNRL] protocols [where these exist] and are accredited to ISO17025 standards where possible.) We did not include data from additional studies outside this three-season programme.

We did not include historical data in this assessment, so have not reported trends in toxicity observed in shellfish. We found that marine biotoxins and toxic algae were present in samples from all monitored areas in England and Wales, Northern Ireland and Scotland during the three years of study, but they did not increase either temporally or spatially. However, the number of samples in which toxins were detected was small, generally fewer than 5% and less than 1% in some years. There were no incidents of human toxicity and thus we conclude that the current monitoring regime within the UK (which monitors the three classes of shellfish toxins responsible for amnesic shellfish poisoning, ASP; diarrhetic shellfish poisoning, DSP; and paralytic shellfish poisoning, PSP – see Figure 2.20) provides sufficient protection for human consumers of shellfish in respect of ASP, DSP and PSP toxins. Other algae occur in UK waters which generate additional toxins; notably the azaspiracids and spirolides

Figure 2.20 Numbers of samples testing positive or negative for the presence of ASP, DSP and PSP toxins in samples from England, Wales and Scotland 2005-2008. Note low frequency of samples testing positive. © Crown copyright 2010: permission granted by Cefas.



Farmed mussels

© Muckairn Mussels

and the toxins responsible for neurotoxic shellfish poisoning. Current monitoring arrangements do not include these, so we need further research to assess the risk they pose and whether they should be monitored in the future.

2.10 Future work

The information presented in this report and the more extensive descriptions detailed in the associated Feeder Report prepared by the Clean and Safe Seas Evidence Group, lead us to conclude that where regulation has been introduced it has generally had a beneficial impact. Reductions in emissions, losses and discharges of hazardous substances have resulted in some reductions in contaminant concentrations in sediment and biota. However, concentrations of some persistent chemicals resident in sediments are declining only very slowly. The timescale for observation of these trends is generally greater than that of the period between *Charting Progress* and the present assessment. The gathering of additional data, extending time series, is resulting in some downwards trends becoming significant.

There are problem areas in UK waters with respect to eutrophication, but these represent a very small total area.

In terms of safe seas, this report has focussed on microbiological contamination and algal toxins. Both issues are regulated and current programmes provide a significant degree of public protection. Thus, in the UK, we continue to move towards a vision of clean and safe seas.

Some of the 'new' issues, such as underwater noise and litter, are going to require greater scrutiny in the future given their incorporation as descriptors of Good Environmental Status under the Marine Strategy Framework Directive. Current monitoring will need to be extended in order to gather evidence in relation to their significance and impacts.

2.11 Assessment summary

The summary table presents our expert opinion on a single comparable assessment of status and trend across all components assessed for clean and safe seas. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. Trend arrows are also provided based on the evidence available, showing whether the state or condition of the component is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (⟷). The confidence rating of each assessment is also estimated in the associated Feeder Report, based on the criteria developed by the UK Marine Climate Change Impacts Partnership (MCCIP). Assessments which have been made with low confidence are indicated in the summary table.

A more detailed discussion of the rationale for this approach, 'traffic-light' status, and the confidence assessment is given in Chapter 1.

The 'traffic light' status assigned originates from the detailed assessments made for each component in the Feeder Report. Each component assessment section in the Feeder Report was accompanied by a table which summarised the findings if sufficient evidence was available.

The Feeder Report assessments provide valuable additional information that can be used to give context to the summary assessments in the summary table and provide a broader understanding of each component, its status and the pressures to which it is subject.

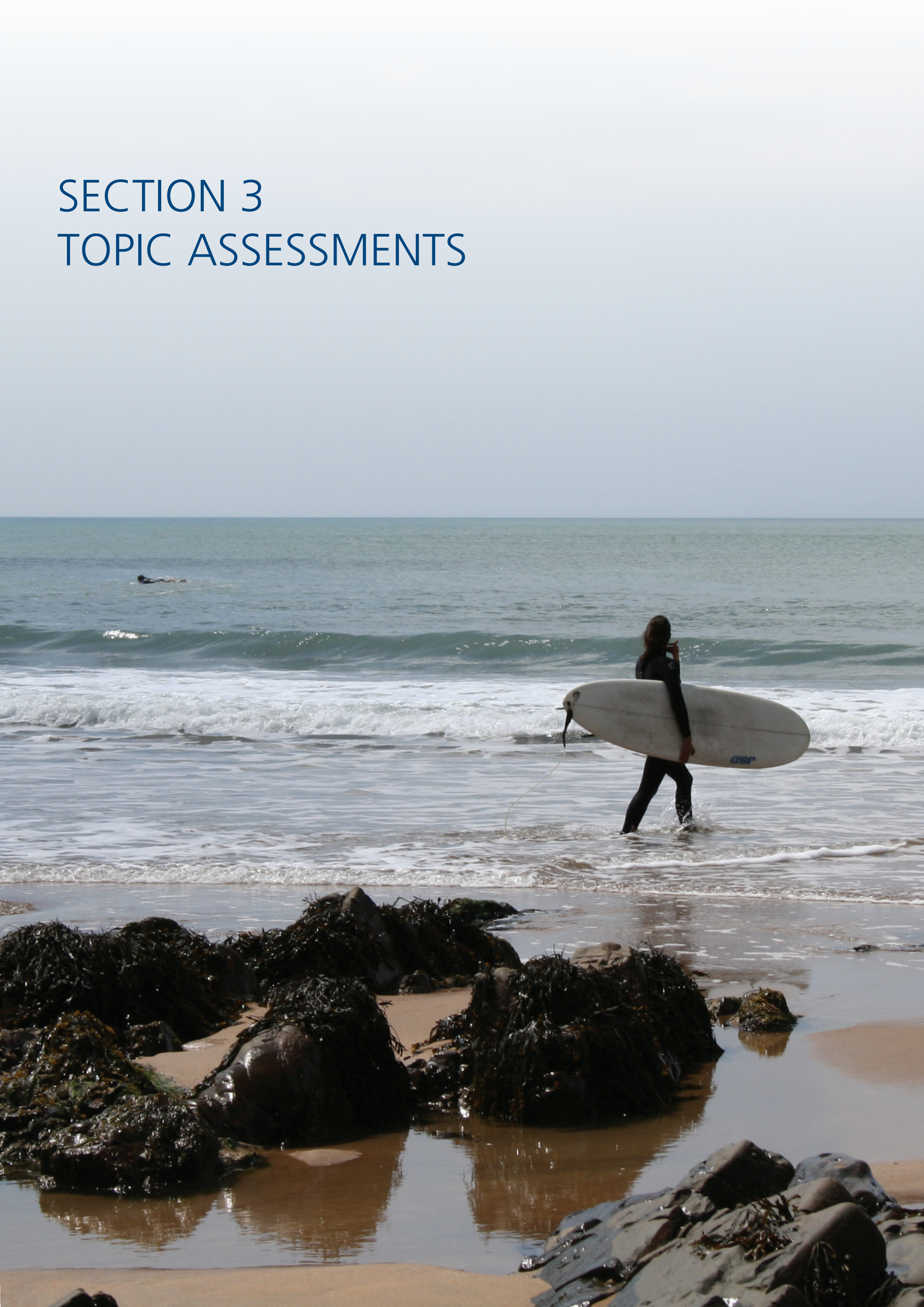
Clean and Safe Seas – Summary Table

Components currently assessed	*Region*							
	1	*2*	*3*	*4*	*5*	*6*	*7*	*8*
Clean Seas								
Hazardous substances	↔	↔	↔	↔	↔	↔	↔	*
	Main sources are inputs from rivers, the atmosphere, various industries and agriculture. These sources are subject to controls. In some limited areas marine biota are at risk, particularly near to the main sources in industrialised estuaries. Reservoirs in sediments due to historical contamination will take many years to dissipate to background concentrations due to persistency of the substances.							
Radio-activity	↔	↔	↔	↑	↑	↔	↔	↔
	Main sources are discharges from the nuclear sector and hospitals and the offshore oil and gas industry which discharges naturally occurring radionuclides. Received doses of radioactivity to both humans and wildlife continue to be well within regulatory limits.							
Eutro-phication	↑	↔	↑	↑	↑	↑	↑	*
	Main sources are inputs of nitrogen (N) and phosphorus (P) from sewage works and agriculture. Ecosystems are at risk if eutrophication occurs. A few very small coastal harbours and embayments with limited water circulation experience eutrophication problems. Nitrogen and phosphorus inputs to these are controlled.							
Oil/ chemical spills								
	Main sources are accidental spills from ships and the offshore oil and gas industry. Ecosystems, habitats and species may be at risk if loads are significant. Where significant tonnages are spilt, regional monitoring programmes are implemented to assess risk. Assessment of cumulative impact is problematic.							
Beach litter	↔	↔	↑	↔	↔			
	Main sources are the general public, fishing, sewage discharges and shipping. Aesthetic, economic and environmental impacts may occur if levels are high. Only limited data available for Regions 6, 7 and 8.							
Offshore litter								
	Main sources are from fishing/shipping and plastics discarded on land and at sea. There is insufficient evidence or criteria to assess impacts and state on a regional basis, but several surveys indicate that marine litter accumulates in certain locations due to sea currents.							
Noise								
	Main anthropogenic sources are explosions used in construction and demolition, shipping, seismic surveys, offshore construction and industrial activities and sonar of various types. There is insufficient evidence or criteria to assess impacts and state on a regional basis, but research indicates that cetaceans and fish may be affected by specific noises.							

Components currently assessed	*Region*							
	1	***2***	***3***	***4***	***5***	***6***	***7***	***8***
Safe Seas								
Micro-biological quality of bathing waters	↔	↔	↔	↔	↔	↔	↔	**NA**
	The main source is bacteria and viruses originating from sewage treatment works. In 2007 96% of bathing waters met the imperative standard of the EC Bathing Waters Directive, which indicates that water quality is acceptable, and should present a minimal risk to human health. However, the standards are being tightened under the new Bathing Waters Directive to give a higher level of protection.							
Micro-biological quality of shellfish growing waters	↔	↔	↔	↔	↔	↔	↔	**NA**
	The main source is bacteria and viruses from sewage treatment works. Concentrations of indicator bacteria in shellfish have not consistently met the most stringent standards. In such cases, the shellfish have to be removed for depuration before placing on the market in order to protect public health or in a small number of cases cannot be harvested.							
Algal toxins	*	*	*	*	*	*	*	**NA**
	Algal toxins from phytoplankton can contaminate seafood. The phytoplankton which are the source of the toxins occur naturally in UK waters. Controls are in place, including occasional closures, to protect public health. No shellfish harvesting in Region 8. Algal toxins were not addressed in CP1.							

(green)	Few or no problems	↔	No overall trend discernable
(amber)	Some problems	↑	State improving
(red)	Many problems	↓	State deteriorating
(chequered)	Lack of evidence and/or robust assessment criteria	*	No trend information available
NA	No significant activity in the region	≡	Low confidence in assessment

SECTION 3
TOPIC ASSESSMENTS



3.1 Hazardous Substances

3.1.1 Key points

i. Introduction

Hazardous substances enter the marine environment from both natural sources (in the case of metals and polycyclic aromatic hydrocarbons (PAHs), for example) and as a result of human activities. The main routes through which hazardous substances can enter the marine environment are through waterborne discharges to rivers and estuaries and via the atmosphere into the sea. The relative importance of these will vary between substances depending on their individual properties and behaviour following release. Their degree and mode of toxicity will likewise vary, and there is a potential for toxicity to both aquatic-dwelling organisms and human seafood consumers.

ii. How has the assessment been undertaken?

Where the necessary assessment criteria have been developed (i.e. within OSPAR) then these have been used to assess the data. Where this is not the case, other criteria have been used, notably the Effects Range-Low (ERL) and Effects Range-Median (ERM) approach developed for the US Environmental Protection Agency and widely used in many countries since then. The approach taken is outlined within each section. Currently, the Clean Seas Environmental Monitoring Programme (CSEMP) directly monitors a limited number of hazardous chemicals, selected on the basis of a risk assessment and subject to there being agreed methodological guidelines, quality assurance procedures and assessment criteria available. For others, such as tributyltin (TBT), we can assess their impacts by studying the biological effects they cause.

iii. Current status of hazardous substances and past trends

Inputs and concentrations of the contaminants (e.g. lindane) which are most commonly monitored in water have declined as a result of controls placed upon their use and concentrations are now generally below UK Environmental Quality Standard (EQS) limit values. Under the EU Water Framework Directive (WFD), monitoring will potentially encompass a wider range of substances. Also, a different approach has been taken in developing EQS limit values under the WFD and in many cases these limit values are lower than the UK EQS values, in some cases substantially so. These proposed WFD standards, adopted in Directive 2008/105/EC and currently being transposed into UK law, aim to provide an enhanced level of environmental protection and their use may result in more exceedances of the EQS limit values. Because concentrations of many contaminants in water are very low and variable (due for example to water movement associated with tides and the varying levels of freshwater run-off into rivers and estuaries), the CSEMP now concentrates on monitoring contaminant levels in biota and sediments, where concentrations are generally higher and less variable. This is particularly important for trend monitoring, both in space and time.

Some of the key points are as follows:

- The downward trend in inputs of contaminants over time reported in *Charting Progress* for rivers, sewage works and industrial discharges has continued for

mercury (Hg), cadmium (Cd), lindane and polychlorinated biphenyls (PCBs) to both the Celtic Seas and the Greater North Sea

- Water samples analysed for the EU Directives on Dangerous Substances (mainly in estuarine waters) and Shellfish Waters (mainly in coastal waters) showed that 98.6% of metal concentrations in water were below the UK EQS values in 2007, indicating very limited toxicological effects
- Metal concentrations in sediments were low in Scotland and the western Irish Sea and higher in England and Wales, with a number of industrialized estuaries, such as the Tees, Tyne, Thames, Severn and Mersey, showing elevated levels which may give rise to toxicological effects
- Mercury levels in fish flesh are elevated in some industrialized estuaries, such as the Mersey and the Thames, although not to levels likely to give rise to risks to human health
- Cadmium levels in fish liver appear not to be of concern either in terms of environmental harm or human health
- Lead (Pb) concentrations in fish liver are elevated in industrialized estuaries, including the Forth, Tyne, Tees, Wear, Severn, Mersey and Bann and in a few other coastal areas, but are unlikely to pose a risk to human health
- Cadmium, Hg and Pb concentrations were slightly elevated in mussels from some industrialized estuaries, such as the Dee, Humber and Thames for Cd, the Thames and Mersey for Hg, and the Tyne, Tees and Forth for Pb, but posing little risk to human health
- Silver concentrations were elevated in mussels in several industrialized estuaries, including the Severn and Thames. Historically, the major inputs were from its use in photography, which should be declining with the growth in use of digital cameras. However, its increasing use in nanoparticles as an antimicrobial agent in textiles may result in increased inputs
- The lowest concentrations for the ICES7 chlorobiphenyls (CBs) in sediments were found at the Scottish offshore sites, and the highest at those in south-east England
- For CB118, the most toxic of the ICES7 CBs, sediment concentrations in most areas are above the Environmental Assessment Concentration (EAC) and therefore of concern
- CB concentrations in fish liver were lowest in the Minch and Malin Sea region and highest in the Anglia region. Concentrations of CB138, CB153 and CB180 were below the respective EACs, while those of the more toxic CB118 were above the EAC, so these are of concern
- Sediment PAH concentrations were elevated in several currently or historically heavily industrialized estuaries (e.g., Tees, Tyne, Wear, Milford Haven) indicating the potential for toxicity to sediment-dwelling organisms. These results are in line with the poor biological sediment quality reported, suggesting that biological effects are being observed where contaminant levels are high
- Biological sediment quality was good where tested, except in parts of the Tees, Wear and Thames estuaries. These observations of adverse biological effects may be linked to elevated sediment PAH concentrations
- PAH concentrations in mussels were below the Background Assessment Concentration (BAC) or the EAC in all UK regions except the Tyne/Tees and Western Channel, at single sites at Teesmouth and off Plymouth Sound
- More data were available for the brominated flame retardant (BFR) compounds polybrominated diphenyl ethers (PBDEs) than

at the time of *Charting Progress*, but not enough to assess trends with time, and no assessment criteria have yet been developed by OSPAR. The highest concentrations in fish were found in industrialized estuaries, such as the Clyde, Tees and Humber, and the lowest off the Scottish coast, in the Western Channel and off the Humber/Wash area in eastern England

- In harbour porpoises from UK waters, a rapid upward trend in concentrations of the BFR hexabromocyclododecane (HBCD) after 2001 has been reversed in later years, probably largely due to closure of two UK plants; one was a manufacturing site for HBCD and the other used HBCD in textile manufacture
- High levels of ethoxyresorufin-*o*-deethylase (EROD) enzyme activity in fish liver, indicating exposure to planar organic contaminants such as dioxins and furans, planar CBs or PAHs, were found off the north-east English coast, on the western edge of the Dogger Bank, close to the Liverpool Bay coastline and at two historical sewage disposal sites close to the Scottish east coast
- Levels of DNA adducts of PAHs detected in fish in industrialized estuaries were similar to those previously reported in *Charting Progress*, indicating that while concentrations of contaminants are not increasing, there is an ongoing risk of carcinogenic exposure at these locations
- In terms of fish disease, the prevalence of liver pathologies (including cancer) are elevated and potentially increasing at certain Irish Sea sites, elevated but static at some North Sea sites, and low and static (approaching or at background levels) at inner North Sea and English Channel sites
- Imposex has declined since 1998, showing that the bans on the use of TBT have been very effective at improving the health of dogwhelk populations, and probably those of epifauna more broadly
- Biological water quality was good or very good everywhere tested, except in the Tees estuary, close to industrial discharges
- Levels of species diversity in soft sediment benthic communities around the UK coastline are what we would expect to observe in both coastal and offshore, soft-bottomed, marine benthic habitats
- Concentrations of vitellogenin (VTG) in offshore species of fish, cod and dab, were found to be at, or close to, background levels.

iv. What has driven change?

Within the hazardous substances area it is generally innovation by manufacturers followed by monitoring and, where appropriate, regulation, which are the main drivers of change. As an example we can consider the case of the PBDEs, which are BFRs added to products in order to slow the spread of fires. BFRs have been manufactured and used since the 1970s. During the 1980s, PBDEs were discovered in the environment in Sweden and became known environmental contaminants. Through the 1990s, environmental monitoring became more widespread and the toxicity of the PBDEs to both humans and wildlife was investigated. The European Commission then began a process of risk assessment for the three BFR products, the penta-, octa- and deca-mix PBDE formulations, in use within the EU. By 2008, the risk assessments were complete, and production and use of all three products were controlled within the EU.

v. What are the uncertainties?

Monitoring data are collected alongside appropriate quality control measures and are reliable. Confidence in the assessments is often constrained by the sparseness of the data, particularly at the regional scale (too few sampling sites to fully characterise a region).

vi. Forward look

Assessment of the data collected under the umbrella of the EU Water Framework Directive will yield a fuller picture of the contaminant status of the seas immediately around the coasts of the UK (to 12 nautical miles from the shore). Within the EU Marine Strategy Framework Directive there is a requirement to assess Good Environmental Status (GES) against a series of descriptors. For hazardous substances, the most relevant of these are Descriptor 8 ...*concentrations of contaminants are at levels not giving rise to pollution effects* and Descriptor 9 ...*contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards*. The criteria and methodological standards by which GES can be assessed are currently under development, and are due to be published by the European Commission in July 2010. It is important to note that there are still significant uncertainties associated with the quality levels that are to be established to determine GES under the Marine Strategy Framework Directive. It is therefore unclear at present how the current assessment will inform the future assessment of GES for the Directive.

OSPAR and the International Council for the Exploration of the Sea (ICES) are currently working on the development of guidelines for integrated chemical and biological effects monitoring and assessment, initially for use within the OSPAR community.

3.1.2 Inputs of hazardous substances

The main routes through which hazardous substances can enter the marine environment are through the aquatic pathway via waterborne discharges and losses, and via the atmospheric pathway through emissions which fall out of the atmosphere and into the sea. Quantitative information on inputs of hazardous substances through these pathways is given in the following sections.

3.1.2.1 Aquatic pathway

3.1.2.1.1 Summary

The downward trend in inputs of contaminants over time reported in *Charting Progress* for rivers, sewage works and industrial discharges has continued for Hg, Cd, lindane and PCBs to both the Celtic Seas and the Greater North Sea. This reflects the control measures which have been introduced for these substances. Use of lindane and PCBs has been phased out, and Hg and Cd are subject to cessation controls under both the EU Water Framework Directive and the OSPAR Hazardous Substances Strategy.

Lead, zinc (Zn) and copper (Cu) results show some reductions but tend to follow the riverine flow patterns with no clear downward trend over time. These elements occur naturally and there will be significant background concentrations, depending on local geology.

3.1.2.1.2 Introduction

The UK Riverine and Direct Discharges Programme has the aim of estimating the loads of targeted contaminants entering the marine environment from over 200 rivers, and from substantial sewage treatment works and industrial installations which discharge directly into the sea. Loads are calculated on the basis of regular monitoring of concentrations and flows at the same locations around the UK. The programme has been in place since 1990. The results provide an indication of how well pollution control regimes for land-based point and diffuse sources are working and give an indicator of the likely impacts on marine biota which can be directly affected through toxic effects, or indirectly affected via bioaccumulation through marine food webs.

Annual reports are required for submission to the OSPAR Riverine Inputs and Direct Discharges (RID) programme, and the information is also used to give an indication of progress towards cessation targets for hazardous substances set by OSPAR and the EU (e.g. in the Water Framework Directive).

The information gathered in the RID programme does not directly measure environmental impacts. However the estimated loads of the various contaminants which are all toxic to some extent give a useful indicator of the impacts which these substances are likely to have on marine ecosystems. They can also be used for modelling purposes, or for targeting areas of specific impact. It also provides an indicator of the broad performance of pollution control measures both nationally and locally.

3.1.2.1.3 Developments since *Charting Progress*

The various sources of the contaminants constitute the key pressures on the environment. The main types are (1) naturally occurring (e.g. Pb, Zn, Cu and Hg which occur through natural geological sources), (2) point source emissions and discharges originating from factories, workshops, large industrial installations and sewage treatment works, and (3) diffuse sources, which originate from agriculture, households, amenity use, transport, run-off from roads, and the use of products containing the contaminant (e.g. Cu and Pb piping, galvanised products). There are also reservoirs of contaminants in aquatic sediments and soil which have built up over the years from various uses.

Nearly all of these sources are controlled by permits (particularly for the point source discharges) or through various product restrictions and marketing and use controls. In general there have been no significant developments in management since *Charting Progress*, although there have been developments at the EU level on the control of Hg in products, and there is an ongoing process for upgrading permits, particularly through the EU Integrated Pollution Prevention and Control (IPPC) Directive.

There have been no significant developments with indicators or assessment methodology. However, detection limits continue to improve as analytical methodology becomes more sophisticated. Furthermore, *Charting Progress* gave an indication of the levels and patterns of change in UK inputs on a national scale for combined riverine and direct discharges. In this report, the load patterns of the riverine, sewage and industrial inputs have been reported separately and rather than reporting on a national scale, the inputs to the Celtic Seas and the Greater North Sea have been presented. Results have not been presented for the eight biogeographical CP2 Regions as the patterns of change are broadly similar to those at the broader scale.

3.1.2.1.4 Presentation of the evidence

This assessment presents the mean annual loads of five heavy metals, lindane (γ-HCH) and PCBs from rivers (riverine inputs) and sewage treatment works and industrial installations (direct discharges) entering the two sub-regions relevant to the UK specified in the EU Marine Strategy Framework Directive. The two sub-regions comprise the 'Celtic Seas' (consisting of the Western Atlantic, the Irish Sea and the Celtic Sea), and the 'Greater North Sea' (consisting of the North Sea and the English Channel).

It should be noted that riverine inputs include the discharges from sewage treatment works and industrial installations upstream of the monitoring points in the various estuaries. Direct discharges reflect the loads from discharges from substantial sewage treatment works and industrial installations downstream of the riverine monitoring point. Estimates of the diffuse source loads from the land areas downstream of the monitoring points are not included.

The levels and patterns of change in inputs of heavy metals, lindane and PCBs to the Celtic Seas and the Greater North Sea between 1990 and 2007 are illustrated in Figures 3.1 and 3.2, respectively. By addressing rivers, sewage treatment works and direct discharges separately, this report enables the contributions and trends for these sectors to be examined more closely.

Quality and reliability of the evidence

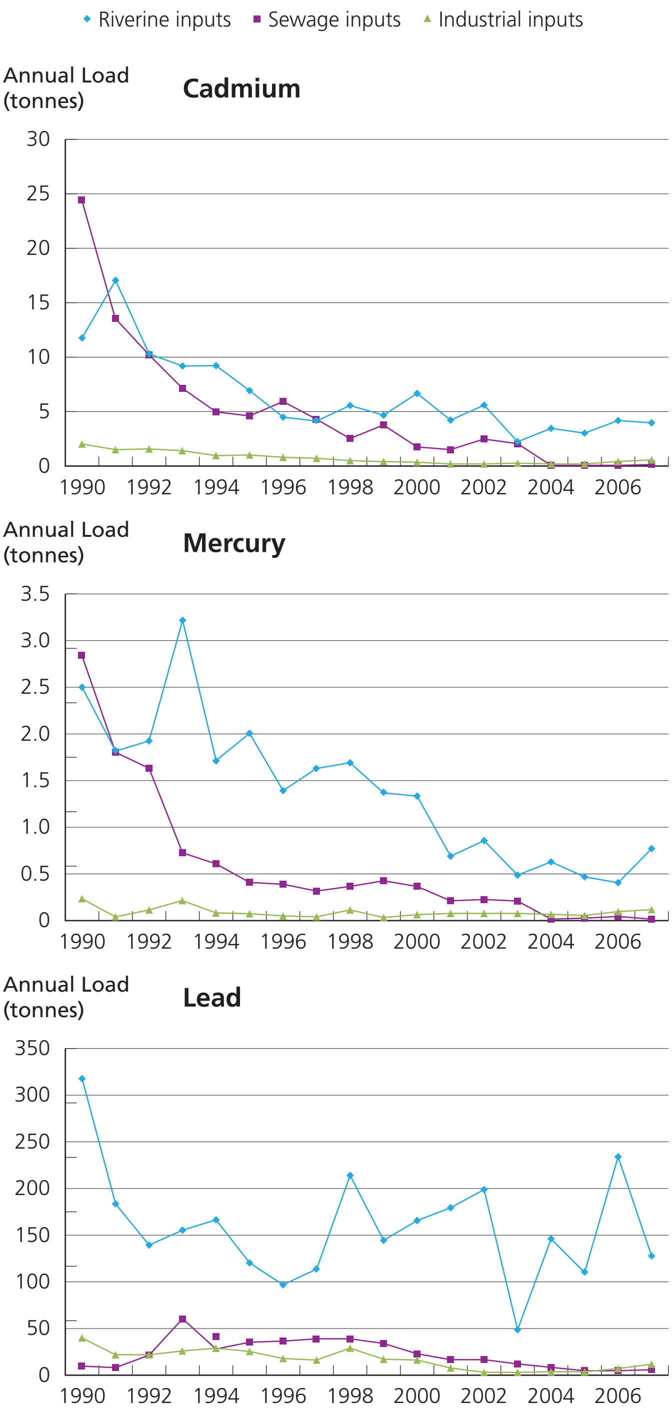
The sampling and analysis has been carried out by the Environment Agency, the Scottish Environment Protection Agency and the Department of the Environment in Northern Ireland using agreed sampling protocols in accredited laboratories. However, since the programme began in 1990, analytical methods have developed and become more sensitive, and detection limits have become lower. Also, particularly for direct discharges, if the value from a particular site has been below the detection limit for a number of years, analysis for that particular determinand may be stopped for financial reasons. When examining the riverine inputs data, it must be borne in mind that the loads are based on the river flows, which are significantly affected by rainfall patterns. High rainfall also brings substances arising from diffuse sources into the rivers through run-off and high flows and associated flood events can also stir up riverine sediments which can hold reservoirs of the substances concerned here and which are all persistent. An indication of the variability of the annual riverine flows is given in Figure 3.3. Such factors mean that the trends and reductions presented in the following sections must be treated with great caution. However, taken as a whole, the information, although not perfect, does give a useful picture of the reductions in hazardous substances entering the marine environment through riverine and direct discharges between 1990 and 2007.

Riverine inputs

Mercury, Cd, lindane and PCB inputs to both the Celtic Seas and the Greater North Sea have declined since 1990, although Cd inputs to the Celtic Seas have not decreased over the past ten years. This reflects the control measures which have been introduced for these substances. The use of both lindane and PCBs has been phased out, and Hg and Cd are subject to cessation targets warranting strict controls.

Copper inputs to the Greater North Sea have also declined since 1990, but there has been no decrease in Cu inputs to the Celtic Seas. Pb

Figure 3.1 Annual loads of heavy metals, lindane and polychlorinated biphenyls into the Celtic Seas between 1990 and 2007. © Crown copyright 2010.



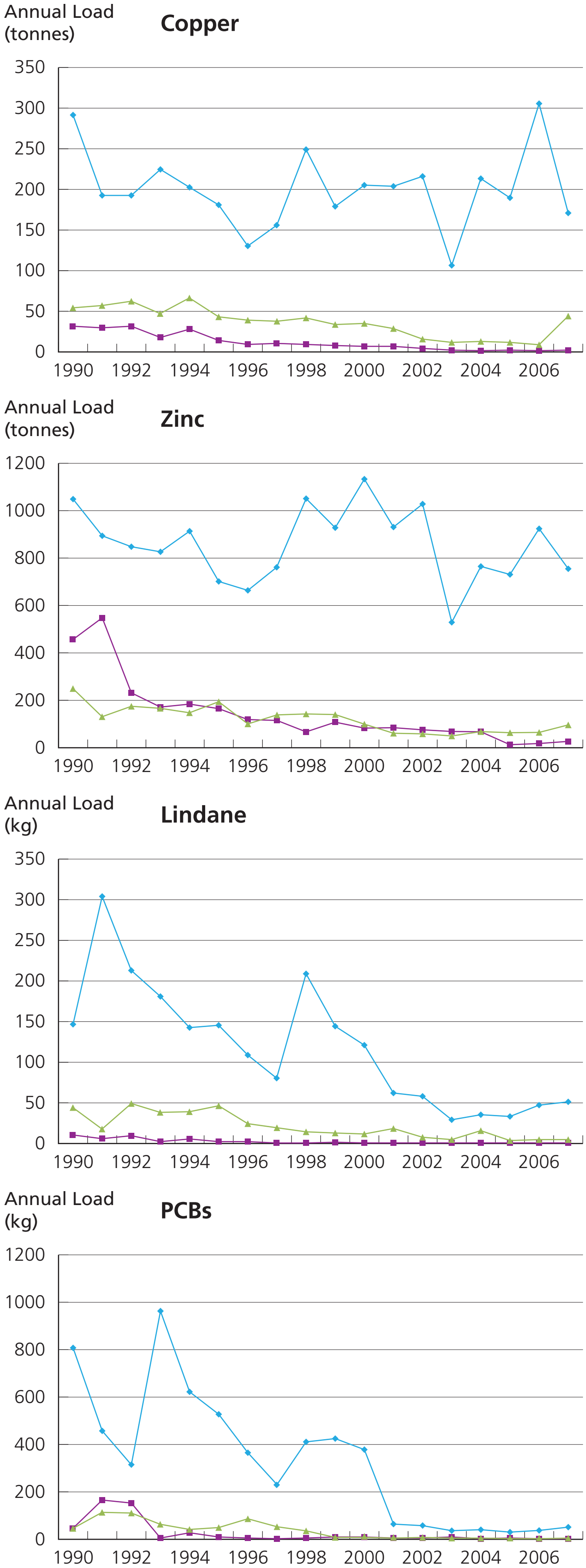
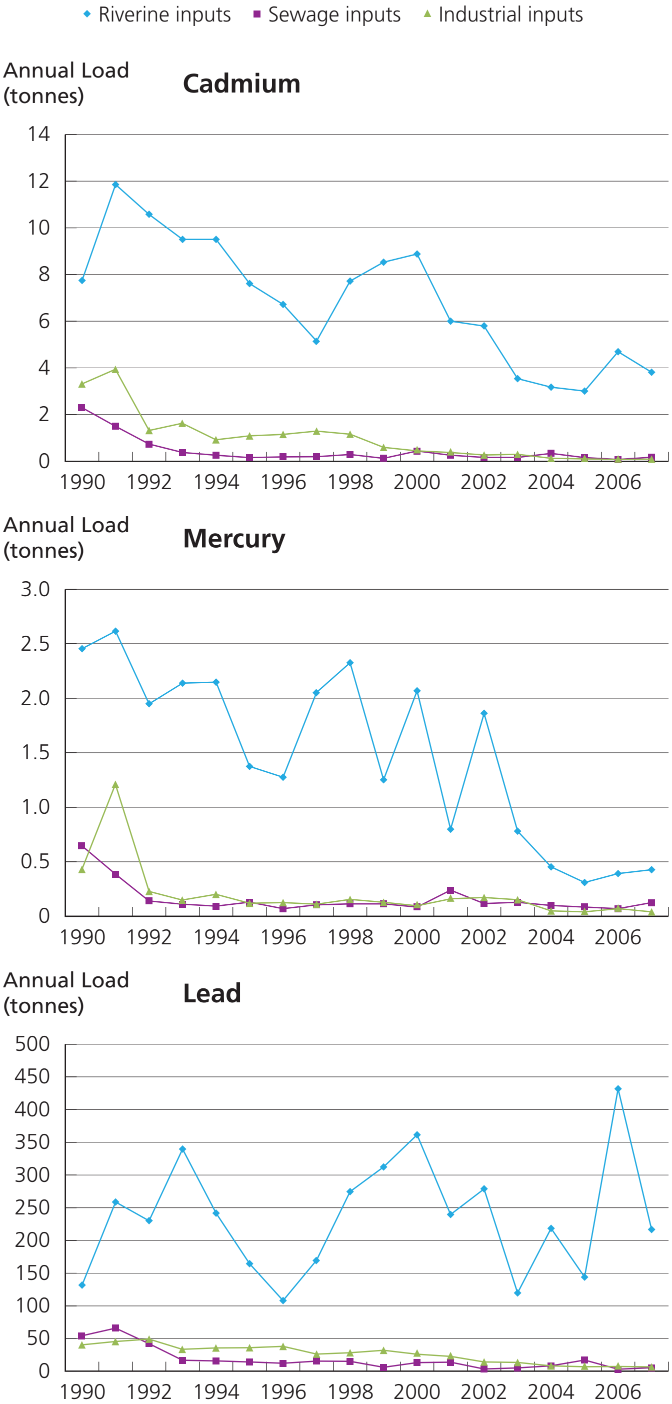


Figure 3.2 Annual loads of heavy metals, lindane and polychlorinated biphenyls into the Greater North Sea between 1990 and 2007. © Crown copyright 2010.



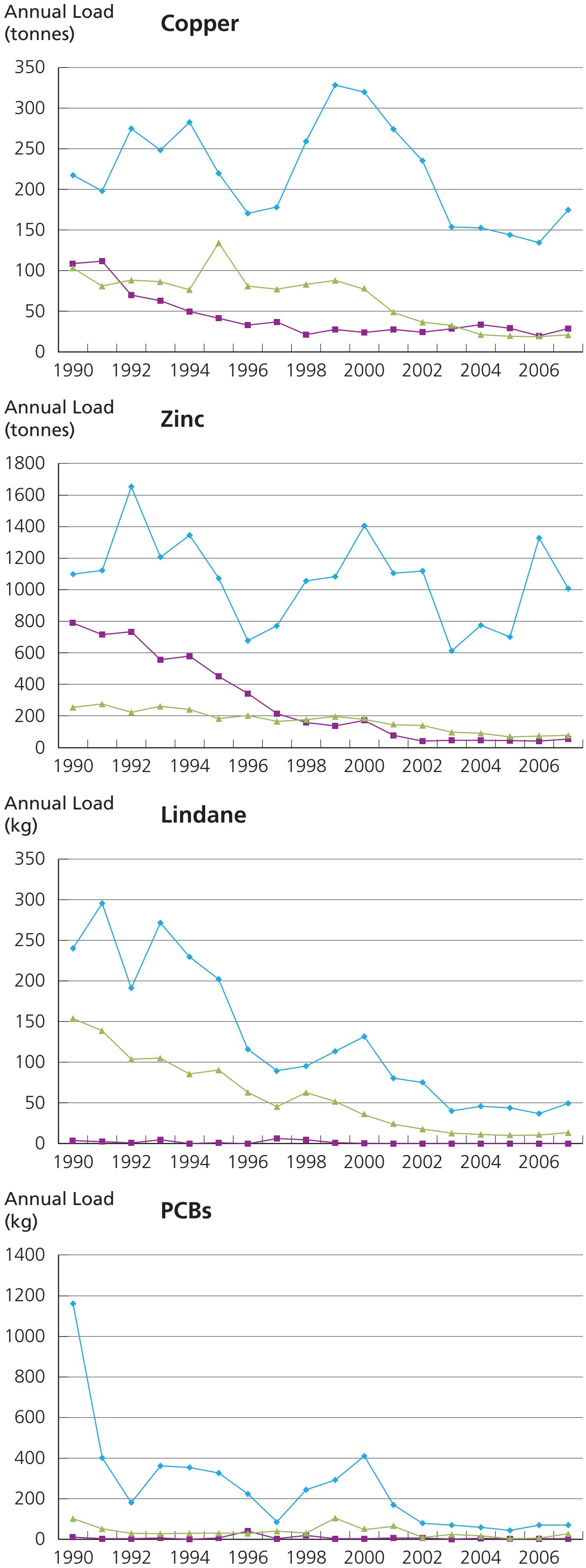
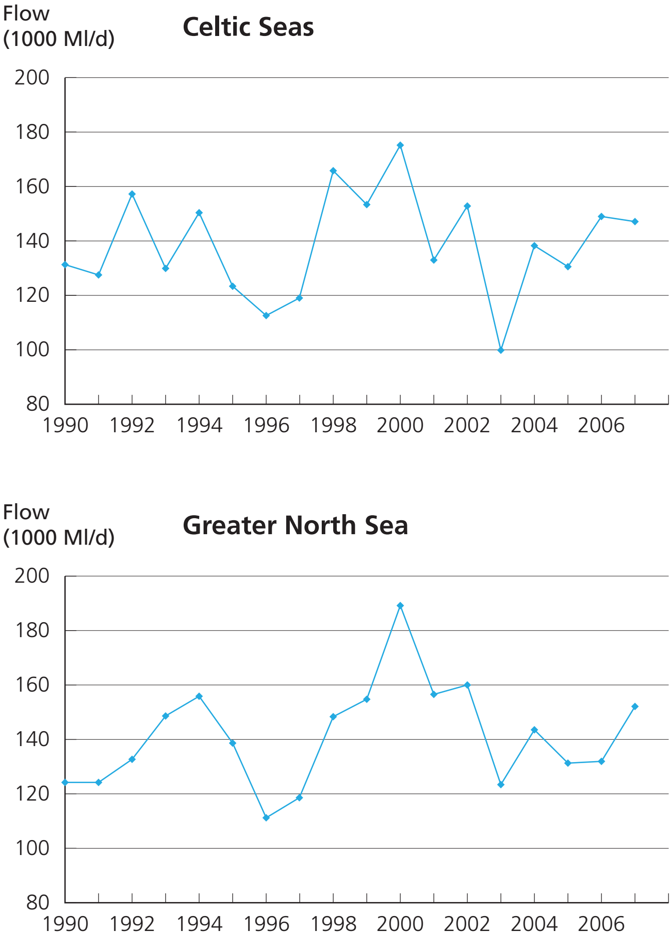


Figure 3.3 Annual riverine flows for the Celtic Seas and Greater North Sea, 1990–2007. © Crown copyright 2010.



inputs to both the Greater North Sea and the Celtic Seas have remained stable since 1990. Having adjusted for flow, there is evidence that Zn inputs to the Celtic Seas have declined since 1990, but have been stable over the past ten years. Zn inputs to the Greater North Sea have been stable since 1990. These elements all occur naturally in the UK and there will be a significant background level in the rivers and a significant reservoir in sediments due to historical use. Also, Cu and Zn have been widely used in materials (e.g. galvanised roofing and water pipes) which will leach significant amounts into the environment for years to come.

Direct discharges

Direct inputs from sewage treatment works and industrial installations are covered together, as the patterns and trends are similar. Generally, the direct discharges are much lower than those from rivers, and the flows are less influenced by rainfall patterns. Currently, the flows for direct discharges are only available for the past few years. In many cases, concentrations are below the limit of detection. In most cases, there have been decreasing inputs since 1990 and in some cases the loads are approaching zero. Over the past ten years, industrial inputs of heavy metals to the Greater North Sea have tended to stabilise, although inputs to the Celtic Seas have continued to decrease. Conversely, sewage inputs to the Celtic Seas have stabilised, while continuing to decrease in the Greater North Sea. These findings reflect the progressive application of best available techniques at industrial installations and permitting controls for sewage effluents, but the results need to be treated with caution due to changing detection limits. In many cases the reduction curves have flattened out, suggesting that further reductions will only occur when there are step-changes in abatement technology.

Differences between the sub-regions

There appear to be no major differences in the patterns and trends between the Celtic Seas and the Greater North Sea.

3.1.2.1.5 Progress towards the vision of clean and safe seas

For the substances covered, the UK vision of clean and safe seas would be achieved if their concentrations in the marine environment were causing no significant adverse effects on populations and that any impacts were restricted. The information on riverine and direct

discharges is not enough to be able to address this question directly, as it deals with loads entering the marine environment. However, the levels and trends of these loads do give an important indication of how the control measures required to deal with the discharges, emissions and losses of hazardous substances are working.

Lead, Cd, and Hg are priority hazardous substances under the EU Water Framework Directive. PCBs are not currently listed as priority substances, but are under consideration as potential candidates under Annex III of 2008/105/EC and lindane is a priority hazardous substance under both OSPAR and the EU Water Framework Directive. Under the OSPAR Hazardous Substances Strategy, this means that they are subject to cessation targets. The target in OSPAR is to make ‘every endeavour to move towards the target of the cessation of discharges, emissions and losses of hazardous substances by the year 2020’. The Water Framework Directive requires Member States to aim to cease or phase out emissions, discharges and losses of priority hazardous substances within 20 years of proposals being adopted bearing in mind relevant product and process controls which will be set up to achieve this (i.e. by 2029).

At this stage it is not possible to forecast whether these targets will be achieved. However, the information presented in Figures 3.1 and 3.2 suggests that at the sub-region level, the discharges, emissions and losses from sewage treatment works and industrial installations are heading in the right direction. The picture for riverine inputs is less clear, particularly for Pb and further work will need to undertaken on how natural background concentrations and reservoirs stored in sediments can be taken into consideration in the achievement of these targets.

3.1.2.2 Atmospheric inputs

3.1.2.2.1 Heavy metals

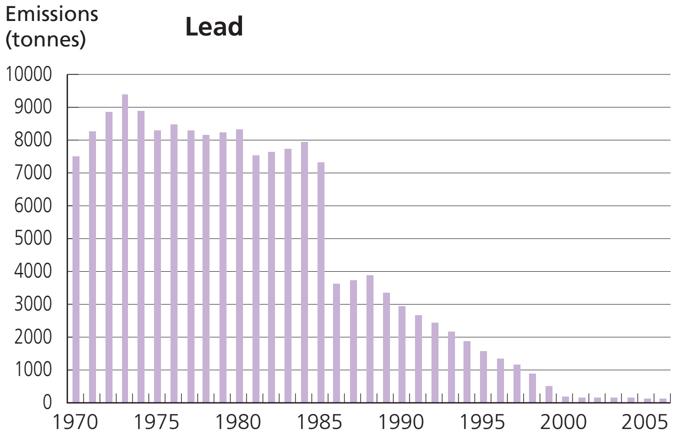
Emissions

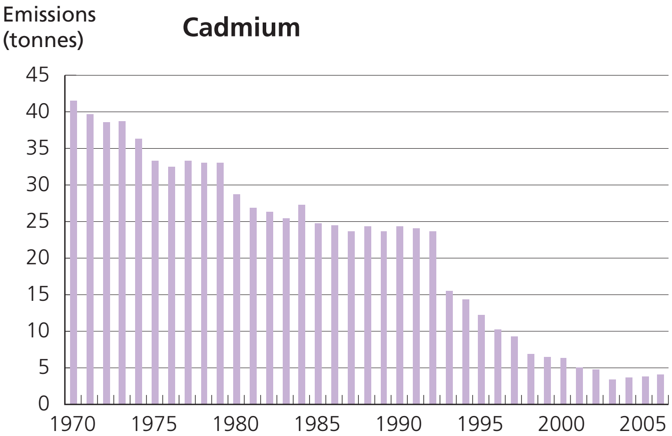
Figure 3.4 illustrates the trend in heavy metal emissions to the atmosphere since 1970. With the exception of Hg, heavy metals are transported through the atmosphere in association with particulates and so deposit in the same manner as the particles with which they are associated. The principal sources of emissions to the atmosphere are fuel use and the metals industry. Improvements in emission control and changes in the economic activity in the UK have driven the change in UK emissions. Most noticeable are the step changes in Pb emission arising from the reduction in the Pb content of petrol in 1995 and the subsequent phasing out of leaded petrol. Since 1970, emissions from coal combustion have decreased for all the metals both as a result of reduced quantities of fuel burnt and improved abatement for most of the remaining combustion plant.

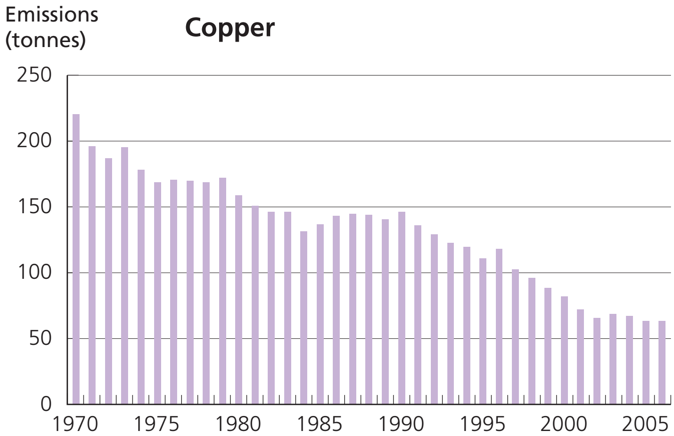
While some Hg is associated with particulates the majority is present in the environment in the vapour phase. Elemental Hg vapour has a very small, approaching zero, net deposition rate and so has a very long atmospheric lifetime. Outgassing of Hg from previous deposition can also occur. Hence away from the major industrial sources (such as the former Pb-Zn smelter at Avonmouth and chloralkali plants) air concentrations of Hg are dominated by the global background level. However emissions from industrial sources have declined markedly since 1970.

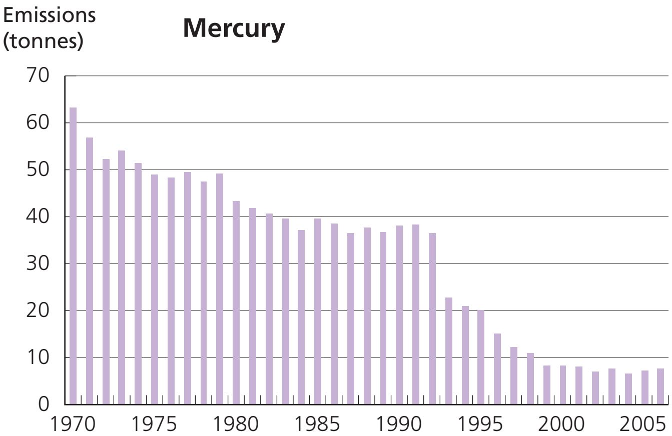
Between 1990 and 2007, anthropogenic emissions of Cd decreased by 84%, Cu by 57%, Pb by 96%, Zn by 55% and Hg by 80%.

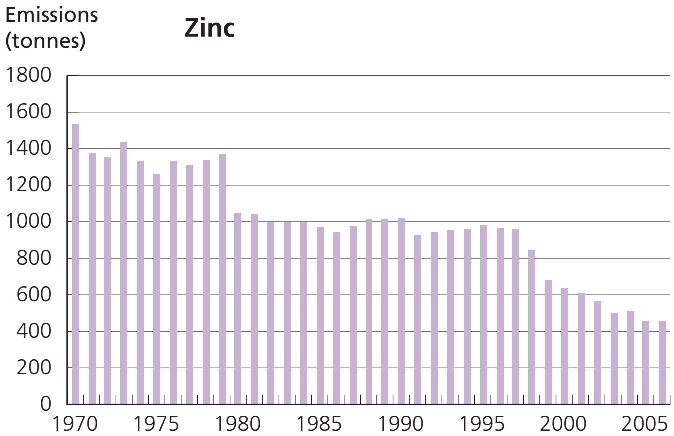
Figure 3.4 Emissions of heavy metals in the UK between 1970 and 2006. © AEA Technology.











Between 2002 (the latest date for information in *Charting Progress*) and 2007, emissions of Cd decreased by 16%, Cu by 3%, Pb by 25% and Zn by 20%. While industrial change is responsible for some of these decreases improved abatement at the remaining sources has also contributed. However emissions of Hg increased by 9% as a result of higher emissions from coal-fired power stations.

Modelling of atmospheric transport

Modelling of the atmospheric transport of Cd, Pb and Hg has been carried out by the UNECE European Monitoring and Evaluation Programme's Meteorological Synthesising Centre – West (MSC-W) based at the Norwegian Meteorological Institute in Oslo. The work has been undertaken on behalf of contracting

parties to the UNECE Convention on Long-Range Transboundary Air Pollution. The model calculates the air concentrations and resulting deposition of metals to Europe and the North West Atlantic based on officially reported emissions inventories and knowledge of the particle size distribution for individual metals. The modelled concentrations are regularly compared with monitoring data. The data outputs provided by the MSC-W were used to estimate the atmospheric inputs of Cd, Pb and Hg into each of nine Regional Reporting areas by interpolating from the 50 km grid to a 1 km grid and then summing the deposition in each area. Estimates were provided of the contribution of the UK source of the metal in question to the deposition at any one point, relative to that from natural sources.

Figure 3.5 shows the mass of Cd, Pb and Hg deposited to each of the sea areas divided between the contribution from the UK and from other sources. It is clear that the deposition rates are dominated by resuspension of natural material. This component is subject to considerable uncertainty. This is also the case for riverine discharges from several UK rivers, for which the input to the marine environment is dominated by geological material mobilised in sediments and as a result is dependent from year to year on changes in river flow and thus the rivers ability to mobilise sediment.

The scale of the inputs to the sea areas from rivers and coastal discharges are compared to the atmospheric inputs in Tables 3.1 to 3.3. It is evident that while for Pb and Hg the atmosphere represents the dominant input for most sea areas, this is less the case for Cd. However UK anthropogenic sources are only more than aquatic inputs for Hg in the Northern North Sea and Southern North Sea and of course for the sea areas without a coastline.

Figure 3.5 Total deposition of cadmium, lead and mercury to UK sea areas in 2006. © AEA Technology.

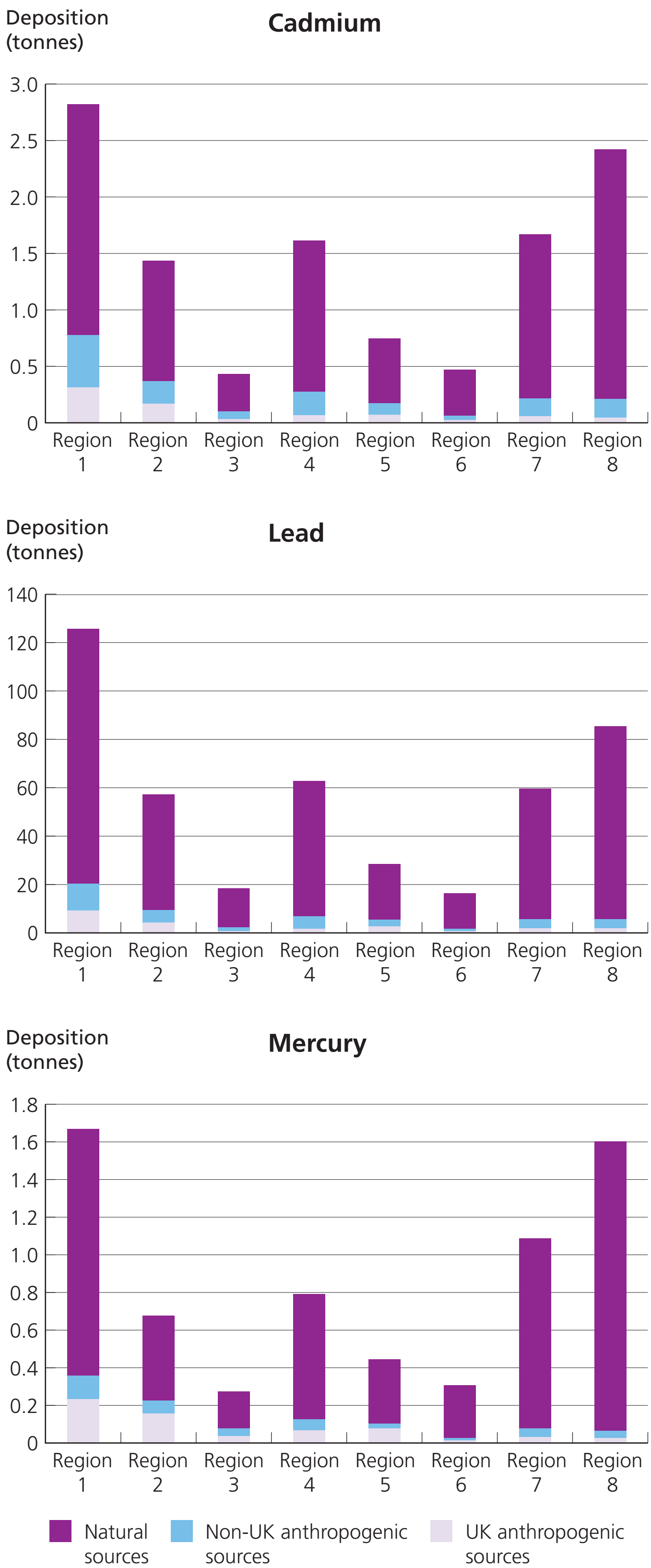


Table 3.1 Comparison of the aquatic and atmospheric inputs of cadmium to the UK sea areas, including inputs from the rivers Bann and Foyle which are shared with the Irish Republic

Sea area	*RID*	*UK anthro-pogenic sources*	*Non-UK anthro-pogenic sources*	*Natural sources*	*Total from air*	*Total input*	*Air as % of total*	*UK sources as % of atmosp-heric inputs*
	t/y	*t/y*	*t/y*	*t/y*	*t/y*	*t/y*		
Northern North Sea	3.53	0.31	0.47	2.03	2.81	6.34	44	11.1
Southern North Sea	0.94	0.17	0.20	1.06	1.43	2.37	60	11.8
Eastern Channel	0.14	0.03	0.07	0.34	0.44	0.58	76	7.0
Western Channel and Celtic Sea	1.54	0.07	0.21	1.33	1.61	3.15	51	4.3
Irish Sea	2.30	0.07	0.10	0.58	0.75	3.04	24	9.8
Minches and Western Scotland	0.31	0.02	0.04	0.40	0.46	0.77	60	4.9
Scottish Continental Shelf	0	0.06	0.16	1.45	1.67	1.67	100	3.4
Atlantic North-West Approaches	0	0.05	0.16	2.20	2.41	2.41	100	2.0

Table 3.2 Comparison of the aquatic and atmospheric inputs of lead to the UK sea areas, including inputs from the rivers Bann and Foyle which are shared with the Irish Republic.

Sea area	*RID*	*UK anthro-pogenic sources*	*Non-UK anthro-pogenic sources*	*Natural sources*	*Total from air*	*Total input*	*Air as % of total*	*UK sources as % of atmosp-heric inputs*
	t/y	*t/y*	*t/y*	*t/y*	*t/y*	*t/y*		
Northern North Sea	379	9.1	467	105.4	582	961	61	1.57
Southern North Sea	49	3.9	198	48.0	250	299	83	1.58
Eastern Channel	7.3	0.4	68	15.3	84	91	92	0.50
Western Channel and Celtic Sea	61	1.3	211	55.5	268	329	81	0.50
Irish Sea	168	2.5	96	23.4	122	289	42	2.07
Minches and Western Scotland	11	0.5	39	14.6	54	64	83	1.03
Scottish Continental Shelf	0	1.5	158	53.9	213	213	100	0.72
Atlantic North-West Approaches	0	1.3	162	79.5	243	243	100	0.54

Table 3.3 Comparison of the aquatic and atmospheric inputs of mercury to the UK sea areas, including inputs from the rivers Bann and Foyle which are shared with the Irish Republic.

Sea area	RID	UK anthro-pogenic sources	Non-UK anthro-pogenic sources	Natural sources	Total from air	Total input	Air as % of total	UK sources as % of atmosp-heric inputs
	t/y	t/y	t/y	t/y	t/y	t/y		
Northern North Sea	0.289	0.232	0.122	1.317	1.670	1.959	85	13.9
Southern North Sea	0.142	0.154	0.074	0.447	0.675	0.817	83	22.9
Eastern Channel	0.083	0.040	0.034	0.198	0.273	0.356	77	14.8
Western Channel and Celtic Sea	0.136	0.065	0.061	0.662	0.788	0.924	85	8.2
Irish Sea	0.313	0.076	0.027	0.346	0.449	0.761	59	16.9
Minches and Western Scotland	0.050	0.016	0.011	0.276	0.303	0.353	86	5.4
Scottish Continental Shelf	0	0.035	0.042	1.008	1.086	1.086	100	3.3
Atlantic North-West Approaches	0	0.026	0.042	1.537	1.605	1.605	100	1.6

Figure 3.6 compares the deposition rate (mass per unit area per unit time) for Hg and Pb. Hg exhibits very limited variation between sea areas as a result of the long atmospheric half-life and global atmospheric transport of Hg. Lead shows a more marked variation between the North Sea and the remote sea areas owing to the more pronounced deposition of the particles with which the Pb is associated nearer the coast.

3.1.2.2.2. Persistent organic pollutants

Emissions

Emissions data are provided in Figure 3.7 for two persistent organic pollutants (POPs); lindane (γ–hexachlorocyclohexane with a purity greater than 99%) and PCBs expressed as the total mass emitted. The major source of lindane over this period is estimated to have been emissions from wood treated before the substance was banned in the mid-1990s. Release of PCBs is thought to be dominated by leakage from closed appliances in which they were contained before new uses were controlled in 1986. The sharp decrease in PCBs emissions in 2000 is based on the need for identified PCBs to have been destroyed, however given the difficulty of identifying components containing PCBs, emission is assumed to continue.

Measurements of concentrations in air

There are currently no measurements in the UK of lindane in the atmosphere. Measurements of PCBs have been carried out since 1991 at a limited number of sites with a gap in monitoring between 1995 and 1999. High Muffles is a rural coastal site in North Yorkshire, data from which is reported to OSPAR for use in their Comprehensive Atmospheric Monitoring Programme (CAMP). The measurement results for this period are shown in Figure 3.8. Congeners below the detection limit are given at the detection limit.

Figure 3.6 Rate of deposition of mercury and lead to UK sea areas. © AEA Technology.

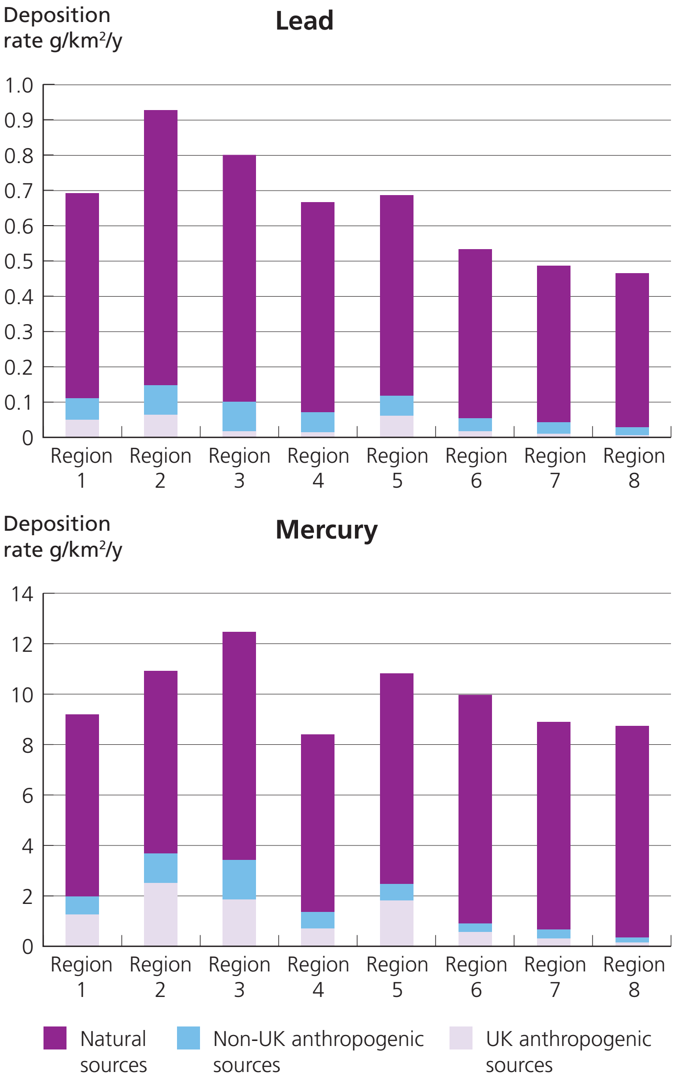
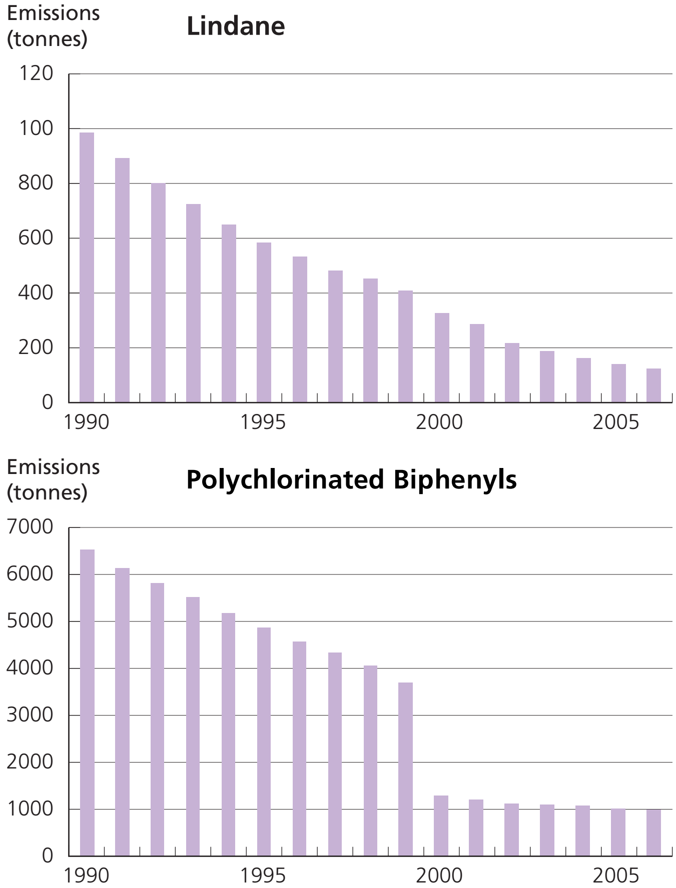


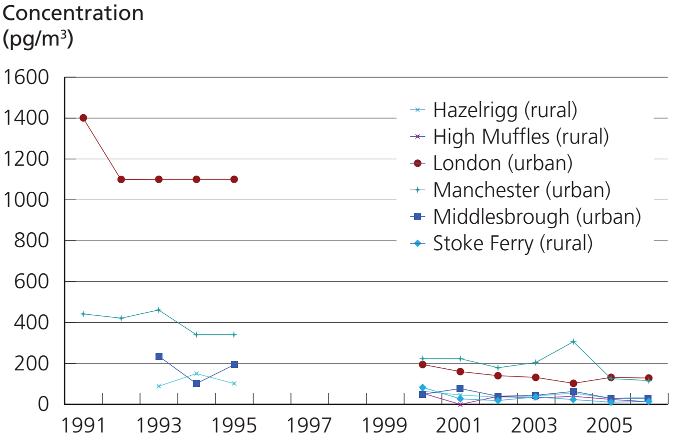
Figure 3.7 Emissions of lindane and polychlorinated biphenyls, 1990–2006. © AEA Technology.



Modelling of atmospheric transport

Modelling of the atmospheric transport of lindane has been carried out by the UNECE European Monitoring and Evaluation Programme's Meteorological Synthesising Centre – East (MSC-E), based in Moscow. The work has been undertaken on behalf of contracting parties to the UNECE Convention on Long-Range Transboundary Air Pollution. The model calculates the deposition of oxidized and reduced nitrogen species to Europe and the North West Atlantic. The data outputs provided by the MSC-E were used to estimate the atmospheric inputs of lindane to each of the eight Regional Reporting areas by interpolating from the 50 km grid to a 1 km grid and then summing the deposition in each area. The results of the atmospheric modelling are shown in Figure 3.9 as the rate of lindane deposition to each sea area and in Figure 3.10 as the total deposition of lindane to each sea area. Figure 3.9 shows that sea areas nearest to the major land masses have the highest deposition rates, especially where the sea area is more southerly. The Scandinavian countries controlled lindane use earlier than the UK and France. The north-westerly sea areas show the lowest deposition rates as a result of their geographic location. However, Figure 3.10 shows that

Figure 3.8 Measured concentrations of the total of a group of selected PCBs at UK monitoring sites, 1991–2006. © AEA Technology.



The selected PCB congeners are the group known as the ICES 7; 2,4,4′ trichlorobiphenyl (CB28), 2,5,2′,5′ tetrachlorobiphenyl (CB52), 2,4,5,2′,5′pentachlorobiphenyl (CB101), 2,4,5,3′,4′ pentachlorobiphenyl (CB118), 2,3,4,2′,4′,5′ hexachlorobiphenyl (CB138), 2,3,4,2′,4′,5′ hexachlorobiphenyl (CB153), 2,3,4,5,2′,4′,5′ heptachlorobiphenyl (CB180).

Figure 3.9 Deposition rate of lindane to each sea area showing the UK and non-UK deposition separately. © AEA Technology.

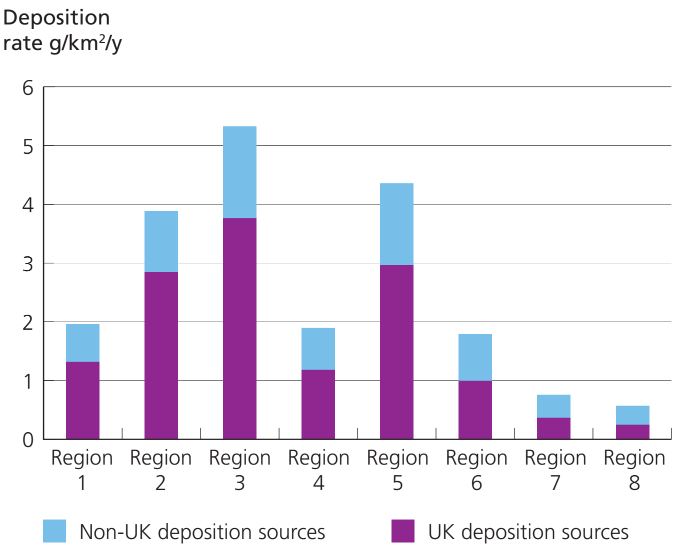
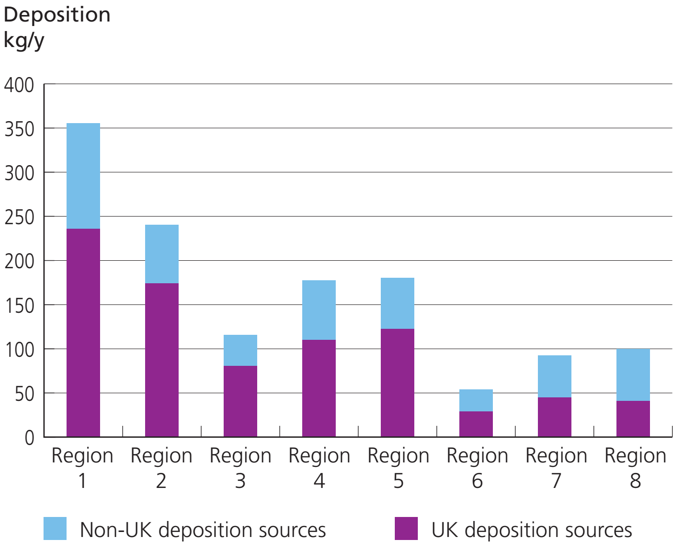


Figure 3.10 Total deposition of lindane to each sea area in 2006. © AEA Technology.



the Northern North Sea has the largest total deposition of lindane as it has the largest area of the coastal sea areas.

Modelling has only been carried out for one of the 209 PCB congeners and so, in the absence of an appropriate comparator, this has not been included.

Lindane deposition from the atmosphere is compared in Table 3.4 with inputs to the relevant sea areas from the coastline and rivers. It is apparent that the atmosphere contributes a small fraction of the lindane inputs to the sea areas, except where there is no significant coastline. However, of the atmospheric input, the majority for all coastal sea areas is estimated to come from the UK. Future emissions to the atmosphere of lindane are projected to decrease as treated materials are taken out of service and destroyed.

Table 3.4 Comparison of the aquatic and atmospheric inputs of lindane to the UK sea areas, including inputs from the rivers Bann and Foyle which are shared with the Irish Republic.

Sea area	RID	UK anthro-pogenic sources	Non-UK anthro-pogenic sources	Total from air	Total input	Air as % of total	UK sources as % of atmosp-heric inputs
	t/y	t/y	t/y	t/y	t/y		
Northern North Sea	22	0.24	0.12	0.36	22.00	1.62	67
Southern North Sea	19	0.18	0.07	0.24	18.74	1.29	73
Eastern Channel	3.2	0.08	0.03	0.12	3.35	3.49	71
Western Channel and Celtic Sea	19	0.11	0.07	0.18	18.78	0.95	62
Irish Sea	21	0.12	0.06	0.18	20.72	0.88	68
Minches and Western Scotland	13	0.03	0.02	0.05	12.99	0.42	56
Scottish Continental Shelf	0	0.05	0.05	0.09	0.09	100	49
Atlantic North-West Approaches	0	0.04	0.06	0.10	0.10	100	41

3.1.3 Concentrations in the Marine Environment

3.1.3.1 Metals in seawater

3.1.3.1.1 Summary

Of the measured metal concentrations in water 98.6% are below the UK environmental quality standards (EQSs) for 2007, indicating very limited toxicological effect. Some industrial areas have higher concentrations of several metals, specifically the Thames Estuary, The Wash and the Dee Estuary. Concentrations should continue to be monitored and necessary action taken to maintain levels well below the EQS values. New standards have now been developed for use in assessments under the EU Water Framework Directive (WFD), these the potential to increase the number of occasions when the EQS limits are not met.

3.1.3.1.2 Introduction

Trace metals are present naturally in coastal waters and are mostly derived from the underlying geology. Anthropogenic inputs of metals to coastal waters are largely derived from industrial discharges and sewage effluents. Metals can induce toxicological effects when present at high concentrations, and in seawater, dissolved metals can easily bioaccumulate in organisms. Of particular concern are shellfish, these are filter feeding organisms which amass chemicals directly from the waters in which they grow. Driven by the EU Shellfish Waters Directive, monitoring takes place with the aim of protecting or improving shellfish waters in order to support shellfish life and growth.

3.1.3.1.3 Developments since *Charting Progress*

There have been no updates to the ranges of BACs for metals in water provided by OSPAR since 1997. Standards have now been developed for use in assessments under the WFD. The use

Table 3.5 BAC and EQS dissolved metal standards. Revised WFD standards are shown for comparative purposes.

Metal	*EQS (annual average, µg/l)*	*WFD Priority Substances Daughter Directive annual average (µg/l)*	*BAC[a] (µg/l)*
Cd	2.5		0.005 – 0.025
Cr	15		0.09 – 0.12
Cu	5		0.05 – 0.36
Hg	0.3	0.05	0.0001 – 0.0005
Ni	30	20 (interim pending Risk Assessment)	0.12 – 0.26
Pb	25	7.2 (interim pending Risk Assessment)	0.005 – 0.02
Zn	40		0.03 – 0.45

a Range of values present over the following inclusive regions: Atlantic Ocean, Northern North Sea, Southern North Sea, English Channel, Celtic Sea.

of shellfish waters data in this assessment allows contaminants to be evaluated over greater spatial scales than was the case for *Charting Progress*.

3.1.3.1.4 Presentation of the evidence

Data collected in 2007 under the EU Shellfish Waters Directive were used for this assessment and compared against the relevant standards for dissolved metals in water. Table 3.5 shows the UK EQS limits as officially adopted by Defra; the revised WFD standards are shown for comparative purposes. The BACs are as adopted by OSPAR and it is the maximum value of each range against which the monitoring data have been compared.

It should be noted that the Cr data are a mixture of Cr(III) and Cr(VI) ions, although it has been assumed that the data comprise Cr(VI) ions only.

Most of the sites were sampled on two occasions; although some stations were sampled more frequently. Where results are below the limit of detection (LOD) of the methodology, the LOD has been compared against standards (Table 3.6). The number of results that were below the LOD is given so that the reader can evaluate the significance of this approach on the assessment. Spatial trends for Cu and Zn are based on the median and maximum value measured at each station. The maximum value (therefore a conservative approach) for each station is shown in Figure 3.11.

Most of the metals investigated (Cr, Cd, Cu, Hg, Ni, Pb, Zn) fall below the recommended EQS values as identified in previous studies (Marine Environment Monitoring Group, 2004; Defra et al., 2005). Figure 3.11 shows that concentrations of Cu and Zn are generally low in Scottish coastal waters and increase towards the south of the UK where a minority of sites have concentrations above the EQS, which may reflect local conditions. Significant relationships between salinity and trace metal concentrations are not observed in the study area, suggesting that other more local factors and differences in the behaviour of trace metals across salinity gradients are of greater significance.

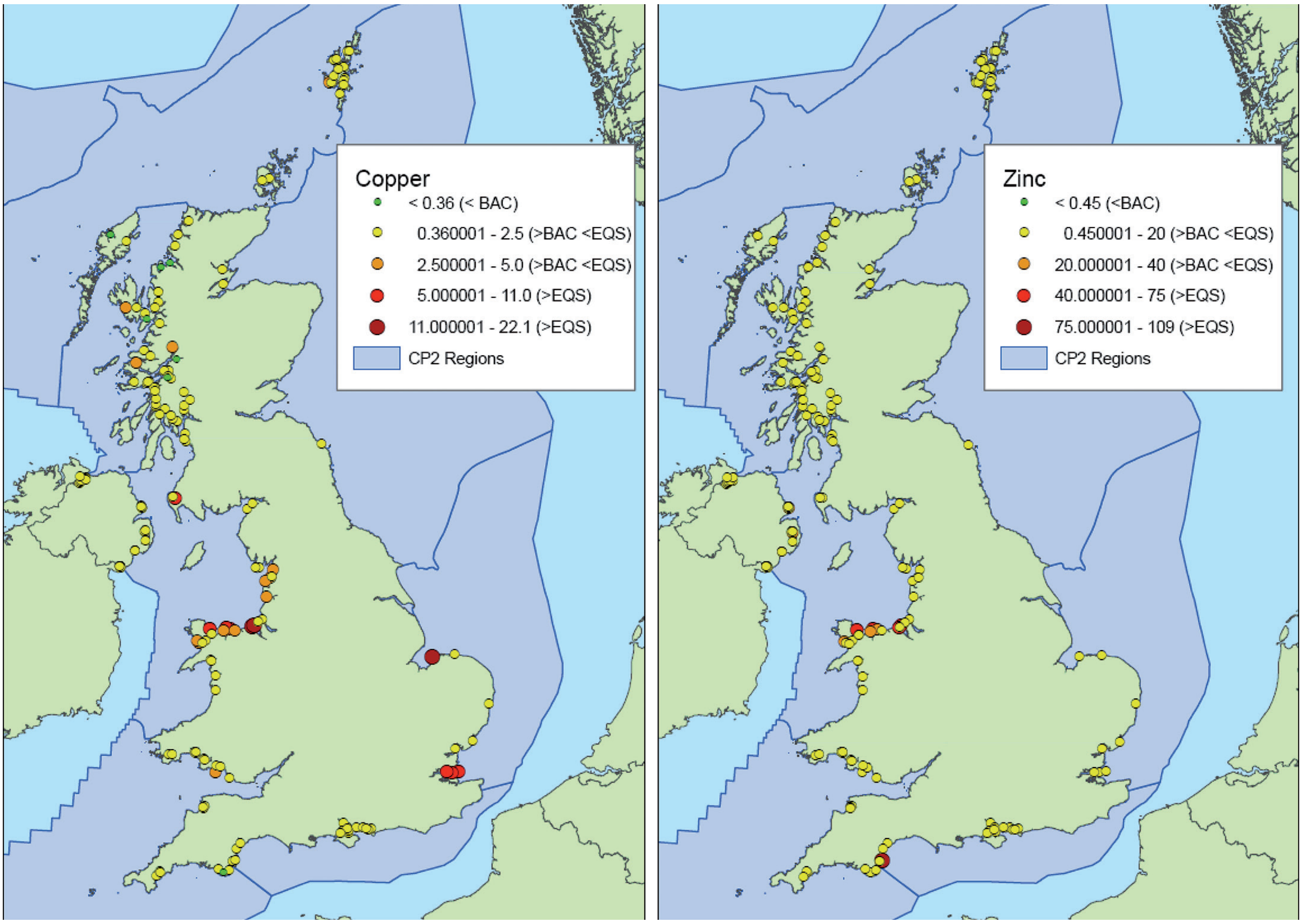
Copper

Copper is the trace metal with the greatest number of results above the EQS (6%). The areas with results above the EQS include one station in the Thames Estuary and one in The Wash and stations near to the Dee Estuary and the North Wales coast (Figure 3.11). Previous

Table 3.6 Assessment summary for metals in shellfish waters in 2007.

Metal	*Total number of results*	*Percentage of results below LOD*	*Percentage of results below BAC*	*Percentage of results between BAC and EQS*	*Percentage of results above EQS*
Cd	340	67	19	81	0
Cr(VI)	211	7	0	99.5	0.5
Cu	334	0	9	85	6
Hg	227	77	0	99	1
Ni	335	4	4	97	0
Pb	335	18	0	99.7	0.3
Zn	332	0.3	0.3	98.2	1.5

Figure 3.11 Copper and zinc concentrations (µg/l) in UK coastal waters in 2007. The plots show the maximum recorded value at each station. Data were collected under the EU Shellfish Waters Directive. © SEPA.



assessments did not identify the northern Welsh coastline as having high concentrations. However, there has been an increase in metal concentrations in sediment in the Dee Estuary, which may have impacted on local waters. Elevated Cu concentrations in The Wash sediments have been observed but a cause has not yet been established.

Zinc

Of the 332 Zn results, 98.2% are above the BAC but below the EQS suggesting that anthropogenic activities do cause increases in concentration but not to levels that could be considered detrimental to the ecosystem. 1.5% of results exceed the EQS, primarily in the Dee Estuary (Figure 3.11).

Chromium

Only one Cr result is above the EQS, and this was recorded at the same station in The Wash where Cu concentrations were also above the EQS.

Cadmium and mercury

Of the Cd results, 81% are above the BAC but below the EQS, indicating anthropogenic influence. The remaining 19% are below the BAC. 77% of Hg results are below the LOD so it difficult to make a true assessment against BACs, however all results are below the EQS with the exception of two at different stations in the Thames Estuary sampled in April 2007. The revised EQS under the WFD is for an annual average of 0.05 µg/l rather than 0.3 µg/l as at present in the UK, so there may be more failures to meet the new standard.

Nickel and lead

Most of the results for Ni and Pb are above the BAC but below the EQS. With the exception of one result, concentrations of Ni and Pb are below the revised interim WFD standards of 20 µg/l and 7.2 µg/l, respectively.

3.1.3.1.5 Need for further work

Further investigation and possible management of the areas identified as having high metal concentrations may be required.

3.1.3.1.6 Overall conclusion and forward look

Concentrations of trace metals in water are below the EQS, with the exception of a minority of sites particularly in the Dee Estuary, the North Wales coast and The Wash. Cu is the only trace metal to have a large number of exceedances of the EQS. New standards have been developed for use in assessments under the WFD, these have the potential to increase the number of occasions when the EQS limits are not met. Given that sampling sites used for this study are in coastal regions it is likely that concentrations at offshore sites are also below concentrations that are likely to cause toxicological effects.

3.1.3.2 Organics in seawater

3.1.3.2.1 Introduction

Organic contaminants are assessed in seawater to establish their distribution in surface waters around the UK coast. Organics are those carbon-based chemicals, such as solvents and pesticides which may enter the marine environment from industrial discharges, run-off from agriculture and pollution events. They include:

- *Polycyclic aromatic hydrocarbons*. PAHs are a group of chemicals that are formed during the incomplete burning of coal, oil, gas, wood, garbage, or other organic substances. PAHs generally occur as complex mixtures (e.g. as part of combustion products such as soot), not as single compounds. PAHs also occur naturally and are present in crude oil.

- *Organophosphorus pesticides*. Most organophosphates are insecticides. Organophosphates were developed about 200 years ago, but their insecticidal properties were only discovered in the 1930s. Some are very toxic, however, they usually are not persistent in the environment.
- *Organohalogens*. Chlorine organohalogens are the most common and are called organochlorines. Some common uses for organohalogens are as solvents, pesticides and coolants, fire-resistant materials, and components of adhesives and sealants. PCBs come under this heading. They are a group of toxic, persistent chemicals used as insulation in electrical transformers and capacitors, and for lubrication in gas pipeline systems. The sale and new use of PCBs were banned in 1981, and their use in existing equipment was banned in 2000. Other organohalogen compounds, brominated flame retardants (BFRs), are added to products in order to make them more fire-resistant.

The main drivers for monitoring organic contaminants in seawater are the EU Water Framework Directive (WFD), the Dangerous Substances Directive (DSD) and the Shellfish Waters Directives (SWD).

Organics monitored under the SWD are γ-HCH (lindane), dieldrin, DDT and parathion. The list for those substances monitored under the DSD is much more extensive and includes those substances that can potentially harm human health, aquatic life and water quality. Only those actually contained within each individual associated discharge are monitored at each location, usually managed through the consenting process. For the WFD, quality standards for surface waters of dangerous chemical substances include 33 priority and priority hazardous substances. Each of these Directive requirements has chemicals in common and existing EQS values are used where possible.

Reviews of contaminant status are made annually under the DSD and SWD, and in each three-year reporting round for the WFD, the most recent being 2009. The WFD classification reporting carried out in 2009 integrates with data collected for the DSD and SWD. The WFD results form the foundation of the *Charting Progress 2* assessment of the status of the UK seas with regard to contaminants.

Environmental impacts

Although many organics are present as naturally-occurring chemicals, many others are man-made and not required for the functioning of the marine ecosystem. In fact, they often interfere with the proper functioning of the ecosystem. Only when certain threshold (EQS) concentrations are exceeded, and if there is no continuing measurable improvement in reducing their concentration levels, are management efforts indicated.

The EQS concentrations depend on the relevant background concentration. The EQS for a substance is based on the toxicity of the substance. It defines a concentration in the water below which we can be confident that the substance will not have a polluting effect or cause harm to plants and animals. If the concentration in the water is lower than the EQS then pollution is considered to be eliminated.

Negative environmental impacts that may be linked to organic contaminants in seawater occur through a number of routes:

- Direct availability for uptake by humans/ wildlife through contact with contaminated seawater
- Increase in contaminants in sediments as contaminants attach to sediment particles
- Availability for uptake by fish, invertebrates, etc. Leading to increasing levels in biota with possible toxic effects
- Indirect availability for uptake by humans through the fish and shellfish food chain leading to possible toxic effects.

Individual impacts will vary with the chemicals in question but it is the degree to which a substance is able to damage an exposed organism that is important. This damage could either occur as an effect on the whole organism and/or as an effect on part of the organism, such as an organ (e.g. the liver) or a cell. Impacts include effects on the reproductive potential of an individual, neurological damage, and carcinogenic impacts, etc. These in turn affect the viability of a population, should the impact extend across a large number of individuals.

Socio-economic impacts

Direct socio-economic consequences of the negative environmental impacts of organic contaminants include loss to fisheries, either through reduced catches in the case of direct fish kills or loss of product in the case of shellfisheries when toxins have rendered the harvested shellfish unfit for human consumption. Decrease in biodiversity overall is a direct impact and not easily linked to economics, but this in turn can indirectly affect fisheries, should it cause a reduction in their food supply and may have amenity / recreational costs.

3.1.3.2.2 Developments since *Charting Progress*

Developments in key pressures, understanding and management measures

Long-term monitoring of UK waters has occurred relatively unchanged for SWD and DSD monitoring. However, as the chemical component of discharges alters in relation to the DSD, monitoring will alter to reflect the ongoing contaminants of concern. SWD monitoring has not altered in recent years as the sites and issues at the sites have remained relatively constant.

The introduction of the EU Habitats Directive required an assessment of the possible impact of discharges on protected sites, known as Natura 2000 sites. This was to ensure that the discharges did not have an adverse effect on these sites. There is a legal duty to ensure that no regulated discharges pose an unacceptable risk to these sites. For existing consents this meant they could be revoked, amended or remain unchanged as a result of these assessments. A number of changes resulted from the review of consents.

Developments in approach and indicators used

EQS values are still under discussion and development for some of the newer priority substances referred to in the WFD. Once established, the full programme of hazardous substances monitoring for organic compounds will be put in place.

Relevant research/activities

The aim of the EU's ambitious Marine Strategy Framework Directive (adopted in June 2008) is to protect more effectively the marine environment across Europe. It aims to achieve good environmental status of the EU's marine waters (out to 200 nautical miles) by 2020. One of the overarching descriptors of good environmental status is that the ...*concentrations of contaminants are at levels not giving rise to pollution effects*. It is expected that the UK Marine Monitoring and Assessment Strategy (UKMMAS) will be the route by which this monitoring and assessment is delivered in inshore waters. Current legislative drivers will not be sufficient to cover the organics monitoring in these waters and additional work is required to ensure offshore areas are adequately assessed.

3.1.3.2.3 Presentation of the evidence

Monitoring period and data sources

Monitoring information from various inshore programs and drivers for the period 2004 to 2008 have been used for the majority of the areas assessed.

Contaminant data provided for the WFD draft reporting in March 2008 are used for this assessment. Data used in these reports were available from three possible sources: DSD, SWD and WFD data. The WFD reports provide pass/fail assessment for chemical status, but are based on a reduced list of organic compounds since EQSs have not been set for all hazardous substances.

Adequacy and confidence of the data

The data were generally for surveys undertaken for existing drivers for the DSD and SWD and as such are not designed for full spatial coverage and have a limited frequency of sampling. Plus, the contaminants analysed vary between locations.

Trends in England and Wales

Not all of the 229 waterbodies for the WFD in England and Wales were provided with chemical classification in March 2008. The reason for this lack of data is that the monitoring strategy employed embraced the risk-based approach advocated in the directive. To this end, data will only be available for waterbodies into which priority substances were known to be discharged in significant quantities. There are therefore a number of waterbodies where no monitoring has been undertaken for priority substances.

The assessments can be split into two assessments of status; Annex VIII provides a generic grouping of polluting substances, while Annex X contains the priority list including priority and priority hazardous substances. Of the Annex VIII substances, 124 waterbodies were monitored for organics. Of these all 124 came out as High Status for Annex VIII concentrations. Of the Annex X organic substances, 116 waterbodies were monitored and 111 came out as of Good Status, with the 5 failing to achieve Good Status being for reasons of: HCH exceedances in the Medway, Mersey, Witham and Leven estuaries and indeno[1,2,3-*cd*]pyrene exceedance in the Stour Estuary.

Trends in Northern Ireland

Where available, the Northern Ireland Environment Agency (NIEA) used five years of data to classify NI Coastal and Transitional waterbodies for WFD chemical and physicochemical status. There were no exceedances of WFD EQSs for organic determinands.

As is the case for England and Wales, a combination of SWD, DSD and dedicated WFD monitoring data were used to make classifications for the draft River Basin

Management Plan. Supplementary data to address gaps and lack of coverage were incorporated in the March 2009 classification. Five years of SWD and DSD monitoring produced only one exceedance in HCH in 2006 in Lough Foyle. NIEA have not detected PCBs in the marine environment for 20 years. Organochlorine pesticides have not been detected at CSEMP water sampling sites since 1999, but will still be monitored once every six years within the marine surveillance suite as part of the river basin plans.

The organophosphorus pesticides simazine, atrazine and diazinon are still being detected on a sporadic basis, especially in the River Bann. Atrazine and simazine have been banned for some time and should ultimately disappear, although similar problems may arise with the herbicides that will be used to replace them.

Some chlorinated solvents are still being detected at CSEMP sites and this is because they are still in common use as degreasing agents and as dry cleaning solvents.

Pentachlorophenol (PCP) has not been detected at any CSEMP sites since 1999 and will be monitored once every six years in a marine surveillance suite.

NIEA are currently concluding year two of a three-year rolling programme to address a suite of 113 emerging pollutants and chemicals of concern, of which approximately 80% are organics.

Trends in Scotland

The Scottish Environment Agency (SEPA) routinely monitors organics in waters for the SWD and if, after two years of monitoring, all the results are below the LOD of the analysis and so there is no risk of failing the Directive standards then monitoring is discontinued (this is allowed under Article 7 of the Directive). Also, all SEPA background DSD and WFD monitoring results for organics are below analytical limit of detection. Therefore SEPA monitoring for trace organic compounds in Scottish waters will concentrate on monitoring in biota and sediments.

3.1.3.2.4 Need for further work

Further investigation and possible management of the areas identified as having high metal concentrations may be required.

3.1.3.2.5 Overall conclusion and forward look

Concentrations of trace metals in water are below the EQS, with the exception of a minority of sites particularly in the Dee Estuary, the north Wales coast and The Wash. Cu is the only trace metal to have a large number of exceedances of the EQS. When the revised WFD standards have been adopted, this may increase the number of occasions when the EQS limits are not met. Given that sampling sites used for this study are in coastal regions it is likely that concentrations at offshore sites are also below concentrations that are likely to cause toxicological effects.

3.1.3.3 Metals in sediments

3.1.3.3.1 Introduction

Sediments may act as a source or sink of trace metals in the environment depending on historical inputs and changes in the sediment regime within a defined area. Many trace metals are essential for normal biological activity; however some may induce toxicological effects when present at high concentrations. The exposure of the organism to trace metals is related to its ecology (such as feeding mechanisms) and the phase of sediment with which the trace metal is associated. Physiological responses to trace metals vary across organisms according to the ability of an individual to regulate body burdens.

3.1.3.3.2 Developments since *Charting Progress*

Further consideration of appropriate indicators against which to compare metal concentrations has been given within Europe and America and the latest criteria are used for this assessment. Many of the criteria have not been applied to metal concentrations in UK coastal sediments previously and their suitability for the underlying geology in the UK has not been fully examined. A new sampling methodology for sediments has been tested and implemented to reduce the effects of benthic heterogeneity and so increase statistical power to identify temporal trends.

3.1.3.3.3 Presentation of the evidence

Surficial sediments are widely used to assess trace metal contamination in shelf seas as they integrate the contamination status of the environment over a period of months. Higher concentrations of metals are usually found in fine-grained sediments because of the greater surface area to volume ratio that is available for interaction. Also, fine-grained sediment in the size range of silt and clay tends to be more reactive than coarse-grained carbonates and quartz sands (Windom et al., 1989). Therefore, when comparing the enrichment of trace metals in sediments over a large study area it is essential to compensate for these 'grain size' effects. Aluminium (Al) is employed as a reference element to 'normalise' for grain size differences because it is a conservative element unaffected by anthropogenic discharges and activities (Schropp et al., 1990) and a major component of the aluminosilicate matrix of clay minerals (Loring, 1991). Sites which are enriched with metals can be identified by high metal/Al ratios. This approach has been previously used throughout the UK to identify spatial trends in trace metal contamination (Balls et al., 1997; Rowlatt and Lovell, 1994; Charlesworth and Service, 2000).

Metal/Al ratios are compared to the BACs to identify if concentrations are 'close to background' and also against the Effects Range-Low (ERL) concentrations and Effects Range-Median (ERM) concentrations to identify if toxicological effects on marine organisms are 'negligible' or 'likely', respectively. The numbers of stations which fall into each category are given in Table 3.7. Figure 3.12 shows spatial trends in normalised metal concentrations in coastal sediments.

Table 3.7 shows that for all metals with the exception of Ni, fewer than 25% of the sites monitored had concentrations above that which would be likely to cause toxicological effects. The broad spatial trend for most metal/Al ratios is low ratios in Scotland and the western Irish Sea and higher ratios in England and Wales with a number of particular sites of concern in industrialized estuaries such as the Tees, Tyne, Thames, Severn and Mersey (Figure 3.12). This reflects higher industrial and riverine inputs to England and Wales and specifically to some estuaries.

Key to Figure 3.12 on Pages 64/65.
Circles are blue if the metal/Al ratio is significantly below the BAC; green if greater than the BAC but below the ERL; amber if above the ERL but below the ERM; and red if above the ERM. Statistically significant upward or downward trends are indicated by triangles; open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data.

Table 3.7 Assessment summary for metals in sediments.

	Number of stations	Percentage of stations below BAC	Percentage of stations between BAC and ERL	Percentage of stations between ERL and ERM	Percentage of stations above ERM
Cadmium	64	22	58	19	2
Mercury	67	3	18	55	24
Lead	69	12	9	59	20
Arsenic	69	43	0	38	19
Chromium	69	17	0	74	9
Copper	69	22	6	67	6
Nickel	69	25	0	16	59
Zinc	69	17	14	46	22

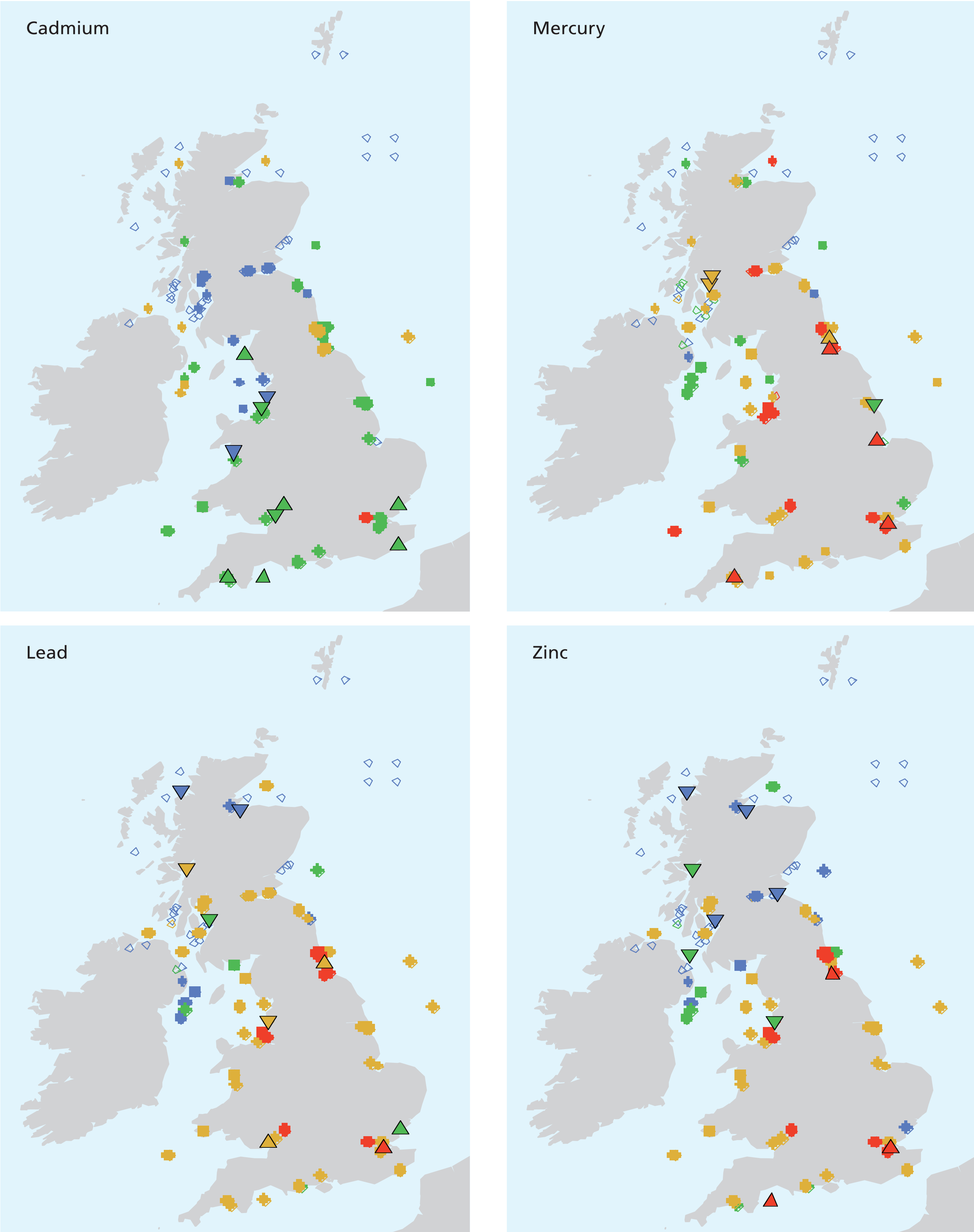
Cadmium

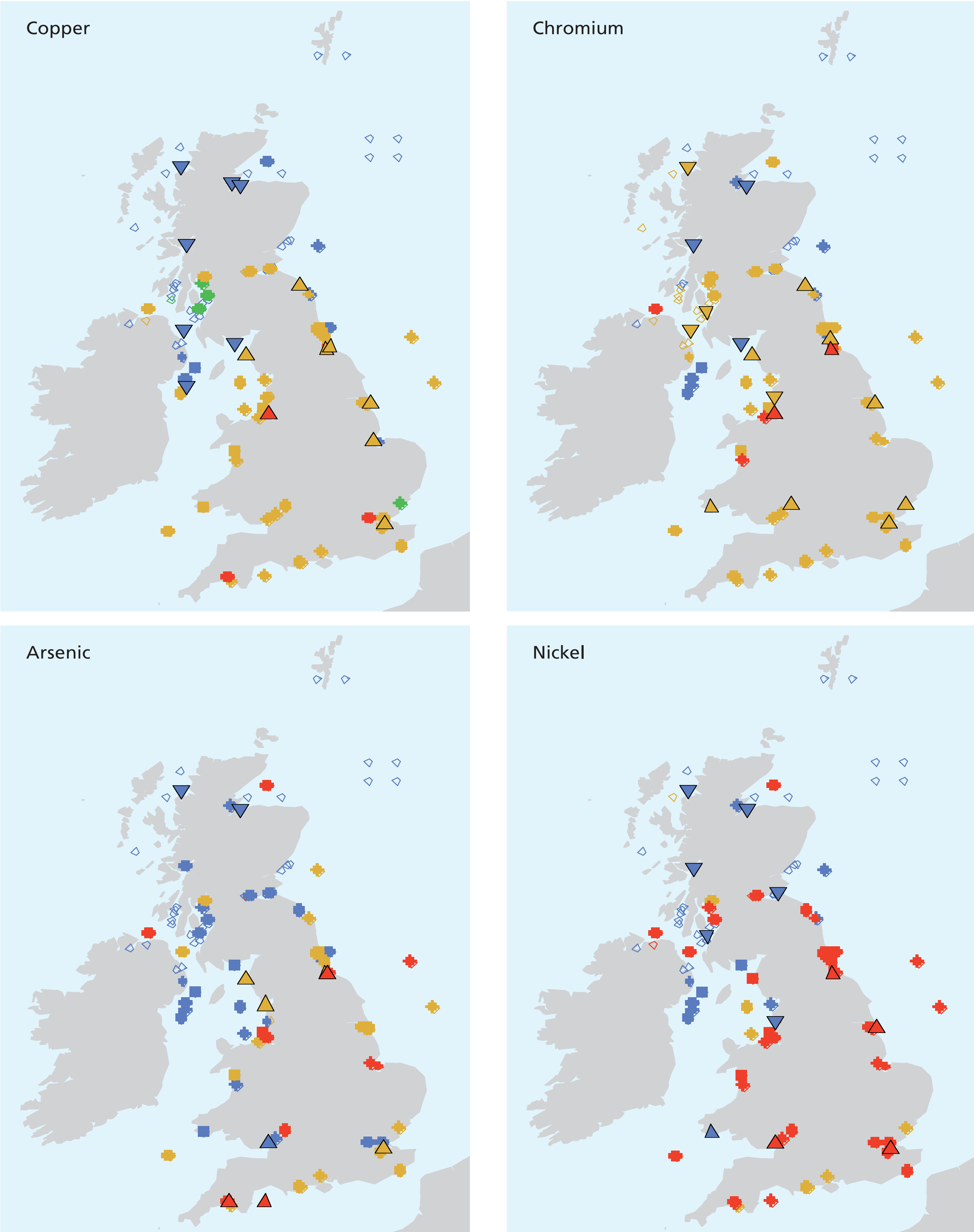
Cadmium is one of three trace metals (the others being Hg and Pb) that are of most concern in the marine environment because of its potential to cause toxicological effects. Of the sites monitored 98% had Cd/Al ratios less than the ERM and 22% of which were below the BAC, suggesting that toxicological effects are unlikely and many sites had concentrations close to background, although the Tees and Tyne estuaries had Cd/Al ratios that are potentially of concern. Significant upward trends were recorded in estuaries in the south of England but there is not a significant upward trend in Cd riverine inputs to the estuaries monitored and in the case of the sites in Plymouth Sound and the Severn Estuary there are other sites nearby that do not have upward trends. Some sites suggest that changes in the sediment regime have occurred and these or unexplained inputs may have caused the upward trends observed. Liverpool Bay and the Ribble Estuary showed a significant decrease in Cd contamination which reflects the decrease in Cd inputs to this region since the mid-1990s.

Mercury, lead and zinc

Mercury, Pb and Zn had similar numbers of stations in each category and are discussed together. Between 20% and 24% of sites monitored had ratios above the ERM and 46% to 59% had ratios between the ERL and ERM. Fewer than 20% of sites had ratios that could be considered close to background. These figures suggest that at most sites a toxicological effect would be unlikely but given that ratios are above the ERL a representative sample of these sites should continue to be monitored. Sites that are most at risk include the Tees, Tyne and Mersey estuaries which have been identified previously as being contaminated (Rowlatt and Lovell, 1994; Marine Environment Monitoring Group, 2004). The most contaminated sites exist close to the coast but there are some notable exceptions such as the outer Moray Firth and a site in the Celtic Sea which are far from riverine and industrial inputs. A number of significant downward trends are observed for Hg, Pb and Zn in Scottish coastal waters and some upward trends in the south and west of England. At some of these sites the results suggest abrupt changes in metal concentrations between

Figure 3.12 Metal/Al ratios in coastal sediments. © Marine Scotland Science. Key on Page 62.




Copper
Chromium
Arsenic
Nickel

years which may be related to changes in the sediment regimes. This would have an effect on the identification of significant temporal trends.

Copper, chromium and arsenic

Most Cu/Al and Cr/Al ratios are between the ERL and ERM. Significant upward trends in these metal/Al ratios were observed along the east coast of England, whereas concentrations in Scotland and the western Irish Sea were mostly below the BAC with some sites showing downward trends. 43% of sites monitored for arsenic (As) have As/Al ratios below the BAC which are primarily to the north and west of the UK. A number of sites in the Tees, Mersey, Wash estuaries and Plymouth Sound had ratios that are likely to lead to toxicological effects which may in turn be related to the industrialized nature of these areas.

Nickel

Of the 69 sites monitored for nickel (Ni) 41 had Ni/Al ratios above the ERM suggesting that effects would be likely throughout the UK. Ni concentrations in sediments have been shown to be a function of concentrations in the underlying geology in Northern Ireland coastal sediments (Charlesworth and Service, 2000) and a previous national assessment using different assessment criteria (Marine Environment Monitoring Group, 2004) did not identify any such risk to the environment from Ni. Previous studies have shown that when Ni concentrations are compared to background concentrations (i.e. pre-industrial levels) there is only minor or no anthropogenic enrichment in UK coastal sediments except for a minority of sites close to known sources of contamination (Irion and Muller, 1990; Balls et al., 1997). It is likely then that the ERL and ERM criteria applied in this case are not appropriate for UK geochemistry and further work in this area is required.

3.1.3.3.4 Need for further work

The criteria used for assessment need continual update as new information becomes available. In particular the suitability of the ERL and ERM for Ni in UK coastal sediments requires further study.

3.1.3.3.5 Overall conclusion and forward look

Broad spatial trends for most metal/Al ratios show low ratios in Scotland and the western Irish Sea and higher ratios in England and Wales with a number of particular sites of concern in industrialized estuaries such as the Tees, Tyne, Thames, Severn and Mersey which may lead to toxicological impacts. There is no overall significant trend in the contamination status of metals, but if metal inputs from rivers, sewage and industry continue to decrease further significant downward trends would be expected in future assessments.

3.1.3.4 Polychlorinated biphenyls in sediments and biota

3.1.3.4.1 Summary

Chlorobiphenyls (CBs) were measured in sediment (79 sites), fish liver (63 sites) and mussels (*Mytilus edulis*) (81 sites) from a total of 16 regions. The CB data were assessed using a recently proposed 'traffic light' system, based on the assessment criteria adopted by OSPAR for use in the 2008 Coordinated Environmental Monitoring Programme (CEMP) assessment (OSPAR, 2008). Concentrations were compared to BACs and EACs for sediment or $EAC^{passive}$ for biota. CB concentrations exceeded EACs (or $EAC^{passive}$) for two or more congeners (mainly CB118 and CB101) in some regions and so indicated that adverse biological effects may occur and, in this case, would therefore be assigned a red 'traffic light' status. Few trends

were detected, but where they were detected for two or more congeners these were mostly downward.

3.1.3.4.2 Introduction

Commercial formulations of PCBs, such as Aroclors, have been widely used in transformers, capacitors, hydraulic fluids and as plasticisers in paints, plastics and sealants. In the marine environment the main sources of PCBs include energy production, combustion industry, production processes and waste (landfill, incineration, waste treatment, disposal). Due to concerns about the environmental impact of PCBs, production in the UK ceased in the 1970s. Authorisation for use in closed systems continued until 1986 when sales of PCB formulations finally stopped in the UK. However, PCBs still enter the marine environment following the destruction and disposal of industrial plants and equipment or from emissions from old electrical equipment and landfill sites.

PCBs are persistent and have the potential for long-range atmospheric transport and are, therefore, ubiquitous in the marine environment. Due to their low solubility and hydrophobic nature, PCBs tend to associate with particulate material. Accumulation of hydrophobic compounds, such as PCBs, in sediments is dependent on type; sediments with a high organic carbon content and a smaller particle size (i.e. larger surface:volume ratio) have a greater potential to accumulate PCBs compared to coarser, sandy sediments. In addition, due to their persistence and lipophilic nature, PCBs have the potential to bioaccumulate, particularly in lipid-rich tissue such as fish liver.

There are 209 possible CB congeners; however, most monitoring programmes require the analysis of only a limited number of these. The seven CBs recommended by the ICES Marine Chemistry Working Group (CB28, CB52, CB101, CB118, CB153, CB138 and CB180) were selected as indicators due to their relatively high concentrations in technical mixtures and their wide chlorination range (3 to 7 chlorine atoms per molecule). The aim was also to allow comparison between datasets in which different sets of congeners were measured, as long as these seven CBs were included in both. Currently, the UK CSEMP specifies that the ICES7 CB congeners should be analysed in sediment, fish liver and mussels as part of the temporal monitoring survey, but additional CBs have been measured at some sites, although these data are not reported here.

3.1.3.4.3 Developments since *Charting Progress*

For *Charting Progress*, only the ICES7 CBs were reported in sediment, shellfish and fish liver. Although a greater number of CBs were reported prior to *Charting Progress 2*, this was limited to just a few sites and so the data were not assessed. More sites have been monitored since *Charting Progress*, particularly for shellfish, with a wider spatial coverage. The data assessed for *Charting Progress* covered the 1999 to 2002 period and so with only one to four years worth of data trends in concentration could not be assessed.

Assessment criteria

CB data were assessed here using the recently proposed 'traffic light' system adopted by OSPAR (OSPAR, 2008). Assigning a green status for a particular contaminant would mean that the environmental concentrations were satisfactory in that they present little or no risk. The first transition point (blue/green, T_0) for a particular contaminant would require concentrations for that contaminant to be at, or close to,

background concentrations and, therefore, the OSPAR BACs were selected as T_0 for biota and sediment. Concentrations below BACs would be considered to have high environmental status (coloured blue in the maps). EACs represent the contaminant concentration in the environment below which it can be assumed that no chronic effects will occur in marine species, including the most sensitive species and therefore were selected as the second transition point (green/red, T_1). Concentrations significantly below EACs (or EAC equivalents) could be considered to have good environmental status (coloured green in maps) and those above bad (coloured red in maps).

BACs have been established for the ICES7 CBs in sediment. Concentrations are expressed in μg/kg dry weight (dw), normalised to 2.5% total organic carbon. For mussels, BACs are available for each of the ICES7 CBs (Table 3.8). EACs are available for all ICES7 CBs in sediment and as they are above BACs, ICES recommends that they may be used for OSPAR assessments. However, EACs for CBs in shellfish or fish are not recommended for use in assessments as they are below the BACs. Therefore an alternative solution is needed for the green to red transition. Recent work on the bioavailability of hydrophobic contaminants in sediment using silicone rubber passive samplers has generally shown the potential for all the burden of CBs in sediments to be mobilised into the sediment pore water, i.e. to be potentially bioavailable (Smedes, 2007). Therefore, partitioning theory can be reliably applied to calculate the concentrations of CBs in lipid in biota that would be in equilibrium with the CBs in the sediment. The biota sediment accumulation factor (BSAF) can be expressed as the ratio between the contaminant concentration in sediment (expressed on the basis of total organic carbon) and the concentration in biological

Table 3.8 Assessment criteria for CBs in sediment, mussels and fish liver.

Compound	*LC*[a]	*BAC (T_0)*	*EAC/EAC*[passive] *(T_1)*
Sediment (μg/kg dry weight)			
CB28	0.05	0.22	1.7
CB52	0.05	0.12	2.7
CB101	0.05	0.14	3.0
CB118	0.05	0.17	0.6
CB138	0.05	0.15	7.9
CB153	0.05	0.19	40
CB180	0.05	0.10	12
Mussels (μg/kg wet weight)			
CB28	0.05	0.15	0.64
CB52	0.05	0.15	1.08
CB101	0.05	0.14	1.2
CB118	0.05	0.12	0.24
CB138	0.05	0.12	3.16
CB153	0.05	0.12	16
CB180	0.05	0.12	4.8
Fish (μg/kg wet weight, EACs lipid weight)			
CB28	0.05	0.10	64
CB52	0.05	0.08	108
CB101	0.05	0.08	120
CB118	0.05	0.10	24
CB138	0.05	0.09	316
CB153	0.05	0.10	1600
CB180	0.05	0.11	480

a LC: low concentration – this is recommended for use when a BAC cannot be set due to lack of data.

material. In cases where the total concentration of a contaminant in sediment is potentially bioavailable, the value of BSAF is close to unity. Therefore, EACs for CBs in sediment were used to calculate concentrations of CBs in mussel tissue in equilibrium with sediment containing CB concentrations equal to the EACs in sediment

Table 3.9 Assessment summary for CBs in mussels. The assessment is based on a total of 38 sites where CB data were available for more than three years.

Region	*No of sites/ strata*	*ΣICES7 CBs (µg/kg ww)*	*Percentage of sites below BAC (i.e. blue status)*	*Percentage of sites below EAC^passive^ (i.e. green status)*	*Percentage of sites above EAC^passive^ (i.e. red status)*
Moray Firth	5	2.35 - 4.44	n/a	n/a	n/a
East Scotland	4	3.52 - 131.7	0	0	100
Forth	7	3.49 - 21.10	0	0	100
Tyne/Tees	4	<7.00 [a]	0	0	100
Humber/Wash	3	4.19 – 6.68	0	0	100
Anglia	3	6.64 – 7.92	0	0	100
Eastern Channel	2	<7.00 [a], 9.03	0	0	100
Western Channel	1	9.19	0	0	100
Severn	1	7.92	0	0	100
Irish Sea	8	1.02 – 16.70	0	43	57
Clyde	15	1.59 – 19.80	0	18	82
Minch/Malin	20	1.36 – 5.08	0	75	25
Hebrides	1	1.45	n/a	n/a	n/a
Northern Scotland	2	1.74, 2.66	n/a	n/a	n/a
West Shetland	4	1.48 – 2.00	n/a	n/a	n/a
East Shetland	1	1.48	n/a	n/a	n/a

a Reflects data where LODs were high (i.e. 1µg/kg ww for each of the ICES7 congeners).

and it was proposed that these calculated values (termed EAC^passive^) be used as the green/red transition for CBs in mussels and fish liver.

The assessment summary tables at the ends of the sections represent our expert opinion on the status and trends for CBs in the marine environment. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

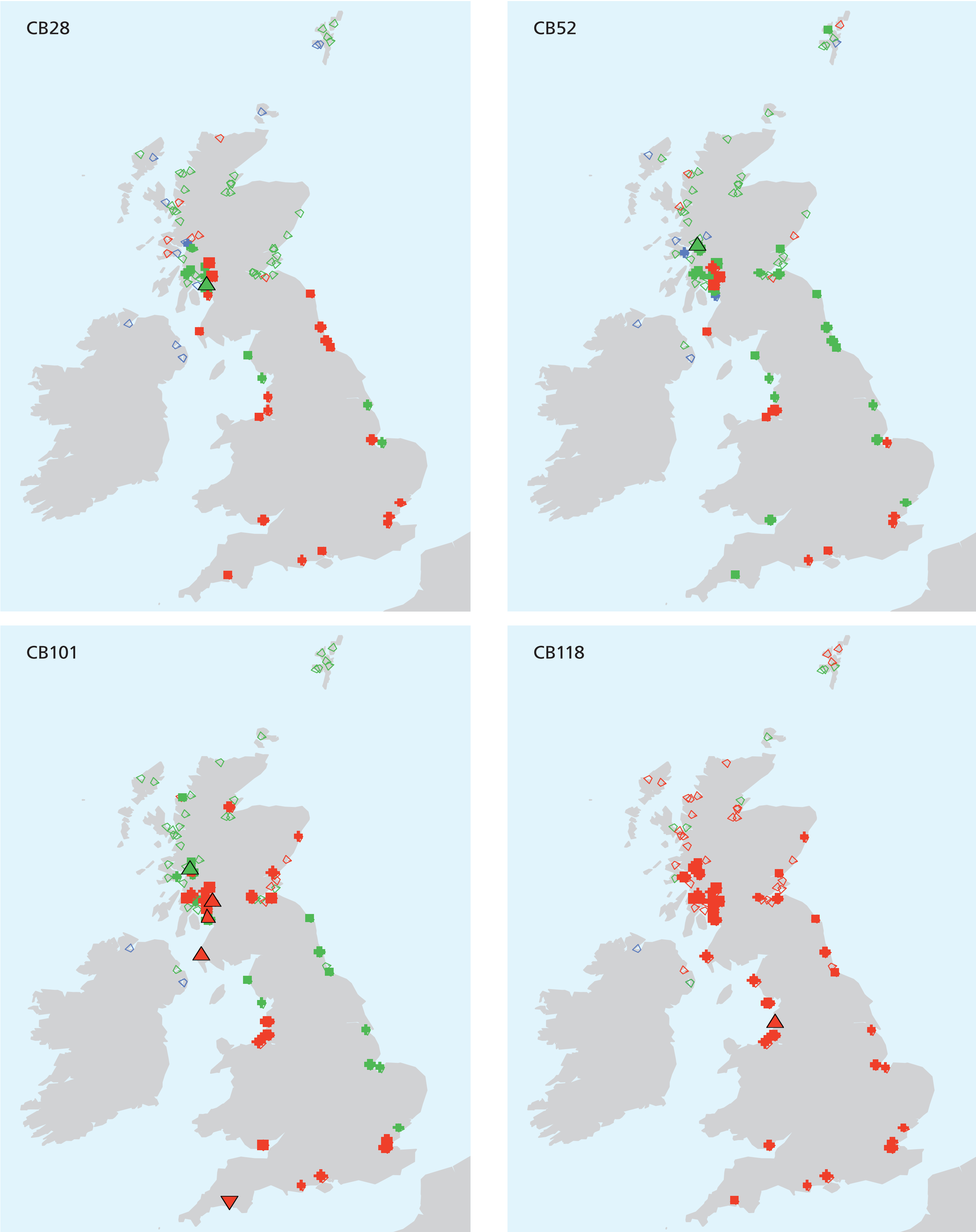
3.1.3.4.4 Presentation of the evidence

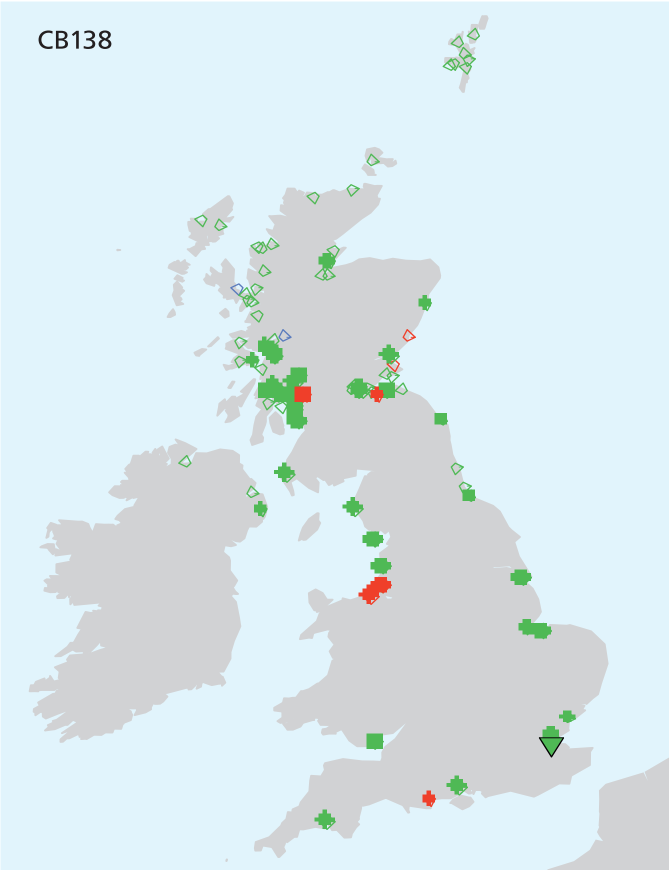
Shellfish

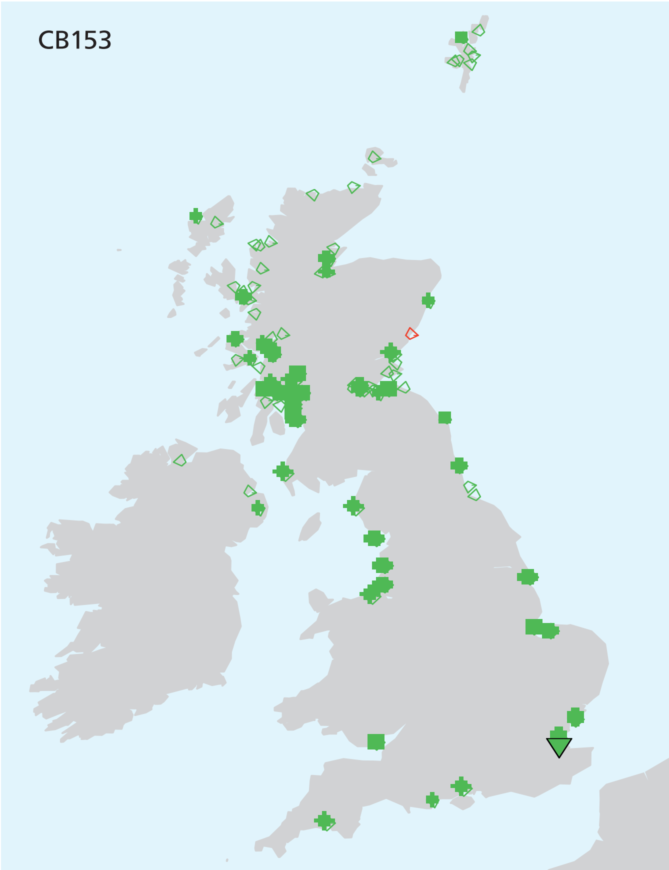
The ICES7 CBs were measured in mussels (*Mytilus edulis*) from 81 sites in 16 regions (15 sites in the Clyde). The results are summarised in Table 3.9 and shown in Figure 3.13. Mean concentrations for the ΣICES7 CBs ranged from 1.0 µg/kg wet weight (ww) in the Irish Sea region to 132 µg/kg ww in East Scotland which was exceptionally high and possibly due to the mussels being sampled on an industrial site.

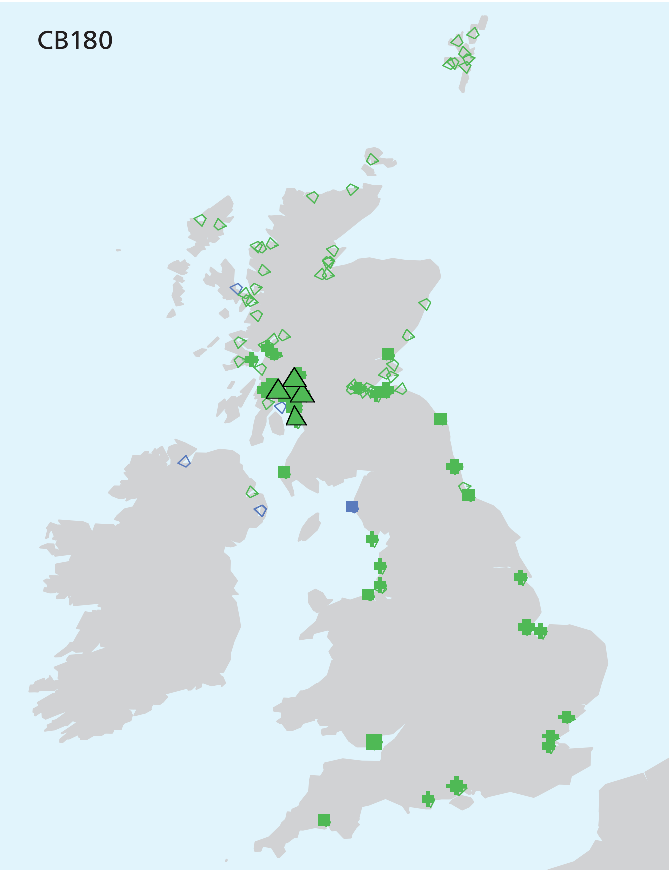
The CB profiles were similar at most sites with the penta- and hexa-CBs (CB101, CB118, CB138, CB153) dominating. In general, the

Figure 3.13 CB concentrations in mussels. © Marine Scotland Science.









Circles are blue if the concentration is significantly below the BAC; green if significantly greater than the BAC but below the $EAC^{passive}$; and red if significantly above the $EAC^{passive}$. Statistically significant upward or downward trends are indicated by triangles; open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data.

proportion of higher chlorinated CBs (≥ 5 chlorines) increases through the food web as they are less volatile, more lipophilic and more resistant to metabolic and microbial degradation.

The 95% upper confidence limits on the mean concentrations were compared to the green/red transition (EAC^{passive}) and BACs for CBs in mussels (with more than three years of data). Thirty-eight sites had more than three years of data for all ICES7 CBs. The Moray Firth, Hebrides, North Scotland, West Shetland and East Shetland regions had CB data for fewer than three years. Concentrations were above BACs at all sites for all congeners. The EAC^{passive} is lowest for CB118 (0.24 µg/kg ww), a mono-*ortho* CB congener and the most toxic of the ICES7 CBs. All 38 sites where data were available for more than three years, except for one site in the Minch and Malin region, gave concentrations for CB118 above the EAC^{passive} and therefore were assigned a red status for this CB, the other site was assigned a green status. CB101 and CB28 also exceeded the EAC^{passive} at a high proportion of sites. Of the sites with more than three years of data for CB153, the least toxic CB congener, concentrations at all sites were below the EAC^{passive} but none were below the BAC. To aggregate results for CB congeners, if two congeners within a region were assigned a red status then the overall CB status for that region would be red. Table 3.9 summarises the percentage of locations that exceeded the EAC^{passive} for two or more congeners in each region.

Trends were assessed on sites with more than five years of data, and only four sites (Anglia, Minch/Malin, and two sites in the Clyde) showed a significant downward trend for two or more ICES7 CB congeners.

An assessment summary for each region, including the 'traffic light' status, is provided in Table 3.10, which presents our expert opinion on the status and trends for CBs in mussels within the UK seas. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

Fish liver

CB data were reported in fish (plaice, dab, whiting, flounder) liver at 63 sites or strata in 13 regions. The ICES7 CBs were measured at all sites. The sum of the ICES7 CBs are summarised in Table 3.11 for each region and shown in Figure 3.14. Concentrations ranged from 93 µg/kg lipid weight (lw) in the Minch and Malin Sea region to 4833 µg/kg lw in Anglia.

The 95% upper confidence limits on the mean concentrations were compared to the green/red transition (EAC^{passive}) for CBs in fish liver (with more than three years of data, 45 sites). The EAC^{passive} is lowest for CB118 (24 µg/kg lw), a mono-*ortho* CB and the most toxic of the ICES7 CBs. A high proportion of sites gave concentrations for CB118 above the EAC^{passive} and, therefore, CB118 was assigned a red status. Two samples in two different regions (Cardigan Bay and Tyne/Tees) gave concentrations below the EAC^{passive} for CB118 and were assigned a green status for CB118. CB101 also exceeded the EAC^{passive} at a high proportion of sites. For CB153, the least toxic CB congener, concentrations were above EAC^{passive} at five sites only (two in Anglia, and one in the Irish Sea, West Channel and Severn regions). None of the sites gave CB concentrations below BACs.

Table 3.10 Assessment summary for CBs in mussels.

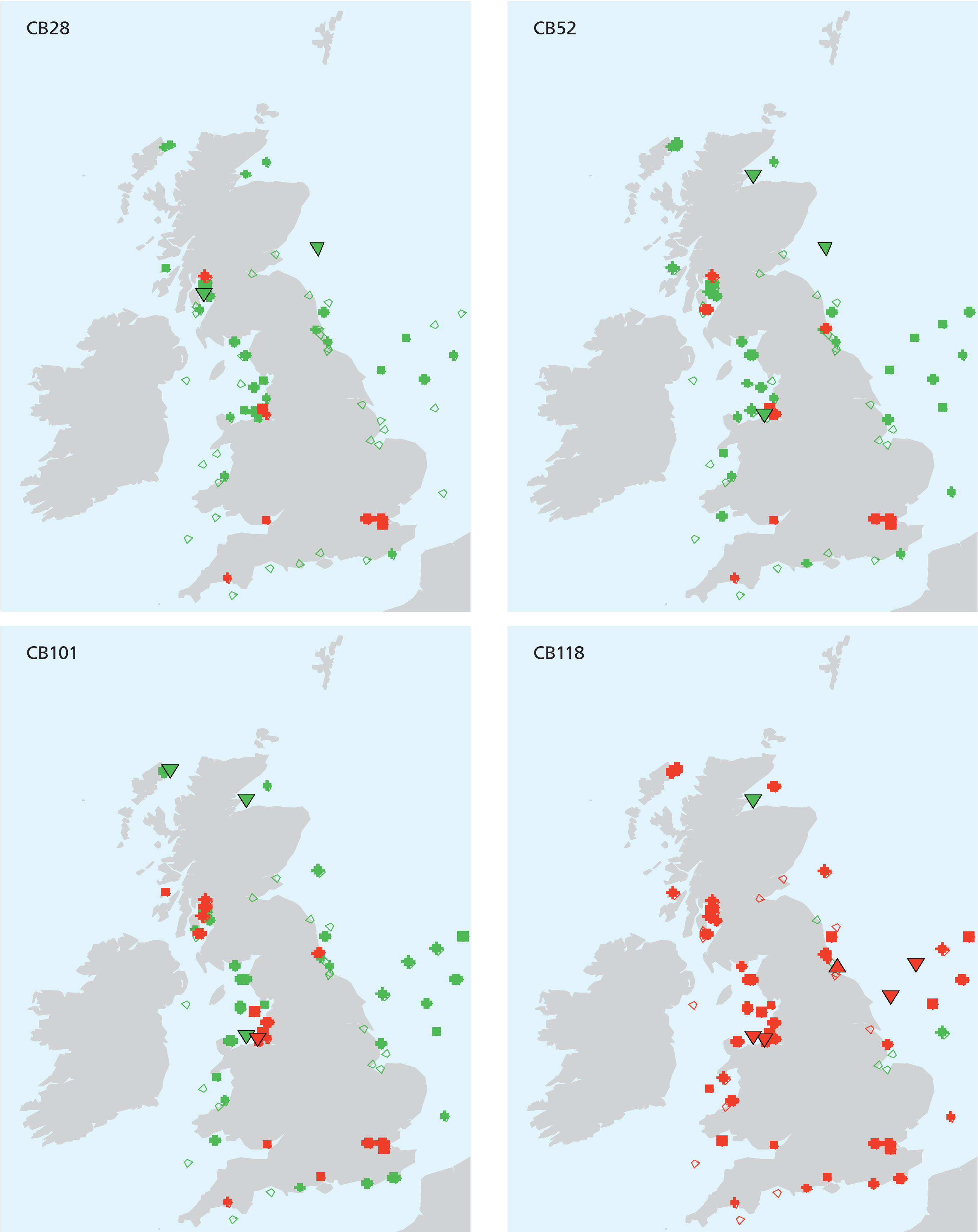
Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
Moray Firth			↔		II	
East Scotland			↔		II	
Forth	Urban area		↔		II	
Tyne/Tees	Urban area; historic industry		↔		II	
Humber/Wash	River inputs		↔		II	
Anglia	River inputs		↔		II	
East Channel			↔		I	
West Channel			↔		I	
Severn	Urban area; historic industry		↔		I	
Irish Sea	Urban area; historic industry		↔		II	
Clyde			↑		II	
Minch/Malin			↔		II	
Hebrides			↔		I	
North Scotland			↔		I	
West Shetland			↔		II	
East Shetland			↔		I	

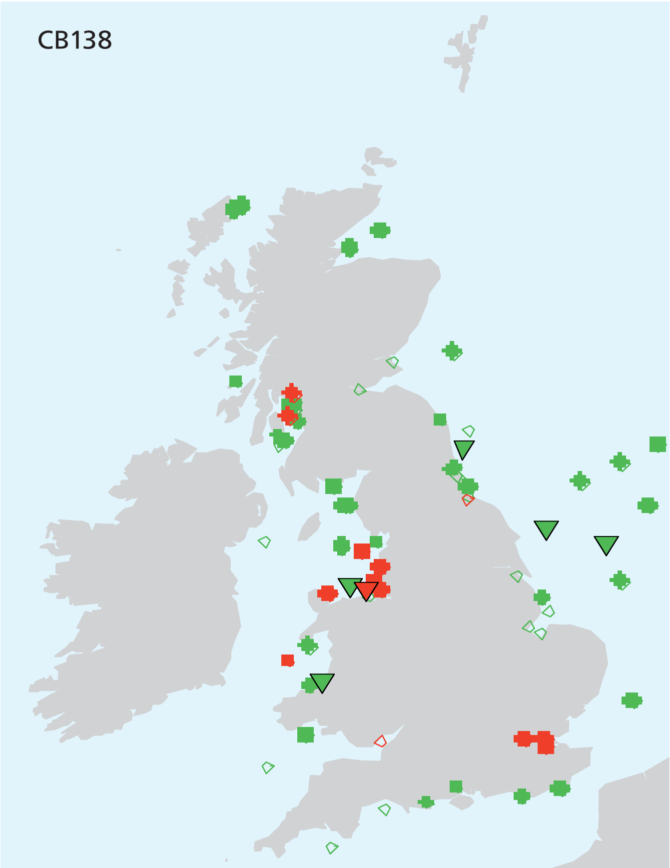
To aggregate results for CB congeners, if two congeners within a region were assigned a red status than the overall CB status for that region was red. Table 3.11 summarises the percentage of sites that exceeded the EAC^{passive} for two or more congeners (normally CB118 and CB101) in each region.

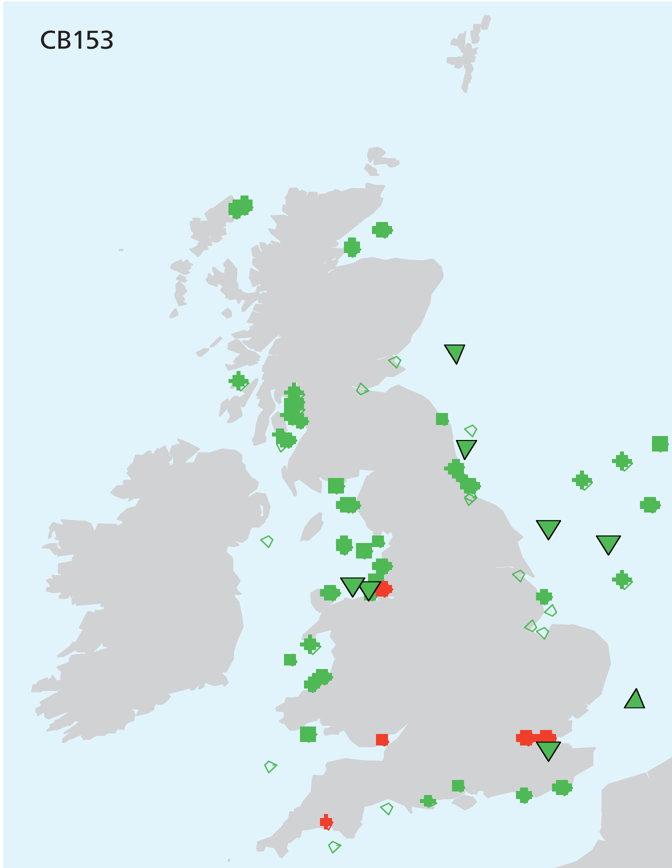
Trends were investigated at sites with more than five years of data for each of the ICES7 CBs. The patterns for each CB tended to be the same. Few trends were detected, only nine stations in seven of the regions (Anglia, Cardigan Bay, East Scotland, Humber/Wash, Irish Sea, Moray Firth, Tyne/Tees) showed significant downward trends for two or more of the ICES7 CBs. None of the sites showed a significant upward trend for two or more CBs.

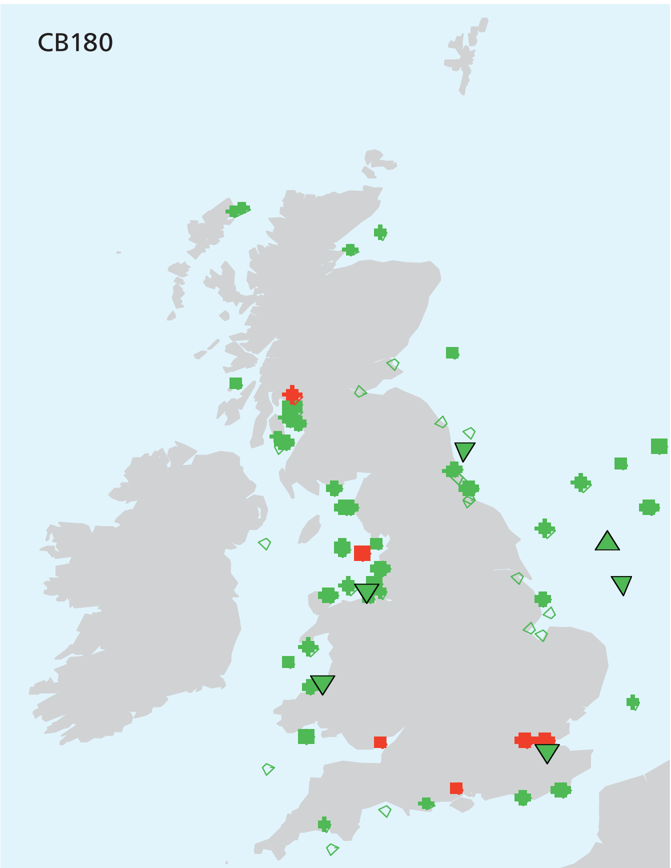
An assessment summary for CBs in fish liver for each region, including 'traffic light' status is provided in Table 3.12, which presents our expert opinion on the status and trends for CBs in fish liver within the UK seas. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (↔). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

Figure 3.14 CBs in fish liver. © Marine Scotland Science.









Circles are blue if the concentration is significantly below the BAC; green if significantly greater than the BAC but below the EAC^{passive}; and red if significantly above the EAC^{passive}. Statistically significant upward or downward trends are indicated by triangles; open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data.

Table 3.11 Assessment summary for CBs in fish liver. Assessment based on a total of 45 sites where data were available for more than three years.

Region	*No of sites/ strata*	*ΣICES7 CBs µg/kg lipid weight*	*Percentage of sites below BAC (i.e. blue status)*	*Percentage of sites below EAC[passive] (i.e. green status)*	*Percentage of sites above EAC[passive] (i.e. red status)*
Moray Firth	2	117, 188	0	100	0
East Scotland	1	138	0	100	0
Forth	1	799	n/a	n/a	n/a
Tyne/Tees	10	133 – 1301	0	80	20
Humber/Wash	8	122 – 1350	0	100	0
Anglia	4	271 – 4833	0	25	75
Eastern Channel	5	166 - 1067	0	100	0
Western Channel	2	137 – 1824	n/a	n/a	n/a
Severn	2	277 – 3422	0	100	0
Cardigan Bay	4	140 – 401	0	100	0
Irish Sea	14	265 – 2963	0	46	54
Clyde	7	369 – 1613	0	43	57
Minch/Malin	3	93 – 191	0	100	0

Table 3.12 Assessment summary for CBs in fish liver.

Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
Moray Firth			↔		I	
East Scotland			↔		I	
Forth	Urban area					
Tyne/Tees	Urban area; historic industry		↔		II	
Humber/Wash	River inputs		↔		II	
Anglia	River inputs		↔		I	
Eastern Channel			↔		I	
Western Channel			↔		I	
Severn	Urban area; historic industry		↔		I	
Cardigan Bay			↔		I	
Irish Sea	Urban area; historic industry		↑		III	
Clyde	Urban area; historic industry		↔		II	
Minch/Malin			↔		I	

Sediment

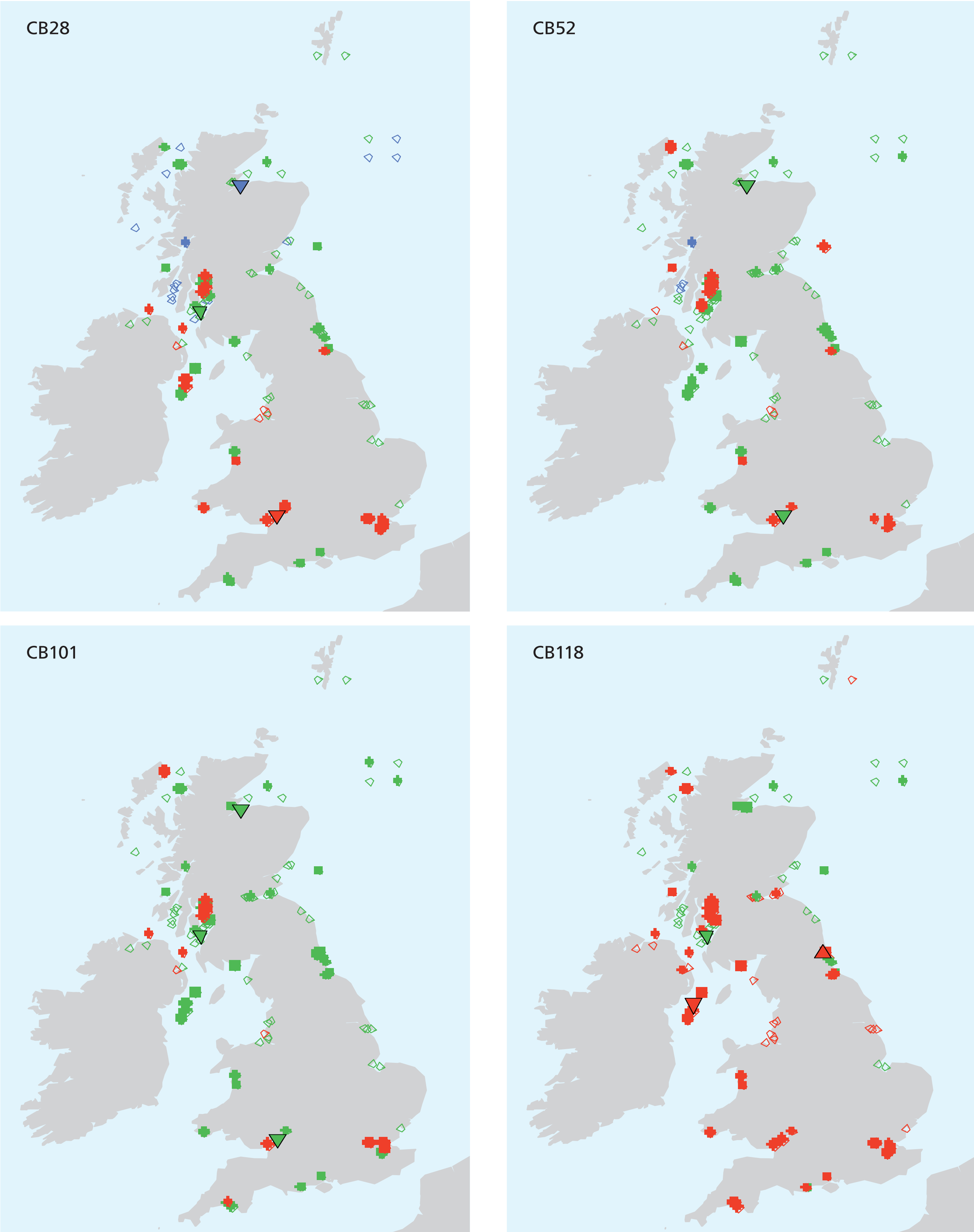
Sediment CB concentrations were reported at 79 fixed or stratified random sites, covering 15 regions. The ICES7 CBs were measured at all sites. The sum of the ICES7 CBs are summarised in Table 3.13 for each region and shown in Figure 3.15. Concentrations ranged from 0.91 µg/kg dw in the Minch and Malin Sea region to 188.0 µg/kg dw in the Irish Sea.

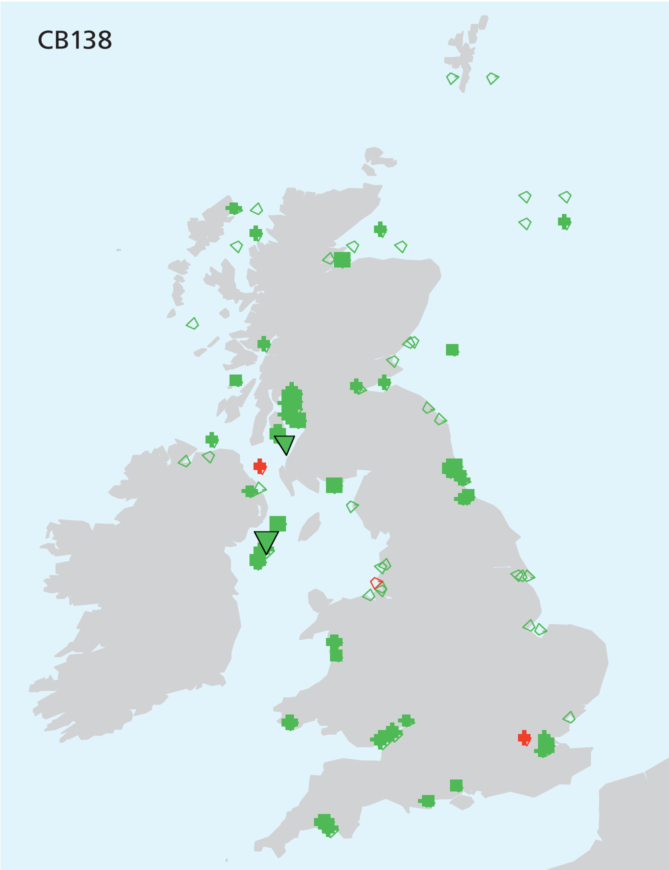
The 95% upper confidence limits on the mean concentrations were compared to the green/red transition (EAC) for CBs in sediment with more than three years of data (55 sites). The EAC is lowest for CB118 (0.6 µg/kg dw), a mono-*ortho* CB and the most toxic of the ICES7 CBs. The Moray Firth, East Scotland, Forth, Humber and Wash, South Irish Sea and Cardigan Bay regions had concentrations at all locations below the EAC for CB118 and were therefore assigned a green status for this CB. Three regions were assigned a red status for CB118 for all sites/strata (Anglia, Severn and West Channel). EACs were also exceeded for CB28, CB52 and CB101 at a number of sites. For CB153, the least toxic CB, only one site from the Minch and Malin Sea and one from the Fladen region were assigned a blue status, two were assigned a red status (Irish Sea, Anglia), and all others were assigned a green status. To aggregate results for CB congeners, if two congeners within a region were assigned a red status than the overall CB status for that region was red. Table 3.13 summarises the percentage of sites that exceeded the EAC for two or more congeners in each region.

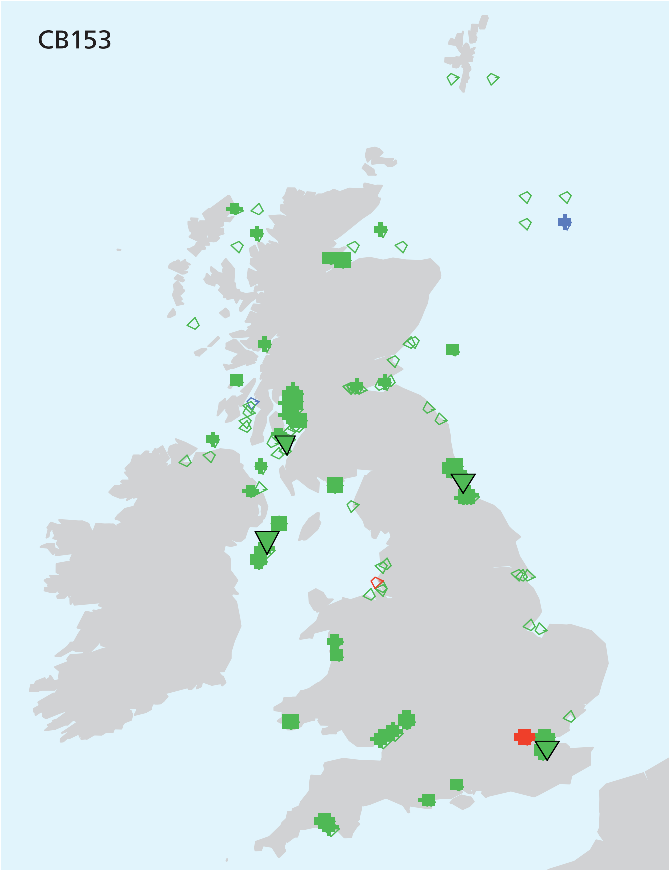
Table 3.13 Assessment summary for CBs in sediments. Assessment based on a total of 55 sites where data were available for more than 3 years.

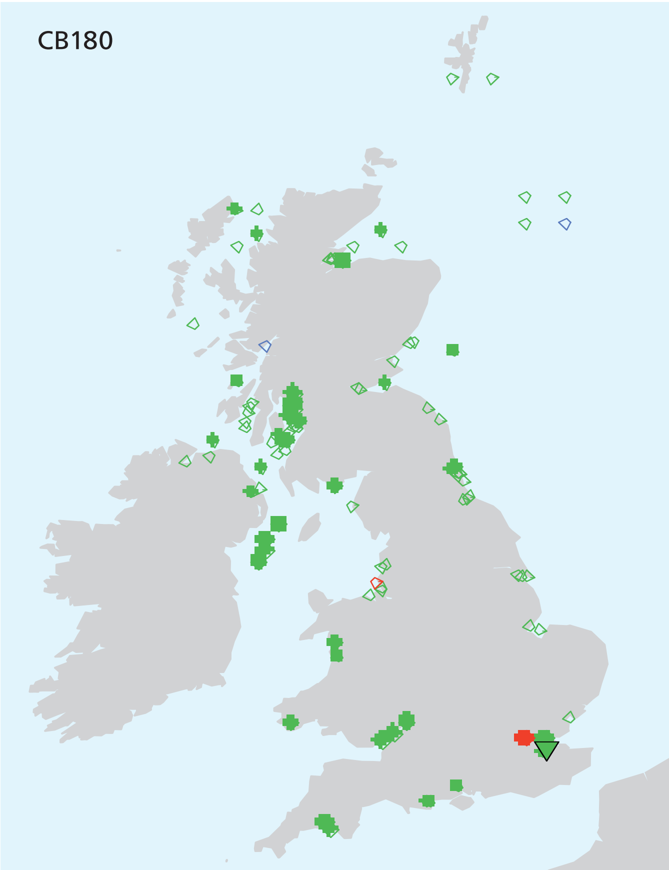
Region	*No of sites/ strata*	*ΣICES7CBs (µg/kg dw)*	*Percentage of sites below BAC (i.e. blue status)*	*Percentage of sites below EAC (i.e. green status)*	*Percentage of sites above EAC (i.e. red status)*
Anglia	5	7.04 – 121.9	0	0	100
Cardigan Bay	2	2.08, 5.19	0	100	0
Clyde	10	2.34 – 16.32	0	50	50
East Scotland	2	3.76, 4.13	0	100	0
East Channel	3	3.02 – 5.08	0	100	0
Fladen	4	1.07 – 2.21	n/a	n/a	n/a
Forth	4	2.24 – 5.72	0	100	0
Humber/Wash	5	2.40 – 10.35	0	100	0
Irish Sea	13	2.32 – 188.0	0	100	0
Minch/Malin	7	0.91 – 30.6	0	67	33
Moray Firth	3	1.13 – 2.52	0	100	0
South Irish Sea	1	3.36	0	100	0
Severn	4	6.56 – 26.1	0	0	100
Tyne/Tees	13	1.70 – 15.32	0	86	14
West Channel	3	8.86 – 13.44	0	100	0

Figure 3.15 CBs in sediment. © Marine Scotland Science.









Circles are blue if the concentration is significantly below the BAC; green if greater than the BAC but below the EAC; and red if significantly above the EAC. Statistically significant upward or downward trends are indicated by triangles; open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data.

Table 3.14 Assessment summary for CBs in sediment.

Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
Anglia	River inputs		↔		II	
Cardigan Bay			↔		I	
Clyde	Urban area; historic industry		↔		II	
East Scotland			↔		I	
Eastern Channel			↔		II	
Fladen			↔		II	
Forth	Urban area		↔		II	
Humber/Wash	River inputs		↔		II	
Irish Sea	Urban areas; historic industry		↔		II	
Minch/Malin			↔		II	
Moray Firth			↔		II	
Southern Irish Sea			↔		I	
Severn	Urban areas; historic industry		↔		II	
Tyne/Tees	Urban areas; historic industry		↔		II	
Western Channel			↔		II	

Trends were investigated at sites with more than five years of data for each of the ICES7 CBs. The patterns for each CB tended to be the same. Few significant trends were detected; only five stations in five different regions (Clyde, Moray Firth, Irish Sea, Anglia, Severn) showed significant downward trends for two or more of the ICES7 CBs. No station had an upward trend for two or more of the CBs.

An assessment summary for CBs in sediments for each region is provided in Table 3.14, which presents our expert opinion on the status and trends for CBs in sediment within the UK seas. Hatching indicates a lack of evidence and/or robust assessment criteria. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (↔). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

3.1.3.4.5 Progress towards the vision of clean and safe seas

CB concentrations continue to be high in some regions, particularly for mussels. Mussels in all regions, except the Minches and Malin Sea, are assigned a red status for CBs. Only three regions were assigned a red status for fish liver; Anglia, Clyde and Irish Sea. Three regions also had high CB concentrations in sediment and were assigned a red status; Anglia, Clyde and Severn. Only the

Clyde and Anglia were assigned a red status for all three matrices. As a result, adverse biological effects may occur in these regions due to CBs. CB concentrations showed little improvement over the monitoring period with very few sites showing any significant downward trend.

3.1.3.4.6 Need for further work

Analysis of a wider suite of CBs including the more toxic 'dioxin-like' CBs should be considered. Currently only one of the 'dioxin-like' CBs is routinely monitored (CB118) at CSEMP sites. The EU Marine Strategy Framework Directive requires that ...*contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards*. Commission Regulation (1881/2006/EC) sets maximum toxic equivalent (TEQ) concentrations for the sum of dioxins and dioxin-like CBs in the muscle meat of fish and fishery products (8 pg/g ww). On 1 July 2008 limits were also set for fish liver (amendment 565/2008) with a maximum TEQ concentration for the sum of dioxins and dioxin-like CBs of 25 pg/g ww. Currently no assessment can be made against these standards as 'dioxin-like' CBs are not currently reported at CSEMP sites. However it may be possible to use published models (Bhavsar et al., 2007) to predict TEQs in fish tissue, using either total CB concentrations or indicator CBs (with *ortho* substitution).

3.1.3.4.7 Overall conclusion and forward look

CB concentrations in biota and sediment continue to be of concern, with some regions being assigned a red status for this contaminant group, particularly in mussels. There has been little change in concentrations since *Charting Progress* and concentrations appear to be relatively stable. The ban on the use of PCBs should result in a decrease in contaminant loading of both biota and sediments with time. However, the slow degradation of CBs means this could take some time and will require continued monitoring.

3.1.3.5 Polycyclic aromatic hydrocarbons in sediments and shellfish

3.1.3.5.1 Summary

Tables 3.15 and 3.16 present our expert opinion on the status and trends for PAHs in mussels and sediment within the UK seas. Trend arrows are provided based on the evidence available, showing whether the state or condition is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (↔). A question mark indicates insufficient data. The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

Overall findings

Individual PAH concentrations in mussels were classified in relation to guideline values (Figure 3.16). The BAC (where set) defined the blue/green transition point, and the EAC defined the green/red transition point. For mussels, the majority of sites were assigned either green or blue status in all but two regions, Tyne/Tees and Western Channel. In each of these cases, confidence in the assessment is low because single hotspot sites are responsible for them having been assigned red status. These sites are Teesmouth (Tyne/Tees) and Plymouth Sound (Western Channel).

Individual PAH concentrations in sediments were classified in relation to guideline values (Figure 3.17). The BAC defined the blue/green transition point, the ERL concentration defined the green/amber transition point and the ERM

Table 3.15 Assessment summary for PAHs in mussels.

Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
East Scotland (inc. Moray Firth)			?		II	
Forth	Urban area		?		I	Better trend assessment with more data
Tyne/Tees	Urban area; historic industry	Tees hotspot	?		I	Stay red
Humber/Wash	River inputs		?		II	
Anglia	River inputs		?		II	
East Channel			?		II	
West Channel		Plymouth Sound hotspot	?		I	Stay red
Severn	Urban areas; historic industry		?		II	
Irish Sea	Urban areas, historic industry		↑		II	
Clyde	Urban area; historic industry		?		II	
Minch/Malin		Low levels of contamination, close to background	↑		I	

Table 3.16 Assessment summary for PAHs in sediment.

Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
East Scotland (inc. Moray Firth)		Low levels of contamination; close to background	↑		II	↔
Fladen	Oil exploration	Clean offshore area, close to background	?		II	↔
Forth	Urban area		?		II	↔
Tyne/Tees	Urban area; historic industry		↓		II	↔
Humber/Wash	River inputs		↓		II	↔
Anglia			?		II	↔
East Channel			↓		II	↔
West Channel			↓		II	↔
Severn	Urban area; historic industry		↓		II	↔
Cardigan Bay			?		II	↔
Irish Sea	Urban area; historic industry		↑		II	↔
Clyde	Urban area; historic industry		↑		II	↔
Minch/Malin			↑		II	↔

Figure 3.16 Site classifications for mussels, assessed for individual PAHs. © Crown copyright 2010: permission granted by Cefas.

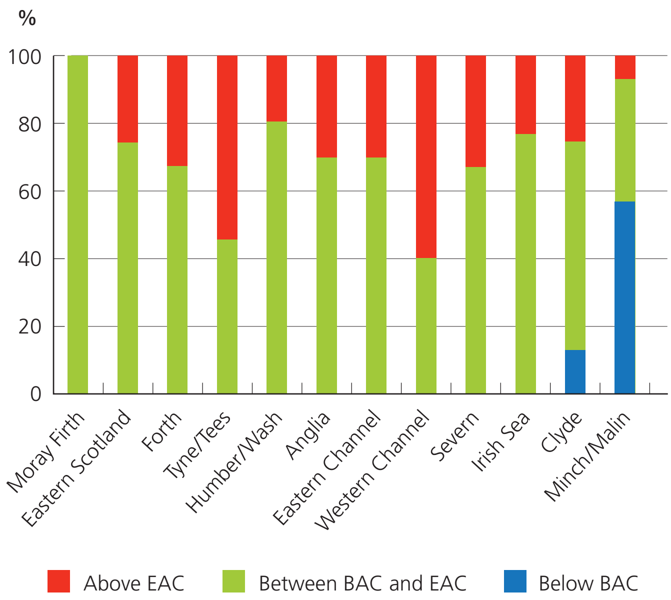
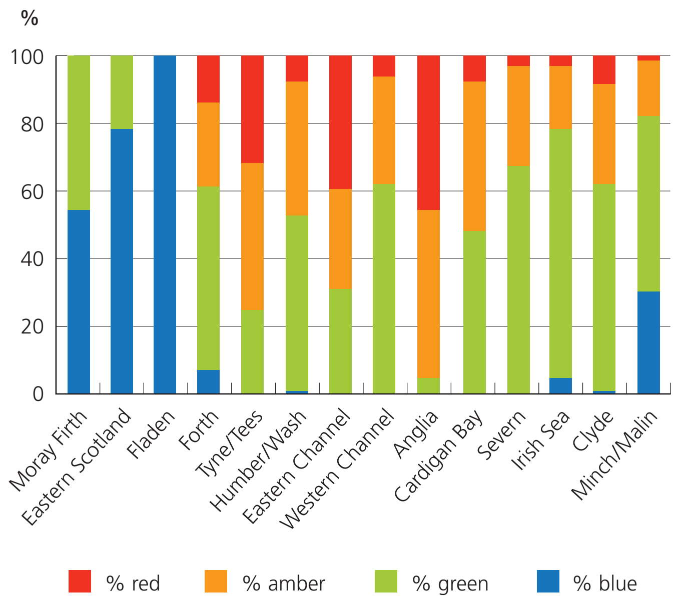


Figure 3.17 Site classifications for sediments, assessed for individual PAHs. © Crown copyright 2010: permission granted by Cefas.



defined the amber/red transition point. For sediments, the majority of sites within the East Scotland and Fladen regions were assigned a blue status (close to background). The Tyne/Tees, Anglia and Western Channel regions have a third or more of their sites assigned a red status.

Contribution to GES descriptors

The PAH data illustrate contamination levels in biota and sediments for an important class of chemicals, which have the ability to harm marine species and human consumers. Within the EU Marine Strategy Framework Directive there is a requirement to assess Good Environmental Status (GES) against a series of descriptors. For PAHs, the most relevant of these are Descriptor 8 *...concentrations of contaminants are at levels not giving rise to pollution effects* and Descriptor 9 *...contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards*.

Future risks

Climate change may alter the relative importance of contaminant transport pathways, for example by increasing the movement of some contaminants to the Arctic, but in the context of PAHs in UK waters this is unlikely to make a significant difference to impacts.

Cross-cutting issues

There are no known cross-cutting issues.

3.1.3.5.2 Introduction

Polycyclic aromatic hydrocarbons are compounds containing carbon and hydrogen based on fused ring systems. They arise as products of incomplete combustion, from both domestic and industrial sources, and are components of crude oil and refined oil products. PAHs derived from oil show a high proportion of the smaller PAHs (including alkylated PAHs) and those from combustion a high proportion of the larger (4- to 6-ring) parent PAHs. In some shellfish (particularly bivalves) they can be accumulated to high concentrations, particularly when in direct contact with contaminated sediments. PAHs are of concern as the smaller PAHs can cause tainting of fish and shellfish, and some of the larger PAHs can be metabolically activated to cancer-causing derivatives following ingestion in fish, marine mammals and human consumers of seafood. The degree of carcinogenicity is closely related to the structure of the specific PAH compounds. Atmospheric inputs have been reduced significantly in recent years. However, in some historically industrialized estuaries, such as the Tyne and Tees estuaries of north-east England, there can be high levels of contamination. As PAHs are persistent (particularly in low oxygen conditions, as found in organic-rich muddy estuarine sediments) these levels of contamination are reducing only slowly.

The UK CSEMP routinely analyses sediments and bivalves (mostly mussels) from dedicated sites for their PAH burdens, on an annual basis. PAH burdens in biota are also monitored under the auspices of the Food Standards Agency, but seldom and using samples from retail outlets whose catching locations are unknown. Within the CSEMP, spatial distributions can be investigated for both parent and alkylated PAHs, but only for ten of the parent compounds are there sufficient years of data for trends to be studied.

Environmental impacts

At some sites in a number of currently or historically heavily industrialized estuaries (Tees, Tyne, Wear, Milford Haven), PAHs are at concentrations in sediments which are likely to be acutely toxic to some sediment-dwelling organisms (Woodhead et al., 1999). A wider range of industrialized estuaries have sediment PAH levels which are likely to cause chronic effects. PAH in sediments degrade only slowly, particularly when they are low in oxygen, and concentrations are likely to decline only slowly.

Socio-economic impacts

Generally, socio-economic impacts would be limited to losses suffered if regulatory limits for PAHs are exceeded and fisheries (particularly shellfisheries) closed as a result.

3.1.3.5.3 Developments since *Charting Progress*

Developments in key pressures, understanding and management measures

Developments in key pressures, understanding and management measures are addressed in Section 3.5 (Oil and Chemical spills).

Developments in approach and indicators used

At the time of *Charting Progress*, routine monitoring of PAH concentrations in sediments had begun under the National Marine Monitoring Programme, but trend data were not available. PAH concentrations in shellfish were available for 20 sites, but were not monitored routinely within that programme at that time. With only 1 to 4 years of data available trends could not be assessed.

Relevant research/activities

Biota

Trends were studied for sites with five years or more of data. In the most recent year for which CSEMP data were available, concentrations of the sums of the ten parent PAH (ΣPAH) in mussels and oysters ranged from 0.6 to 603

µg/kg ww. The ΣPAH concentrations for each region are summarised in Table 3.17. The lowest concentration occurred in a sample from the Orkney Islands, and the highest in one from the Clyde Estuary, both in Scotland. In comparison, cultivated rope-grown mussels from remote locations around Scotland were found to have total PAH concentrations (2- to 6-ring parent and alkylated PAHs) of below 50 µg/kg ww during earlier studies (Webster et al., 2003). In the mussel sample from Sun Pier in the River Medway, only low concentrations of ΣPAH were found (58 µg/kg ww), despite the site showing the highest sediment PAH concentration. As sediments are taken subtidally and mussels gathered intertidally, there is no direct link between them.

To give a visual indication of the PAH concentrations, the data were colour-coded using a modified 'traffic light' system. Where OSPAR BACs had been set for individual parent PAHs in biota and the concentrations were below the BAC, sample sites were assigned a blue status, if concentrations were above the BAC but below the EAC they were assigned a green status, and if they were above the EAC they were assigned a red status. These are shown in Table 3.17 and mapped in Figure 3.18. On the maps, triangles indicate significant downward trends. Other than for chrysene/triphenylene and indeno[1,2,3-*cd*]pyrene, and to a lesser extent, fluoranthene and pyrene, the concentrations generally fall within the green band. There are no EACs for chrysene/triphenylene and indeno[1,2,3-*cd*]pyrene and so these cannot be green. Also, no BACs have been set for naphthalene or anthracene, hence these cannot be blue. The concentrations at sites with three or more year's data were compared to the assessment criteria. This is summarised by region in Table 3.17. Significant downward trends were seen at two sites: for fluoranthene in the Ribble estuary, NW England; and for 6 of the 10 PAH at Loch Etive, Scotland.

Sediments

Concentrations of mean ΣPAH in surface sediments ranged from 21 µg/kg dw at a station off western Scotland to 53 600 µg/kg dw (all concentrations are normalised to 2.5% total organic carbon content) at Sun Pier in the River Medway, southeast England. The ΣPAH concentrations for each region are summarised in Table 3.18. Concentrations above 10 000 µg/kg dw were also observed in sediments from the Thames Estuary and Liverpool Bay. Earlier studies (Woodhead et al., 1999) had shown high levels of PAHs in the historically heavily industrialized estuaries of northeast England (Tees, Tyne, Wear), primarily from combustion sources. At some sites these may be acutely toxic to some sediment dwelling organisms, and chronic effects are likely to be more widespread.

For assessing PAH concentrations in sediments a slightly different system was used. The BAC was used to define the blue/green transition point, but the green/amber and amber/red transition points were set to equal the ERL and ERM concentrations, respectively. These adverse effects assessment criteria were derived for the US National Oceanic and Atmospheric Administration using data from multiple marine and estuarine sites in the USA and Canada (Long et al., 1995) and have been widely used in many countries since then. Figure 3.19 shows the colour-coded status for each site and any significant trends. The lowest concentrations occur in the north of Scotland, away from the major centres of population – known industrial and domestic sources of PAH contamination. For benz[*a*]anthracene and benzo[*ghi*]perylene (the two PAH showing the most red locations)

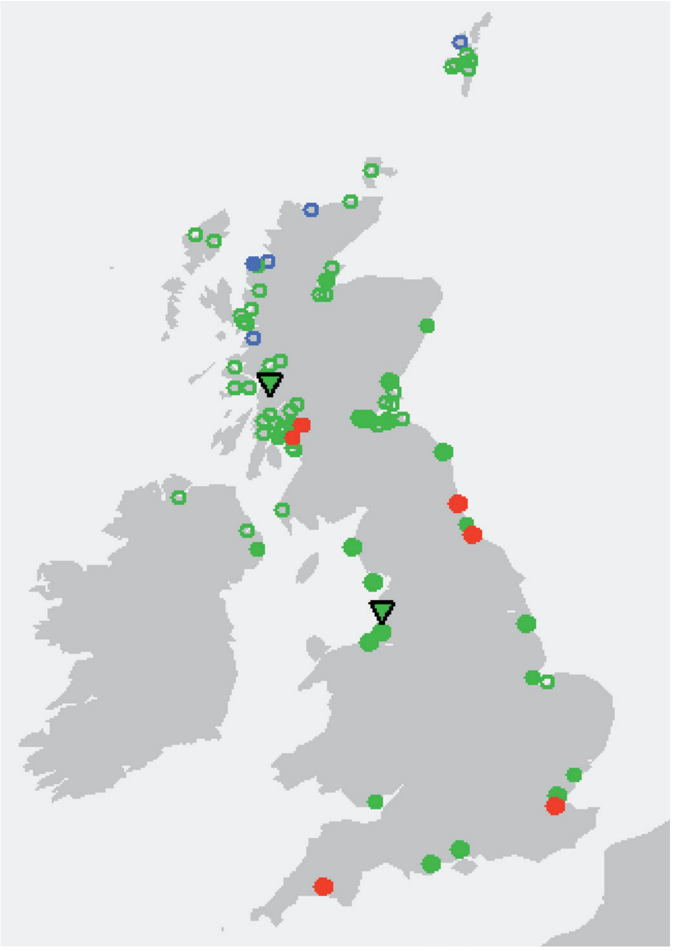
Table 3.17 Assessment summary for PAHs in mussels.

Region	*No. of sites*	*Number of trends upward*	*Number of trends downward*	*Range ΣPAH (µg/kg, ww)*	*Percentage of sites below BAC (i.e. blue status)*	*Percentage of sites below EAC (i.e. green status)*	*Percentage of sites above EAC (i.e. red status)*
Moray Firth	5			10 - 40		100	
East Scotland	3			0.9 - 31		75	25
Forth	2			36 - 50		67	33
Tyne/Tees	4			48 - 297		46	54
Humber/Wash	3			36 - 89		80	20
Anglia	3			41 - 137		70	30
East Channel	2			57 - 100		70	30
West Channel	1			84		40	60
Severn	2			12 - 193		67	33
Irish Sea	10		1	5 - 147		77	23
Clyde	5			0.7 - 603	13	62	25
Minch/Malin	3		6	0.6 - 50	57	36	7

Table 3.18 Assessment summary for PAHs in sediments.

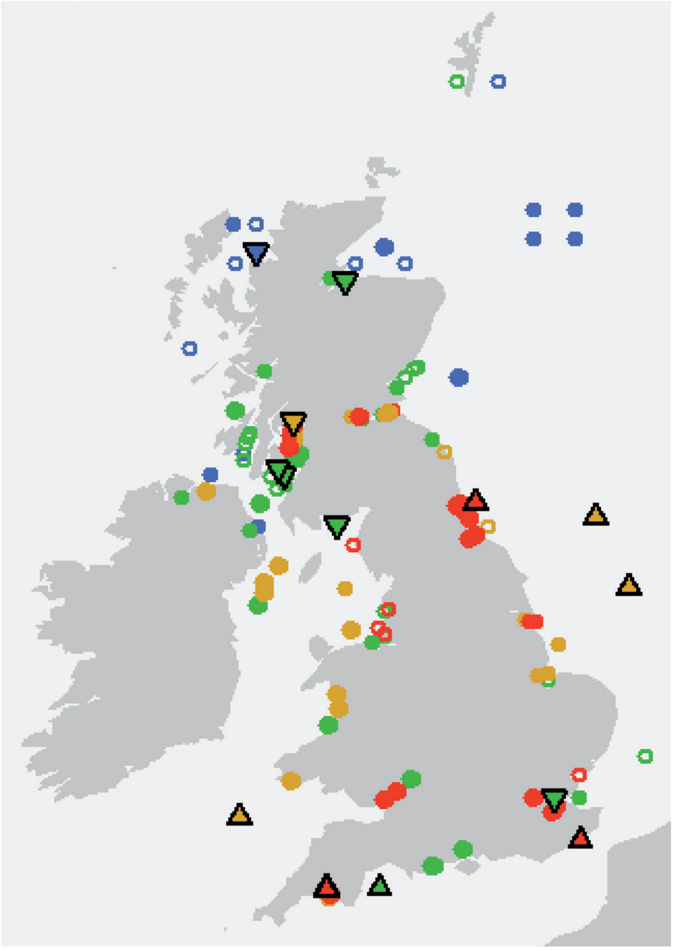
Region	*No. of sites*	*Number of trends upward*	*Number of trends downward*	*Range ΣPAH (µg/kg dw)*	*Percentage of sites below BAC (i.e. blue status)*	*Percentage of sites below ERL (i.e. green status)*	*Percentage of sites below ERM (i.e. amber status)*	*Percentage of sites above ERM (i.e. red status)*
Moray Firth	3		4	102 – 424	54	46		
East Scotland	2			92 & 120	78	22		
Fladen	4			70 – 106	100			
Forth	4			590 – 1830	7	54	25	14
Tyne/Tees	15	3		713 – 8960		24	43	31
Humber/Wash	9	13		349 – 4210	1	52	39	8
Anglia	7	3	5	517 – 53600		31	29	40
East Channel	5	11	5	398 – 1240		62	32	6
West Channel	4	16		3440 – 9610		5	49	46
Severn	5	8	1	640 – 3230		48	44	8
Cardigan Bay	3			866 – 1330		67	30	3
Irish Sea	17		2	315 – 42700	5	74	19	3
Clyde	10		13	344 – 2450	1	61	29	9
Minch/Malin	7		3	21 – 1290	30	52	16	2

Figure 3.18 PAHs in mussels and oysters. © Marine Scotland Science.



Data concern those PAHs for which both a BAC and an EAC have been set. Circles are blue if the mean concentration of all but one of the compounds is significantly below the BAC; green if all but one is below the EAC; and red otherwise. Open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data. Statistically significant upward or downward trends are indicated by triangles (trends were assessed only where there were five or more years of data).

Figure 3.19 PAHs in sediments. © Marine Scotland Science.



Data concern those PAHs for which a BAC, ERL and ERM have been set. Circles are blue if the mean concentration of all but one of the PAH compounds is significantly below the BAC; green if all but one is significantly below the ERL; amber if all but one is significantly below the ERM; and red otherwise. Open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large circles denote 5 or more years of data. Statistically significant upward or downward trends are indicated by triangles (trends were assessed only where there were five or more years of data).

high concentrations (> ERM) were seen in the Clyde, Forth, Tyne, Tees, Humber, Thames and Mersey estuaries, and in the Bristol Channel. The majority are derived from combustion sources.

Trends were only assessed at sites with five years or more of data. In terms of trends for individual parent PAH, where significant upward trends have been observed these are in waters around England and Wales, including some offshore waters. For ΣPAH, two-thirds of the observed trends were upwards and one-third downwards. PAH concentrations in sediments continue to be of concern, particularly as many of the significant trends observed are upward (Table 3.18). For the 21 sites for which upward or downward trends were detected, all the PAH concentrations showing significant trends were moving in the same direction.

Adequacy and confidence of the data

CSEMP PAH data are fully quality assured and we can have confidence in them. Overall, the number of sites sampled is inadequate for full regional assessments to be made with any degree of confidence.

3.1.3.5.4 Progress towards the vision of clean and safe seas

While PAH concentrations are low in many locations, some historically industrialized and contaminated estuaries have concentrations high enough to cause toxicity to sediment-dwelling organisms. These concentrations are declining only very slowly, if at all, as shown by data derived from the analysis of dredged material considered for sea disposal in England and Wales.

3.1.3.5.5 Need for further work

More data are needed for alkylated PAHs so that an assessment can be made for those compounds as well as for the parent compounds. A wider range of carcinogenic PAHs merit study, particularly those of 278 and 302 Daltons molecular weight which occur in contaminated marine sediments (Law et al., 2002). More sampling sites for biota and sediments would improve the ability to make meaningful regional assessments.

3.1.3.6 Brominated flame retardants in sediment and biota

3.1.3.6.1 Summary

Table 3.19 presents our expert opinion on the status and trends for brominated diphenyl ethers (BDEs) in sediments, shellfish and fish from UK waters, respectively. Trend arrows are provided based on the evidence available, showing whether the state or condition is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). A question mark indicates insufficient data. The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

Overall findings

There are no assessment criteria currently available for BDEs and, for the majority of the stations, there are only one or two years of data so trends cannot be determined. Data are available for fish for the UK and for sediment and shellfish for Scottish waters only. The highest UK concentrations for most BDE congeners in fish are found in the Clyde, Irish Sea, Tyne/Tees and Humber/Wash regions – where there are discharges from urban areas and/or industry. The lowest concentrations are

Table 3.19 Assessment summary for BDEs in sediment, shellfish and fish.

Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
Sediment						
East Shetland			?		I	
West Shetland			?		I	
East Scotland			?		I	
Fladen Ground	Oil exploration and production; shipping		?		I	
Moray Firth	Oil production; windfarms; shipping		?		I	
Clyde	Urban area; historic industry; shipping		?		I	
Minch/Malin			?		I	
Shellfish (mussels)						
East Scotland			?		I	
Forth	Urban area		?		I	
Clyde	Urban area; historic industry		?		I	
Minch/Malin			?		I	
Fish						
East Scotland			?		I	
Moray Firth			?		I	
Forth	Urban area		?		I	
Tyne/Tees	Urban area; historic industry		?		I	
Humber/Wash	River inputs		?		I	
Anglia	River inputs		?		I	
East Channel			?		I	
West Channel			?		I	
Severn	Urban areas; historic industry		?		I	
Irish Sea	Urban areas, historic industry		?		I	
Clyde	Urban area; historic industry		a		II	
Minch/Malin			?		I	

Note that the colours in this table are those of the quartiles and not assessed status.

0% 25% 50% 75% 100%

found in the Minch/Malin region, on the Scottish East Coast, Moray Firth, Western Channel and offshore Humber/Wash regions. The highest Scottish concentrations for sediment are found in the Clyde and offshore of the Tay Estuary and the lowest in the Minch/Malin region.

Contribution to GES descriptors

The BDE data illustrate contamination levels in biota and sediments for an emerging, important class of chemicals, which have the ability to harm marine species and human consumers. Within the EU Marine Strategy Framework Directive there is a requirement to assess Good Environmental Status (GES) against a series of descriptors. For BDEs, the most relevant of these are Descriptor 8 ...*concentrations of contaminants are at levels not giving rise to pollution effects* and Descriptor 9 ...*contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards*.

Cross-cutting issues

There are no known cross-cutting issues.

Future risks

Climate change may alter the relative importance of contaminant transport pathways, for example by increasing the movement of some contaminants to the Arctic, but in the context of BDEs in UK waters this is unlikely to make a significant difference to impacts.

3.1.3.6.2 Introduction

For polybrominated diphenyl ethers, two acronyms are in common use: PBDE denotes the formulations which are produced and used industrially, while BDE denotes individual congeners (chemical compounds) within these products. It is BDEs that are generally determined in environmental samples.

Polybrominated diphenyl ethers are one of the most widely used groups of brominated flame retardants (BFRs), although they are now banned in EU countries. BFRs reduce fire hazards by interfering with the combustion of polymeric materials (OSPAR, 2001a; Webster et al., 2006). Commercial PBDE mixtures are classified according to the degree of bromination (OSPAR, 2001a). The penta-mix contains mainly tetra- and penta-BDEs, the octa-mix mainly hexa- to hepta-BDEs and the deca-mix containing mainly deca-BDE. The penta-BDE product is mainly used in furniture and upholstery, the octa-PBDE product in plastics and the deca-PBDE product in textiles (OSPAR, 2001a). On 30 January 2001, the European Commission issued a proposal to ban the penta- technical mixture. This limitation on use was finally put in place on 15 August 2004, restricting the use of the penta- and the octa- technical mixtures to a limit of 0.1% by mass for all articles placed in the market according to the European Directive 2003/11/EC, 24th amendment of 76/769/EEC. Furthermore, the Restriction of Hazardous Substances in Electrical and Electronic Equipment (RoHS) Directive (2002/95/EC), restricting the marketing of electrical and electronic equipment containing hazardous substances, which includes PBDEs, became effective on 1 July 2006. Deca-BDE was exempted under this ruling but the ban was reinstated on 30 July 2008. PBDEs can be released to the environment during their production and while manufacturing other products, as well as during disposal of products containing these chemicals. In addition, PBDEs may continue to leak out of treated material. Penta- and octa- formulations of PBDEs were manufactured by the Great Lakes Chemical

Company at Newton Aycliffe, County Durham in the northeast of England. Manufacture of PBDEs at this plant ceased in 1996.

There is a limited amount of chronic and acute toxicity data for BFRs. The lower brominated BDEs are more toxic and more likely to bioaccumulate. Deca-BDE is considered to be the least toxic of all PBDEs, mainly due to its large molecular weight, which should reduce its tendency to bioaccumulate although there are concerns that it may break down to the more harmful tetra- and penta-congeners in the environment (Darnerud et al., 2001). There is little information on the health effects of BDEs in humans although animal studies have shown that BDEs are endocrine disruptors, affect thyroid hormone functions and can impair the developing central nervous system and brain (Darnerud et al., 2001; Zou et al., 2002).

PBDEs have the potential for long-range atmospheric transport, and are found in the Arctic at levels higher than those which would be associated with any local use. In addition, release from products containing PBDEs will constitute a diffuse source of these compounds to the environment.

There are 209 possible PBDE congeners. However, PBDE technical mixtures contain only a few of these congeners (~20). Nine of these (BDE28, BDE47, BDE66, BDE100, BDE99, BDE85, BDE154, BDE153, BDE183) were selected, taking into account their occurrence in the environment and their toxicity, to be routinely determined as part of the OSPAR CEMP (OSPAR, 2007a).

Environmental impacts

The environmental impacts of PBDEs are not fully understood at present due to the lack of data, particularly toxicological information.

Socio-economic impacts

Brominated flame retardants help to reduce death and property loss by retarding fires. However, any negative socio-economic impacts they may have are currently not known.

European legislation and regulation

European legislation and regulation concerning PBDEs includes the OSPAR list of Priority Hazardous Substances; the EU Water Framework Directive, the EU Restriction of Hazardous Substances in Electrical and Electronic Equipment (RoHS) Directive (2002/95/EC) and the EU Directive 2003/11/EC, 24th amendment of 76/769/EEC.

3.1.3.6.3 Developments since *Charting Progress*

Developments in key pressures, understanding and management measures

A suite of BDE congeners has recently become mandatory determinands for OSPAR countries (OSPAR, 2008) which means that more data should become available in the future.

Developments in approach and indicators used

This is an emerging group of contaminants, for which *Charting Progress* reported a limited data set for some regions. Although there has been an increase in the spatial and temporal coverage of these substances, in most places this is limited to one or two years of data.

3.1.3.6.4 Presentation of the evidence

As there are no assessment criteria for BDEs in sediment and biota (fish and shellfish) the data were summarised by dividing the reported concentrations for the UK stations into

quartiles (see Table 3.20) and representing them graphically using the colours shown in the key to Table 3.19. Some stations will be in the upper quartile (75% to 100%) for all congeners. But unlike the data where assessment criteria are available, these stations are not automatically problem areas – classification in the upper quartile signifies only that these are the areas with the highest UK concentrations. For sediment and shellfish these are the highest Scottish concentrations as data in these matrices are not available for England and Wales.

Owing to the lack of assessment criteria the 'traffic light' colours as applied to other contaminants cannot be applied here. The overall concentration of the BDE congener for the UK (fish) and Scotland (sediment and shellfish) is divided into quartiles and coloured as shown in the key to Table 3.19 and the values for each BDE congener are given in the relevant tables. Because of the lack of assessment criteria, in particular BACs, this is an arbitrary division and therefore those stations in the highest quartile are not necessarily problem areas.

Sediment

The nine BDE congeners (BDE28, BDE47, BDE66, BDE100, BDE99, BDE153, BDE154, BDE85 and BDE183) were detected in sediments from sites around the Scottish coast and also from the offshore Fladen Ground. Except for part of the Clyde Estuary all congeners except BDE154 were below the limit of quantification (LoQ).

For the Clyde Estuary sites, the dominant congeners were BDE47, BDE99 and BDE154, with maximum concentrations of 0.74, 2.21 and 1.23 μg/kg dw respectively (all concentrations are normalized to 2.5% total organic carbon). There are as yet no assessment criteria for any of the BDE congeners in sediment so the significance of these concentrations cannot

Table 3.20 BDE concentrations defining sediment, shellfish and fish quartiles.

	0%	*25%*	*50%*	*75%*	*100%*
Sediment (μg/kg dw normalised to 2.5% total organic carbon) (for Scotland only)					
BDE28	0.03	0.08	0.13	0.21	0.74
BDE47	0.07	0.09	0.15	0.26	0.74
BD100	0.07	0.08	0.13	0.21	0.74
BDE99	0.22	0.26	0.46	0.63	2.21
BDE66	0.05	0.09	0.13	0.21	0.74
BDE85	0.03	0.13	0.21	0.35	1.23
BD154	0.12	0.25	0.31	0.43	1.23
BD153	0.07	0.08	0.13	0.21	0.74
BD183	0.03	0.08	0.13	0.21	0.74
Shellfish (mussels) quartiles (μg/kg ww) (for Scotland only)					
BDE28	0.10	0.10	0.10	0.10	0.10
BDE47	0.10	0.10	0.20	0.20	0.77
BD100	0.10	0.10	0.10	0.10	0.40
BDE99	0.10	0.10	0.10	0.17	0.55
BDE66	0.10	0.10	0.10	0.10	0.10
BD153	0.10	0.10	0.10	0.10	0.10
BDE85	0.10	0.10	0.10	0.10	0.10
BD154	0.10	0.10	0.10	0.10	0.11
BD183	0.10	0.10	0.10	0.10	0.10
Fish quartiles (μg/kg lw) for the UK					
BDE28	0.29	1.02	4.56	16.81	41.18
BDE47	1.18	9.61	28.49	49.16	188.82
BD100	0.86	4.58	8.36	19.43	38.51
BDE99	0.27	1.05	7.26	16.37	41.18
BDE66	0.25	0.83	5.82	15.43	41.18
BD153	0.27	0.90	5.27	16.94	41.18
BDE85	0.24	0.45	4.34	15.49	41.18
BD154	0.40	1.82	6.75	12.99	28.09
BD183	0.27	0.62	5.79	41.69	176.47

be assessed. Assessment criteria need to be developed.

The data for individual BDE congeners are shown in Figure 3.20. The concentrations are expressed in µg/kg dw normalised to 2.5% total organic carbon. One problem with this is that in many cases the LoQ was normalised because the actual congener concentration was below the LoQ. For areas with low levels of total organic carbon and BDE concentrations below the LoQ, this approach can give an apparently significant contaminant loading.

No trend assessments could be made for BDEs in sediment as five or more years of data were not available.

Shellfish

The nine BDE congeners (BDE28, BDE47, BDE66, BDE100, BDE99, BDE153, BDE154, BDE85 and BDE183) were measured in farmed and wild blue mussels from long-term monitoring sites in Scotland (Figure 3.21). At present there are data available from 2006 and 2007 for each site. The dominant congeners were BDE47, BDE99, BDE100 and BDE154. Concentrations for BDE47 ranged from below the LoQ to 0.77 µg/kg ww. The Aberdeen Harbour site (East Scotland) had the highest concentrations.

Fish

There are more data available on the concentrations of BDEs in fish in UK waters (Figure 3.22) but again there are no assessment criteria and, for many of the sites there are only data for one or 2 years. The exceptions are the Clyde, for which there are 5 years of data and the Irish Sea, Cardigan Bay, Severn, Humber/Wash and Tyne/Tees regions where there are three to four years of data. The fish species analysed were *Limanda limanda* (dab – England and Wales), *Platichthys flesus* (flounder – Scotland, two sites) and *Pleuronectes platessa* (plaice – Scotland all other sites). The dominant congener was BDE47.

The areas with the highest concentrations for BDE47 were the Clyde (47.8 – 189 µg/kg lw), Tyne/Tees (13 – 111 µg/kg lw), Humber/Wash (14.5 – 64.8 µg/kg lw) and the Irish Sea (19 – 106 µg/kg lw). These are all near areas of concentrated industrial activities onshore. In addition, in the Clyde there is a former sewage sludge dump site at Garroch Head and this contributes to the higher concentrations there. The Tyne/Tees stations are downstream from the former PBDE manufacturing plant at Newton Aycliffe (Allchin et al., 1999).

Only one site in the Clyde had enough years of data for a trend assessment and this indicated a slight downward trend in concentrations.

Monitoring period

BDEs are a ‘new’ or emerging contaminant group and so there is only a limited quality assured dataset available. In particular, BDE data for sediments is limited to only one year of data, from sites in Scotland only. There are more data available for BDEs in fish, particularly for areas such as the Tyne/Tees, Irish Sea and Clyde Estuary regions. Shellfish data are available for five sites in Scotland only, for 2006 and 2007.

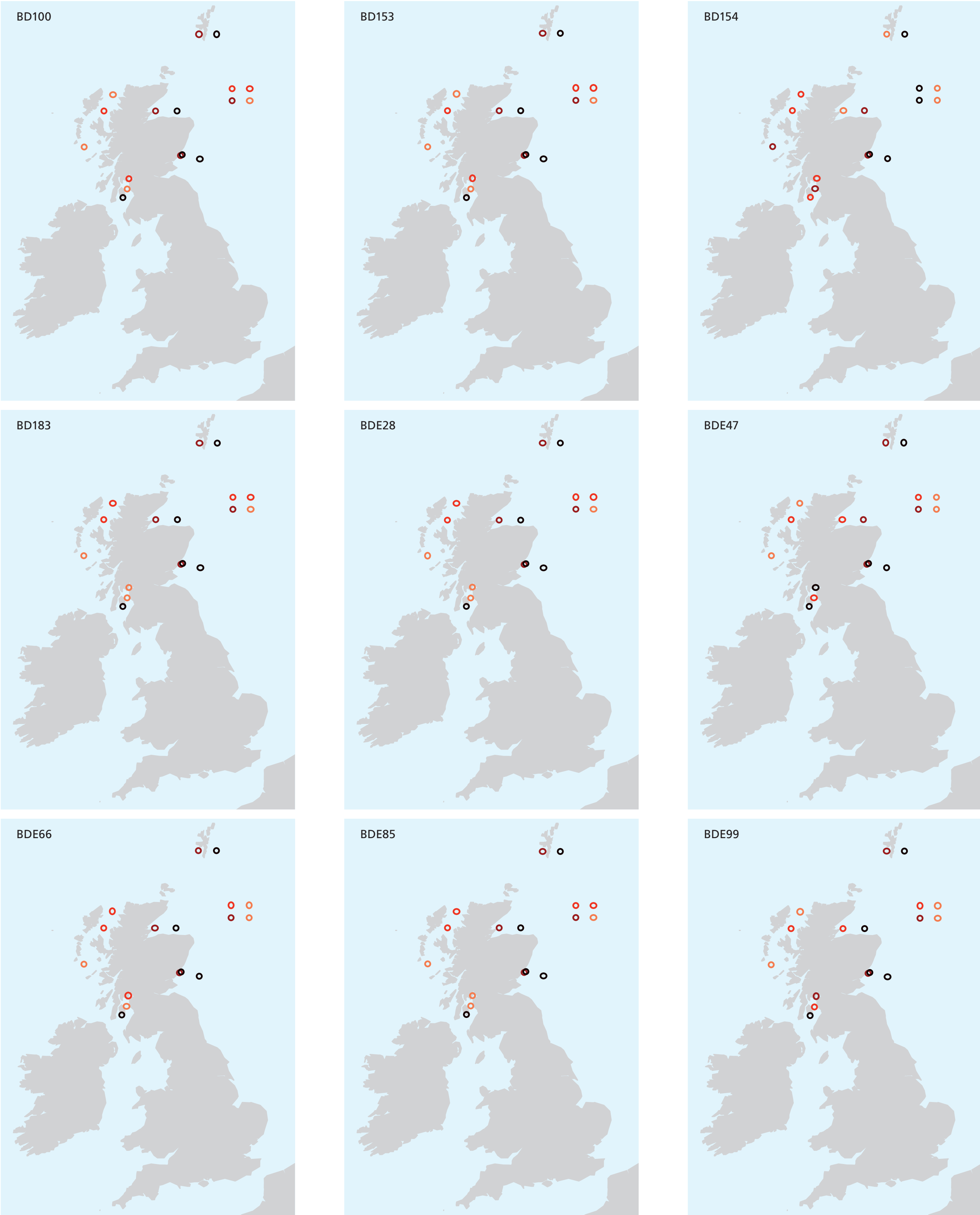
Data sources

The MERMAN database at the British Oceanographic Data centre holds the CSEMP data.

Adequacy and confidence in the data

CSEMP data for BDEs are fully quality assured but the number of sites sampled and the timescale over which they are sampled are

Figure 3.20 Assessment summary for PBDEs in Scottish sediments. © Marine Scotland Science.



Colours are as specified in the key to Table 3.19. Open circles denote 2 years of data only. Concentrations are expressed in µg/kg dw normalised to 2.5% total organic carbon.

Figure 3.21 Assessment summary for PBDEs in shellfish (mussels). © Marine Scotland Science.

Colours are as specified in the key to Table 3.19. Concentrations are expressed in µg/kg ww.

inadequate for full regional assessments to be made with any degree of confidence.

Trends

No overall trends could be identified as there were insufficient data and no assessment criteria.

Gaps

Additional data are needed, as well as assessment criteria, before a robust assessment can be made.

Contribution to GES descriptors

The BDE data illustrate contamination levels in biota and sediments for an emerging, important class of chemicals, which have the ability to harm marine species and human consumers. This addresses GES Descriptors 8 and 9 in the EU Marine Strategy Framework Directive.

3.1.3.6.5 Progress towards the vision of clean and safe seas

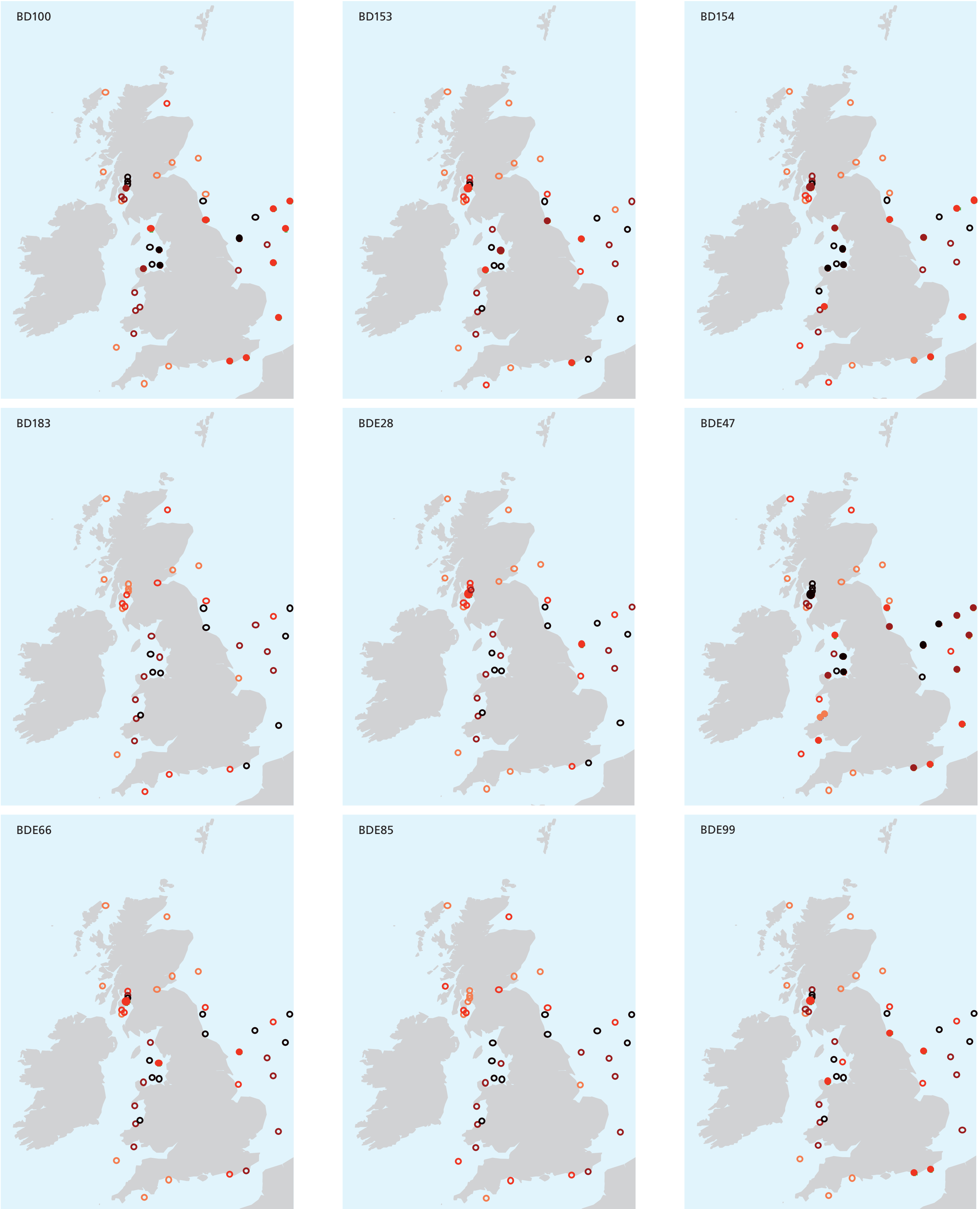
Areas of possible concern for BDEs are those traditionally associated with industry, such as the Mersey, the Clyde and the Tyne/Tees area. Because there are no assessment criteria available for BDEs and it is difficult to assess the significance of the concentrations found. Of all the stations for which there are data available, only that of the Middle Clyde Estuary had sufficient data for a trend to be ascertained and this trend appears to be downward.

3.1.3.6.6 Forward look and need for further work

BDEs are expected to persist in the marine environment, even though their use has been banned within the EU, and they exhibit similar toxic properties to the CBs (OSPAR, 2001a). There are limited data available on the levels required to affect human health and on what these human health effects may be. Of increasing concern is the effect of mixtures of persistent organic pollutants and also of the effects of long-term, low-level exposure to these pollutants.

The lack of assessment criteria makes it impossible to determine whether the concentrations found in the UK environment are significant. Background concentrations for hazardous substances included in the CEMP have been developed by OSPAR, with input

Figure 3.22 Assessment summary for BDEs in fish. © Marine Scotland Science.



Open circles denote 1–2 years data, small closed circles indicate 3–4 years data, large closed circles denote 5 or more years of data..

from various ICES Working Groups. Observed concentrations are said to be 'near background' if the mean concentration is statistically significantly below the corresponding BAC. EACs represent the contaminant concentration in the environment below which no chronic effects are expected to occur in marine species, including the most sensitive species. Therefore the priority for further work must be to provide such assessment criteria in order that an assessment of PBDE monitoring data in the UK marine environment can be carried out.

3.1.3.7 Metals in biota

3.1.3.7.1 Introduction

Fish and shellfish take up contaminants from their surroundings. Contaminants that cannot be excreted are accumulated over time as a 'body burden', integrating the fluctuations in concentrations in surrounding waters, particulate material and in their prey species.

Sampling is limited to a few species to reduce complications caused by inter-species variability. Fish recommended for contaminant monitoring include dab (*Limanda limanda*), flounder (*Platichthus flesus*) and plaice (*Pleuronectes platessa*), with whiting (*Merlangius merlangus*) allowed where these species are not available. For shellfish, the blue mussel (*Mytilus edulis*) is preferred.

3.1.3.7.2 Developments since *Charting Progress*

Whereas *Charting Progress* covered data for the 3-year period 1999 to 2001, this report includes data up to 2007, generally for the most recent three to eight years.

The most substantive change in this report compared to *Charting Progress* is in the assessment criteria used to set the results in context. In *Charting Progress*, OSPAR Background Reference Concentrations (BRCs) were used to assess concentrations of Hg in fish, and Hg, Cd, Pb, Cu and Zn in mussels. However, in this report, the BRCs have been replaced by the Background Assessment Concentrations (BACs) proposed by ICES for Hg, Cd and Pb in mussels on the basis of 'low concentrations' typical of remote areas, and derived by OSPAR on a statistical basis for fish.

Background to BRCs and BACs

In 1998, OSPAR set out its Hazardous Substances strategy with an ultimate aim of achieving concentrations in the marine environment near background values for naturally occurring substances and close to zero for man-made synthetic substances (OSPAR, 1998a). In order to assess concentrations of contaminants in the marine environment, OSPAR produced a set of background reference concentrations (BRCs) as a range of concentrations observed in remote areas. Concentrations less than twice the maximum BRC were described as 'close to background' (MON, 2003).

In 2008, however, ICES pointed out that there is a difficulty with a unique concept of a natural background concentration for individual contaminants in biota for the entire OSPAR convention area (the North-East Atlantic, the North Sea and the Baltic) due to the differences across the convention area, for example geochemical differences, oceanographic factors such as upwelling, and different transport pathways (ICES, 2008). Furthermore, ICES stated that there is no sound methodology to determine natural background (pre-industrial) concentrations of these contaminants in biota.

To expedite the 2008 data assessment for the OSPAR CEMP, ICES proposed the use of Background Assessment Criteria (BACs) based on 'low concentrations' of metals in mussels from coastal areas in various OSPAR countries, including Spain, Greenland, Shetland, the Faroe Islands and the west of Ireland. The BACs are set as a multiple of the mean 'low concentrations', depending on the variability in the data (see Table 3.21), and are used in this report for metals in mussels.

ICES recommended to OSPAR that the high variability in the limited data available for fish due to biological factors such as diet, size and age precludes making any recommendation for low or background concentrations for fish, and recommended that OSPAR use a statistical approach to derive BACs such as that in the 2007 MON report should be used. This approach has been adopted for this report.

Use of EU Limit Values to protect human health

Another change in this report compared to *Charting Progress* is the replacement of the Ecotoxicological Assessment Criteria (EACs) used in *Charting Progress* as assessment criteria by limit values derived by the European Commission to protect human health, set in EU Regulation No. 1881/2006 (EC, 2006) as maximum levels for contaminants in fish and shellfish as foodstuffs. Limit values have been set for Cd, Hg and Pb in bivalve molluscs and fish muscle. However, Cd and Pb are measured in fish liver, and since contaminant concentrations are higher in fish liver than in fish flesh, the use of the fish flesh limit value is not appropriate. Instead, it was decided to use the EU limit values set for bivalve mussels for the assessment of Cd and Pb in fish liver, in order to more closely match the matrices and hence provide more appropriate

Table 3.21 ICES 'low concentrations' and OSPAR BACs for metals in mussels.

Metal	*ICES range of 'low concentrations', as µg/kg ww*	*ICES derived 'low concentrations', as µg/kg ww*	*OSPAR BACs, as µg/kg ww*
Cadmium	45–730	120	192
Mercury		10	18
Lead	70–280	160	260
Copper	910–1500	1100	

limit values. In the absence of reference concentrations or limit values for the other metals in mussels, these data are presented as quartiles.

3.1.3.7.3 Presentation of the evidence

Metals in fish

The OSPAR CEMP guidelines require Hg to be measured in fish flesh, and Cd and Pb to be measured in fish liver.

Mercury in fish flesh

In *Charting Progress* (data for 1999–2001), Hg concentrations in fish flesh were compared to BRCs of 30 to 70 µg/kg ww. In this report, results are compared to the OSPAR BAC of 35 µg/kg ww set on the basis of the statistical method outlined above, and to the EU limit value of 500 µg/kg ww for fish muscle and bivalves as foodstuffs, set to protect human health (see Table 3.22).

For this report (data to 2007), mean Hg concentrations were in the range 8 to 276 µg/kg ww, with levels at most sites above the BAC but below the EC limit value (shown as green circles in Figure 3.23). However, this homogeneity indicates that the assessment criteria do not have sufficient sensitivity to

identify more contaminated sites compared to less contaminated sites. Slightly elevated concentrations were observed in dab from Cardigan Bay and the Irish Sea at 112 to 117 µg/kg ww, in flounder from the Medway Estuary at 112 µg/kg ww, and in plaice from the Irish Sea at 127 µg/kg ww. More elevated concentrations were measured in dab from Liverpool Bay and the Irish Sea in the range 196 to 276 µg/kg ww, and in flounder from the Severn, Thames, Mersey and Ribble estuaries at 138, 198, 195 and 271 µg/kg ww, respectively. These concentrations are similar to the range of 200 to 230 µg/kg ww reported in *Charting Progress* for the Mersey Estuary, Liverpool Bay and Thames Estuary for the period 1999 to 2001. Figure 3.23 indicates that downward trends were identified at only three sites and upward trends at two sites.

In summary, concentrations of Hg in fish flesh are elevated in some industrial estuaries, although there do not appear to be risks to human health. Monitoring should be maintained in areas where elevated concentrations have been found.

Cadmium in fish liver

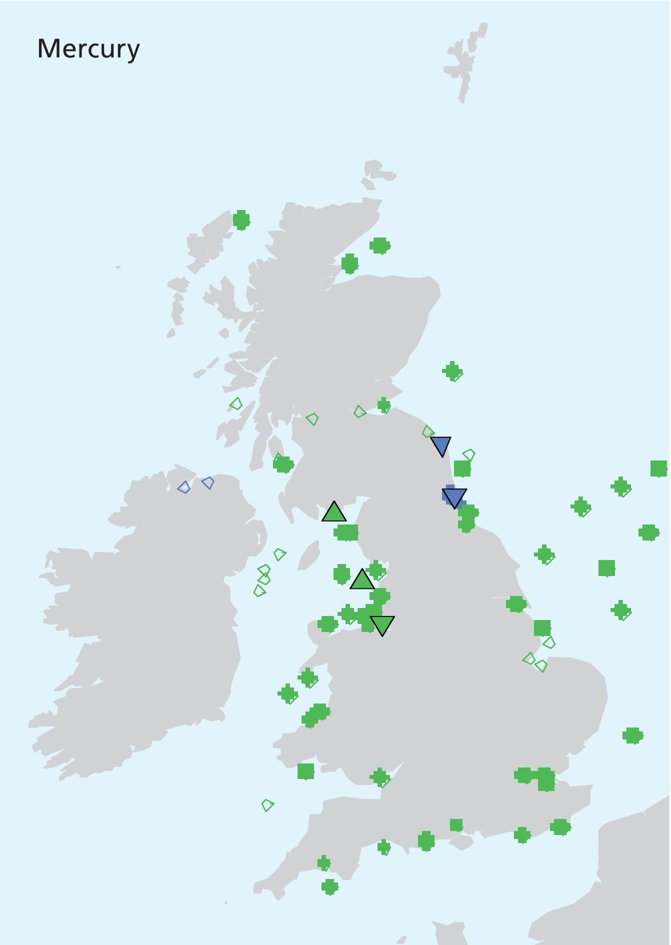
For *Charting Progress*, there were no BRCs, BACs or other assessment criteria available for Cd in fish lever. In contrast, concentrations in this report are compared to the BAC derived by OSPAR and to the EC limit value for mussels, set to protect human health (see Table 3.22).

Charting Progress (data for 1999–2001) reported concentrations of Cd in fish liver as being mostly below 150 µg/kg ww, with higher concentrations in the Moray Firth, West Dogger area, Thames Estuary and Inner Cardigan Bay, at 175, 343, 180 and 335 µg/kg ww, respectively.

Table 3.22 BACs and EC limit values for metals in fish.

Metal	*OSPAR BAC (µg/kg ww)*	*EC limit value (µg/kg ww)*
Cadmium – fish liver	26	1000 (bivalves)
Mercury – fish flesh	35	500 (fish muscle)
Lead – fish liver	26	1500 (bivalves)

Figure 3.23 Mercury in fish flesh. © Marine Scotland Science.



Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

For this report (data to 2007), mean Cd concentrations in fish liver ranged from 15 µg/kg ww in Poole Harbour and in the Dee Estuary in North Wales to 389 µg/kg ww at the Humber/Wash Open Sea NE site. Mean concentrations were below 150 µg/kg ww at 35 of the 59 sites sampled and with 95th percentile confidence limits intermediate between the BAC and the EC limit value (shown as green circles in Figure 3.24). However, this homogeneity indicates that the assessment criteria do not have sufficient sensitivity to identify more contaminated sites compared to less contaminated sites. Mean concentrations for the period to 2007 for the areas reported in *Charting Progress* as having higher concentrations were respectively 253, 277, 131 and 187 µg/kg ww, indicating that concentrations were broadly similar to those previously reported.

In summary, concentrations of Cd in fish liver appear not to be of concern either in terms of environmental harm or human health. Monitoring should be maintained in areas where elevated concentrations have been found.

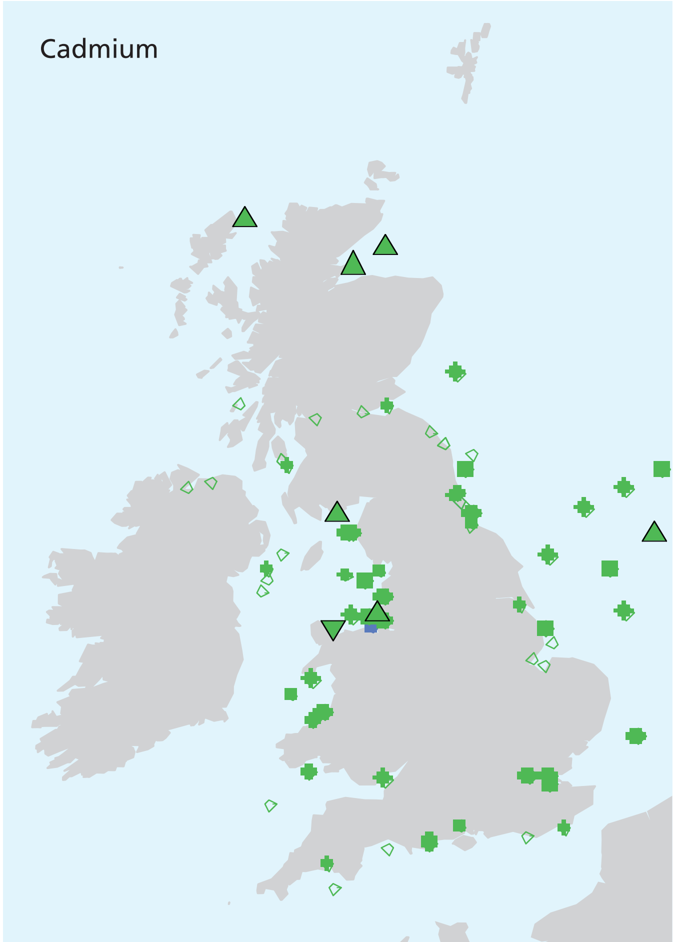
New assessment criteria for Cd in fish liver are needed, to provide better discrimination between sites.

Lead in fish liver

For *Charting Progress*, there were no BRCs, BACs or other assessment criteria available for Pb in fish liver. In contrast, concentrations in this report are compared to the BAC derived by OSPAR and to the EC limit value for mussels, set to protect human health (see Table 3.22).

Charting Progress (data for 1999–2001), reported that Pb concentrations in fish liver were below 200 µg/kg ww at most sites, with higher concentrations in the range 200 to 700 µg/kg ww in dab or flounder from sites in the Tyne, Tees, Wear and Humber estuaries.

Figure 3.24 Cadmium in fish liver. © Marine Scotland Science.



Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

For this report (data to 2007), mean Pb concentrations were again below 200 µg/kg ww at most sites, with higher concentrations in the range 200 to 874 µg/kg ww in dab or plaice from industrialized estuaries including the Forth, Tyne, Tees, Wear, Severn, Mersey and Bann.

Concentrations were also elevated in Cardigan Bay, the Irish Sea, the Moray Firth, Morecambe Bay and the outer Firth of Clyde. Most sites had mean concentrations between the BAC and the EC limit value (shown as green circles in Figure 3.25). However, at some sites the 95th percentile concentrations were above the EC limit value (shown as red circles in Figure 3.25). These were in Cardigan Bay, the Dogger Bank area, the Irish Sea and the Mersey Estuary, with 95th percentile concentrations of 1500, 1562, 1864 and 2361 µg/kg ww respectively at individual sites. These levels of contamination were found at individual sites in each area and not at every site in these areas, and for this reason and because fish flesh is consumed rather than fish liver, these concentrations are likely to pose low risks to human health.

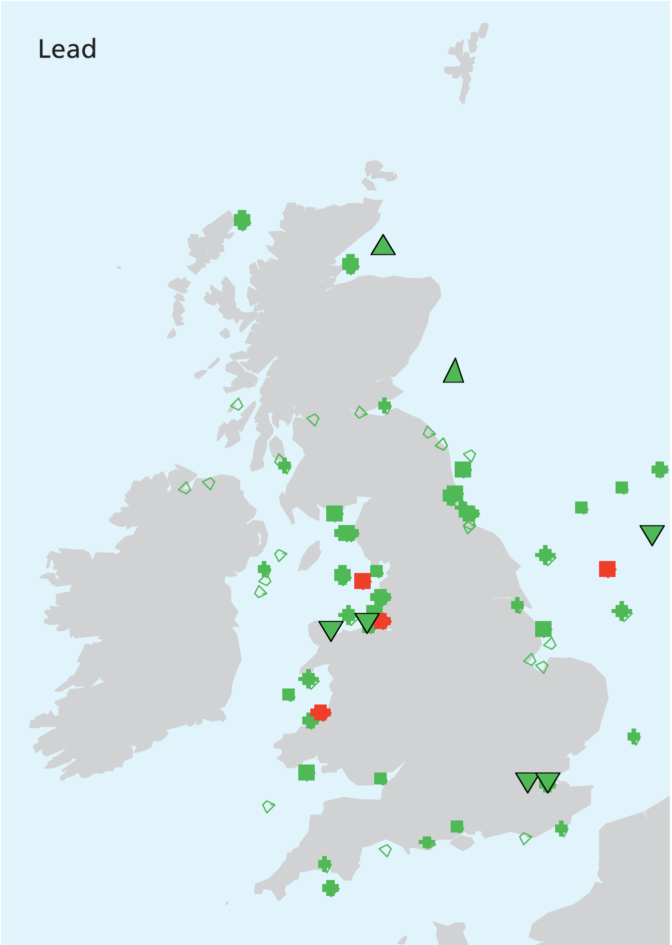
In summary, Pb concentrations in fish liver are elevated in industrialized estuaries and in a few other coastal areas, but are unlikely to pose a risk to human health. Monitoring should be maintained in areas where elevated concentrations have been found.

New assessment criteria for Pb in fish liver are needed, to provide better discrimination between sites.

Other metals in fish flesh

Data for metals other than Hg, Cd and Pb in fish flesh are sporadic, with varying numbers of sites sampled for each metal. There are sufficient data available only for As, Cu and Zn to make these data worth showing (see Figure 3.26). However, there are no BACs or EC limit values available for these metals, so the data are shown as quartiles. It is difficult to assess these data objectively in terms of their significance.

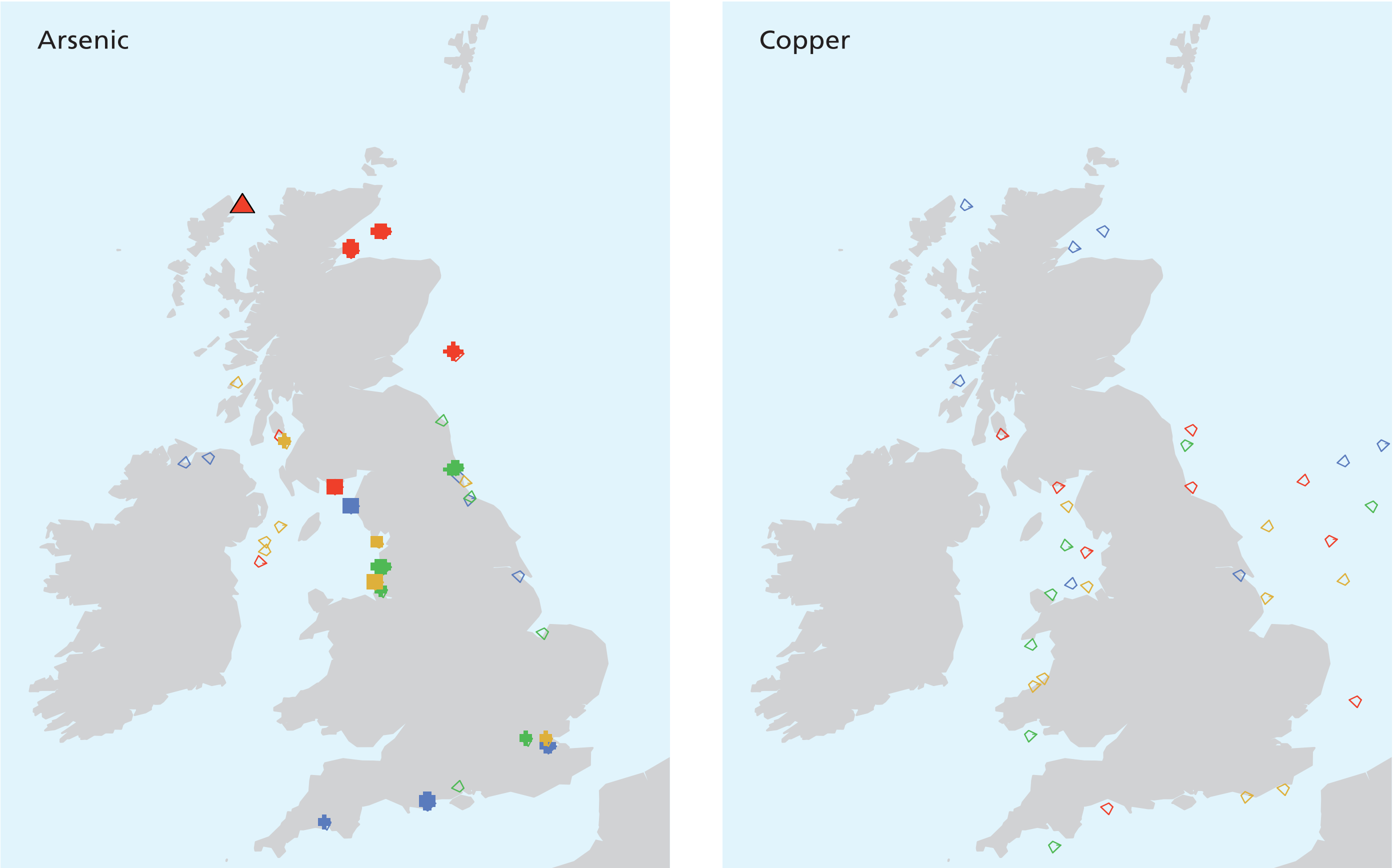
Figure 3.25 Lead in fish liver. © Marine Scotland Science.

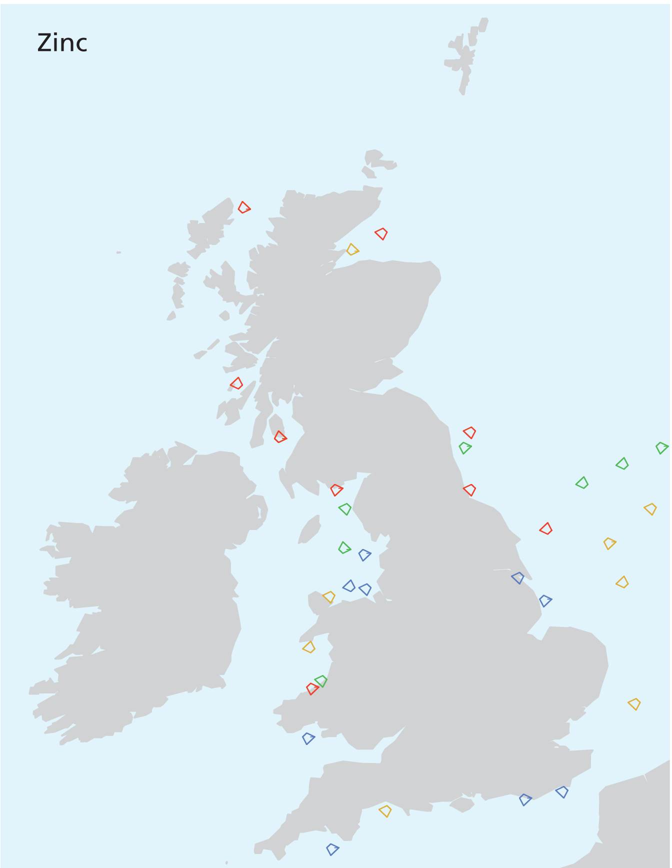


Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

The limited amount of data reported for metals other than Cd, Hg and Pb is probably due to the lack of a regulatory driver, and has contributed to the lack of suitable assessment criteria against which to compare the data. As a result, unless concentrations are elevated locally due to known metal inputs, the merits in continuing to collect data for metals other than Hg, Cd and Pb in fish flesh are questionable.

Figure 3.26 Arsenic, copper and zinc in fish flesh. © Marine Scotland Science.





Blue circles represent the lowest 25% of concentration data, green circles the next 25% of concentration data, amber circles the next 25% of concentration data, and red circles the highest 25% of concentration data. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

New assessment criteria for metals in fish flesh are needed, because the current ones are based on food limits set for the protection of human consumers rather than ecotoxicological criteria, and because BACs are based on low concentrations rather than background concentrations.

Metals in mussels

In *Charting Progress*, concentrations of nine metals (Cd, Hg, Pb, As, Cu, Cr, Ni, silver [Ag] and Zn) were assessed against OSPAR Background Reference Concentrations (BRCs). In this report, concentrations are reported for these metals as well as for selenium (Se). Assessment criteria used here are the OSPAR BACs and the EC limit values for Cd, Hg and Pb (see Table 3.23).

Mercury in mussels

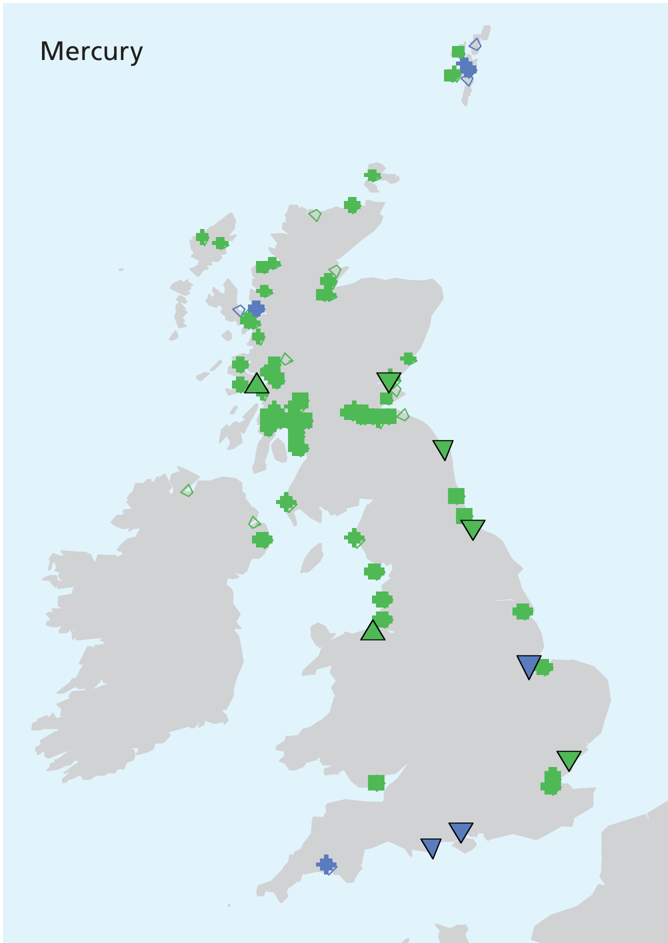
In *Charting Progress* (data for 1999–2001), the lowest Hg concentration in mussels was reported as 14.5 µg/kg ww in the Tamar Estuary. Concentrations above 50 µg/kg ww were reported for the Northumberland and Durham coasts, Morecambe Bay, and the Blackwater, Thames and Mersey estuaries.

For this report (data to 2007), the lowest mean Hg concentrations were in the range 10 to 15 µg/kg ww in a few voes and firths in Shetland, in Loch Ewe, Loch Carron and Loch Striven on the west coast of Scotland, in Loch Leurbost in the Western Isles, in the Outer Clyde Estuary and Loch Long, and in The Wash, Poole Harbour, Southampton Water and Plymouth Sound. Concentrations for all other sites were above the BAC but well below the EU limit value (shown by green circles in Figure 3.27). Concentrations were elevated to over 50 µg/kg ww in industrialized estuaries such as the Forth, the Wear, the Thames, the Mersey and Liverpool Bay (68, 67, 58, 62 and 56 µg/kg ww, respectively),

Table 3.23 BACs and EC limit values for metals in mussels.

Metal in mussels	*OSPAR BAC (µg/kg ww)*	*EC limit value (µg/kg ww)*
Cadmium	192	1000
Mercury	18	500
Lead	260	1500

Figure 3.27 Mercury in mussels. © Marine Scotland Science.



Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

but the highest concentration of 163 µg/kg ww was observed in the Severn Estuary. Elevated concentrations were also found in Cumbria at 56 µg/kg ww, and in Morecambe Bay at 76 µg/kg ww. Although significant downward trends were observed at seven sites, and upward trends at two sites (see Figure 3.27), concentrations at most sites were in a similar range to those previously reported.

In summary, concentrations of Hg in mussels are slightly elevated in some industrial estuaries, but there do not appear to be risks to human health. Monitoring should be maintained in areas where elevated concentrations have been found.

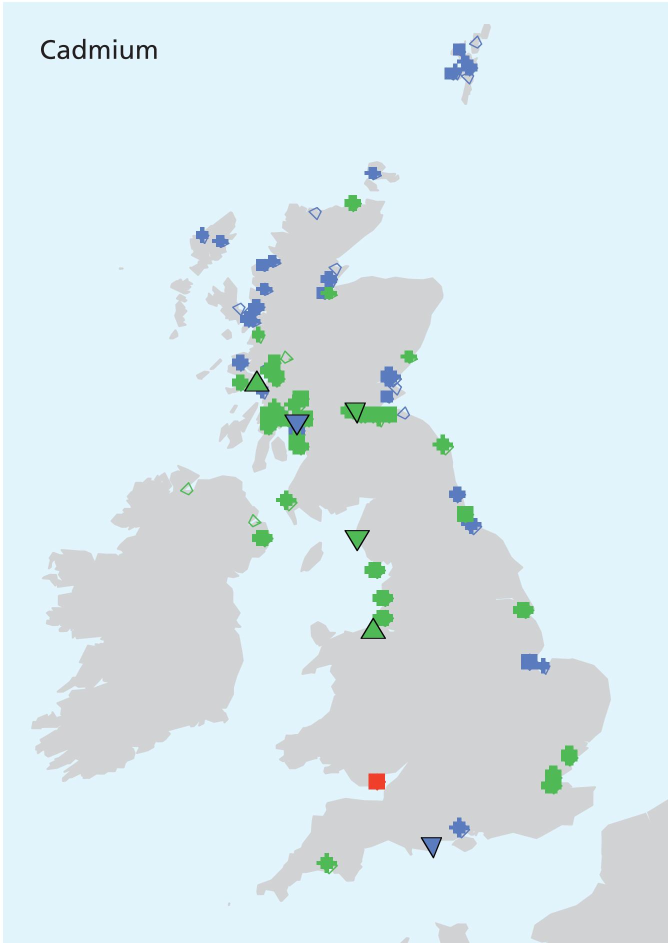
Cadmium in mussels

In *Charting Progress* (data for 1999–2001), the lowest Cd concentrations were reported as 102 µg/kg ww in The Wash, with low concentrations also found in mussels from the Forth and Tay estuaries, in the Firth of Clyde and Poole Harbour. Median concentrations above 500 µg/kg ww were reported in the Humber and Thames estuaries, with the highest concentration at 1178 µg/kg ww in the Medway Estuary.

The recent data for this report (data to 2007) show that Cd concentrations were low and 'close to background' at sites on the east and north-west coasts of Scotland, in The Wash and on the south coast of England (see Figure 3.28). Concentrations were below 200 µg/kg ww and hence intermediate between the BAC and the EC limit value at most other sites (shown by green circles in Figure 3.28). Mean concentrations were above 500 µg/kg ww in the Dee Estuary, Liverpool Bay, Humber and Thames estuaries at 506, 555, 564 and 690 µg/kg ww, respectively, while the highest mean concentration by far was 2420 µg/kg ww in the Severn, where the 95th percentile on the mean was above the EC limit value (shown by a red circle in Figure 3.28). Significant downward trends were identified at only three sites and upward trends at two sites, and concentrations showed little overall change since *Charting Progress*.

In summary, concentrations of Cd in mussels are elevated in some industrial estuaries; although apart from in the Severn Estuary, there do not

Figure 3.28 Cadmium in mussels. © Marine Scotland Science.



Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

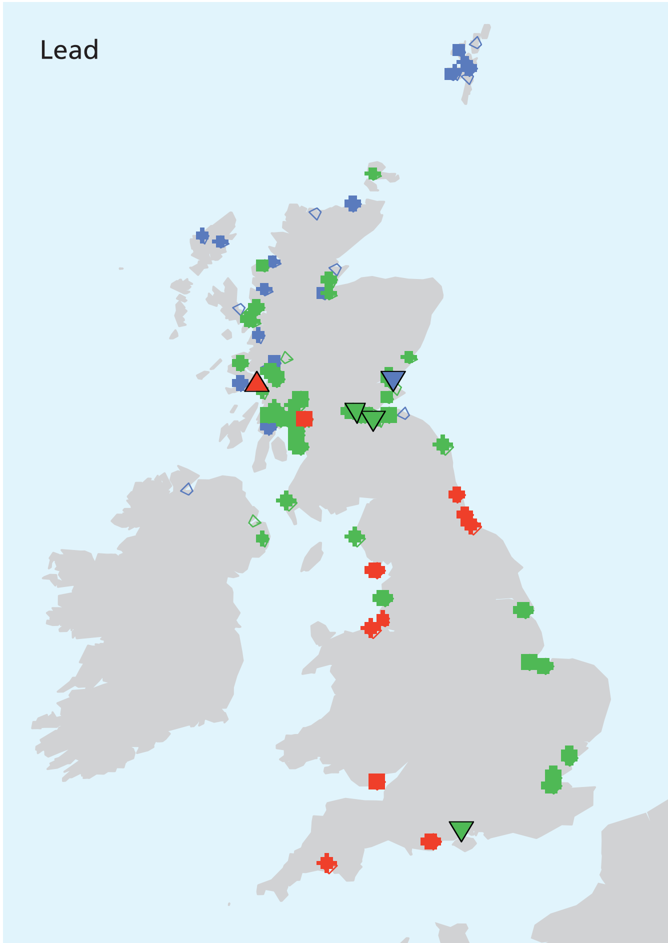
appear to be risks to human health. Monitoring should be maintained in areas where elevated concentrations have been found.

Lead in mussels

In *Charting Progress* (data for 1999–2001), the lowest Pb concentration was 147 µg/kg ww in Lough Foyle. Concentrations were close to background at only two sites. Concentrations were above 1000 µg/kg ww in the Tyne, Tees and Forth estuaries, and on the Northumberland and Durham coast, in the Tamar Estuary and in the Clyde Estuary and Morecambe Bay.

The recent data for this report (data to 2007) show that the lowest Pb concentrations were similar to those reported previously, in the range 138 to 146 µg/kg ww in mussels from Loch Scridain, Loch Linnhe and Little Loch Broom on the north-west coast of Scotland and at Cat Voe on Shetland. Mean concentrations at a further 15 sites were below 200 µg/kg ww and had 95th percentiles below the BAC of 304 µg/kg ww. These were mainly in the north and north-west of Scotland and in Shetland, and may be described as having concentrations 'close to background' (shown by blue circles in Figure 3.29). Pb concentrations were intermediate between the BAC and the EC limit value at most sites (shown by green circles in Figure 3.29), with mean concentrations in the range 500 to 1000 µg/kg ww in industrial estuaries such as the Forth, Humber, Medway, Thames, and Clyde, and also at Broadford Bay in Skye and Loch Spelve in Mull. These elevated levels may be due to local mineralisation. High mean Pb concentrations in the range 1000 to 1885 µg/kg ww, some exceeding the EC limit value, were found in mussels at several sites in industrialized estuaries (shown by the red circles in Figure 3.29). These include the Tees, Tyne, Wear, Poole Harbour, Plymouth Sound, Severn

Figure 3.29 Lead in mussels. © Marine Scotland Science.



Circles are blue if the concentration is significantly below the BAC; green if significantly below the EC limit value; and red if significantly above the EC limit value. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

Estuary, Dee Estuary, Liverpool Bay, Mersey Estuary, Morecambe Bay, and the Clyde Estuary. The highest concentrations were in mussels from the Tyne, at 3380 µg/kg ww. It should be noted that CSEMP sampling is targeted at wild mussel populations, and does not concentrate on those which are commercially exploited.

In summary, Pb concentrations in mussels are elevated in some industrial estuaries, presumably as a result of localised sediment contamination. Potential risks to human health exist in several areas if mussels were consumed. Steps should be taken to protect the public including the erection of warning signs and the continued monitoring and control of discharges. Monitoring of mussels should be maintained in areas where elevated concentrations have been found.

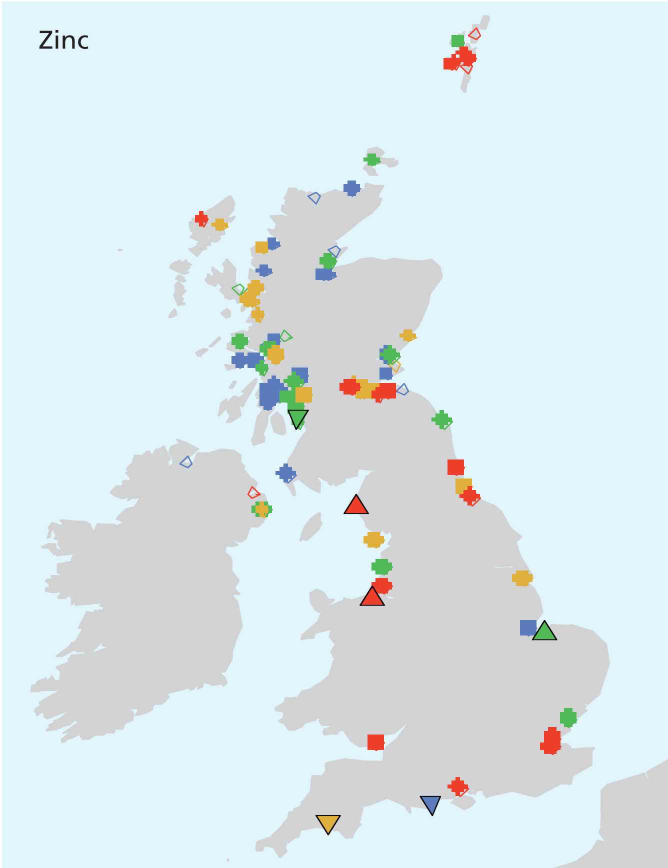
Other metals in mussels

Data are also reported for Ag, As, Cr, Cu, Ni, selenium and Zn. There are no assessment criteria available for these metals in mussels, so the data are presented in quartiles. The values used for each of these thresholds are shown in Table 3.24.

Metals data are available for a wider geographical area than was the case for *Charting Progress*, and for more metals, and this provides more information on the effects of point source inputs, especially for silver (Ag; see following section). Any conclusions drawn need to take into account that mussels have 'good' indicator potential for Cr, 'moderate' potential for Ni but are 'unreliable' for Cu and Zn, except at high concentrations (Miller, 1986).

In general terms, the data show that most of the sites with concentrations in the lower quartiles are found in the north and west of Scotland, whereas most of the sites with elevated concentrations are found in industrialized estuaries throughout the UK, as has been reported previously (Miller, 1986). See Figure 3.30 for chromium, copper, nickel, selenium and zinc in mussels.

Figure 3.30 Chromium, copper, nickel, selenium and



Blue circles represent the lowest 25% of concentration data, green circles the next 25% of concentration data, amber circles the next 25% of concentration data, and red circles the highest 25% of concentration data. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

Silver in mussels

No data for Ag in mussels were reported in *Charting Progress* (data for 1999–2001),. However, Ag was included in the EC Shellfish Growing Waters Directive as a Mandatory substance, and so data are now available from various monitoring programmes.

Historically, silver found in the aquatic environment mainly originated from sewage treatment plants, due to its use in photographic

zinc in mussels. © Marine Scotland Science.

Table 3.24 Threshold values for quartiles for metals in mussels, as μg/kg ww.

Metal	*0% (Min)*	*25%*	*50%*	*75%*	*100% (Max)*
Ag	1	6	15.0	28.4	473
As	1028	1927	2266	2882	5417
Cr	106	267	333	441	2200
Cu	544	828	1029	1349	11 408
Ni	157	234	339	438	1369
Se	179	408	538	753	1851
Zn	7408	12 744	15 774	21 385	262 144

Note: some of the maximum concentrations are extremely high in comparison to the rest of the dataset, and these values should be treated with caution.

chemicals (Rozan and Hunter, 2001). In recent years, however, increasing use has been made of silver in nanoparticles added to clothes during their manufacture, due to the antimicrobial properties of nanosilver (Luoma, 2008). Although silver itself is classed as an environmental hazard because it is toxic, persistent and liable to bioaccumulate, there are no examples of adverse effects from nanosilver technologies in the environment, at present. The environmental risk from silver itself may be mitigated by a tendency of the silver ion to form strong complexes that are apparently of very low bioavailability and toxicity. However, it is also possible that nanoparticles could shield nanosilver from forming complexes, so that free silver ions may be able to enter the membranes or cells of organisms through a 'Trojan horse' mechanism (Luoma, 2008).

The environmental fate of nanosilver will depend on the nature of the nanoparticle. Nanoparticles that aggregate and/or associate with dissolved or particulate materials are likely to end up deposited in sediments or soils, being re-suspended in particulates and available for uptake by organisms.

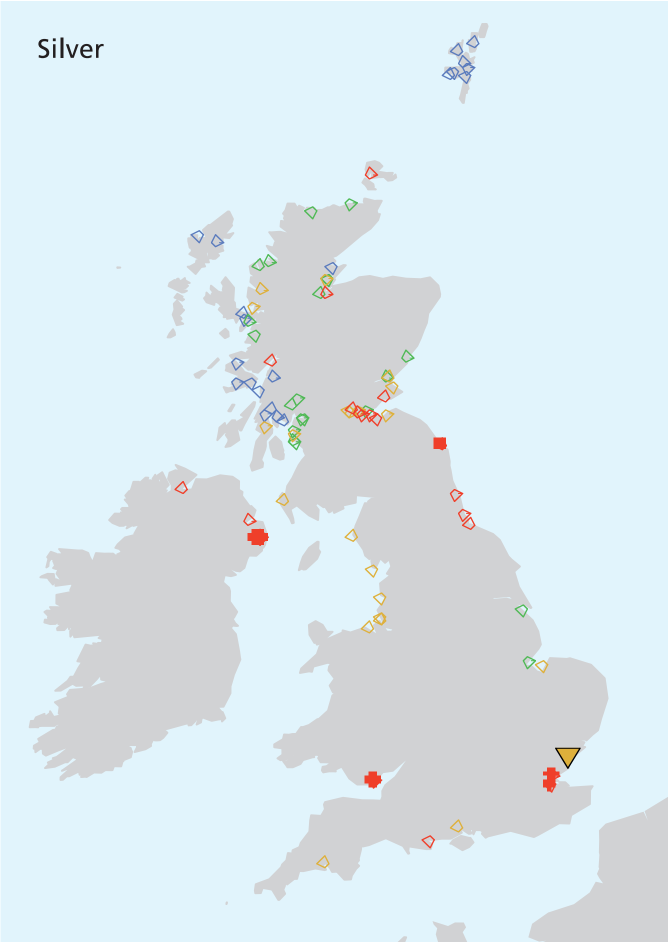
It has also been reported that while the element is found in very low concentrations in the dissolved phase in estuarine and coastal waters, formation of a stable chloro-complex enhances its bioavailability and toxicity to biota (Luoma et al., 1995; Bryan and Langston, 1992).

While there is no doubt that more work is required on the toxicity, behaviour and fate of silver and nanosilver in the terrestrial and marine environments, elevated concentrations have been found in mussels collected close to waste water treatment works discharges (see Figure 3.31), confirming that these are current sources of the metal and that it is available for uptake and bioaccumulation.

The data for Ag collected as part of the UK CSEMP programme and for the EC Shellfish Growing Waters Directive shows this to be the case in that while concentrations in mussels are generally very low, at or below 10 μg/kg ww, concentrations are elevated at sites where mussels have been exposed to effluents from sewage treatment works.

In industrialized estuaries, Ag concentrations were elevated to 30 to 57 μg/kg ww in the Forth, 38 μg/kg ww in the Tyne, 50 μg/kg ww in The Wash, 56 μg/kg ww in the Medway, 100 μg/kg ww in Poole Harbour, 338 μg/kg ww in the Severn and 473 μg/kg ww in the Thames. Interestingly, concentrations of 30 and 37 μg/kg ww were measured in mussels from the Moray Firth from near Inverness and from Loch Linnhe near Fort William, in north-west Scotland respectively, and 107 to 121 μg/kg ww in Strangford Loch, Northern Ireland.

Figure 3.31 Silver in mussels. © Marine Scotland Science.



Blue circles represent the lowest 25% of concentration data, green circles the next 25% of concentration data, amber circles the next 25% of concentration data, and red circles the highest 25% of concentration data. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

The new evidence tells us that mussels are exposed to a complex mixture of contaminants from a variety of sources, highlighting the need for the continued development and application of biomarkers or suites of biomarkers to assess the sub-lethal impacts on local mussel populations (see Hagger et al., 2008). It may be that increasing use of digital photography may have caused a decline in discharges of silver, but the increasing use of nanosilver as an antibacterial agent in clothes has resulted in elevated concentrations of silver in mussels collected close to waste water treatment discharges.

Silver concentrations should be routinely measured in sewage and other effluents in order to establish whether controls need to be set in place on specific discharges, to protect marine organisms from silver contamination.

There is a need for the environmental risks from silver and nanosilver to be thoroughly assessed, in view of their increasing use in clothes and other applications. There is a need for the development of assessment criteria so that the implications of the elevated Ag concentrations can be assessed, and for the development and application of biomarkers, in conjunction with traditional contaminant monitoring.

Arsenic in mussels

No data for As in mussels were reported in *Charting Progress* (data for 1999–2001). However, As was included in the EC Shellfish Growing Waters Directive as a Mandatory substance, and so data are now available from various monitoring programmes.

It is important when considering the concentrations of As in marine biota to take into account the form or species of As present, and to know whether the analytical method has allowed every As species to be quantified. This is because while As may be taken up by biota in the inorganic form, it is quickly converted to organo-arsenic compounds which are relatively non-toxic, in contrast to other organo-metallic compounds such as methyl-Hg or tributyltin (TBT), which are more toxic than their inorganic forms. Data are reported here for total As; the proportion of As present in the organic forms may vary and be as high as 85% or more. It has been suggested that As concentrations may show natural variability due to changes in water salinity, due to arsenobetaine having a role as an acquired osmolyte in mussels.

In terms of its sources, As is present at low concentrations in food, and can enter the marine environment as a result of the weathering of rocks, so local mineralisation has to be considered. In contrast to silver, As concentrations are difficult to attribute to one source, and are due to a combination of anthropogenic point source inputs and natural diffuse sources. It is difficult to draw firm conclusions about the reasons for the distribution in observed As concentrations in mussels from around the UK. As concentrations in mussels were in the range 1028 to 5417 mg/kg ww, but much of this may be present as organo-arsenic species, which are non-toxic. On this basis, the evidence suggests that As concentrations in mussels present neither environmental harm nor human health risks. Data and trends are illustrated in Figure 3.32.

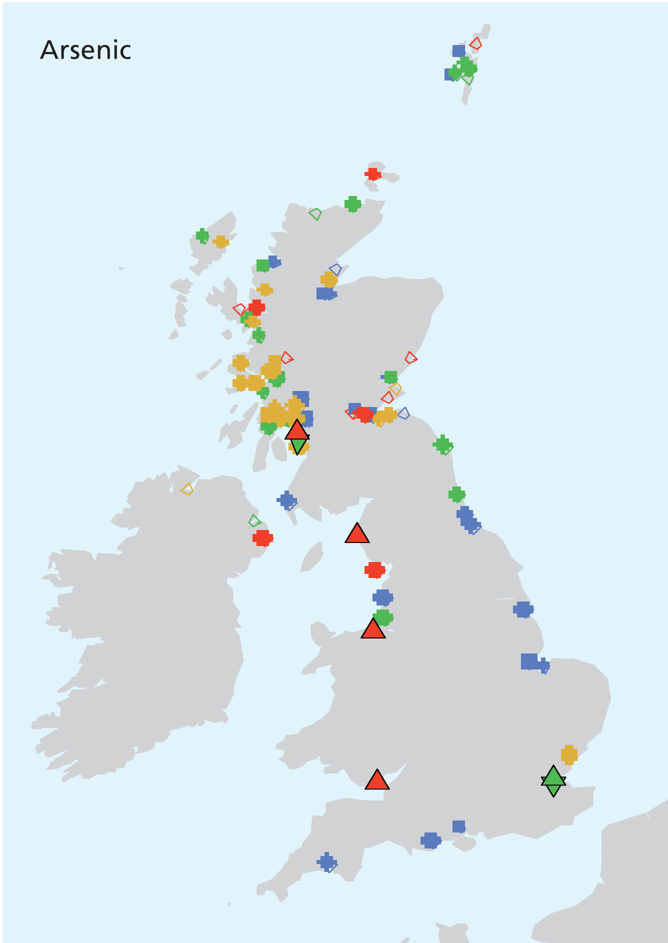
It would be precautionary to carry out a comprehensive survey of the total As concentrations in mussels and the speciation of the organic complexes present, to verify the above statements. Techniques are available for such determinations using anion- and cation-exchange chromatography to separate the organic and inorganic As species in a nitric acid digest followed by ICP-MS, with UV photo-oxidation as an alternative method for sample decomposition (Goessler and Pavkov, 2003).

3.1.3.7.4 Need for further work

Metals concentrations in fish were generally low and not of concern, either in terms of causing harm to the fish or risks to human health. However, standards were not available for Cd and Pb in fish liver, so these could be developed to provide greater confidence in this assessment. Consideration should be given to reducing the monitoring of metals in fish.

For metals in mussels, although it was possible to assess the relative concentrations of each metal in each area, it was not possible to make an objective assessment of the degree of risk to

Figure 3.32 Arsenic in mussels. © Marine Scotland Science.



Blue circles represent the lowest 25% of concentration data, green circles the next 25% of concentration data, amber circles the next 25% of concentration data, and red circles the highest 25% of concentration data. Open circles denote 1–2 years of data and small closed circles indicate 3–4 years of data. Statistically significant upward or downward trends are derived from 5 or more years of data and indicated by triangles.

human health presented by the consumption of mussels collected from intertidal shores, due to the unavailability of suitable standards for most of the metals in mussels. These need to be developed, in order for this assessment to be made.

3.1.3.7.5 Overall conclusion and forward look

Metals concentrations in fish were similar to those previously reported, generally low and not of concern, either in terms of causing harm to the fish or risks to human health. Consideration should be given to reducing the monitoring of metals in fish.

Metals concentrations in fish were elevated in industrialized estuaries and low in remote areas, as previously reported. Consideration should be given to the development of suitable standards to allow the assessment of the risks to human health presented by the consumption of mussels collected from intertidal shores.

3.1.3.8 Contaminants in marine mammals

3.1.3.8.1 Summary

As top predators in marine food chains, marine mammals accumulate high concentrations of many contaminants. Since *Charting Progress*, additional data have become available for 'emerging' contaminants, specifically some brominated flame retardants and perfluorinated compounds. The significance of these compounds cannot currently be assessed due to a lack of comprehensive toxicological information and assessment criteria.

Contribution to the GES descriptor

Within the EU Marine Strategy Framework Directive there is a requirement to assess Good Environmental Status (GES) against a series of descriptors. The data reported in this section illustrate contamination levels in biota species of high conservation importance. This addresses Descriptor 8 *...concentrations of contaminants are at levels not giving rise to pollution effects.*

Future risks

There are likely to be shifts in the geographical range of some marine mammal species as a result of climate change, and of their prey species. Likely impacts cannot yet be foreseen. A wider range of brominated flame retardants than those studied to date are currently in use and have been found in environmental samples. Their levels and possible impacts in UK marine mammals are unknown.

Cross-cutting issues

Contaminants may have effects at the individual or population level. If the latter, they would add to other recognised pressures such as marine mammal by-catch in commercial fisheries.

3.1.3.8.2 Introduction

The Cetacean Strandings Investigation Programme (CSIP) is funded by Defra as a part of its commitments under ASCOBANS (the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas). More than 20 species of marine mammal have been studied within the CSIP, but most attention has been focussed on the harbour porpoise (*Phocoena phocoena*). This is because it is the most widely distributed of the small cetaceans, and occurs in waters all around the UK. Marine mammals stranded in England, Wales and Scotland or retrieved directly following entanglement in commercial fishing gear (by-catch) are taken for autopsy using standardised methodology (Law et al., 2006a). Contaminant levels in selected animals have then been determined, using methodologies

which have delivered consistent data since CSIP's inception in 1989 and with appropriate analytical quality control. Contaminants determined have included both historic (e.g. organochlorine pesticides, polychlorinated biphenyls (PCBs) and heavy metals) and 'emerging' compounds (e.g. brominated flame retardants and perfluorooctane sulphonate).

Environmental impacts

Impacts of contaminants on marine mammals include effects on hormone and immune systems, which can render individuals more susceptible to infectious disease and consequent mortality.

Socio-economic impacts

Disappearance of marine mammal species as a result of pollution impacts or climate change would have an impact on whale-watching activities underway in various parts of the UK. This is not a high value industry at present.

3.1.3.8.3 Developments since *Charting Progress*

One of the main aims of the CSIP has been to investigate potential links between contaminants levels in porpoises and death due to infectious disease. This arose as a topic for study following the 1988 morbillivirus epidemic in the North Sea, which killed 18 000 common seals. At the time, it was suggested that elevated burdens of contaminants such as PCBs in the animals had suppressed their immune response and made them more vulnerable to infection (Hall et al., 1992). For PCBs, higher concentrations have been reported for porpoises dying from infectious disease than from physical trauma (including by-catch) (Jepson et al., 2005), in many cases above a threshold for adverse health effects in marine mammals derived from experimental data of 17 mg/kg lw of total PCB. In a subsequent study (Hall et al., 2006) the increased risk was quantified. For each 1 mg/kg lw increase in the concentration of PCBs in porpoise blubber, there is a 2% increase in the risk of mortality due to infectious disease in the porpoise population in UK waters. Elevated PCB levels have also been shown to be one of the determining factors in the degree of nematode worm infestation in porpoise tissues (Bull et al., 2006). In grey seal pups sampled between 1998 and 2000, summed PCB concentrations ranged from 0.1 to 93 mg/kg lw (Kalantzi et al., 2005). For harbour seals, the highest summed CB concentrations were found in the Islay/Jura area (SW Scotland; 8.2 mg/kg lw) and lowest in Orkney (1.3 mg/kg lw). Intermediate levels were found in animals from the Tay Estuary and The Wash. Despite controls on the production and use of PCBs in the UK over 20 years ago, they are still present at concentrations which result in chronic impacts in marine mammals, and the concentrations in the blubber of porpoises are declining only slowly (Law et al., 2010). Other marine mammals found in UK waters, such as bottlenose dolphins and killer whales (Law et al., 2006a; McHugh et al., 2007), can have higher PCB burdens as a result of their greater size, longevity and diet, but possible impacts on these animals cannot be assessed at present due to a lack of data.

In terms of 'emerging' contaminants, concentrations of two persistent fluorinated organic compounds, PFOS (perfluorooctane sulphonate) and PFOA (perfluorooctanoic acid) have been determined in liver of porpoises from UK coasts. PFOA was not detected in any of the 58 livers analysed, but PFOS was detected at concentrations up to 2420 µg/kg ww, a similar range to that seen in other European countries (Law et al., 2008a). These data form a baseline against which OSPAR can, in the future, assess the effectiveness of limitations on the

use of PFOS and the success of its Hazardous Substances Strategy for this compound. For the brominated diphenyl ethers (BDEs), a family of compounds arising from commercial BFR products, the highest concentrations observed in a recent study were in the range 10 to 20 mg/kg ww sum BDEs in a bottlenose dolphin and a killer whale (Law et al., 2005). Low but detectable concentrations were found in a range of other species, including baleen whales which feed at a low level in the marine food chain and a beaked whale which feeds offshore (remote from land sources) and in deep water. BDE183, considered a marker for the octa-mix technical product, was not detected in any samples, suggesting that contamination arises from earlier use of the penta-mix product and is widespread. Summed BDE concentrations in grey seals from the North Sea were in the range 45 to 1500 µg/kg lw (Kalantzi et al., 2005), and in harbour seals were up to 630 µg/kg lw with the highest concentration being seen in The Wash (Eastern England) (Hall and Thomas, 2007). In both grey and harbour seals from the UK, BDEs were shown to interfere with thyroid homeostasis (Hall et al., 2003; Hall and Thomas, 2007). All three polybrominated diphenyl ether products are now banned within the EU. Two further BFRs have also been studied in porpoises, HBCD (hexabromocyclododecane) and TBBP-A (tetrabromobisphenol-A) (Law et al., 2006b, 2008b). TBBP-A was detected at low concentrations in some of the porpoises in the earlier study. However, following the development and application of a confirmatory analytical technique, similar concentrations initially found in the later study were not confirmed, and it is possible that the earlier data were overestimates. TBBP-A is a reactive flame retardant used in printed circuit boards, which means that it is less likely to leach out into the environment than the other, additive BFRs, as it is firmly bound into the products. In addition, its major area of application in electronics will be in Asia, reaching Europe primarily in finished products. For HBCD, studies of a possible time trend in HBCD concentrations in porpoise blubber over the period 1994 to 2003 indicated a sharp increase from about 2001 onwards. HBCD is a BFR which is used primarily in expanded and extruded polystyrene for thermal insulation in buildings, with a secondary application in textiles manufacture. Subsequently, another study extended this timeline to 2006, with data for 223 porpoises in all. This study indicated that concentrations had begun to decline between 2003 and 2004. Neither trend was confounded by other factors, such as sex, age or location. For BDEs, a decline in porpoise blubber concentrations is now evident as a result of controls on production and use within the EU (Law et al., 2010b).

3.1.3.8.4 Presentation of the evidence

Monitoring period

Contaminants in marine mammals have been determined during the period 1989-2007.

Data sources

The data used were derived from post-mortem studies and tissue contaminant analyses undertaken within the CSIP.

Adequacy and confidence of the data

The data are collected using established protocols and with appropriate quality assurance in place. We can therefore be confident in the data obtained.

Trends

No data have been gathered on the classical organochlorine pesticides in the past few years due to a reduction in CSIP funding and this prevents trends being determined for these compounds. Concentrations of HBCD in porpoise blubber have begun to decline following a steep rise, as a result of factory closure and industry initiatives to reduce discharges. Concentrations of BDEs are also falling. In contrast, CB concentrations are declining only very slowly despite controls having been in place for 30 years.

3.1.3.8.5 Progress towards the vision of clean and safe seas

Despite controls applied during the 1970s, concentrations of PCBs in marine mammal blubber seem to be declining only slowly and are at levels likely to injure the health of individuals. Marine mammals are also exposed to additional threats from a wide range of industrial chemicals via their food chains.

3.1.3.8.6 Need for further work

Work is needed to determine concentrations of the classical organochlorine compounds in order to fulfil the OSPAR aim that they be demonstrably 'close to zero' at some point in the future. Also, we need to conduct smaller-scale surveys for specific persistent, bioaccumulative and toxic compounds (candidate POPs [persistent organic pollutants] in the future) in order to assess the risks they pose.

3.1.3.8.7 Potential impact of climate change

The major impact of climate change is likely to be changes in the geographic distributions of different marine mammal and prey species. For instance, in UK waters, white-beaked dolphins are currently mainly distributed towards the northern part of the North Sea and off NW Scotland (Figure 3.33; Reid et al., 2003), and may move further northward as waters warm and the distribution of their prey (mainly clupeoid and gadoid fish – herring and cod species – also squid and crustaceans) alters. Changes in the relative importance of different contaminant pathways may also alter contamination levels in particular species, depending on their individual habitats and feeding preferences.

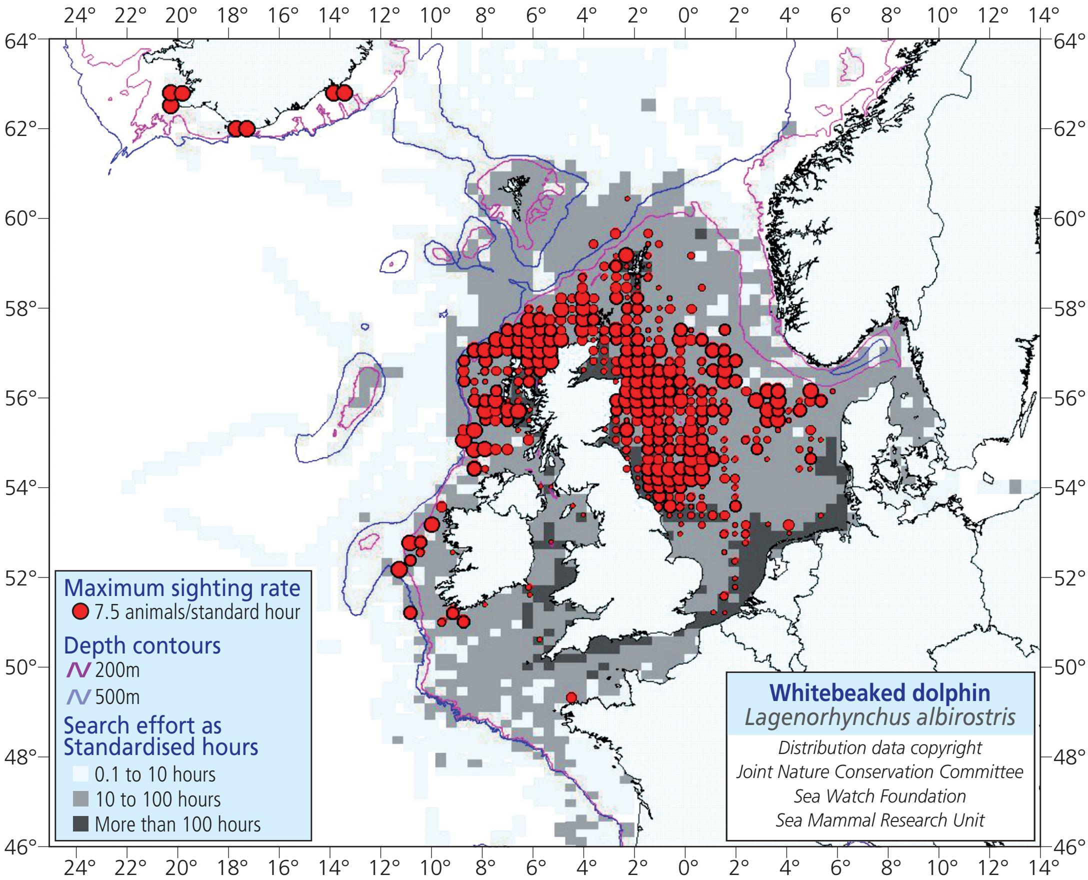
3.1.4 Contaminant-specific biological effects

3.1.4.1 Liver pathology

3.1.4.1.1 Key points

- Clinical fish disease and liver pathology can be utilised as high-level indicators of marine ecosystem health. Disease is directly and indirectly linked to biomarkers of contamination and to contaminant burdens
- Internationally agreed quality assurance protocols are in place for the measurement of fish disease
- For thus report, external diseases and liver pathology (including cancer) data were assessed over the period 2002 to 2007
- The prevalence range of each disease was assigned to a score representing 'background', 'good', 'adverse effects possible' or 'adverse effects probable'
- Cumulative scores represent an overall assessment of 'harm' to fish populations from particular sites
- Grading of harm scores into 'Site Types' (A, B, C) provides a broad overview of the relative health status of fish populations sampled from sites and CP2 Regions

*Figure 3.33 Sightings of white-beaked dolphins (*Lagenorhyncus albirostris*) in NW European waters. © JNCC.*



- Certain sites in the Irish Sea and North Sea have elevated and potentially increasing levels of liver pathology (including cancer). Other sites are elevated and stable, or low and stable (e.g. the English Channel).

3.1.4.1.2 Introduction

General

Several studies on marine epizootics in the past decade have led to suggestions that the number and extent of disease outbreaks in marine organisms is increasing (Lafferty et al., 2004). Also, fish diseases of infectious (pathogens) and non-infectious (e.g. cancer) types provide one of the clearest high-level indicators of the general health status of a given population. In this context, fish diseases have been recorded in national marine and estuarine monitoring programmes for many years.

Within the North-East Atlantic region, guidelines for biological effects of contaminants monitoring include the measurement of externally visible fish diseases for general (non-specific) monitoring and of liver nodules (tumours) and liver histopathology for PAH-specific effects. The disease status of the flatfish species dab (*Limanda limanda*) at offshore sites and flounder (*Platichthys flesus*) at inshore and estuarine sites is monitored for the UK CSEMP (OSPAR JAMP, 1998a,b). Data have been used to detect long-term trends in disease prevalence at given locations and in combination with other biomarkers of exposure, are used to provide greater confidence in the use of fish diseases as indicators of contaminant effects (Lang and Dethlefsen, 1996; Wosniok et al., 2000).

Liver cancer

Specifically, the presence of liver tumours has been classified as a direct indicator of historic contaminant exposure, particularly to chemicals that initiate and promote carcinogensis (Myers et al., 1990, 1991, 1992, 1994; Schiewe et al. 1991; Reichert et al., 1998). At some offshore sites in the North Sea, liver tumour prevalence in flatfish has exceeded 10% in recent years (Feist et al., 2004) while prevalence in estuarine species can be even higher (Stentiford et al., 2003; Koehler, 2004). These figures are significantly elevated over those observed in other wildlife populations (e.g. Fowler, 1987; Harshbarger, 2004). In addition to the assessment of grossly visible tumours, histopathological assessment of liver samples allows for the diagnosis of microscopic lesions not visible during whole fish assessments. The lesions recorded using this approach include those thought to precede the development of benign and malignant tumours and give an early warning of detrimental health effects. The diagnosis of these lesion types in dab and flounder liver follows the guidelines set out by Feist et al. (2004).

Availability of data

We have analyzed major datasets pertaining to external disease and liver pathology in fish collected from UK CSEMP in English, Welsh and Scottish waters visited annually over the period 2002 to 2006. Prevalence data range for each disease variable has been used to create a simple grading system for 'site types' that broadly defines the level of harm due to disease inflicted on fish populations therein.

3.1.4.1.3 Developments since *Charting Progress*

Charting Progress stated that the prevalence of disease in dab at some Irish Sea locations was probably increasing and that levels of most fish disease in the North Sea were generally static. It also reported on evidence for feminisation in some estuarine flounder populations. Data reported for *Charting Progress 2* is significantly more comprehensive and has benefited from quality assurance protocols that are now embedded into the UK monitoring programme for fish disease.

Overall, the prevalence of liver pathologies (including cancer) appears to be elevated and potentially increasing at certain Irish Sea sites (e.g. Cardigan Bay, Lyons et al., 2006), elevated but static at some North Sea sites (e.g. Dogger Bank) and low and static (approaching or at background levels) at Inner North Sea and English Channel sites. External fish diseases appear to have decreased to low levels at several Irish Sea sites (e.g. Liverpool Bay) though remain relatively elevated and static at sites on the Dogger Bank, North Sea. The prevalence of external diseases in the English Channel and in Scottish populations remains low and static. During the reporting period, the first incidences of intersex (feminization) were reported in dab from the Dogger Bank (Stentiford and Feist, 2005).

3.1.4.1.4 Presentation of the evidence

Representative images of the external diseases measured are shown in Figure 3.34. Representative images of normal liver and the five lesion category groups (non-neoplastic toxicopathic, inflammatory, foci of cellular alteration, benign neoplasm and malignant neoplasm) are shown in Figure 3.35. Liver histopathology data were generated by

*Figure 3.34 Representative images of external diseases of dab (*Limanda limanda*) according to ICES criteria (Bucke et al., 1996). (A) No visible external diseases. (B) Epidermal papilloma on skin (arrow). (C) Skin hyperpigmentation. Multi-focal hyperpigmented regions on skin (arrows). (D) Skin ulceration (arrow). (E) Lymphocystis. Cluster of infected epidermal cells (arrow). (F) Single large liver nodule (arrow) and adjacent apparently normal liver (asterisk). © Crown copyright 2010: permission granted by Cefas.*

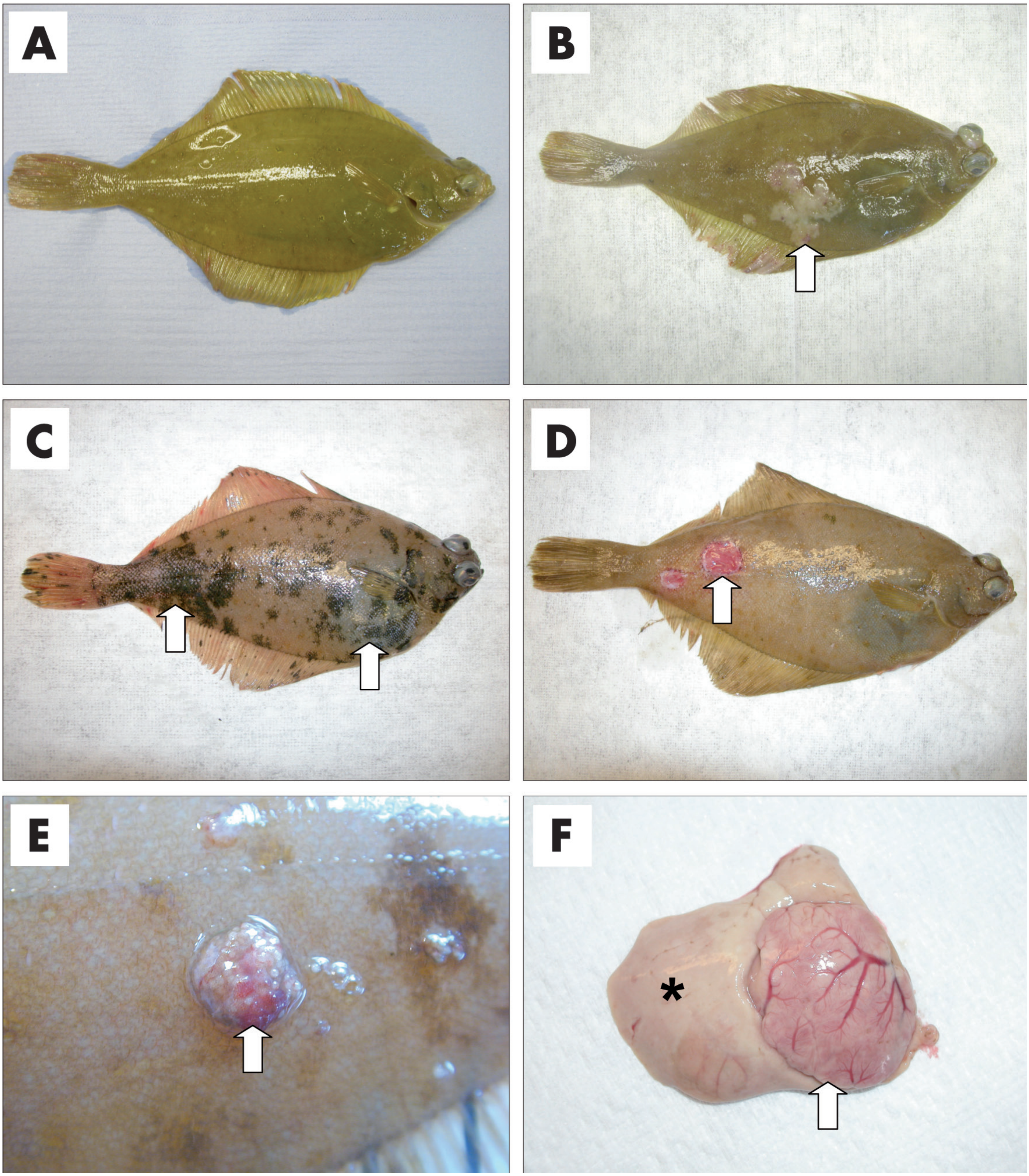
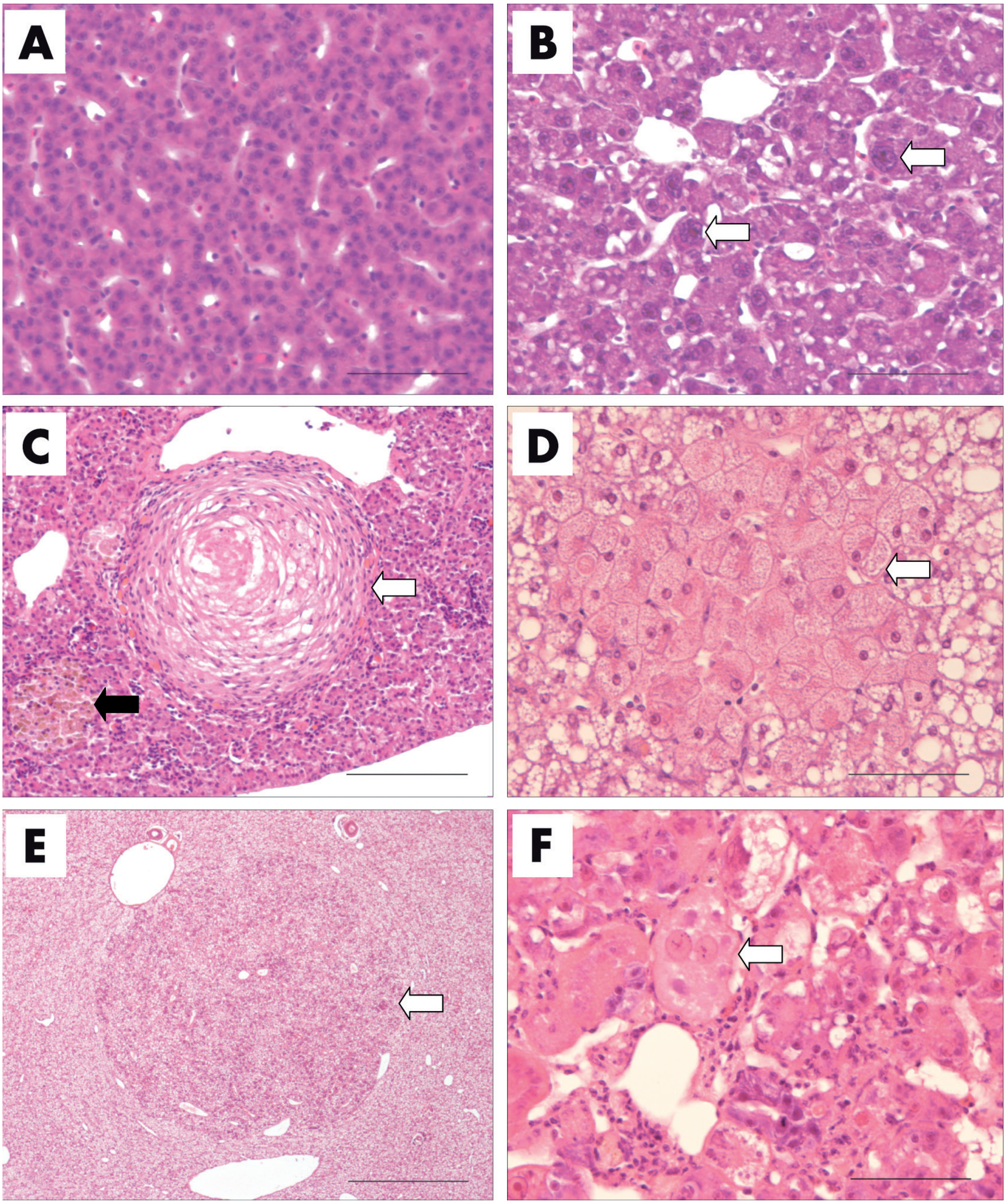


Figure 3.35 Representative liver pathologies and categories according to Feist et al. (2004) (pathology category in italics and parentheses). (A) Normal liver (No Abnormality detected); Bar = 100 μm. (B) Nuclear pleomorphism (white arrows) (Non-specific toxicopathic lesions); Bar = 100 μm. (C) Granuloma (white arrow) and melanomacrophage centres (black arrow) (Inflammatory lesions); Bar = 200 μm. (D) Clear cell focus of cellular alteration (white arrow) (Foci of cellular alteration); Bar = 50 μm. (E) Hepatocellular adenoma (white arrow) (Benign neoplasms); Bar = 200 μm. (F) Hepatocellular carcinoma with atypical cellular and nuclear profiles (white arrow) (Malignant neoplasms) Bar = 50 μm. All Haematoxylin and Eosin staining. © Crown copyright 2010: permission granted by Cefas.



allocating the 32 individual liver pathologies measured to one of the five liver pathology categories (Table 3.25). Each external disease and liver pathology category for each site/year was assigned a score (0 to 3) relating to: 'background', 'good', 'adverse effects possible' or 'adverse effects probable', respectively (Table 3.26). Cumulative scores for each site/year formed an overall 'harm score' (this score further divided into two to represent harm associated with external diseases and liver pathologies). An arbitrary division of these two harm scores into three categories defined a broad classification system of 'Site Types' A, B and C according to the external and liver disease profiles of flatfish found at those sites (Table 3.27). Site types and harm scores for each CSEMP station in each year are shown in Table 3.28. The predominant site type within a given region was used to generate an overall assessment for that region (Table 3.28).

3.1.4.1.5 Progress towards the vision of clean and safe seas

The disease profile for several sites remains similar to that reported in *Charting Progress*. However, elevation and apparent increase of liver cancer prevalence at some Irish Sea sites (e.g. Cardigan Bay) and ongoing elevation of liver cancer prevalence at sites on the Dogger Bank is noteworthy. Lack of correlation between liver cancer prevalence and certain biomarkers at the Cardigan Bay sites has been reported (Lyons et al. 2006).

Higher liver cancer prevalence at Cardigan Bay and Dogger Bank sites cross-correlates to an elevated and rising prevalence of skin hyperpigmentation in these regions though the relationship (if any) between these two conditions is not understood.

Liver cancer prevalence in fish from certain UK marine sites is significantly higher than the prevalence of this disease in terrestrial wildlife populations, and in humans. Co-factors such as age and size explain some of this elevated prevalence but not all since cancer onset appears to occur significantly earlier in North Sea than Irish Sea sites.

External diseases are largely stable and at low prevalence in dab from several UK sites. Lowest prevalence occurs in Scottish and English Channel populations.

3.1.4.1.6 Need for further work

There is a need for further work in several areas.

- To continue to consider fish disease as one of the highest-level indicators of health in marine ecosystems
- To separate the measures of external disease (indicating broader range impacts and population level health status) and liver pathology (anthropogenic contaminant aetiology) in data analysis to allow for concentration of effort on understanding the link between liver cancer and anthropogenic inputs
- To align relevant R&D programmes on cancer induction in sentinel fish with disease monitoring programmes
- To consider the UK marine flatfish cancer model as a valuable tool to study cancer induction in wildlife (and human) populations
- To undertake integrated analysis of disease, biomarker and chemistry data.

Table 3.25 Flatfish liver pathology categories and specific pathologies as listed by BEQUALM (see Feist et al. 2004). Another category, 'No abnormality detected' (NAD) is assigned to individual specimens that do not exhibit any of the pathologies listed in this table. Not all specific pathologies were observed during the 2002-2006 sampling period.

Non-specific and inflammatory	*Non-neoplastic toxicopathic*	*Foci of cellular alteration*	*Benign neoplasms*	*Malignant neoplasms*
Coagulative necrosis	Phospholipoidosis	Clear cell	Hepatocellular adenoma	Hepatocellular carcinoma
Apoptosis	Fibrillar inclusions	Vacuolated	Cholangioma	Cholangiocarcinoma
Steatosis	Polymorphism	Eosinophilic	Hemangioma	Pancreatic carcinoma
Hemosiderosis	Hydropic degeneration	Basophilic	Pancreatic adenoma	Hepatobiliary carcinoma
Variable glycogen	Spongiosis hepatis	Mixed		Hemangiosarcoma
Melanomacrophages				Hemangiopericytic sarcoma
Inflammation				
Granuloma				
Fibrosis				
Regeneration				

Table 3.26. Prevalence ranges for 10 disease variables measured over the five-year study period. The prevalence range of each disease over the period has been divided into quartiles and a score assigned to each quartile (0 to 4). A zero score is taken to represent the five-year baseline prevalence for each disease. Higher scores depict higher prevalences. Scores assigned to the prevalence of each disease at each site in each year allow an overall 'harm score' to be assigned.

	Low	*High*	*Range*	*Baseline*	*Low-Mid*	*High-Mid*	*High*
TOXICO	0	26	26	0 to 6.5	6.6 to 13	13.1 to 19.5	above 19.5
FCA	2	58	56	0 to 16	16.1 to 30	30.1 to 44	above 44
BENIGN	0	24	24	0 to 6	6.1 to 12	12.1 to 18	above 18
MALIGNANT	0	8	8	0 to 2	2.1 to 4	4.1 to 6	above 6
INFLAMM	42	100	58	0 to 56.5	56.6 to 71	71.1 to 85.5	above 85.5
LY	0	14	14	0 to 3.5	3.6 to 7	7.1 to 10.5	above 10.5
U	0	38	38	0 to 9.5	9.6 to 19	19.1 to 28.5	above 28.5
EP	0	14	14	0 to 3.5	3.6 to 7	7.1 to 10.5	above 10.5
HYP	0	52	52	0 to 13	13.1 to 26	26.1 to 39	above 39
LN	0	18	18	0 to 4.5	4.6 to 9	9.1 to 13.5	above 13.5
			Score	**0**	**1**	**2**	**3**

*Table 3.27 Simple marine site classification scheme based upon disease profiles and derived harm scores in populations of dab (*Limanda limanda*) captured at those sites. Free text provides a broad outline of the likely disease profile within each site type.*

Type A	*Type B*	*Type C*
Generally low prevalence (<10%) of ICES external diseases and almost complete absence of skin hyperpigmentation	Appearance of higher prevalence of ICES external diseases (incl. Lymphocystis, ulceration and hyperpigmentation)	Highest levels of ICES external diseases (including up to 50% prevalence of skin hyperpigmentation)
Up to 50% of fish with no indication of BEQUALM liver pathology categories	Less than 30% of fish with no indication of BEQUALM liver pathology categories	Less than 20% of fish with no indication of BEQUALM liver pathology categories
Low prevalence (<5%) of fish with toxicopathic liver lesions and approximately 50% prevalence of inflammatory liver lesions	Low prevalence (<10%) of fish with toxicopathic liver lesions but an elevated prevalence of inflammatory liver lesions (up to 90%)	Similar prevalence of toxicopathic liver lesions to Type B sites but a consistently high prevalence (up to 100%) of inflammatory liver lesions
Low prevalence of fish with liver FCA (<10%) and benign liver tumours (<5%). Malignant liver tumours very rare or absent.	Prevalence of FCA can exceed 15%. Benign liver tumour prevalence around 10%. Malignant liver tumours more common than in Type A (up to 6%)	High prevalence (up to 50%) of FCA with benign tumour prevalence often exceeding 15%. Malignant liver tumours still comparatively rare though generally comprise a larger proportion of observed liver tumours than Type B (up to 8%).
Liver pathology score <5 and/or disease score < 5	**Liver pathology score >5<10 and/or External disease score >5<10**	**Liver pathology score >10 and/or External disease score >10**

3.1.4.1.7 Overall conclusions and forward look

The current study has reported on the application of quality assured monitoring principles to the standardised recording of external diseases and liver pathology in marine fish from UK waters. Recent studies have demonstrated how disease profile patterns in this sentinel species are temporally and spatially stable, at least during the period under investigation (Stentiford et al., 2009) – the first step in assessing the applicability of disease as a reliable marker of population health and suggestion that it has a defined underlying basis. In the context of marine monitoring, such investigations should include the measurement of inherent biological features in the populations of concern (e.g. age, diet, migration, population genetics), known or unknown abiotic factors (e.g. temperature, salinity) or anthropogenic factors (e.g. chemical pollution, fishing pressure).

While intuitive that the higher prevalence of liver neoplasia in dab from certain UK CSEMP sites identified in the current study may be associated with elevated levels of carcinogenic contaminants (such as PAHs) in sediment and tissues at those sites, or even elevated biomarker responses (such as DNA adducts – see Section 3.1.3.5), demonstration of these relationships has proved more elusive, with less clear links between causal contaminants and disease at offshore locations (Lyons et al., 2006; Cefas, 2006, 2007; and see Section 3.1.3.5).

Current work in our laboratory is attempting to interpret this apparent paradox by investigation of potential confounding factors that may alter the expression of biomarkers and the prevalence of disease. Recent work on the

Table 3.28 Allocation of site/year combinations into site types (A, B and C) according to liver pathology data (Table A) and external disease data (Table B). Specific harm scores for each data set (range 0 to 15) for each site in each year are also shown. Overall assessment of the region is defined by the predominant site type within the given region in the final year of assessment (2006 for liver pathology data and 2007 for external disease data).

Table A

Site	*Region*	*2002*	*2003*	*2004*	*2005*	*2006*	*Region*
Amble	Tyne/Tees			3	3	4	
Flamborough	Tyne/Tees	5	4	3	5	5	A
Tees Bay	Tyne/Tees		4	0	3	3	
C Dogger	Humber/Wash		4	5	8	8	
Indefatigable Bank	Humber/Wash		2	2	5	5	
N Dogger	Humber/Wash	8	8	6	7	10	
NE Dogger	Humber/Wash			9	12	9	B
Off Humber	Humber/Wash	4	3	3	6	5	
W Dogger	Humber/Wash	7	6	6	7	10	
Wash	Humber/Wash	0					
Rye Bay	Eastern Channel	1	0	1	1	1	A
S Eddystone	Western Channel				1	3	A
Carmarthan Bay	Severn		2	2	1	3	A
Inner Cardigan Bay	Cardigan Bay		5	10	7	8	
N Cardigan Bay	Cardigan Bay	6	2	7			B
S Cardigan	Cardigan Bay		3			10	
Burbo Bight	Irish Sea	4	2	3	2	6	
Liverpool Bay	Irish Sea	4	5	7	4	7	
Morecambe Bay	Irish Sea	1	1	1	4	5	
Red Wharf Bay	Irish Sea	2	0	1	1	5	B
SE Isle of Man	Irish Sea		0	2	7	7	
St Bees	Irish Sea	5		4	5	10	

Table B								
Site	*Region*	*2002*	*2003*	*2004*	*2005*	*2006*	*2007*	*Region*
Amble	Tyne/Tees			3	0	3	3	
Flamborough	Tyne/Tees	5	1	1	2	1	4	A
Tees Bay	Tyne/Tees		0	1	0	2		
C Dogger	Humber/Wash		3	6	4	2	4	
Indefatigable Bank	Humber/Wash		1	1	3	1	4	
N Dogger	Humber/Wash	9	6	4	3	7	7	
NE Dogger	Humber/Wash			5	7	10	6	B
Off Humber	Humber/Wash	3	2	0	3	1	2	
W Dogger	Humber/Wash	11	7	2	5	4	5	
Wash	Humber/Wash	0						
Rye Bay	Eastern Channel	1	2	0	0	0	0	A
S Eddystone	Western Channel				0	1	0	A
Carmarthan Bay	Severn		1	2	0	1	0	A
Inner Cardigan Bay	Cardigan Bay		5	2	2	1	1	
N Cardigan Bay	Cardigan Bay	3	3	2				A
S Cardigan	Cardigan Bay		4			4	2	
Burbo Bight	Irish Sea	4	1	2	1	3	3	
Liverpool Bay	Irish Sea	7	1	4	3	1	1	
Morecambe Bay	Irish Sea	2	3	1	1	1	1	A
Red Wharf Bay	Irish Sea	6	0	0	0	1	0	
SE Isle of Man	Irish Sea		5	2	4	3		
St Bees	Irish Sea	4		3	0	3	4	
St Abbs Head	East Scotland Coast	2	1	1	2	2	2	A
Bell Rock	East Scotland Coast	0	0	1	1	3	3	
SE Fair Isle	North Scotland Coast	1	0	1		0	2	A
Moray Firth	Moray Firth	0	0	0	0	2	2	A

Type A
Type B
Type C

genetics of dab around the coast of the UK and the North Sea has investigated the presence of population sub-structuring (Tysklind et al., 2009). Studies of this type highlight how ecological data will play an increasingly important role in interpretation of data collected for the purposes of assessing status of the marine environment. Effects of other confounding factors that may affect disease prevalence, such as the age, sex and migrational tendencies of fish also need consideration. Recognising the contributing role of such factors will significantly refine our approaches to marine monitoring and will lead to a greater understanding of cause-effect pathways, particularly with respect to the biological effects of contaminant exposure (Stentiford et al., 2005; Ward et al., 2006; Hines et al., 2007; Bignell et al., 2008).

For the purposes of health status classification based upon disease, the ICES Working Group on Pathology and Diseases of marine Organisms has recently proposed a 'Fish Disease Index' (FDI) that will be utilised for defining disease trends in fish captured from open ocean monitoring sites (ICES, 2007). The introduction of the FDI represents the final phase in the development of robust quality assurance for the use of fish disease measurement in defining the health status of fish stocks.

3.1.4.2 Endocrine disruption in estuarine and offshore areas

3.1.4.2.1 Key points

- Chemicals that have the potential to interfere with the normal action of endogenous hormones of aquatic organisms can lead to steep population declines, as was seen in some dogwhelk populations as a result of exposure to tributyltin (TBT) from antifouling paints
- A suite of specific biomarkers in flatfish has indicated the widespread occurrence of oestrogenic chemicals in the UK estuaries in the 1990s. Since then no further monitoring of biological responses has taken place creating a gap in knowledge on the current state of contamination from xenoestrogens
- The limited new evidence suggests that the problem of oestrogenicity is also present offshore
- Further work needs to be done on developing suitable surrogate species in order to assess the effects of endocrine disrupting chemicals on sustainability of fish populations.

3.1.4.2.2 Introduction

A number of natural and man-made chemicals have the potential to interfere with the endocrine system of fish. These so-called endocrine disrupting chemicals (EDCs) are diverse in nature. In freshwater systems, natural and synthetic steroidal oestrogens originating mainly from domestic sewage discharges account for the largest part of oestrogenic activity. Industrial chemicals such as alkylphenols are also contributing in total oestrogenic activity; they are weak oestrogen agonists but are produced in high volumes. It is of major concern however, that we know very little of the nature of EDCs in estuarine and marine sediments. A TIE (Toxicity Identification Evaluation) approach, using an *in vitro* screen for oestrogens, revealed that 91% of the UK estuarine sediments tested were oestrogenic and that 99% of this activity was due to unknown chemicals (Thomas et al., 2004).

In terms of biological effects, post EDMAR, the large research programme that was conducted between 1998 and 2001, there has been no further systematic monitoring of oestrogenic activity in estuaries. The diagnostic

end-points used widely for detecting activity are the presence of the egg-yolk protein vitellogenin (VTG) in immature and male fish and the incidence of intersex (ovo-testes). The former is the most robust, extensively tested and internationally recognised biomarker for xenoestrogens while the latter has a more complex aetiology although oestrogens appear to be implicated in the majority of experimental work published thus far. It is true that both VTG and intersex incidence in flounder revealed some downward trends in UK estuaries between 1996 and 2001 (Kirby et al., 2004), most likely due to government pressure to reduce emissions in light of the early biological effects data. The lack of monitoring data post 2001 however precludes any firm statement on the current status. In one occasion the VTG levels in flounders caught in the Mersey Estuary in 2004 were measured and it was found that the mean VTG value was more than two-fold higher than in 2001. The Tyne Estuary fish, however, remained at low VTG values as in 2001 (Kirby et al., 2006). These results suggest that additional monitoring in estuaries should be considered, either to highlight any newly emerged problems or to provide reassurance regarding environmental status in respect to EDCs.

3.1.4.2.3 Developments since *Charting Progress*

It is not possible to report on changes since *Charting Progress* as no further monitoring has taken place in UK estuaries.

3.1.4.2.4 Presentation of the evidence

In terms of signs of endocrine disruption in the open sea, reports are less numerous but include elevated VTG or zona radiata protein (Zrp) in male swordfish (Fossi et al., 2001, 2002, 2004; Desantis et al., 2005), tuna (Fossi et al., 2002; Barucca et al., 2006), cod (Scott et al., 2006a) and dab (Scott et al., 2007). The cod study revealed two significant facts: first, that there was no association between VTG levels and site of capture (sampling locations included the North Sea and the English Channel, the Irish Sea and the Shetland box), and second, that VTG was elevated in large male fish (a closer analysis of the data revealed that there was no correlation between the VTG titres and the age of the male cod but there was a significant correlation between the VTG and the weight of the fish – see Figure 3.36). On the basis of these data the following hypothesis was formed: the fish are picking up potential EDCs via their food rather than from water and the inclusion of bottom-dwelling organisms in the diet of large cod may account for the difference in VTG titres with age. Further investigations began early in 2008.

A follow-up study was conducted in 2004/05 to determine whether dab caught in offshore waters show any evidence of exposure to oestrogenic endocrine disrupters (Scott et al., 2007). The results were positive with a number of male dab registering significant VTG titres. Again, there was not a clear pattern of VTG levels in relation to point of capture but there was a strong positive correlation between size of fish and VTG levels (as in cod and tuna). Although there was no clear distinction between North Sea and Irish Sea in VTG levels there were clear differences between individual sites, suggesting that in the case of dab the levels of exposure to EDCs vary between different areas. The Dogger Bank area for example was registering significantly higher VTG levels in male dab. Although two intersex dabs were found in the Dogger Bank area in the 2003 survey (Stentiford and Feist, 2005; Figure 3.37), no intersex fish were found in the 2004/05 surveys. In addition, VTG concentrations in dab were typically 10 to 20 times lower than those found

Figure 3.36 Concentrations of VTG in plasma of male cod caught in three different areas around the UK plotted against the body weight of the fish with a) VTG concentrations shown on a linear scale and b) VTG concentrations shown on a logarithmic scale (with regression line). © Crown copyright 2010: permission granted by Cefas. ■ North Sea (n=82); ▲ Shetland Box (n=85); ● Irish Sea (n=99).

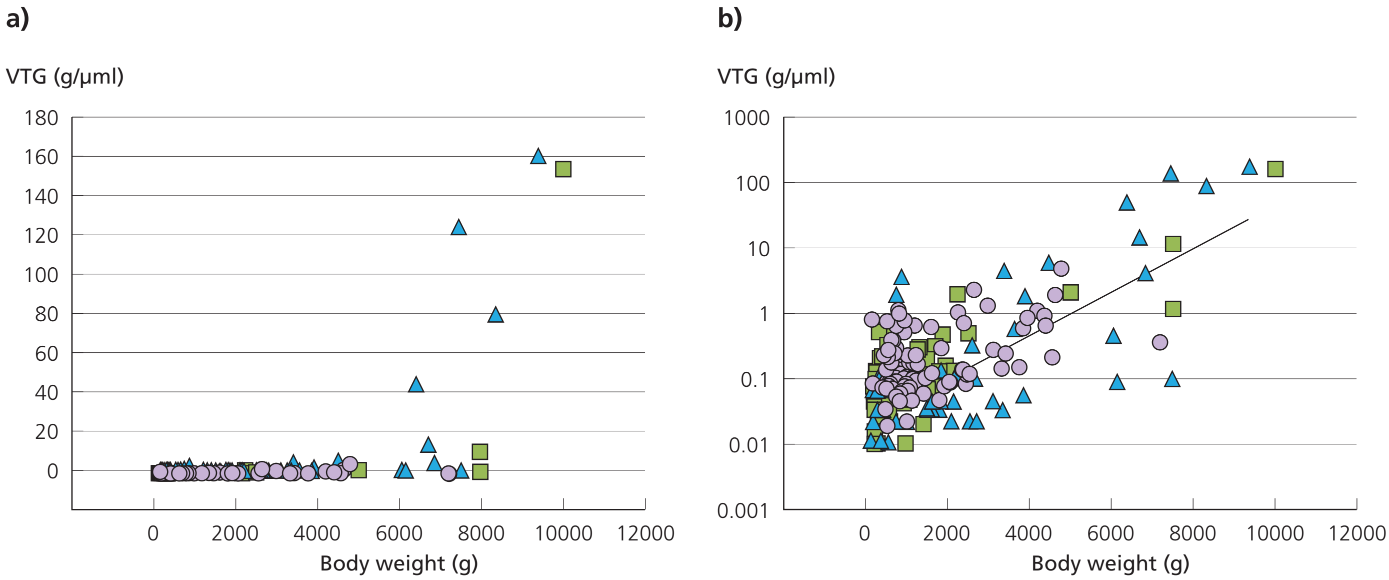
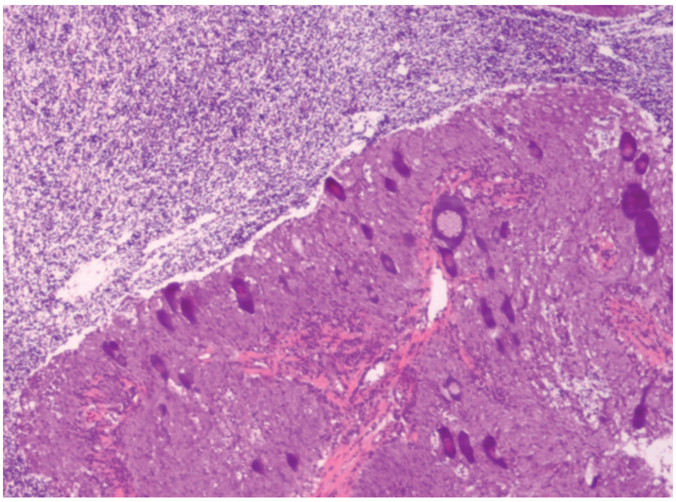


Figure 3.37 A specimen of intersex dab caught on the Dogger Bank in the 2003 survey. © Crown copyright 2010: permission granted by Cefas.



in cod. Since dab are lower down in the food chain than large cod, this finding supports the hypothesis that the main route of exposure is via food.

In terms of preferred sentinel species, although cod appears to be more sensitive, the scarcity of large fish at least in the North Sea rules out the possibility of adopting cod as a sentinel for EDCs effects. Dab on the other hand, although presenting a number of advantages, appears to be less sensitive so a great deal of effort should be placed in method standardisation before VTG measurements are used for monitoring purposes.

3.1.4.2.5 Progress towards the vision of clean and safe seas

There are many difficulties in defining what is normal for basal VTG concentration in wild male fish. This is largely due to the difficulty in defining a reference or control site because,

to some extent, every place on the planet has already been affected by pollution. Robinson and Scott (2006) suggested that one way of resolving this problem would be to assign basal levels by looking at the available data. There are a limited number of environmental and laboratory data already available for plasma VTG concentrations in flounder, dab and cod and up to seven other species (Hiramatsu et al., 2006). A background concentration of 0.13 µg/ml has been calculated based upon the 90th percentile of all of the male VTG concentrations (range = < 0.01 to 0.17 µg/ml, n = 95) in fish collected at a UK reference estuary (River Alde) between 1996 and 2001. The dab data suggest that 0.1µg/ml is an acceptable basal level. In a caged cod study (Scott et al., 2006b), VTG concentrations in males from 'reference' sites in the North Sea were determined to be in the range < 0.01 to 1.35 µg/ml (n = 69). Based upon the 90th percentile of these data, a provisional background concentration of 0.22 µg/ml was proposed for cod (Robinson and Scott, 2006). All these concentrations are close to the background concentration of 0.1 µg/ml proposed for Japanese flounder and Japanese common goby (Hiramatsu et al., 2006).

The implications of oestrogen exposure to fish health are more difficult to define on the basis of available data. One would expect that as a direct consequence of high VTG in male plasma (i.e. more than 10 000 µg/ml), which was often the case in estuarine flounders caught in the late 1990s, kidney failure is an adverse outcome. This has been demonstrated before (Folmar et al., 2001). The potential effects of EDCs however can be far more reaching than individual organ pathology. They may compromise the ability of the individuals to reproduce, resulting in population level changes. On the basis of the results so far it appears that, offshore at least, the concentrations of VTG are generally low (albeit more than the assigned baseline), hence of low biological significance.

There are two issues associated with this statement. One is that even though the production of VTG is in the range of low µg/ml, the fish are using some energy to produce this protein that has no biological role for them, hence wasting potentially important energy resources. The second issue is relevant to the nature of the causative compounds: if they are cumulative and persistent, then VTG induction might just be the tip of the iceberg in terms of biological and, ultimately, population effects. Several experimental studies (admittedly using freshwater fish) have shown that prolonged laboratory exposure to oestrogenic EDCs at environmentally relevant concentrations (Nash et al., 2004; Mills and Chichester, 2005) may reduce the reproductive success of the fish and so potentially adversely affect populations. Although it is difficult to prove similar effects in the field, there has been a recent report (Johnson et al., 2005) that female English sole (*Parophrys vetulus*) in Puget Sound, USA (where males had elevated VTG concentrations) underwent precocious vitellogenesis and had a markedly slower rate of subsequent gonadal development (presumably due to a negative feedback mechanism operating between the brain and the gonadal axis) than fish from sites with no evidence of oestrogenic endocrine disruption. Since the spawning periods of many fish species that reproduce in temperate waters maximise the availability of food for the growing larvae, one can speculate from this observation that, in temperate seas, where the plankton blooms are precisely timed and often short-lived, oestrogenic EDCs could have a negative impact on population size by simply delaying the time of spawning.

3.1.4.2.6 Need for further work

Unfortunately, the reproductive cycles of model species that are recommended by the OECD (Organisation for Economic Co-operation and Development) for screening potential EDCs are very different to those of the majority of fish species that live in temperate waters. Aquaria species tend to have a continuous reproductive cycle while the majority of wild fish have an annual cycle, hence the conclusions on potential implications of EDC exposure to reproductive success and recruitment resulting from OECD tests are of limited value. Consideration should be given to the need to develop a marine fish screening test that is relevant to risk assessment of EDCs in the marine environment.

In addition, there is an urgent need for continuous monitoring of estuaries and offshore areas for signs of endocrine disruption. When no data are available it is impossible to draw conclusions on environmental status. In addition, further work should support the development of internationally standardised tests in aid of generating comparable data across laboratories.

3.1.4.2.7 Overall conclusions and forward look

In view of the lack of recent data, there is a need for further monitoring in estuaries of VTG presence in male fish before it can be concluded that EDCs are an historic problem. Although the levels of VTG present in wild fish caught offshore are generally low, the potential implications of oestrogen exposure to population sustainability should not be neglected. A marine species representative of the reproductive cycle of most temperate water fish could be used as a surrogate to model the potential effects of EDCs in the marine environment.

3.1.4.3 Imposex and tributyltin

3.1.4.3.1 Background

Tributyl tin oxide (TBT) was first used as a biocide in marine antifouling paints for yachts and large ships in the mid-1980s. It is an extremely effective biocide and when first introduced its environmental impact was unknown. Subsequently, TBT-specific effects became apparent, most notably, thickening of shells of Pacific oysters (*Crassostrea gigas*) and the development of imposex in dogwhelks (*Nucella lapillus*) and intersex in periwinkles (*Littorina littorea*). Imposex is the imposition of male sexual characteristics on female gastropods and has been found to be the most sensitive indicator of TBT exposure. The effect is dose-related and severe imposex can lead to sterility in females and detrimental reproductive effects on individuals and populations.

TBT-specific biological effects monitoring was first established in the mid-1980s. The data collected by 1987 provided the evidence for environmental damage and subsequently, led to a UK ban on the use of TBT on small boats and aquaculture facilities. In 1989, the EU imposed a similar ban (EU, Council Directive 76/769/EEC).

Over the past fifteen years, extensive surveys have been conducted to measure the prevalence of imposex in the UK. In 1992, in preparation for the 1993 North Sea Quality Status Report, Fisheries Research Services (FRS), Aberdeen, conducted a survey around the North Sea. To complement this study a further survey was conducted using a similar sampling strategy, by laboratories in the countries around the Celtic Sea with the added emphasis on including some ‘hot spot’ monitoring. In 1998, FRS conducted a further survey which was reported in *Charting Progress*.

An IMO (International Maritime Organization) International Convention on the Control of Harmful Anti-fouling Systems agreed at a Diplomatic Conference, in October 2001, to prohibit the application or re-application to ships of organotin compounds as biocides in antifouling systems from 1 January 2003. This has now been implemented in the EU by Council Directive 2002/62/EC and, in the UK, approvals for the use of organotin compounds acting as biocides in antifouling systems have been revoked. More recently, the IMO ban has included the prohibition of use (or presence) on any ship's hull and this will take effect from 1 September 2008.

3.1.4.3.2 Surveys conducted since *Charting Progress*

In the light of the EU ban and the revocation of approvals in the UK for the application of TBT, a UK-wide baseline survey of the effects and residual concentrations of TBT in UK waters was conducted in 2004. These data were to provide a baseline for further monitoring (trends), and provide the basis for a UK-wide strategy to be developed following the total prohibition of use on the hull of any ship (IMO ban). A further survey was conducted in 2007 in advance of the total ban that took place in 2008.

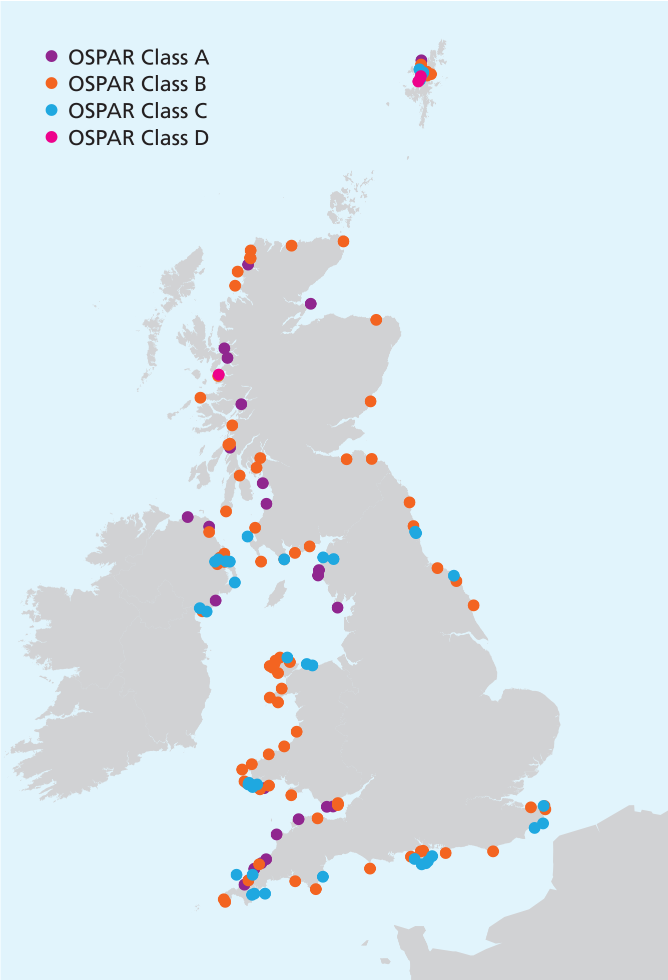
The development of imposex in *N. lapillus* may be divided into six stages, depending upon the developmental state of both the penis and vas deferens in the female (Gibbs et al., 1987). Each of the seven stages of imposex is known as a Vas Deferens Sequence (VDS) stage and calculation of the mean VDS for a group of females (of around 20) provides the Vas Deferens Sequence Index (VDSI) that may be used to compare the reproductive competency of different populations.

Data reported here include the 2004 and 2007 imposex survey on dogwhelks and compare this with the historical survey data from 1992 and 1998 (see Table 3.29). The data are expressed as VDSI as defined by the OSPAR biological effects assessment criteria for imposex in *N. lapillus* (OSPAR, 2004).

- **Class A.** The level of imposex is close to zero (0% to ~30% of females have imposex) indicating exposure to TBT concentrations close to zero, which is the objective in the OSPAR Hazardous Substances Strategy
- **Class B.** The level of imposex indicates exposure to TBT concentrations below the EAC derived for TBT. For example, adverse effects in the more sensitive taxa of the ecosystem caused by long-term exposure to TBT are predicted to be unlikely to occur
- **Class C.** The level of imposex indicates exposure to TBT concentrations higher than the EAC derived for TBT. For example, there is a risk of adverse effects, such as reduced growth and recruitment in the more sensitive taxa of the ecosystem caused by long-term exposure to TBT
- **Class D.** The reproductive capacity in the populations of *N. lapillus* is affected as a result of the presence of sterile females, but some reproductively capable females remain. For example, there is evidence of adverse effects, which can be directly associated with the exposure to TBT.
- **Class E.** Populations of *N. lapillus* are unable to reproduce. The majority, if not all females within the population have been sterilised
- **Class F.** Populations have died out.

Figure 3.38 illustrates the latest OSPAR classifications assigned to each site on the basis of VDSI results.

Figure 3.38 OSPAR classification of UK sampling sites, 2007. © Marine Scotland Science.



The data in Table 3.29 summarise the status of TBT-induced imposex around the coastline of the UK over a 12-year period from 1992 to 2007. In total 241 sites have been sampled at some point in time; 60 in England, 118 in Scotland, 44 in Wales and 19 in Northern Ireland. The sampling design of the programme in 2004 and 2007 included an increased number of sampling locations to take account of knowledge gaps and improve ‘hot spot’ sampling, such as around Dover and Southampton Water. It is clear from from the data as a whole there has been a decrease in the severity of imposex (see Table 3.30). In 1992, 55% of dogwhelks sampled were classified as OSPAR Class D (i.e. showing reproductive impairment at the population level), whereas in 2007 this had declined to 2.6%. Correspondingly, in 1992 the percentage of sites in OSPAR Classes B and A were 5% and 0% and in 2007 this had changed to 52.2% and 16.8% for Classes B and A respectively.

Of the 241 sampling sites, 136 have been sampled on two or more of the sampling years. In these instances it is possible to assess the changes in imposex in more detail. This is shown in Table 3.29 by the arrows indicating change in the OSPAR class: at 3 sites the class increased by one, at 51 sites the class remained the same, at 52 sites there was a decrease of one class, and 20 sites showed a decrease of two OSPAR classes. A closer look at the three sites where an increase in the OSPAR class was observed (indicating an increased effect of TBT-induced imposex) suggests that the measurements of imposex were very close to the cross-over point between classes and only a small increase in VDSI was observed.

3.1.4.3.3 Summary and conclusion

Studies conducted during the 1980s, showed that TBT from antifouling paints had a severe effect on populations of the most sensitive gastropod species such as the dogwhelk. This provided the impetus for UK legislation to prohibit the use of TBT–based antifouling paints on small boats. However, the 1992 survey showed that 55% of populations sampled had individuals severely affected by TBT-induced imposex, resulting from historic exposure, residual contamination in the environment (e.g. sediments) or effects from TBT antifouling paints from shipping. The legislation introduced by the IMO in 2003 and 2008 is likely to have had an important impact in reducing TBT concentrations in the marine environment and so reducing imposex in gastropods. This was borne out

Table 3.29 Imposex measurements at sites in England, Wales, Scotland and Northern Ireland (listed in alphabetical order for each region). Data shown are VDSI values and OSPAR assessment class. A decrease in OSPAR class represents a reduction in pollution-induced imposex, and so an environmental improvement.

Class		*Change in OSPAR classification over the reporting period*
A	↗	Increase of one class
B	→	No change in class
C	↘	Decrease of one class
D	↓	Decrease of two classes
E & F		

Site Name	*1992*		*1997/98*		*2004*		*2007*		*Change in OSPAR classification over the reporting period*
	VDSI	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	
ENGLAND									
Amble					2.5	C			
Bembridge, Isle of Wight							2.56	C	
Portland Bill	4.0	D	-	F			0.41	B	↓
Blyth					3.54	C			
Blyth Ferry	3.96	C	3.86	C			0.44	B	↘
Boulmer	3.97	C	3.52	C	0.9	B	0.77	B	↘
Bovisand	4.69	D	4.03	D	2.4	C	1.26	B	↓
Brixham	4.33	D	4.0	D	3.65	C	3.11	C	↘
Coombe Martin			2.36	C	1.14	B	0	A	↓
Dumpton Gap							1.21	B	
Eastbourne	4.1	D	4.08	D	3.76	C	0.88	B	↓
East Cowes							1.19	B	
Folkestone	4.36	D	4.03	D	3.63	C	1.24	B	↓
Gurnard Bay							1.27	B	
Gwennap Head	3.74	C	3.71	C	2.35	C	1.0	B	↘
Hanover Point							2.1	C	
Hartland Quay			2.27	C	0.79	B	0.24	A	↓
Hartlepool					3.63	C	1.56	B	↘
Hayling Island	2.09	C							
Heysham			4.0	D					
Horse Ledge, Shanklin							3.35	C	
Holywell Bay (Whitehills Bay)			1.88	B					
Maryport			3.9	C	2.36	C	2.29	C	→

Site Name	1992		1997/98		2004		2007		Change in OSPAR classification over the reporting period
	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	
Parsons Rock, N. Sunderland							3.55	C	
Palm Bay							1.72	B	
Perranporth			1.42	B			0.71	B	→
Polzeath			3.68	C	2.95	C	0.92	B	↘
Porlock Weir			2.56	C	2.0	C	1.1	B	↘
Porth					3.63	C			
Porthcorthan			1.92	B	0.33	B	0	A	↘
Porthtowan			1.2	B	0.15	A	0	A	↘
Porthquin			2.19	C			0.13	A	↓
Roa Island			3.0	C	1.79	B	0.06	A	↓
Robin Hood's Bay							1.26	B	
Saltburn							0.85	B	
Selsey Bill	4.14	D	3.9	C			0.64	B	↘
Sennen Cove			2.92	C	2.35	C	0.5	B	↘
Sewerby	3.88	C	3.86	C	2.3	C	0.7	B	↘
St. Agnes			1.7	B					
St. Bee's Head			1.5	B	1.43	B	0.21	B	→
St Catherine's Point							2.0	C	
St. Columb Minor			3.8	C			3.3	C	→
St. Margaret's Bay							2.62	B	
St. Mary's Lighthouse					3.2	C			
St. Mawes							2.59	C	
Start Point	4.0	D	3.77	C	3.08	C	1.11	B	↓
Swanpool							2.78	C	
Towan Head			2.38	C			2.45	C	→
Totland Bay							1.76	B	
Trewentworth Sand			2.12	C			0	A	↓
Trenance			0.88	B					
Trevone			2.83	C					
Tyne entrance							2.2	C	
Ventnor							3.29	C	
Walpole Bay							3.25	C	

Site Name	1992		1997/98		2004		2007		Change in OSPAR classification over the reporting period
	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	
West Bay	3.97	C	3.0	C	1.05	B			↘
West Pentire			2.42	C					
Whitby	4.04	D	4.16	D	2.71	C	2.15	C	↘
Whitehaven			1.96	B			0.19	A	↘
Whitley Bay							3.19	C	
WALES									
Abermawr			1.25	B	1.95	B	1.5	B	→
Aberystwyth			0.6	B	0.19	B	0.53	B	→
Angle Bay			4.33	D			3.73	C	↘
Barry Island							0.24	A	
Cable Bay			1.47	B			0.81	B	→
Caernarfon			0.50	B					
Cemaes Head			1.63	B	1.15	B	0.5	B	→
Cemlyn Bay			1.67	B					
Church Bay (Porth Swaton)			2.44	C			1.17	B	↘
Dale Fort			4.75	D	3.57	C	2.63	C	↘
Freshwater West			4.29	D	3.5	C	2.61	C	↘
Great Ormes Head			2.83	C			2.06	C	→
Lavernock Point			4.0	D	2.55	C	0.32	B	↓
Llanbadrig			3.35	C	1.05	B	0.62	B	↘
Llaneilian							2.27	C	
Limpert Bay							0.17	A	
Manorbier			2.81	C	2.2	C	0	A	↓
Marloes Sands			2.26	C	2.35	C	0.18	A	↓
Martin's Haven			2.0	C	1.0	B	1.69	B	↘
Moelfre			2.47	C			0.38	B	↘
Monk Haven			4.29	D	3.0	C	2.33	C	↘
New Quay			1.82	B	1.31	B	1.0	B	→
Penarth							0.67	B	
Pennar Jetty			3.05	C					
Pennar Point			3.88	C			3.45	C	→
Point Lynas			1.87	B					

Site Name	1992		1997/98		2004		2007		Change in OSPAR classification over the reporting period
	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	
Port Eynon			2.8	C	1.55	B	0.68	B	↘
Porth Colman			1.21	B	0.27	A	1.9	B	→
Puffin Island			2.13	C					
Renny Slip			2.86	C	2.0	C			→
Rhos on Sea			1.77	B	2.55	C	2.44	C	↗
Sandy Beach			2.57	C	2.1	C	1.67	B	↘
Sam Bach			1.68	B			1.9	B	→
South Stack			2.36	C			1.06	B	↘
Stackpole Quay			2.64	C	2.0	C	1.37	B	↘
Tenby			2.25	C	3.24	C	0.56	B	↘
Trearddur			2.27	C			0.74	B	↘
Trefor			1.95	B			0.75	B	→
Watwick Bay			4.31	D	3.15	C	2.93	C	↘
West Angle Bay			4.33	D	3.9	C	2.65	C	↘
West Dale Bay			3.76	C			0.74	A	↓
West of Holyhead Harbour			2.22	C					
Westdale Bay			3.82	C					
Whitesands Bay			1.94	B	2.14	C	1.0	B	→
SCOTLAND									
Craignish Point			0.5	B					
Aird			0.47	B	0.4	B	0.35	B	→
Ardnamurchan					0.76	B	0.37	B	→
Ardrossan			2.0	C	1.0	B	0.29	A	↓
Back of Keppoch			0.53	B					
Badcall			1.8	B	0.14	A	0	A	↘
Ballantrae			1.13	B	0.4	B	0.74	B	→
Bettyhill					0.38	B	0.47	B	→
Billia Skerry – Sullum Voe					1.79	B	1.2	B	→
Bressay, Kirkabister					3.92	C			
Bressay, Tain of Ham					4.0	D			
Broadford					3.64	C			
Burgo Taing – Sullum Voe					0.28	A	0.5	B	↗

Site Name	*1992*		*1997/98*		*2004*		*2007*		*Change in OSPAR classification over the reporting period*
	VDSI	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	
CairnRyan (1N)					3.8	C			
CairnRyan (6S)					3.7	C			
CairnRyan (8S)					2.5	C			
Cluas Deas (Stoer)			1.95	B					
Cockenzie 1					3.65	C			
Cockenzie 2					3.21	C			
Cockenzie 3					3.67	C			
Cockenzie 4					3.47	C			
Cockenzie 5					3.45	C			
Cockenzie 6					3.56	C			
Craignish 10					0.29	A	0.05	A	→
Craignish 12					1.05	B	0.35	B	→
Croivie					1.44	B	1.14	B	→
Cuil			0.8	B	0.29	A	0.29	A	↘
Cullen					0.06	A			
Droman			1.14	B	0.72	B	1.0	B	→
Dunure			2.5	C	0.26	A	0.29	A	↓
East of Ollaberry – Sullum Voe					1.57	B	0.7	B	→
Easterwick – Sullum Voe					0.36	B	0.15	A	↘
Eyemouth 1					4.0	D			
Eyemouth 2					3.75	C			
Eyemouth 3					3.63	C			
Eyemouth 4					3.73	C			
Eyemouth 5					3.44	C			
Eyemouth 6					2.82	C			
Fara					2.23	C			
Fearnmore			0.32	B	0.24	A			↘
Ferryden			4.28	D					
Flotta Pier					3.44	C			
Fraserburgh, Bath Street					4.0	D			
Fraserburgh, Broadsea					4.0	D			
Fraserburgh, Cairnbulg					3.71	C			

Site Name	1992		1997/98		2004		2007		Change in OSPAR classification over the reporting period
	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	
Fraserburgh, Kinnaird Head					3.96	C			
Fraserburgh, Sudan					4.06	D			
Gallanach			1.79	B	1.32	B	1.0	B	→
Glenelg			0.38	B	0.14	A	0.15	A	↘
Grunn Taing – Sullum Voe					3.46	C	3.5	C	→
Isle of Whithorn			2.0	C					
Keil Point			0.65	B	0.53	B	0.68	B	→
Kilmory			1.65	B					
Kinlochbervie					3.83	C			
Kinlochbervie Harbour					3.95	C			
Kinnairds Head	4.1	D	3.87	C					↘
Kirkwall – Hatston Pier					1.19	B			
Laxford 7					0.19	A	0.42	B	↗
Laxford 8					0.91	B	0.5	B	→
Lerwick – South Ness					4.0	D			
Little Roe – Sullum Voe					3.63	C	3.56	C	→
Lochinver – outside harbour					4.06	D			
Mallaig					4.02	D			
Mallaig Bheag			2.14	C					
Mallaig Harbour			4.36	D	3.96	C	4.0	D	→
Maryport			0.5	B	0.42	B	0.33	B	→
Mavis grind – Sullum Voe					4.25	D	4.11	D	→
Montrose	4.04	D							
Morar			0.73	B					
Moss Bank/Grunna Taing – Sullum Voe					1.47	B	1.08	B	→
North Cairn			1.78	B	2.8	C	2.05	C	↗
Norther Geo – Sullum Voe					0.08	A			
Northward – Sullum Voe					4.18	D	4.08	D	→
Noust of Burraland – Sullum Voe					4.05	D	4.0	D	→
Orfassary – Sullum Voe					1.12	B	0.83	B	→
Port Seton	4.23	D	4.08	D	2.55	C	1.7	B	↓
Port-an-eorna			0.62	B	0.3	B	0.25	A	↘

Site Name	*1992*		*1997/98*		*2004*		*2007*		*Change in OSPAR classification over the reporting period*
	VDSI	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	*VDSI*	*OSPAR Class*	
Portnaluchalg			0.61	B					
Portree					4.13	D			
Rattray Head					0.82	B			
Reif			2.95	C	0.83	B	1.33	B	↘
Rhuba nan Sasan			0.92	B	0.74	B			→
Rockcliffe			1.25	B	0.84	B	0.45	B	→
South of Mallaig			1.6	B	1.54	B	0.38	B	→
Samphrey/The Helliack – Sullum Voe					0.4	B	0.2	A	↘
Sandhaven, East					4.0	D			
Sandhaven, Pitullie					2.25	C			
Scarf Stane – Sullum Voe					4.07	D	2.57	C	↘
Skateraw	3.66	C	3.57	C	2.82	C	0.9	B	↘
Skateraw 1					2.7	C			
Skateraw 2					1.8	B			
Skateraw 3					1.78	B			
Skateraw 4					1.5	B			
Skateraw 5					2.14	C			
Skatie Shore					3.46	C			
Skaw Taing – Sullum Voe	3.66	C	3.57	C	2.82	C	0.9	B	↘
Skipness			1.07	B	0.53	B	0.38	B	→
Skirza	2.3	C	1.78	B	0.32	B	0.75	B	↘
Stoer					1.38	B	0.83	B	→
Stranraer (1W)					3.4	C			
Stranraer (3W)					2.5	C			
Stranraer (4W)					3.3	C			
Strathan					3.98	C			
Stromness Harbour					4.15	D			
Strone Point			2.25	C	1.3	B	0.62	B	↘
Tarbat Ness	1.84	B	2.03	C	0.26	A	0.09	A	↓
Tarbert					1.9	B			
The Brough – Sullum Voe					0.89	B			
The Kames – Sullum Voe					4.33	D	3.9	C	↘

Site Name	1992		1997/98		2004		2007		Change in OSPAR classification over the reporting period
	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	VDSI	OSPAR Class	
Tivaka Taing – Sullum Voe					4.0	D	3.58	C	↘
Toward Point			2.46	C	0.58	B	0.72	B	↘
Tynemouth					3.13	C			
Usan			4.0	D	3.0	C	0.4	B	↓
Uynarey – Sullum Voe					1.17	B	0.82	B	→
Voxter Ness – Sullum Voe					4.1	D	3.78	C	↘
West of Mallaig Harbour			2.48	C					
Whithorn – inside					3.25	C	2.6	C	→
Whithorn – outer					0.84	B	0.52	B	→
NORTHERN ISLAND									
Bloody Bridge			3.96	C	0.67	B	0.19	A	↓
Carlingford					2.74	C	1.78	C	→
Carrickfergus Quay			2.9	D					
Carrickfergus – Belfast Lough					4.0	D	2.6	C	↘
Clachan Rock – Belfast Lough					2.59	C	1.46	B	↘
Cloughfin Port – Belfast Lough					1.64	B	0.85	B	→
Cultra			3.1	C	2.19	C	1.21	B	↘
Cushendun			1.65	B	1.14	B	0.25	A	↘
Giant's Causeway			1.67	B	1.92	B	0.04	A	↘
Greencastle			4.13	D	4.0	D	2.88	C	↘
Helens Bay – Belfast Lough					3.58	C	1.78	B	↘
Lukes Point			4.0	D	3.91	C	2.81	C	↘
Portavogie			4.3	D	2.67	C	2.09	C	↘
Red Bay			2.57	C	2.0	C	1.76	B	↘
Ross Rock – Belfast Lough					3.89	C	2.33	C	→
Sandy Bay – Belfast Lough					3.27	C	2.12	C	→
Smelt Mill Bay – Belfast Lough					3.75	C	1.31	B	↘
Swinley Point – Belfast Lough					3.5	C	1.22	B	↘
Warren Point			4.0	D	3.83	C	2.35	C	↘

Table 3.30 Percentage of sites in each OSPAR class for imposex in dogwhelks for each reporting year: 1992, 1998, 2004 and 2007.

Year	Percentage of sites in each OSPAR class				
	Class E	Class D	Class C	Class B	Class A
1992	0	55.0	40	5	0
1997/8	0.8	17.9	43.9	37.4	0
2004	0	12.7	47.7	41	7.6
2007	0	2.6	28.4	52.2	16.8

by the data presented here for the surveys conducted in 2004 and 2007.

Imposex in dogwhelks is a very sensitive and highly specific indicator of exposure of animals to TBT. As a result, any imposex populations with an OSPAR class assessment of B and above will demonstrate exposure of the dogwhelks to TBT. In 1997, this amounted to 100% of the populations sampled, but in 2007 this had reduced to 83.2% with a marked decrease in the more severe effects (i.e. Class D). The UK-wide surveys of imposex have provided conclusive evidence of the effectiveness of the legislation brought in by the UK, the EU and latterly the IMO. In addition, the data provide a baseline for monitoring the effectiveness of the legislation in the future and for establishing Good Environmental Status under the EU Marine Strategy Framework Directive. These surveys fulfil the UK's obligation under OSPAR to report on the incidence of imposex in UK waters on a regular basis and, in addition, the data will be used by OSPAR for the Quality Status Report 2010 (OSPAR, 2010). It should also be noted that, although dogwhelks provide a suitable sentinel organism for monitoring the impact of TBT pollution, many other species may also be affected, as demonstrated in studies of the recovery of the Crouch Estuary in Essex following the initial ban on the use of TBT on small vessels in 1987 (Rees et al., 2001). Case Study 1 illustrates the spatial and temporal variations in the incidence of imposex at the Sullom Voe oil terminal in Shetland.

3.1.4.4 Cytochrome P4501A activity (EROD) and PAH bile metabolites in flatfish

3.1.4.4.1 Key points

- A data set for 35 offshore sites around the UK is presented over a period of eight sampling years (2000 – 2007)
- The data have been classified according to assessment classes derived from the overall data set available of 2400 data points.
- Some sites consistently give elevated responses: ten sites for EROD and four sites for bile metabolites. These sites with the exception of two were close to industrialised and/or densely populated conurbations
- Away from coastal areas the biomarker responses were all close to background with the exception of the West Dogger and Off Humber locations
- In a case study for Scottish inshore waters EROD activity was elevated at sites of known historical contamination, but the response was decreasing
- The relationship of biomarker response to contaminants in fish and sediments needs to be investigated
- The data need to be assessed with other higher biological effect responses. For example, applying the OSPAR chemical-biological effect integrated strategy when it becomes available.

Case Study 1:
Spatial and temporal trends of TBT-specific effects on dogwhelks at Sullom Voe oil terminal, Shetland

The oil terminal at Sullom Voe in Shetland has been operating since 1978 and receives a large number of oil and gas tankers. The use of TBT antifouling paints on these tankers has been a source of TBT contamination in Sullom Voe and the surrounding area for over 20 years. Since 1987, FRS in conjunction with the Shetland Oil Terminal Environmental Advisory Group (SOTEAG) has monitored populations of dogwhelks in Sullom Voe and the surrounding Yell Sound area for imposex.

Dogwhelks from sites close to the oil terminal in Sullom Voe show the highest levels of imposex (larger dots on the map) and some female dogwhelks have become sterilised (VDSI >4) at several of the sites within Sullom Voe (dark blue dots Figure 3.39). Since 1987, the level of imposex in dogwhelks from sites within Sullom Voe has decreased (Figure 3.40) due to changes in the types of TBT antifouling paints used and decreases in the numbers of tankers visiting the terminal. Monitoring of gastropods at these sites will continue to assess how they recover after the use of TBT as an antifoulant was banned on large vessels in 2008.

Figure 3.39 OSPAR classification of sampled sites in Yell Sound and Sullom Voe, 2007. © Marine Scotland Science.

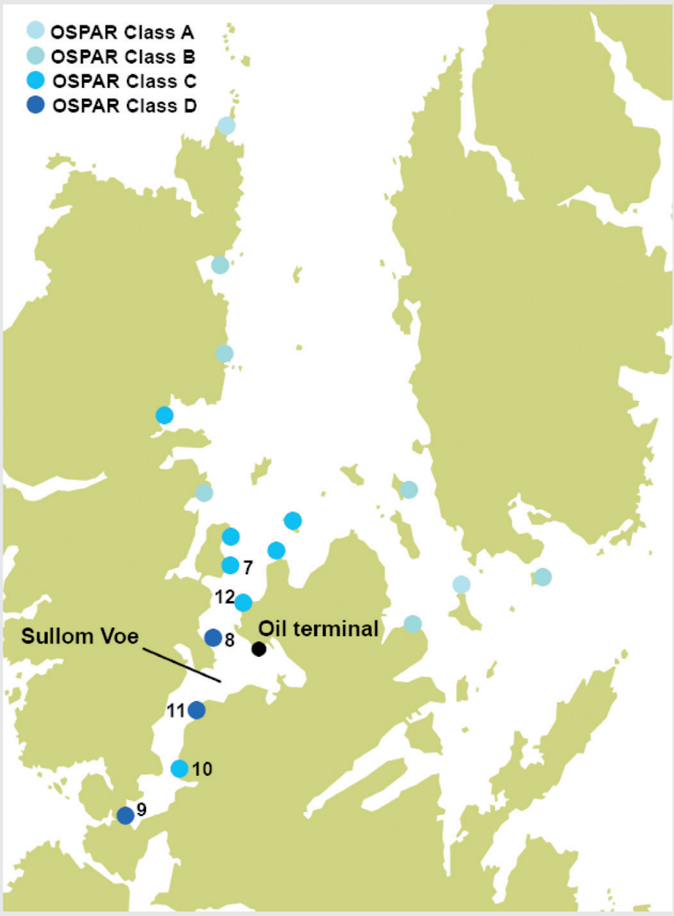
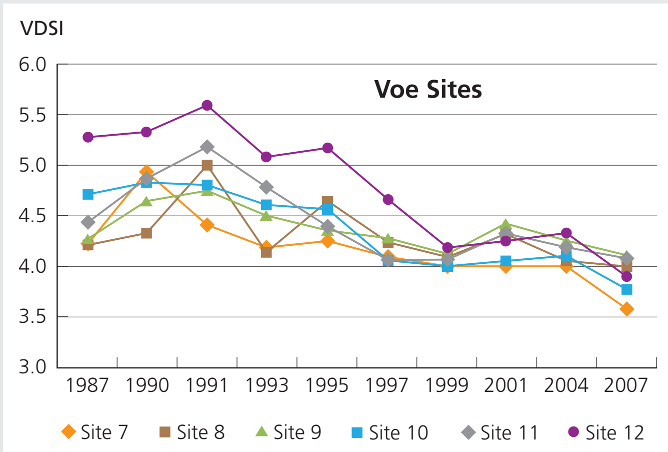


Figure 3.40 Trend in imposex (VDSI) of dogwhelks from sites within Sullom Voe between 1987and 2007. © Marine Scotland Science.



3.1.4.4.2 Introduction

The induction of cytochrome P450 enzymes (specifically EROD [7-ethoxyresorufin-*O*-deethylase] activity, measured as pmol/min/mg protein) in fish liver was first suggested as an indicator of environmental contamination in the 1970s by Payne (1976). This enzyme system is particularly important in the metabolism of many pollutants such as certain planar halogenated hydrocarbons (PHHs) and polycyclic aromatic hydrocarbons (PAHs) and other structurally similar compounds. Fish may be exposed to these compounds, through the food they eat and through exposure to contaminated sediments or less likely via the water column.

CYP1A is useful for biomonitoring purposes because its expression and activity increase upon exposure of fish to contaminants such as dioxins, planar PCBs and PAHs (benzo[*a*] pyrene). The value of EROD as a predictor of higher level health effects is unclear but it is known that the mechanism of CYP1A induction produces highly reactive intermediates of certain contaminants that may be responsible for higher level detrimental effects, including the onset of carcinogenesis, apoptosis and embryonic mortality.

The technique is now used widely (e.g. Förlin and Haux, 1990; Goksøyr et al., 1991; George et al., 1995; Whyte et al., 2000) and was one of the first biomarker techniques along with PAH bile metabolites to be incorporated into the OSPAR Joint Assessment and Monitoring Programme in 2000. In this respect they were designated as techniques to be used to address the OSPAR question ...*are PAHs present in the marine environment at sufficient concentrations to cause adverse biological effects?*

The measurement of PAH metabolites in fish bile has been used as a biomarker of exposure to PAH contamination since the early 1980s. When a fish is exposed to PAHs, its liver will detoxify and eliminate these compounds. The presence of metabolites in bile (and in urine) is the final stage of the biotransformation process whereby lipophilic compounds are transformed to a more soluble form and then passed from the organism in bile or urine. The analysis of PAH metabolites is relatively straight forward, either by direct fluorescence or by HPLC/F or GC-MS SIM (Jonsson et al., 2003; Ariese et al., 2005; Lin et al., 2006; Aas et al., 2000a,b). The data presented here were generated by direct fluorescence and are reported as 1-OH pyrene equivalents.

The induction of PAH bile metabolites occurs within days of exposure of a fish to PAH-type contaminants whereas EROD induction occurs over days to weeks. Both biomarker responses are reversible.

3.1.4.4.3 Developments since *Charting Progress*

Charting Progress presented preliminary data for EROD measurements in fish, *Limanda limanda* (commonly known as the dab). Seventeen sites were sampled in the years 1999, 2000 and 2001. Since *Charting Progress*, sampling activity has continued every year and the number of sites has increased to 35 to reflect a greater geographical coverage. In addition, bile metabolite measurements were made on the same fish.

Presentation of the evidence offshore for dab

The data presented here have been collected over nine years from 2000 to 2008 as part of the UK National Marine Monitoring Programme

(NMMP) and since 2007 as the UK Clean Seas Environment Monitoring Programme (CSEMP). The protocols used here follow those required by OSPAR (OSPAR, 2003). The target fish species is dab and samples were collected on research vessels during single Cefas and FRS cruises each year, and samples of liver and bile taken concomitantly from male fish. The sampling locations are given in Figure 3.41.

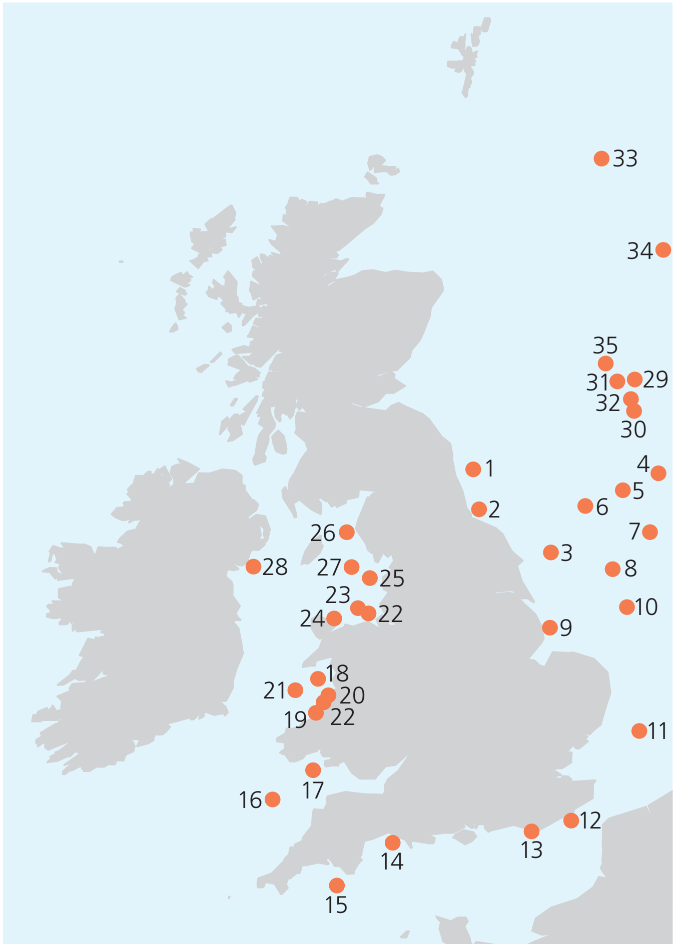


Figure 3.41 Fish sampling locations (for station numbers and geographical coordinates see Tables 3.32 and 3.34). © Crown copyright 2010: permission granted by Cefas.

Data assessment

As a general approach for *Charting Progress 2*, assessment criteria for biological effects should define the boundaries between three levels: the Background Response Range (BRR), the Elevated Response Range (ERR) and the High Effect Response Range (HERR). Formal internationally agreed assessment criteria for EROD and PAH bile metabolites have yet to be derived, although a background value of 40 pmol/min/mg protein has been suggested for EROD activity in male dab through the OSPAR/ICES Workshops on Integrated Monitoring of Contaminants and their Effects in Coastal and Open-Sea Areas (WKIMON) (OSPAR WKIMON, 2008).

For the purpose of this report an assessment strategy has been used based on the whole data set collected in the UK over a period of up to eight years. The following process was used to derive BRR, ERR and HRR values (see also Boxes 3.1 and 3.2). In addition, a fourth assessment category has been derived – Close to Background Response Range (CBRR):

- BRR is the biomarker response expected at sites that may be regarded as clean or reference sites
- CBRR identifies a response that may be slightly raised above the BRR but that is close to background, and both BRR and CBRR may be regarded as identifying sites where there is minimal impact from contaminants for the specific biomarker
- ERR identifies a response where the impact from contaminants is possible
- HRR identifies a response where the impacts from contaminants are probable.

EROD data

A total of 35 sites were sampled for dab in the UK, 28 in England and 7 in Scotland. For 14 sites the sampling took place annually between 2000 and 2008. Best coverage of the sampling

stations occurred between 2006 and 2008. Each of the four response ranges (HRR to BRR) was assigned a colour between dark blue and light blue and the data for each sampling occasion was appropriately allocated. The data are shown in Table 3.32 as a heat map of response. The EROD response was variable from year to year at any one site. At ten sites (Amble, Tees Bay, Off Flamborough, West Dogger, Off Humber, Burbo Bight, Morecambe Bay, St Bees, St Abb's Head, Bell Rock) the response range was at ERR and HRR for 50% or more of the sampling occasions (CFI – Cause For Investigation – in Table 3.32). At 18 of the 35 sites there were no occasions where the response range on any sampling year was above BRR or CBRR; these sites were all offshore, for example, Dogger Bank and in Scotland. Visually there does not appear to be any obvious trends but the data are yet to be analysed statistically.

PAH bile metabolites

A total of 28 sites were sampled in England, no data were available for Scottish waters. The year on year sampling was more intermittent than for the EROD with no samples analysed in 2003. Best coverage of the sampling stations occurred in the period 2006 to 2008. Each of the four response ranges (HRR to BRR) was assigned a colour between dark blue to light blue and the data for each sampling occasion was appropriately allocated. The data are shown in Table 3.34 as a heat map of response. At no sites were any mean responses recorded in the HRR. Only at nine sites, and over all sampling years were any mean responses recorded in the ERR. At four sites (Outer Humber, Burbo Bight, Morecambe Bay, St Bees Head) the response range was at ERR for 50% or more of sampling occasions (CFI – Cause For Investigation – in Table 3.34). At 19 sites there were no occasions when the response range on any sampling year was above BRR or CBRR; as with EROD these sites were all offshore, for example, Dogger Bank. There was insufficient data for an analysis of trends.

Box 3.1: Calculation of EROD response ranges

The data set consisted of 2400 data points which were distributed logarithmically. The 10th percentile value of 31 pmol/min/mg protein was used to calculate the response ranges shown in Table 3.31. The cut off for the upper BRR was calculated as the 10th percentile value × 3.3. The basis for using semi-log intervals for the assessment thresholds was that EROD activity has a logarithmic response and the use semi-log intervals resulted in matching activity levels at known offshore 'background' or 'reference' sites (e.g. Rye Bay) for BRR and CBRR.

Table 3.31 EROD response ranges.

Criteria	*Assessment calculation of upper range value for each criteria*	*Assessment criteria range; values as pmol/min/mg protein (LV = lowest value)*
BRR	10th percentile value × 3.3	LV – 109
CBRR	10th percentile value × 10	110 – 310
ERR	10th percentile value × 33	311 – 1089
HRR	> 10th percentile value × 33	> 1089

Presentation of the evidence inshore for flounder and plaice: a case study from Scottish Waters

To assess the effects of planar organic contaminants in inshore waters, biological effects measurements have been undertaken annually on flatfish species at key inshore locations in the Firth of Clyde and Firth of Forth. Where possible, monitoring has been

Table 3.32 Mean EROD values (pmol/min/mg) and associated assessment classes.

Background response range - LV up to 109 (pmol/min/mg)
Close to background response range 110 – 310 (pmol/min/mg)
Elevated response range 311-1089 (pmol/min/mg)
High effect response range greater than 1089 (pmol/min/mg)
CFI = Cause For Investigation (response range was at ERR and HRR for 50% or more of the sampling occasions)

Site Location with Lat. Long. position	*Site positions in Fig 1*	*2000*	*2000*	*2001*	*2001*	*2002*	*2002*	*2003*	*2003*	*2004*	*2004*	*2005*	*2005*	*2006*	*2006*	*2007*	*2007*	
Amble 55 16.01 N 01 15.26 W	1	952		446		1692		731		483		432		865		173		CFI
Tees Bay 54 45.25 N 01 08.31 W	2	1399		255		336		179		187		128		459		173		CFI
Off Flamborough 54 14.72 N 00 29.91 E	3	1534		175		279		522		262		704		441		90		CFI
North East Dogger 55 18.05 N 02 53.82 E	4					126				168		234		77		81		
North Dogger 55 04.08 N 02 05.40 E	5					170		105		257		129		87		63		
West Dogger 54 46.76 N 01 17.69 E	6	1403		342		174		458		467		258		176		241		CFI
Central Dogger 54 30.00 N 02 42.53 E	7	180				323		220		97		158		307		75		
Off humber 54 03.92 N 01 47.46 E	8	1161		290		416		590		591		272		1152		100		CFI
Outer Humber 53 19.37 N 00 25.47 E	9			27		119												
Indefatigable Bank 53 33.40 N 02 04.92 E	10			58				161		208		164		162		39		
Outer Gabbard 52 01.86 N 02 06.57 E	11	279		38		159		125				113		105		54		
Rye Bay 50 46.74 N 00 46.83 E	12	115		56		66		121		234		235		206		65		
Off Newhaven 50 45.59 N 00 00.00 E	13											172		229		37		
Inner Lyme Bay 50 36.86 N 02 55.82 W	14											276		120		42		
South Eddystone 50 06.44 N 04 06.06 W	15											185		102		68		
West Lundy 51 09.79 N 05 26.67 W	16	409						215				146				23		
Camarthen Bay 51 32.82 N 04 35.13 W	17							91		86		225		210		22		
North Cardigan Bay 52 42.44 N 04 32.29 W	18	152				107		80		93		209		71		102		
South Cardigan Bay 52 10.90 N 04 29.87 W	19											125		28		12		
Inner Cardigan Bay 52 18.00 N 04 16.35 W	20	203		40		180		39		111		187		18		105		
Outer Cardigan Bay 52 23.76 N 04 53.72 W	21			74		264		69										
Burbi Bight 53 28.24 N 03 20.47 W	22	750		231		522				298		774		238		562		CFI
Liverpool bay 53 28.32 N 03 41.91 W	23	198		143		531		120		389		604		203		86		
Red Warfe Bay 53 22.46 N 04 12.84 W	24	310		49		411		153		269		140		255		152		
Morecambe Bay 53 55.31 N 03 23.23 W	25	778		287		1076		273		430		1329		197		257		CFI
St Bees Head 54 30.71 N 03 47.63 W	26					536				328		333		156		110		CFI
SE Isle of Man 54 03.36 N 03 52.47 W	27			105		364		112		230		542		170		109		
Outer Dundrum Bay 54 04.81 N 05 37.29 W	28	795		1213				284				185						
Bell Rock 56 25.50 N 02 09.60 E	29			523		105		238		341		304		647		294		CFI
St Abb's Head 56 04.50 N 02 07.68 E	30			552		222		218		265		358		491		602		CFI
Marr Bank 56 25.00 N 01 50.00 E	31			386		74		119		181		273		282		96		

Site Location with Lat. Long. position	*Site positions in Fig 1*	*2000*	*2000*	*2001*	*2001*	*2002*	*2002*	*2003*	*2003*	*2004*	*2004*	*2005*	*2005*	*2006*	*2006*	*2007*	*2007*	
Wee Bankie 56 16.25 N 02 06.25 E	32			210		81		128		107		254		279		288		
SE of Fair Isle 59 13.00 N 01 30.00 E	33			234		92		72		215				260		125		
Beatrice Oilfield 58 08.00 N 03 01.00 E	34			130		27		74		76		145		181		45		
Montrose Bank 56 40.00 N 01 30.00 E	35													219		82		

Box 3.2: Calculation of bile metabolite response ranges

The data set consisted of 700 data points which were distributed linearly. The 10th percentile value of 54 ng/g 1-OH pyrene equivalents was used to calculate the response ranges as shown in Table 3.33.

Table 3.33 Bile metabolite response ranges.

Criteria	*Assessment calculation of upper range value for each criteria*	*Assessment criteria range; values as ng/g (LV = lowest value)*
BRR	10th percentile value	LV – 54
CBRR	10th percentile value × 4	55 – 216
ERR	10th percentile value × 8	217 – 432
HRR	> 10th percentile value × 8	> 433

undertaken using flounder as the sentinel species, but at several sites in the Clyde plaice have been used as an alternative. The EROD activity results from this temporal trend monitoring programme are assessed here using the same assessment framework as for offshore dab.

Plaice have been sampled annually since 2001 in the Clyde from Garroch Head, Holy Loch, Hunterston, Skelmorlie and Irvine Bay and from Broad Bay, Colonsay and Pladda as reference sites. In addition, EROD activity has also been determined in flounder from the upper Firth of Clyde (Bowling), from the upper Firth of Forth (Blackness) and, as a reference site, St Andrews Bay, on the East coast of Scotland. These sites are shown in Figure 3.42.

The assessment framework for EROD activity in plaice and flounder was developed using the same approach as for offshore dab (see Box 3.2). Based on a dataset of 466 observations (excepting for those below the Limit of Detection and including data from reference sites) for plaice, a 10th percentile value of 5.26 pmol/min/mg (protein) was used to calculate the response ranges in Table 3.35.

Owing to the relatively small available dataset for flounder EROD activity and because flounder and plaice respond in a similar manner, the response ranges developed for plaice were applied for the assessment of EROD data for both species.

EROD activities in plaice at reference sites were < 20 pmol/min/mg (protein). All other determinations were > 20 pmol/min/mg (protein) with the higher activities determined in areas where greater exposure to CYP1A inducing contaminants would be expected (see Table 3.36). The EROD response displayed some variability from year to year at any one site. At the reference sites (Broad Bay, Colonsay, Pladda) the response range was at BRR on all occasions. At the Irvine Bay, Hunterston and

Table 3.34 Mean 1-OH values (ng/g) for each year with associated assessment class; see text for explanation.

Background response range - LV up to 54 (ng/g)
Close to background response range 55 – 216 (ng/g)
Elevated response range 217-432 (ng/g)
High effect response range greater than 433 (ng/g)
CFI = Cause For Investigation (response range was at ERR for 50% or more of the sampling occasions)

Site Location with Lat. Long. position	*Site positions in Fig 1*	*2000*	*2000*	*2001*	*2001*	*2002*	*2002*	*2003*	*2003*	*2004*	*2004*	*2005*	*2005*	*2006*	*2006*	*2007*	*2007*	
Amble 55 16.01 N 01 15.26 W	1	153.1										81.5		83.9		89.3		
Tees Bay 54 45.25 N 01 08.31 W	2	180.5		217.4		198.6										125.7		
Off Flamborough 54 14.72 N 00 29.91 E	3	124.5		193.9		169.3						92.7		42.9		113.5		
North East Dogger 55 18.05 N 02 53.82 E	4					141.1				64.2		95.9		53.7		50.5		
North Dogger 55 04.08 N 02 05.40 E	5									86.9		70.9		12.6		86.0		
West Dogger 54 46.76 N 01 17.69 E	6	138.7		163.7		166.0				76.0		92.1		59.9		125.2		
Central Dogger 54 30.00 N 02 42.53 E	7	124.1				132.8				112.3		94.0		63.0		80.7		
Off humber 54 03.92 N 01 47.46 E	8					260.9				107.4		148.1		47.4		112.8		
Outer Humber 53 19.37 N 00 25.47 E	9			266.7		270.2												CFI
Indefatigable Bank 53 33.40 N 02 04.92 E	10			155.6						155.2				112.3		115.2		
Outer Gabbard 52 01.86 N 02 06.57 E	11			194.1								114.4		109.8		169.9		
Rye Bay 50 46.74 N 00 46.83 E	12			186.1		211.7				103.4		97.7		10.8		98.1		
Off Newhaven 50 45.59 N 00 00.00 E	13											96.8		78.9		150.3		
Inner Lyme Bay 50 36.86 N 02 55.82 W	14											134.3				120.3		
South Eddystone 50 06.44 N 04 06.06 W	15											131.1				112.7		
West Lundy 51 09.79 N 05 26.67 W	16	108.7										83.4				141.1		
Camarthen Bay 51 32.82 N 04 35.13 W	17									126.2		128.6				113.9		
North Cardigan Bay 52 42.44 N 04 32.29 W	18	156.4				195.6						133.4		41.6		76.7		
South cardigan Bay 52 10.90 N 04 29.87 W	19											168.7				59.4		
Inner Cardigan Bay 52 18.00 N 04 16.35 W	20	102.5		221.6		158.3						121.9				36.5		
Outer Cardigan Bay 52 23.76 N 04 53.72 W	21			197.8		209.2												
Burbi Bight 53 28.24 N 03 20.47 W	22	186.3		307.3		414.0				217.0		410.7				194.1		CFI
Liverpool bay 53 28.32 N 03 41.91 W	23	124.6		182.9		186.4				140.0		127.9				175.2		
Red Warfe Bay 53 22.46 N 04 12.84 W	24	140.3		195.8		291.3				106.6		82.8				80.8		
Morecambe Bay 53 55.31 N 03 23.23 W	25	360.3		306.4		394.3				194.4		255.3		190.0		256.9		CFI
St Bees Head 54 30.71 N 03 47.63 W	26					286.1				158.1		171.8		303.4		297.1		CFI
SE Isle of Man 54 03.36 N 03 52.47 W	27			195.7		242.4				126.6		121.5		63.5		135.1		
Outer Dundrum Bay 54 04.81 N 05 37.29 W	28	77.1		117.4								131.1						

Figure 3.42 Locations in the Clyde (6), Forth (1) and reference sites (4) sampled for plaice and flounder annually since 2001. © Marine Scotland Science.

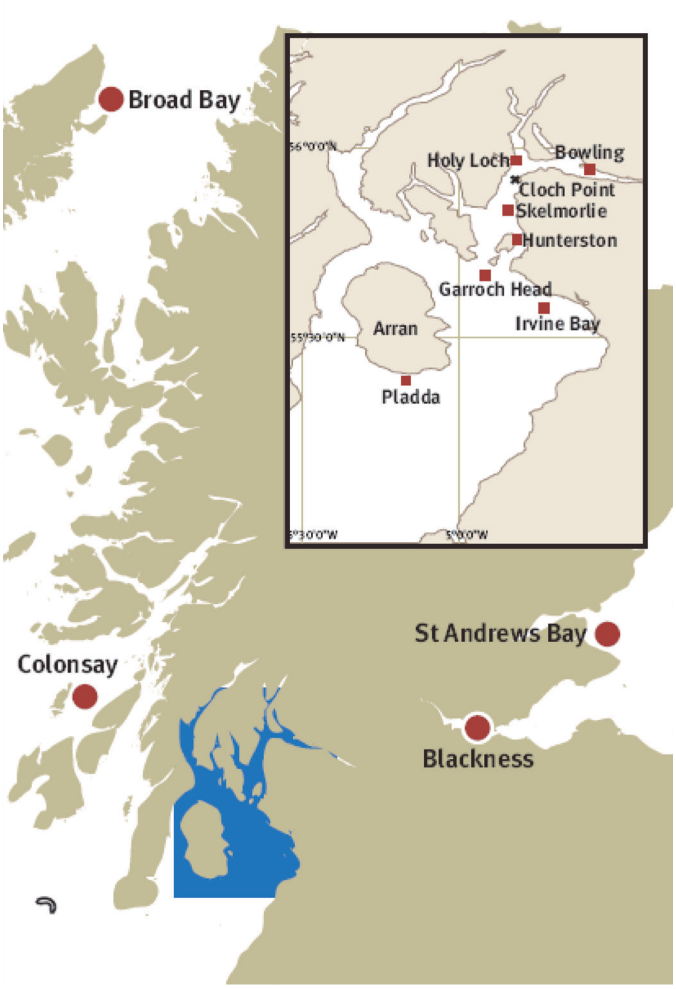


Table 3.35 Response ranges used for classification of EROD activity in plaice and flounder

Criteria	*Assessment calculation of upper range value for each criteria*	*Assessment criteria range; values as pmol/min/ mg protein (LV = lowest value)*
BRR	10th percentile value × 3.3	LV – 17
CBRR	10th percentile value × 10	18 – 52
ERR	10th percentile value × 33	53 – 174
HRR	> 10th percentile value × 33	> 175

Skelmorlie sites, there were no occasions where the response range in any sampling year was above CBRR. The response range at the Garroch Head site was ERR between 2001 and 2005, becoming CBRR in 2006 and 2007.

The response range for flounder indicated an ERR in the upper Firth of Forth in 2001, reducing to CBRR and then BRR between 2004 and 2007. At the upper Clyde (Bowling) site there was no occasion when the response range in any sampling year was above CBRR. The response range at the St Andrews Bay reference site was CBRR in 2001 and 2002, becoming BRR during the remainder of the sampling period.

3.1.4.4.4 Progress towards the vision of clean and safe seas

Based on evidence offshore for dab

The sampling sites used in this monitoring programme provide extensive geographical coverage of offshore areas around the UK, and together with the eight-year sampling period, provide a picture of the status and trend of responses relating to EROD and PAH bile metabolites. As detailed above sites were classified as cause for investigation (CFI) when the EROD response range was at ERR and HRR for 50% or more of the sampling occasions and for bile metabolites, when the response range was at ERR for 50% or more of the sampling occasions. CFI would indicate that the responses are sufficiently elevated to suggest that this may be the result of exposure of the organism to contaminants and that this should be investigated in relation to contaminants in sediment, biota and other biomarker responses in samples from the same sampling locations. For the EROD response, the sites identified for CFI were; three sites close to the NE Coast (Amble, Tees and Off Flamborough – industrialised and densely populated), two

Table 3.36 Mean EROD values (pmol/min/mg) and associated assessment classes.

Background response range - LV up to 17 (pmol/min/mg)
Close to background response range 18-52 (pmol/min/mg)
Elevated response range 53-174 (pmol/min/mg)
High effect response range greater than 175 (pmol/min/mg)

Site Location with Lat. Long. position	*Species*	*2001*	*2001*	*2002*	*2002*	*2003*	*2003*	*2004*	*2004*	*2005*	*2005*	*2006*	*2006*	*2007*	*2007*
Broad Bay 57 17.94N 006 44.46W	Plaice	6.0				13.9		8.5		7.8		6.4		5.3	
Colonsay 56 07.50N 006 04.68W	Plaice	5.7		6.4		14.0		6.2		6.7		14.1		4.7	
Pladda 55 21.54N 005 11.34W	Plaice	3.9						4.2		10.2		7.4		9.4	
Garroch Head 55 40.62N 005 01.26W	Plaice	96.0		91.1		78.2		68.0		60.9		33.0		32.0	
Irvine Bay 55 34.02N 004 46.38W	Plaice	23.1						24.2		20.0				12.5	
Hunterston 55 46.98N 004 52.86W	Plaice	47.9		29.4		26.3		25.2		19.7				19.5	
Skelmorlie 55 51.72N 004 54.36W	Plaice	48.8		38.1		38.0		45.4		43.4				15.8	
Holy Loch 55 59.04N 004 53.04W	Plaice	54.7		54.8		116.6		50.1		39.7		46.5		46.6	
Upper Clyde (Bowling) 55 55.74N 004 29.10W	Flounder	31.7		32.2		49.3		21.3		36.9		22.2		17.8	
Upper Firth of Forth (Blackness/Tancred) 56 00.60N 003 28.80W	Flounder	64.9		24.0				16.4		16.9		5.5		8.3	
St Andrews Bay (East Coast reference) 56 23.04N 002 44.94W	Flounder	24.7		22.4		11.2		15.1		14.9		11.2		10.5	

sites on the western edge of the Dogger Bank (West Dogger and Off Humber), three sites close to the Liverpool Bay coastline (Burbo Bight, Morecambe Bay and St Bees Head) and two historical disposal sites close to the Scottish East coast (St Abb's Head and Bell Rock). It is notable that with the exception of West Dogger and Off Humber that the CFI locations are all close to coastal areas that receive contaminants from industrial and densely populated conurbations. A similar observation can also be made with the bile metabolite biomarker where four sites were classified as CFI; St Bees, Morecambe Bay, Burbo Bight and Outer Humber.

There was no clear visible evidence of any trends in the data over the eight year sampling period but the data have not been subject to statistical analysis for trends.

3.1.4.4.5 Need for further work

The data offshore for dab suggest that there are elevated responses in both EROD and bile metabolite biomarkers and that these may be related to contaminant inputs and this should be investigated by looking at estuarine input data and hydrographic models.

Biomarkers provide evidence of exposure and effect and it is important that the two biomarker responses reported here are used in an integrated assessment with chemical contaminant concentration in sediment and biota and with other higher biological effect responses e.g. apply the OSPAR chemical-biological effect integrated strategy when it becomes available.

There is a clear indication from the inshore data for flounder and plaice that the EROD response measured at Garroch Head and Holy Loch may be related to a decrease in contaminant availability at these sites (i.e. closure of disposal site and submarine base). It would be of value to relate these responses to sediment contaminant data over the same sampling period. Furthermore, as with the offshore data for dab, the EROD responses reported here could be used in an integrated assessment framework with chemical contaminant concentration in sediment and biota and with other higher biological effect responses.

3.1.4.4.6 Overall conclusions and forward look

The inshore data for flounder and plaice show that where changes in the EROD response range have occurred over time, it is a result of decreasing EROD activity. This is particularly apparent at one of the most affected sites in the Clyde, Garroch Head where sewage sludge disposal ceased in 1998 and the evidence from EROD data suggests that exposure of plaice in this area to planar organic contaminants has been decreasing consistently since 2001.

3.1.4.5 DNA adducts

3.1.4.5.1 Overview

DNA adducts have been used extensively in environmental monitoring studies to investigate the genotoxic and carcinogenic risks posed by environmental contaminants. The formation of DNA adducts is considered to be one of the key initiating steps in the chemical carcinogenesis pathway and therefore they are widely used as a biomarker of carcinogen exposure (Reichert et al., 1998). Here we report on the findings of two major UK surveys of DNA adduct prevalence in wild fish species, which were conducted in the past six years. In 2003, a survey of estuarine locations was undertaken using the European flounder (*Platichthys flesus*) and in 2004, an extensive survey of DNA adduct analysis was undertaken as part of the UK CSEMP offshore monitoring programme using dab (*Limanda limanda*) as the sentinel species.

DNA adducts are a CEMP PAH-specific biomarker, as such they should be viewed in relation to other PAH-specific biomarkers such as EROD activity and bile metabolites. When elevated they signify exposure to carcinogenic contaminants. The risk of such exposures should be evaluated using fish liver histopathology.

3.1.4.5.2. DNA adducts as markers of PAH exposure and carcinogenic risk assessment

Polycyclic aromatic hydrocarbons are a ubiquitous and large group of environmental contaminants, some of which are known to cause genetic toxicity through the formation of DNA adducts. Over the past 25 years a growing body of research has investigated the uptake, bioaccumulation and metabolism of PAHs and there is now extensive experimental and field-based evidence supporting their role in the initiation and progression of chemical carcinogenesis through the formation of DNA adducts. Many field studies in both North America and Europe have established a correlation between PAH sediment concentrations and the prevalence of hepatic tumours in fish (Malins et al., 1985; Myers et al., 1991; Baumann, 1998). For example, liver and skin neoplasia in brown bullheads (*Ictaluvus nebulosus*) from the Black River, Ohio (USA) have been shown to be strongly correlated with PAH sediment contamination (Baumann, 1998). Further work carried out in Puget Sound (USA) has also found positive correlations between hepatic lesions including neoplasia (hepatocellular carcinomas and cholangiocellular carcinomas) and foci of cellular alteration (pre-neoplastic lesions) in English sole (*Parophrys vetulus*) and sediment PAH contamination (Malins et al., 1985). Therefore, the measurement of DNA adduct levels in marine organisms is an important step in assessing risk from exposure to environmental carcinogens. DNA adducts are included in the CEMP as a PAH-specific biomarker.

DNA adducts can be removed by cellular repair processes or by cell death, but during chronic exposures they often reach steady-state concentrations in carcinogen target tissues such as the liver. As a consequence, DNA adducts have several important features which make them suitable as biomarkers of PAH exposure:

1. DNA adducts are a quantifiable measurement of the biologically effective dose of a contaminant reaching a critical cellular target and therefore a useful epidemiological biomarker for detecting exposure to environmental carcinogens.
2. DNA adduct levels integrate multiple toxicokinetic factors such as uptake, metabolism, detoxification, excretion and DNA repair in target tissues, therefore they act to assimilate the response of other PAH-specific exposure biomarkers, such as EROD and bile metabolites.
3. DNA adducts are relatively persistent once formed (may last several months) and therefore provide an assessment of chronic exposure accumulated over many weeks rather than a few days, as afforded by other PAH biomarkers such as EROD induction or the presence of bile metabolites.
4. Studies from North America have shown that risk factors for certain lesions can be generated by correlating the level of DNA damage with lesion occurrence, thus allowing the use of a relatively simple biomarker in predicting risk. Where elevated levels of DNA adducts are detected environmental managers should review the fish disease data for that site as they indicate an increased cancer risk in the exposed population.

3.1.4.5.3 Ecological relevance and validation for use in the field

The field validation of a biomarker of exposure, such as DNA adducts is essential in establishing their credentials when used in routine monitoring programmes. In North America the technique has been widely used (>30 marine and freshwater species) and guidelines for implementation are published in an ICES Times technical document (Reichert et al., 1999). Across the OSPAR maritime area the DNA adduct assay has been used in several biological effects monitoring programmes using a range of indicator species including blue mussels, *Mytilus* spp., perch (*Perca fluviatilis*), dab (*Limanda limanda*), European flounder (*Platichthys flesus*), eelpout (*Zoarces viviparous*) and cod (*Gadus morhua*) (Ericson et al., 1998, 2002; Lyons et al., 1999, 2000; Aas et al., 2003; Akcha et al., 2004; Lyons et al., 2004a,b; Balk et al., 2006). Studies from both North America and Europe have clearly demonstrated that when using non-migratory fish the levels of DNA adducts strongly correlate with the concentration of PAH sediment contamination (Van der Oost et al., 1994; Ericson et al., 1999; Lyons et al., 1999). For example, studies using eel (*Anguilla anguilla*) demonstrated a significant relationship between the level of DNA adducts and PAH contamination of the sediment (Van der Oost et al., 1994). Laboratory studies have demonstrated

that fish exposed to PAHs accumulate hepatic DNA adducts in both a time- and a dose-dependent manner (French et al., 1996). It is known from experimental studies using both fish and shellfish that such DNA adducts may persist for many months once formed and are therefore particularly suited to monitoring chronic exposure to genotoxic contaminants (Stein et al., 1990; French et al., 1996; Harvey and Parry, 1998). Significantly, field-based studies have investigated the relationship between DNA adduct formation and neoplastic liver disease and it has been shown that at certain contaminated sites the prevalence of DNA adducts is associated with the prevalence of toxicopathetic lesions including foci of cellular alteration and neoplasia (for review see Reichert et al., 1998).

Studies from North America and Europe suggest that DNA adduct levels are not markedly influenced by factors such as age, sex, season or dietary status, which are known to confound the interpretation of other biomarkers (e.g. EROD). However, validation of any biomarker, including DNA adducts in a species of interest is essential to ensure against any unforeseen species-specific responses (Reichert et al., 1999). While there is no evidence to suggest that environmental factors such as salinity and temperature significantly affect the formation of DNA adducts these factors should always be considered, as it is known that cellular detoxification systems (e.g. Cyp1A as measured by EROD activity) are influenced by changes in environmental variables (Sleiderink et al., 1995).

3.1.4.5.4 Assessment criteria and significance

It is recognised that setting baseline/background response levels has an important role in integrating biological effect parameters into environmental impact assessments of the marine environment. The general philosophy is that an elevated level of a particular biomarker, when compared with a background response, indicates that a hazardous substance has caused an unintended or undesirable level of biological effect. Therefore, in order to understand and apply DNA adducts as a biomarker of genotoxic exposure it is of fundamental importance to gain information on the natural background levels in non-contaminated organisms. A number of studies have now examined fish collected from pristine areas (as supported by chemical and biomarker analyses) and the typical ^{32}P-postlaballing generated DNA adduct profiles either exhibited no detectable adducts or very faint diagonal radioactive zones (DRZs) (Figure 3.43A), suggesting minimal PAH exposure (Ericson et al., 1998; Reichert et al., 1998; Lyons et al., 2000; Aas, et al., 2003; Balk et al., 2006). In contrast, DNA adduct profiles in fish exposed to a complex mixture of PAHs will form DRZs on the chromatogram (Figure 3.43B), which is a composite of multiple overlapping PAH-DNA adducts. Figure 3.43C shows an example of the benzo[*a*]pyrene-DNA adduct profile produced by exposure to a single PAH compound.

By using such studies it has been possible to define reference locations and through the activities of WKIMON develop background response ranges for DNA adducts (OSPAR WIKIMON, 2008). Using a similar approach to that adopted by the US Environmental Protection Agency we have developed Effects Range (ER) values for DNA adducts, which will be used to assess data (Figure 3.44). The ER-Low (ERL) value is defined as the lower tenth percentile of the effect concentration and the ER-Median (ERM) as the median of the effect concentration.

Figure 3.43 Representative hepatic DNA adducts profiles produced following ^{32}P-postlabelling. (A) DNA adduct profile obtained from a site with a low level of PAH contamination. A faint DRZ is visible, indicating a low level of DNA adducts representative of a clean reference location. (B) DNA adduct profile displaying a clear DRZ of ^{32}P-labelled DNA adducts, indicating that the fish has been exposed to a complex mixture of genotoxins. (C) Positive control consisting of benzo[a]pyrene labelled DNA (115 nucleotides per 108 undamaged nucleotides) run with each batch. Figure adapted from Lyons et al. (2004b). © Crown copyright 2010: permission granted by Cefas.

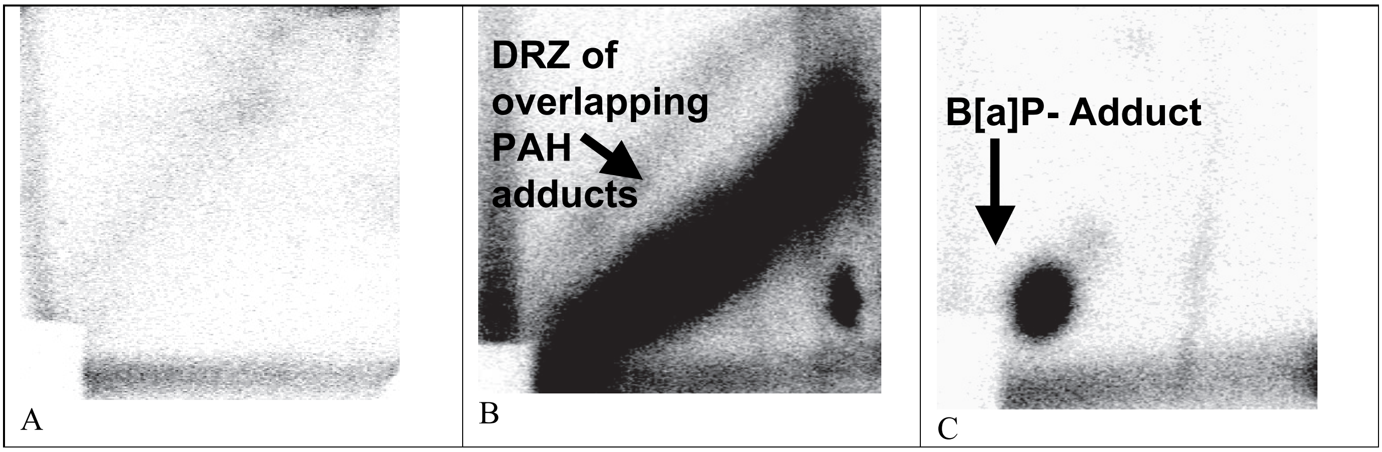
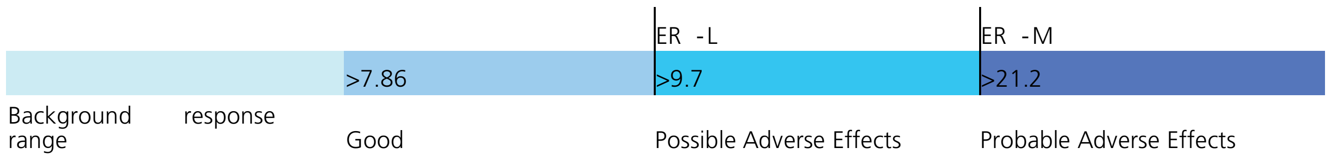


Figure 3.44 Effects Range (ER) values for DNA adducts (adducted nucleotides per 108 normal nucleotides). © Crown copyright 2010: permission granted by Cefas.



3.1.4.5.5 Survey approach and results

Fish were collected in support of the UK's ongoing biological effects monitoring requirements as stipulated under the CEMP for coastal and estuarine waters (OSPAR, 1995).

Dab (offshore). As part of the 2004 CSEMP biological effects cruise, samples of dab (*Limanda limanda*) were collected from 15 offshore locations and liver samples taken for DNA adduct analysis.

Flounder (estuarine). As part of the DMECS (Development of a National Marine Ecotoxicological Analytical Control Scheme) programme samples of European flounder (*Platichthys flesus*) were collected during autumn 2002 from eight UK estuaries and liver samples taken for DNA adduct analysis.

Dab (offshore)

The total hepatic DNA adduct levels from samples of European flounder are shown in Table 3.37. Overall, the levels of DNA adducts detected were similar to those reported previously for dab from offshore locations (Lyons

et al., 2000). In nearly all sites sampled, with the exception of S.E. Isle of Man, the levels detected are considered to be within the background range of concentrations proposed by WIKIMON. Qualitative assessments of the DNA adduct profiles obtained support this with little or no evidence of the presence of PAH-related DNA adducts.

Flounder (estuarine)

The total hepatic DNA adduct levels from samples of European flounder are displayed in Table 3.38. Overall, the levels of DNA adducts detected were similar to those reported previously for European flounder collected from UK estuaries (Lyons et al., 1999). DNA adduct levels varied between the sample sites with the English control site (Alde) and Belfast displaying the lowest levels of carcinogenic exposure. However, the results from Belfast should be treated with caution as only a limited number (n = 5) of fish were obtained from this site.

At the majority of contaminated sites (Southampton, Thames, Clyde, Tyne, Mersey) the predominant DNA adduct profile consisted of DRZs, which is the characteristic profile obtained following exposure to complex mixtures of aromatic and/or hydrophobic genotoxins, such as those formed by PAHs (Figure 3.43B). In contrast, flounder collected from the Forth, Alde and Belfast lacked DRZs with only background levels of DNA damage being observed (Figure 3.43A). European flounder samples collected from Southampton contained the highest levels of DNA adducts. No intra-site related differences in DNA adduct level were detected between male and female fish in this study.

3.1.4.5.6 Discussion

This feeder report addresses the concern that aquatic organisms inhabiting both offshore and estuarine environments are exposed to potentially carcinogenic contaminants. Using dab (offshore) and European flounder (estuarine) as sentinel species we have detected differing levels of DNA adduct levels, which would indicate that depending on location levels of carcinogenic exposure range from background to those where adverse effects are probable.

DNA adducts are biomarkers of carcinogenic exposure (offshore)

The levels of DNA adducts observed offshore at generally at levels, which are considered to be close to background (Table 3.37). This is in agreement with similar studies conducted across Europe which have failed to detect elevated levels of DNA adducts offshore (Aas et al., 2003; Lyons et al., 2000). Apart from the obvious increase in sediment bound PAH concentrations close to disposal sites or offshore oil installations the data provided here indicated that there is little or no carcinogenic risk from PAHs at offshore locations. This is in agreement with observed PAH sediment loadings (Woodhead et al., 1999; Cefas, 2005).

DNA adducts are biomarkers of carcinogenic exposure (estuarine)

The results presented in this report build on previous assessments (Lyons et al., 1999, 2004b) that have highlighted the fact that fish populations in certain industrialized UK estuaries are being exposed to complex mixtures of genotoxic and potentially carcinogenic contaminants. Hepatic DNA adduct profiles characteristic of exposure to complex mixtures of aromatic/hydrophobic genotoxins were detected in European flounder collected from the

Southampton, Thames, Clyde, Mersey and Tyne estuaries (Table 3.38). These findings support previous studies, which have demonstrated that European flounder populations inhabiting industrialized UK estuaries are exposed to high levels of sediment PAH, and that a proportion of the bioavailable PAHs are being metabolised to carcinogenic metabolites (Lyons et al., 1999). The links between detecting DNA adduct profiles, characterized by DRZs, and PAH exposure has been supported by other studies utilizing benthic fish species. For example, similar DNA adduct profiles have been observed in experimental studies exposing English sole (*Pleuronectes vetulus*) to PAH contaminated sediment (French et al., 1996). Of significance to this current study, Bann et al. (1994) demonstrated the formation of hepatic DNA adducts in European flounder kept in mesocosms containing PAH-spiked sediment. Furthermore, their results correlated the levels of DNA adducts with the degree of PAH contamination within each mesocosm and reflected the profile of hepatic pathology reported by Vethaak et al. (1996).

European flounder collected from Southampton contained the highest levels of DNA adducts. The reasons behind the elevated level of DNA adducts at this site are not immediately clear. Previous surveys of sediment PAH contamination (Σ15 PAH dry weight) in UK estuaries have ranked Southampton water (750 μg/kg), behind the Tyne (10 790 to 43 470 μg/kg), Mersey (1811 to 5740 μg/kg) and Thames (597 to 5350 μg/kg) (Woodhead et al., 1999). Furthermore, 1-hydroxypyrene bile metabolite data (Gubbins, unpublished data) collected from the fish used in this current study rank Southampton behind the Mersey, Tyne and Thames in terms of PAH contamination. However, one must consider that in this case the ranking was only based on pyrene metabolites and it is possible that other carcinogenic PAHs (along with other aromatic non-PAH genotoxins) were present at elevated levels at Southampton during the current survey. This assumption is further supported by the work of Thomas and co-workers, who used the mutagenic screening assay Mutatox™ to assess the genotoxic activity associated with sediment samples collected from five UK estuaries, including the Tyne, Thames, Mersey and Southampton (Thomas et al., 2002). In this study the authors identified at least one sediment sample (organic extract) from each estuary that contained potential genotoxins. A bioassay-directed fractionation procedure was then used to identify genotoxins including PAHs, alkyl-substituted PAH, nitro-polycyclic aromatic compounds (PACs), polycyclic aromatic ketones and oxygenated-PACs. There remained a proportion of the extracts in which the potentially genotoxic contaminants could not be identified. It is highly probable that a proportion of these compounds (those not routinely screened for) are also responsible for the overall levels of DNA adducts detected in this current study. Such findings highlight the importance of not restricting monitoring surveys purely to analytical chemistry, which only screen for a limited number of contaminants.

3.1.4.5.7 Key summary points

Offshore, the level of hepatic DNA adducts in dab is generally low and approaching concentrations deemed to be close to background. This suggests that for the sites investigated there is little or no carcinogenic threat posed by PAHs (or related compounds) at offshore locations. This is in agreement with previous assessments undertaken as part of the CEMP.

Table 3.37 DNA adduct levels and assessment of data for dab collected offshore as part of the CSEMP. Levels of DNA adducts expressed as adducted nucleotides per 108 normal nucleotides ± SE.

Location	*CSEMP stn*	*CP2 Region*	*DNA adduct levels*	*Assessment ranking*
St Bees	768	Irish Sea	1.39 ± 0.64	Close to background
S.E. Isle of Man	850	Irish Sea	14.34 ± 8.24	Good
Liverpool Bay	715	Irish Sea	6.24 ± 4.23	Close to background
Burbo Bight	705	Irish Sea	5.61 ± 4.40	Close to background
Red Wharf Bay, Carmarthen	776	Irish Sea	3.67 ± 0.79	Close to background
North Cardigan	NA	Cardigan Bay	3.16 ± 1.52	Close to background
Inner Cardigan	656	Cardigan Bay	1.83 ± 0.67	Close to background
Camarthen Bay	NA	Severn	2.36 ± 0.76	Close to background
Rye Bay	486	Eastern Channel	3.90 ± 1.94	Close to background
Off Humber	346	Humber/Wash	0.82 ± 0.36	Close to background
Central Dogger	287	Humber/Wash	5.51 ± 2.68	Close to background
West Dogger	286	Humber/Wash	5.84 ± 2.47	Close to background
Flambrough	344	Tyne/Tees	4.74 ± 2.40	Close to background
Off Tees	295	Tyne/Tees	3.86 ± 1.23	Close to background
Amble	244	Tyne/Tees	2.28 ± 0.83	Close to background

Table 3.38 DNA adduct levels and assessment of data for European flounder collected in estuarine areas as part of the CSEMP.

Location	*CP2 Region*	*DNA adduct levels*	*Assessment ranking*
Clyde	Clyde	36.2 ± 15.8	Adverse effects probable
Mersey	Irish Sea	17.4 ± 5.9	Adverse effects possible
Belfast	Irish Sea	4.2 ± 0.5	Close to background
Southampton	Eastern Channel	93.9 ± 37.0	Adverse effects probable
Thames	Anglia	51.1 ± 19.2	Adverse effects probable
Alde	Anglia	5.2 ± 1.4	Close to background
Tyne	Tyne/Tees	13.7 ± 7.9	Adverse effects possible
Forth	Forth	9.3 ± 2.7	Good

Certain UK estuaries do contain elevated levels of carcinogenic contaminants and fish residing at these locations are exposed to biologically available carcinogens. Levels of DNA adducts detected appear to be similar to those previously reported, indicating that while concentrations of contaminants do not appear to be increasing there is an ongoing risk of carcinogenic exposure at these locations.

Several sites studied (including Southampton Water, Forth, Clyde) contain levels of adducts at levels where 'probable adverse effects' would be expected. However, at present no routine biological effects monitoring surveys are currently in place for estuarine sites, therefore at present there is a paucity of data (e.g. fish histopathology) from which to establish the consequences of such carcinogenic exposure.

3.1.4.6 Non-specific biological effects

3.1.4.6.1 Oyster embryo bioassay

Introduction

The oyster embryo bioassay has been used for many years to measure the general water quality status of UK marine waters. Embryos of the oyster *Crassostrea gigas* are exposed to discrete water samples for 24 hours and their success in developing to a specific and easily identifiable stage ('D' hinged larvae) provides a measure of general biological water quality. The exposure period, although short, encompasses a period of intense cellular activity during which time a number of critical physiological and biochemical processes may be impaired and cause abnormal growth or development of embryos.

Results of oyster embryo bioassays carried out in marine waters before 1999 indicated that poor water quality was only observed in estuarine waters. Monitoring effort has since been focussed on the assessment of estuaries. *Charting Progress* presented the results of oyster embryo bioassays undertaken on water samples taken from 17 English and Welsh estuaries between 1999 and 2001. The results indicated that biological water quality as measured by oyster embryo-larval development was generally very good (for the sites sampled) but that conclusions could not be drawn regarding the water quality of an entire estuary (owing to the limited number of sites sampled in each estuary), or on trends.

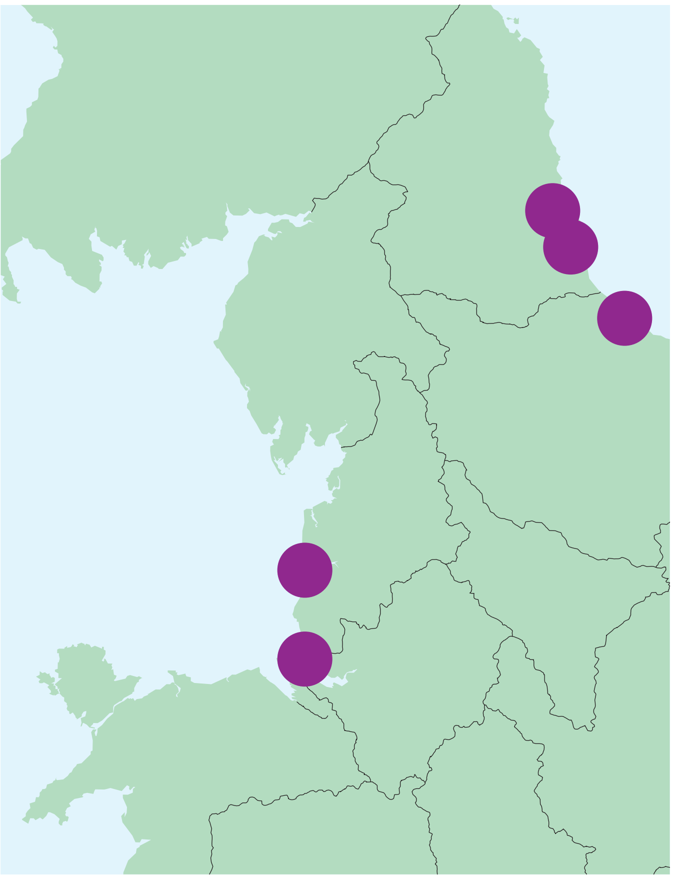
Quarterly analysis of samples taken from one or two sites within each of 12 English and Welsh estuaries continued from 2002 to 2005. In addition, extensive spatial studies (8 to 22 sites) were undertaken on four estuaries during 2004 and 2005.

Results and discussion

The results of estuarine water quality monitoring in English and Welsh estuaries using the oyster embryo bioassay between 2002 and 2005 are shown in Figure 3.45. Each site has been classified according to the mean percentage net response (PNR) achieved for all the samples analysed in each year. The data for 2002 to 2005 indicate that biological water quality as measured using the oyster embryo bioassay remains very good across the English and Welsh estuaries. All the sites sampled were classified as having biological water quality which was either 'close to background' (<10 PNR) or 'good' (10 to 20 PNR) in terms of toxicity to oyster embryos, and no deterioration from the good quality observed in the 1999 to 2001 surveys is evident.

The results of more extensive spatial surveys undertaken in four English estuaries using the oyster embryo bioassay (2004 and 2005) are presented in Figure 3.46. With the exception of the Tees, the results of the more extensive

Figure 3.45 Oyster embryo-larval development; English and Welsh Estuaries 2002-2005. © Crown copyright 2010: permission granted by Cefas.



spatial surveys support the data obtained from the quarterly sampling of these four estuaries. Analysis of samples taken from all sites in the Thames, Medway and Ribble estuaries indicated that biological water quality as measured by the oyster bioassay was 'close to background' (<10 PNR). While biological water quality was indicated to be 'close to background' or 'good' for stretches of the Tees Estuary, some 'hotspots' of significant toxicity were observed. These were generally close to industrial areas with direct discharges of chemicals to the estuary.

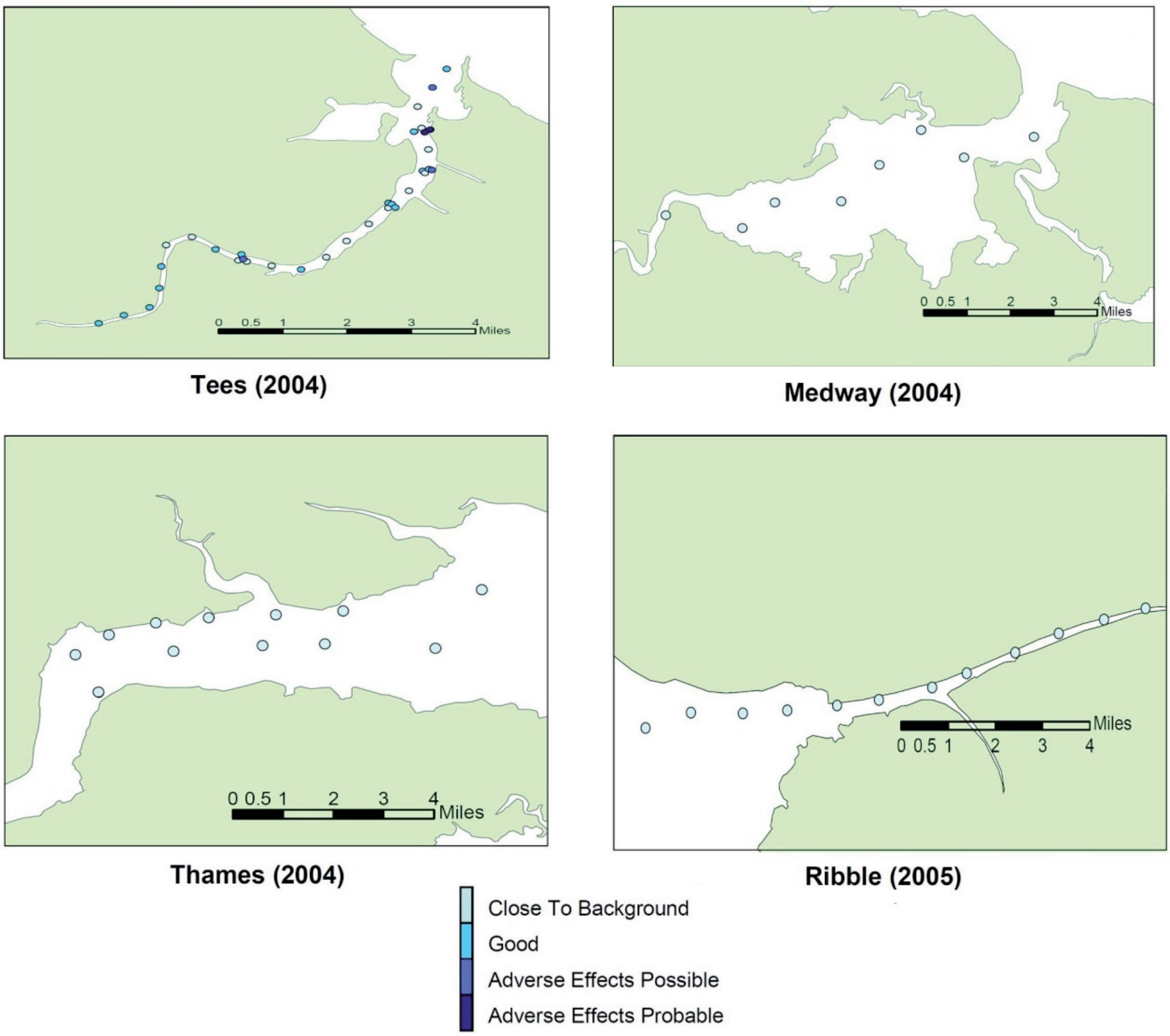
Conclusion and forward look

English and Welsh estuarine biological water quality measured by toxicity to oyster embryo-larval development is generally very good. Poor water quality is evident in parts of the Tees Estuary, particularly in the direct vicinity of known inputs of industrial chemicals.

The more extensive spatial studies employed in the 2004 and 2005 surveys (particularly on the Tees) have demonstrated the limitations of basing estimates of the water quality of an entire estuary on the basis of results obtained for one or two sites, however frequently those sites may be sampled. Water quality issues will often be localised and only by gaining a robust understanding of the spatial extent of toxicity can any temporal trends be adequately investigated.

Future monitoring using the oyster embryo bioassay in English and Welsh estuaries will initially be focussed on completing full spatial surveys of the remaining major estuaries in England and Wales (2006 to 2008). Temporal trend monitoring will then recommence using the spatial surveys as a guide to determining the subsequent sampling frequency and locations for each estuary. Resources are likely to be targeted toward more frequent surveys on estuaries (e.g. in the Tees) where effects remain evident, and with the aim of demonstrating the effectiveness of any actions taken to improve the biological water quality in those areas (e.g. more effective control of effluent discharges using Direct Toxicity Assessments).

Figure 3.46 Oyster embryo-larval development; English Estuaries 2004-2005. © Crown copyright 2010: permission granted by Cefas.



3.1.4.6.2 Sediment bioassays

Introduction

Sediment bioassays measure the toxicity of sediment-bound contaminants to sediment dwelling organisms. Two bioassays have been developed for this purpose using the polychaete *Arenicola marina* and the crustacean *Corophium volutator* (Thain and Bifield, 2001; Thain and Roddie, 2001). Both animals live in the sediment: *A. marina* ingests sediment while *C. volutator* grazes on sediment particles. In both bioassays, the animals are exposed under controlled conditions to collected sediments, and mortality is measured after 10 days. Feeding behaviour is also monitored for *A. marina*.

Charting Progress presented the results of spatial surveys undertaken on four English estuaries in 2001 and 2002. Localised sediment toxicity was recorded in all four estuaries (Mersey, Southampton Water, Tees, Tyne). Between 2003 and 2005, further spatial studies (8 to 22 sites) for sediment toxicity were undertaken on the Dee (2003), Wear (2003), Tees (2004), Thames (2004), Medway (2004) and Ribble (2005) estuaries.

Results and discussion

A summary of the results of spatial surveys undertaken in six English estuaries using the sediment bioassays (2003 to 2005) are presented in Table 3.39. In general, sediment quality across the estuaries surveyed was 'close to background' or 'good'. A few sites in each estuary exhibited some lethal sediment toxicity (20% to 50% effect) to both *C. volutator* and *A. marina*. Sub-lethal measurements for *A. marina* (inhibition of feeding) indicated more widespread effects, most notably in the Wear, Tees and Thames estuaries. The results of the full surveys taken in the Wear and Tees estuaries are shown in Figures 3.47 and 3.48.

While biological sediment quality was indicated to be 'close to background' in the Wear Estuary as measured by lethal effects in the sediment assays, sub-lethal adverse effects (as measured by inhibition of feeding in *A. marina*) were indicated to be possible or probable along the whole of the estuary (Figure 3.47).

Effects on feeding behaviour were also observed on the Tees Estuary (Figure 3.48) although these appeared to be localised in stretches close to industrial areas with direct discharges of chemicals to the estuary. *Corophium* mortality was also observed at levels indicating probable adverse effects in roughly similar parts of the estuary as the sub-lethal *A. marina* effects (although not all sites exhibited effects in both assays).

The Tees represents the only estuary in which any form of temporal assessment of sediment bioassay effects can be attempted. Surveys on the Tees Estuary were carried out in 2001 (reported in *Charting Progress*) and 2004. The 2001 survey was specifically designed to investigate effects in sediments at sites close to known inputs of industrial effluents and indicated that both gross toxicity (mortality) and sub-lethal effects (inhibition of feeding) were evident in these areas. The 2004 survey provided a more extensive spatial investigation of the estuary but suggests that the areas near to industrial outfalls remain the primary concern and this corresponds with results of biological water quality as measured using the oyster embryo bioassay.

Conclusions and forward look

English estuarine biological sediment quality measured using sediment bioassays is generally good. Poor sediment quality is, however, evident in parts of the Tees, Wear and Thames estuaries, particularly in the direct vicinity of known or historical inputs of industrial chemicals. The quality of sediments in the Tees Estuary does not appear to have significantly improved between 2001 and 2004. While it is accepted that any improvement in sediment quality is likely to occur slowly following the reduction in toxicity or number of toxic inputs (e.g. under the Environment Agency's Direct Toxicity Assessment programme), biological water quality at the same sites (measured using the oyster embryo bioassay) would appear to suggest that there remain significant toxicity issues in the stretches receiving direct industrial discharges.

Table 3.39 Sediment bioassay results: English estuaries. Close to background (0% – 10% effect); good (11% – 20% effect); adverse effects possible (21% – 50% effect); adverse effects probable (51% – 100% effect).

Estuary	*Year*	*Bioassay*	*Total number of sites*	*Sites 'Close to Background'*	*Sites 'Good'*	*Sites 'Adverse Effects Possible'*	*Sites 'Adverse Effects Probable'*
Dee	2003	*Corophium* mortality	**21**	**15**	**4**	**2**	**0**
		Arenicola mortality		**12**	**3**	**6**	**0**
		Arenicola inhibition of feeding		**18**	**3**	**0**	**0**
Wear	2003	*Corophium* mortality	**21**	**21**	**0**	**0**	**0**
		Arenicola mortality		**21**	**0**	**0**	**0**
		Arenicola inhibition of feeding		**1**	**2**	**15**	**3**
Tees	2004	*Corophium* mortality	**21**	**12**	**4**	**5**	**0**
		Arenicola mortality		**21**	**0**	**0**	**0**
		Arenicola inhibition of feeding		**9**	**5**	**6**	**1**
Medway	2004	*Corophium* mortality	**8**	**6**	**1**	**1**	**0**
		Arenicola mortality		**8**	**0**	**0**	**0**
		Arenicola inhibition of feeding		**3**	**0**	**3**	**2**
Thames	2004	*Corophium* mortality	**12**	**7**	**5**	**0**	**0**
		Arenicola mortality		**9**	**3**	**0**	**0**
		Arenicola inhibition of feeding		**2**	**1**	**6**	**3**
Ribble	2005	*Corophium* mortality	**12**	**7**	**3**	**2**	**0**
		Arenicola mortality		**12**	**0**	**0**	**0**
		Arenicola inhibition of feeding		**8**	**1**	**2**	**1**

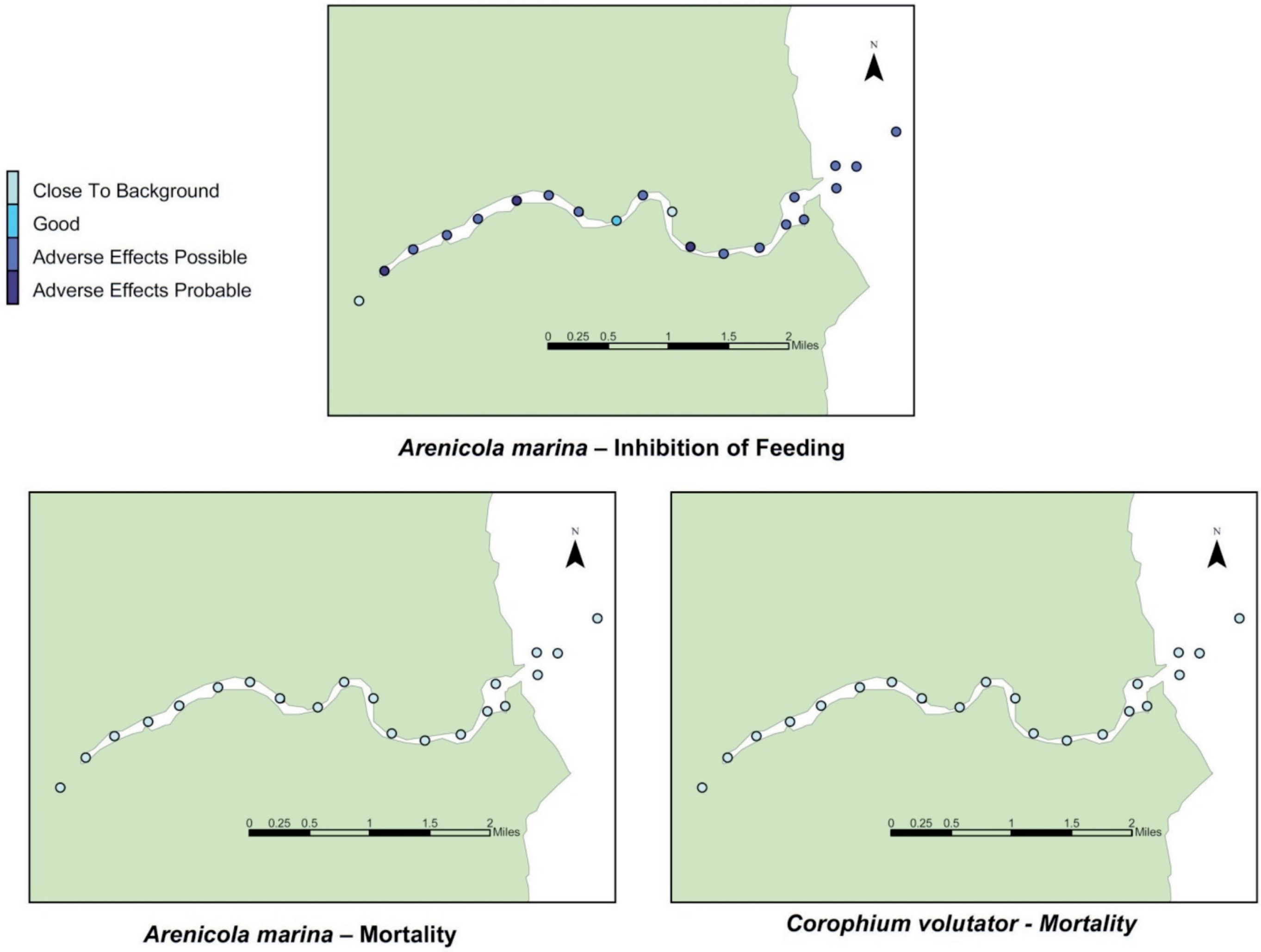
Future monitoring using the sediment bioassays in English and Welsh estuaries will initially be focussed on completing full spatial surveys of the remaining major estuaries in England and Wales (2006 to 2008). Temporal trend monitoring will then commence using the spatial surveys as a guide to determining the subsequent sampling frequency and locations for each estuary. Resources are likely to be targeted toward more frequent surveys on estuaries (e.g. the Tees) where effects remain evident, and with the aim of demonstrating the effectiveness of any actions taken to improve the biological water quality in those areas (e.g. more effective control of effluent discharges using Direct Toxicity Assessments).

3.1.4.6.3 Microarray analysis for biological effects measurements

Introduction

Current biological effects techniques are an assemblage of bioassays, assays for specific inhibition of enzymes, induction of proteins, pollutant metabolites, DNA adducts, physiological responses and pathology. These utilise a wide variety of techniques and

Figure 3.47 Sediment bioassays: Wear Estuary. © Crown copyright 2010: permission granted by Cefas.

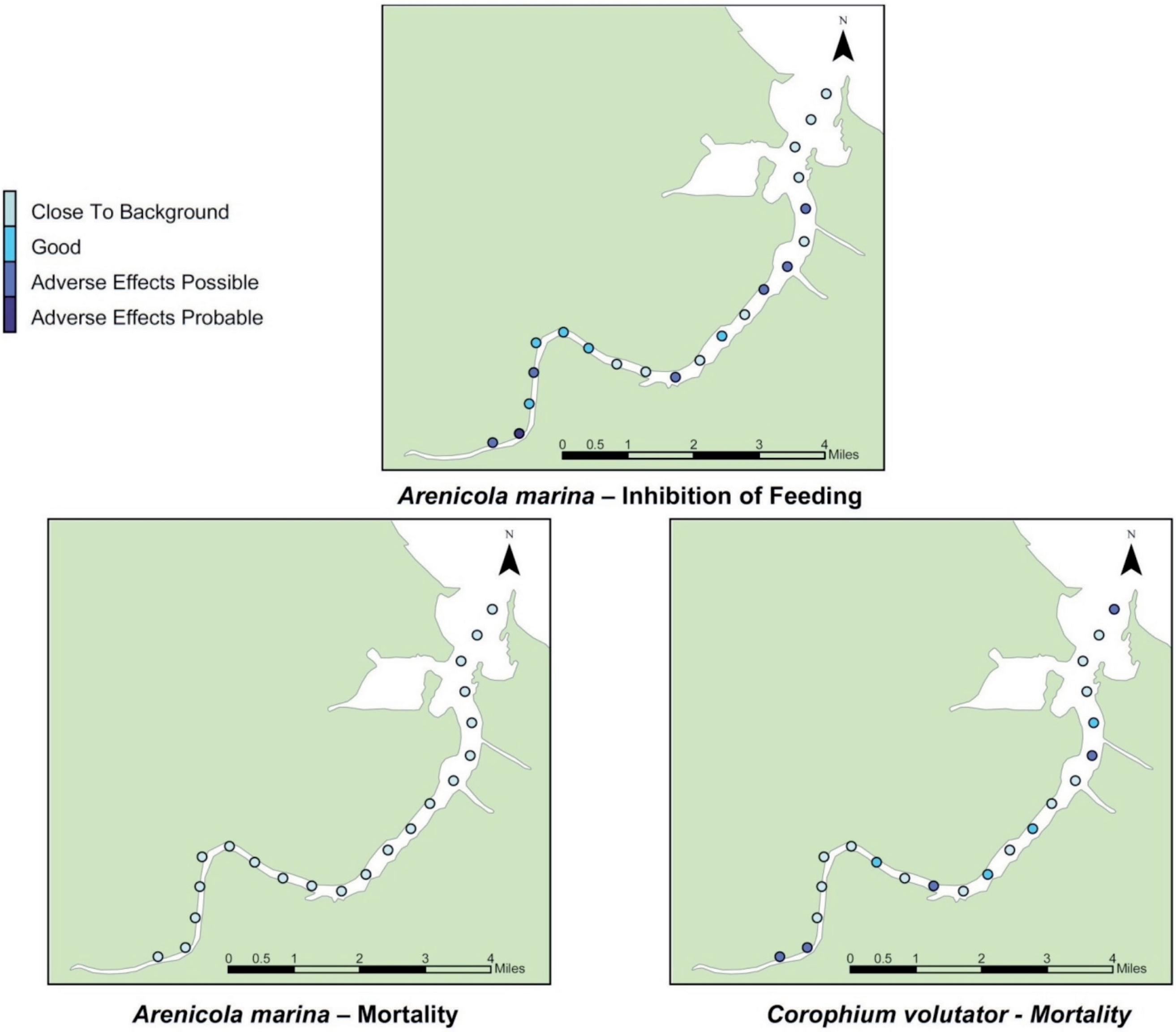


instruments and require staff with expertise in many disciplines. In human disease and pharmacological diagnosis many cannot be conducted because they are too invasive and also contradict the 3R's policy, thus many of those tests have been replaced by the application of 'omics' techniques, including genomics (transcriptomics and population genetics), metabolomics and proteomics. All of these techniques are currently being investigated in an integrated ecotoxicology programme using flounder and stickleback.

Use of transcriptomic analysis (microarrays) for pollutant impact assessment

Changes in cellular steady state resulting in a deterioration in health of the organism are invariably accompanied by changes in gene expression, thus determination of these changes can be used to diagnose sub-lethal changes in cellular physiology, and other adverse health effects such as changes in metabolism, endocrine disruption, carcinogenesis, mutagenesis and stimulation of unscheduled cell death. Depending on the degree of diagnostic comprehensivity required, two techniques for

Figure 3.48. Sediment bioassays: Tees Estuary. © Crown copyright 2010: permission granted by Cefas.



determining mRNA expression are available. The first, using a defined set of diagnostic genes utilises a PCR array, while the second utilizes a high density DNA microarray. Both have been successfully developed for use in European flounder.

Analysis of whole genome responses via microarray technology, in association with the necessary gene ontology and bioinformatic analyses, offers enormous power for dissecting out gene expression patterns most intimately associated with physiological stress and contaminant toxicity. Thus it offers great potential for both investigative monitoring and site-specific risk assessment. However, such approaches require a strong bioinformatic component to facilitate the complex computational analyses to identifying characteristic 'expression fingerprints' specifically related to toxic responses and to identify the genetic factors that determine susceptibility and resistance of individual fish and populations

in polluted environments. Genomic platforms are particularly suited to the study of organism response following exposure to mixtures of environmental contaminants. Furthermore, as they are an open system (i.e. no prior knowledge of the mechanistic action of the contaminant is required) they are ideally suited to the study of novel or emerging/novel contaminants (e.g. nanomaterials). The ability to study the response of thousands of biomarker genes in one single experiment, rather than the traditional one or two key biomarkers available at present (e.g. EROD and VTG), will greatly enhance our understanding and ability to assess environmental risk. This point is very well illustrated by activities of the PAH-metabolising CYP enzymes (as measured by the EROD assay) which are not only inhibited by various metals and organometallics but whose transcription is also suppressed by oestrogens. Microarrays have already been applied to these synergistic and antagonistic responses and it is likely they will contribute significantly to biological effects research over the next ten years (Sheader et al., 2006; Geoghegan et al., 2008).

Use of transcriptomic analysis for assessment of pollutant responses in flounder

The full suite of current biomarker analyses, including those for PAH- and metal-specific effects (EROD and MT assays respectively) and endocrine disruption (unscheduled vitellogenin synthesis) were carried out on flounders collected during the DEMECS programme. RNA was also extracted from these fish and analysed by PCR and cDNA microarray. Results for three UK sites: the Alde with low pollutant status, two polluted sites in the Tyne, and a polluted site in the Elbe (Germany) are presented as an illustration of the application of these techniques.

The three specific responses: i) to PAHs/PCBs, induction of CYP1A determined by mRNA expression or the catalytic activity of the expressed protein (EROD); ii) to Group 2B heavy metals, induction of metallothionein (MT) mRNA or the [Cd/Zn] protein; and iii) to estrogenic substances, unscheduled induction of the egg protein, vitellogenin mRNA in liver, or the protein in plasma in male fish, were determined in ten male fish from each site using the current BEQUALM accredited techniques for the proteins and by Q-PCR for the mRNAs. The results (Figure 3.49) show that as expected the same response was not observed on an individual fish basis since the kinetics of transcriptional and translational responses are different, both techniques resulted in the same impact assessment.

RNA samples from livers of flounder obtained from the Alde, Tyne and Elbe were also analysed using the GENIPOL high density cDNA microarray containing some 14 000 elements, representing 3336 discrete identified genes. Principal component analysis of genes whose expression differed significantly showed that the transcriptomic profiles of genes expressed in the livers of the fish from each of the sites could be distinguished (Figure 3.50) indicating that there are environmental differences.

A variety of computational approaches were then used to identify diagnostic genes. The best diagnostic set was derived from those genes which were characteristic of acute pollutant responses after laboratory exposure to a set of individual toxicants – a PAH, a PCB mixture (Aroclor 1254), a chlorinated hydrocarbon (lindane), an organic hydroperoxide, cadmium, estrogen, a peroxisomal proliferator or an infective agent – indeed the final model only required the responses of 17 genes to provide site diagnosis. Probes for these were developed

Figure 3.49 Biomarker responses in male flounder from three UK Estuarine sites. Comparison of results obtained by q-PCR for mRNA expression with current TIMES methods for estimation of their protein products. EROD is the activity of the CYP1A protein, note the increased sensitivity of the mRNA determination of interindividual differences. © University of Stirling.

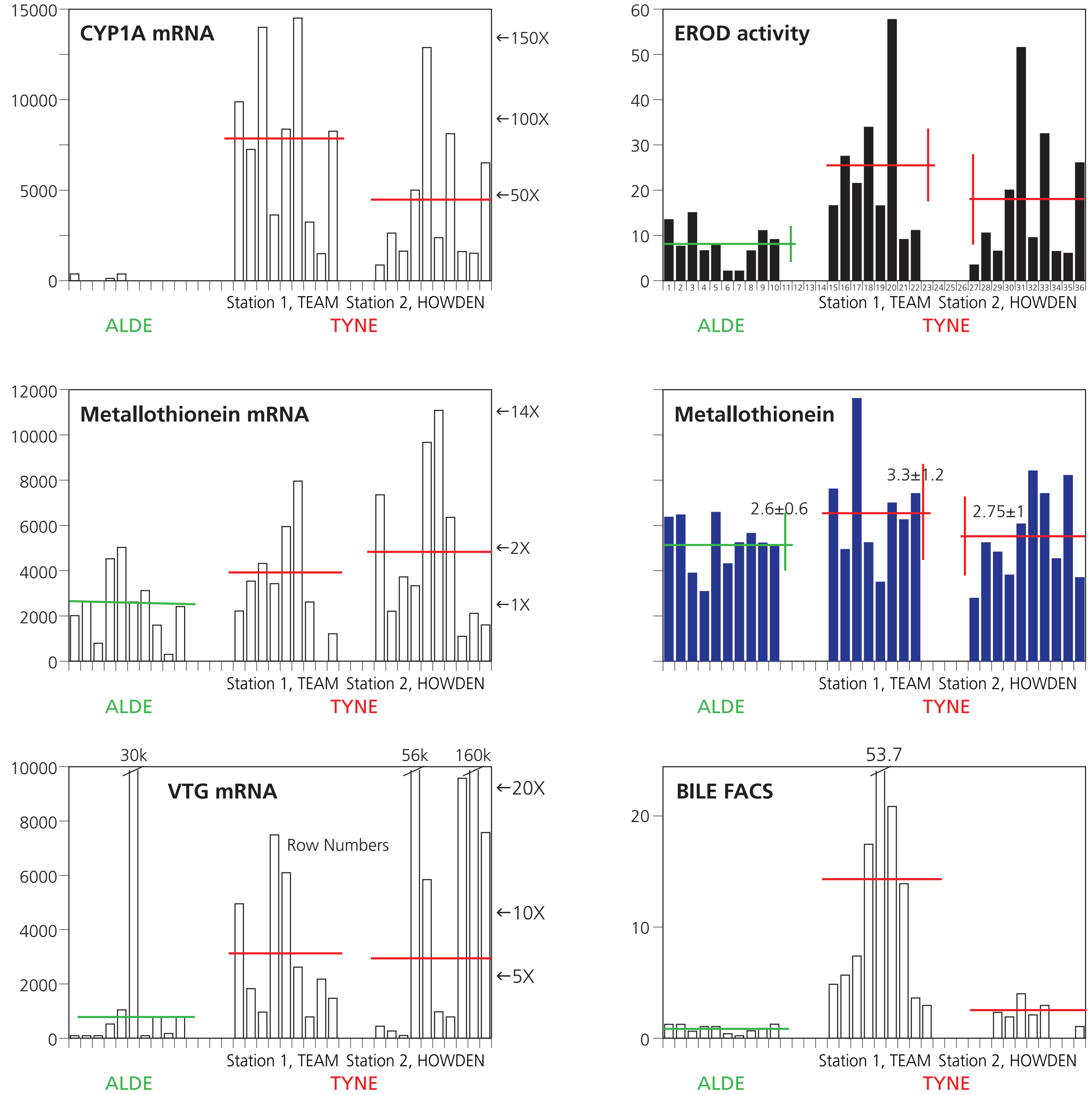
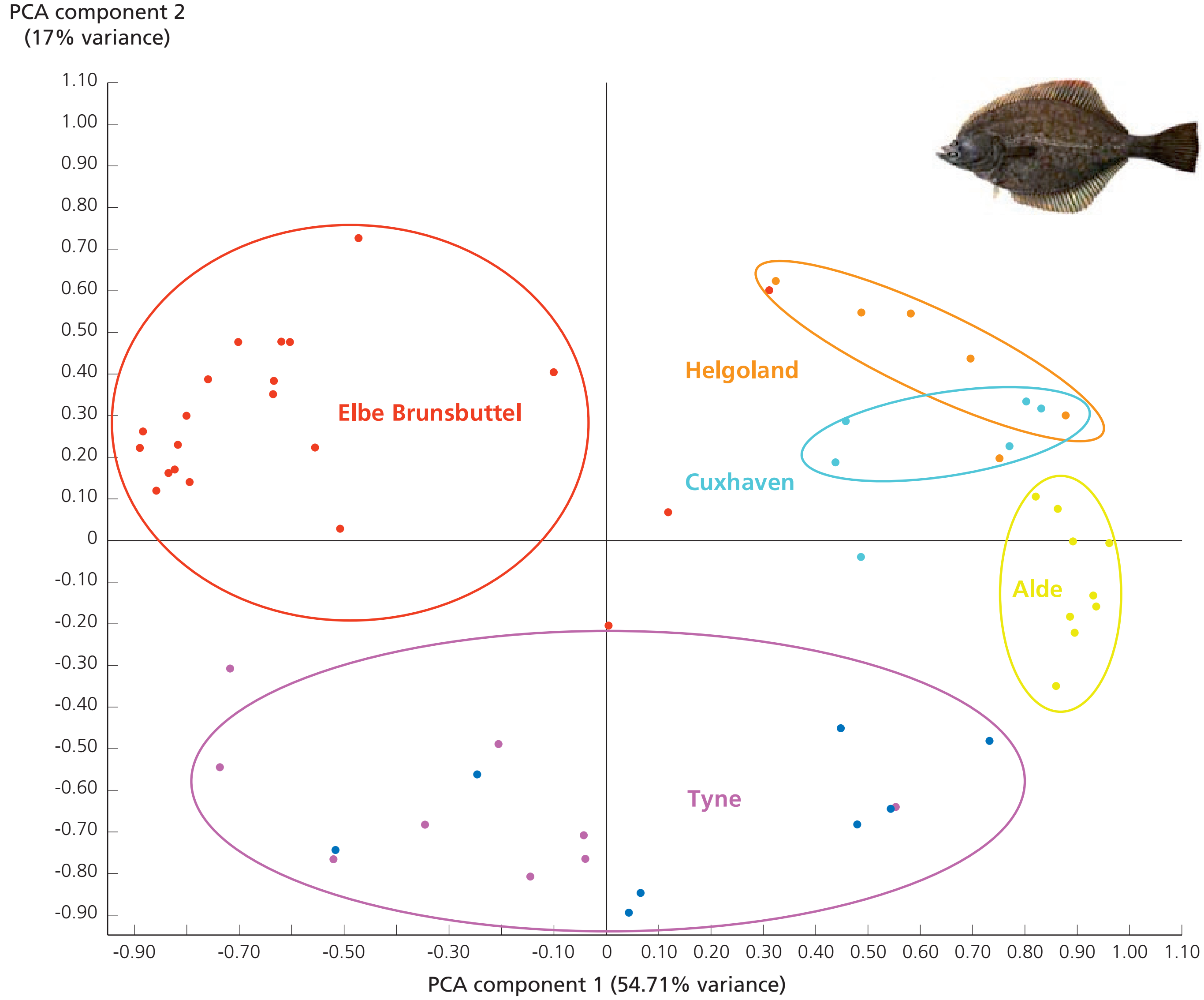


Figure 3.50 Flounder liver microarray analysis. Principal component analysis of top 200 responding genes showing site distinction. © Crown copyright 2010: permission granted by Cefas.



and assembled into a PCR array which was used to analyse an independent set of fish from three of the sites. Cluster analysis of the results (Figure 3.51) showed that the origin of the fish was identified with 100% fidelity. Since the genes utilised are all stress-related genes, they reflect differences in stressors between the sites.

Attempts are currently being made to extend this diagnosis to identify which pollutant classes are responsible for these responses; however, these should always be viewed with caution due to potential pollutant interactions. For example, as previously highlighted it is known that both cadmium and estrogen suppress induction of CYP1A. Analysis of the responses of flounder liver genes after acute laboratory exposure to individual pollutants showed that characteristic profiles are shown for each compound and that these can be distinguished by principle component analysis. For example, Figure 3.52

shows distinction between a PAH (3MC), Aroclor, lindane and a brominated flame retardant mixture (penta-mix PBDE formulation).

Sets of diagnostic genes have been compiled for each treatment and these have been used to assess the fit of responses observed in the field-sampled fish to those observed in the individual single acute pollutant treatments. One such analysis is shown in Figure 3.53 and from the p values of these fits we have made a subjective assignment of status for each chemical class which on simple inspection appears to fit the known chemical status of these sites reasonably well.

These studies with flounder represent an attempt to deploy transcriptomic analysis in a monitoring context in a species whose genome is uncharacterised. The use of cDNA microarrays is more expensive than qPCR arrays, however, it does have the advantage that it is not species specific and the flounder array can be satisfactorily used with all other flatfish species. While the results obtained in these pilot studies are encouraging, it must be stressed that suitable computational methods still have to be developed for analysis of such data to extend its utility beyond health assessment to confident identification of specific contaminant responses.

3.1.4.7 Benthic ecology

3.1.4.7.1 Introduction

Benthic macrofauna are invertebrate organisms that inhabit the seabed. Due to their relatively sessile existence and position within the marine food web they have for many years been used as indicators of state and change of marine ecosystems (Borja et al., 2003, 2007). Benthic macrofauna from soft sediments in which contaminants are known to accumulate (e.g. Woodhead et al., 1999) are monitored to assess relationships between community metrics and levels of contamination.

Changes in benthic indices such as diversity may be caused by natural events such as changes in environmental conditions, for example, salinity and water temperature (Reiss and Kroncke, 2005) and also by anthropogenic impacts which include fishing (Jennings and Kaiser, 1998) and dredge material disposal at sea (Whomersley et al., 2007). Levels of diversity may also be dependent on the ecosystem being studied. Coastal and estuarine habitats are generally found to be lower in diversity than offshore sites due to increased natural and anthropogenic disturbance. Marine habitat types which range from soft mud to coarse gravel and boulder fields will also influence benthic communities.

3.1.4.7.2 Developments since *Charting Progress*

Since the production of *Charting Progress*, much progress has been made toward assessing the health of our regional seas. EU Directives such as the Marine Strategy Framework Directive (2008/56/EC) are putting in place structures and methodologies which will facilitate a consistent approach to marine monitoring and therefore permit the comparison of marine habitats at the regional, national and international scale.

These Directives have necessitated a refocusing of our monitoring programmes, requiring information at greater spatial resolution. To fit with this changing need CSEMP is being redesigned, as previously its focus had been on understanding trends rather than status. During this realignment, data have been collected from stations with greater spatial coverage (Figure 3.54), alongside those from longstanding temporal stations. In conjunction these two approaches will allow the state and relative level

Figure 3.51 Cluster analysis of gene expression profiles for 17 diagnostic genes obtained by qPCR array in flounder from 3 field sites showing correct site assignments. © Crown copyright 2010: permission granted by Cefas.

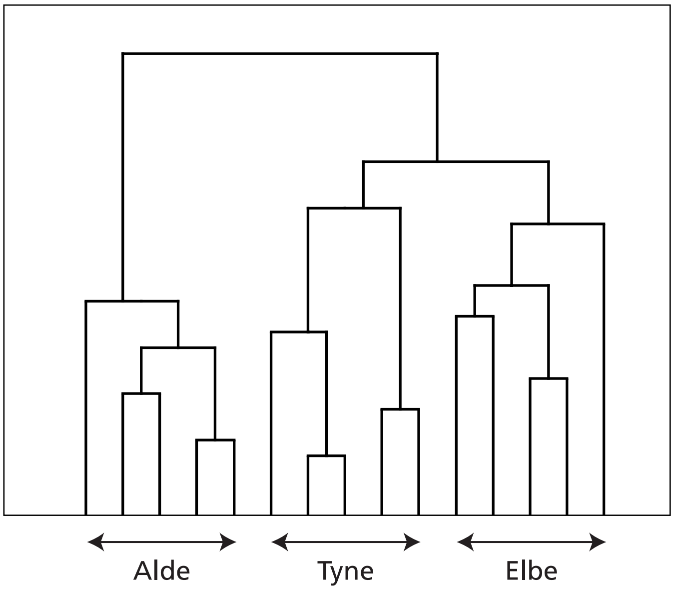


Figure 3.52 Flounder liver microarray analysis of hydrocarbon responses. Principal component analysis of responding genes showing distinction between treatments. © Crown copyright 2010: permission granted by Cefas.

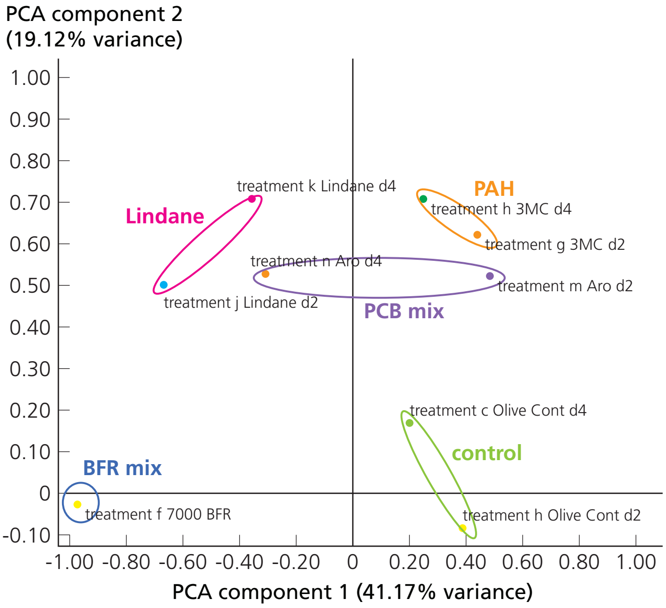


Figure 3.53 Tentative status assessment of UK and German Bight Flounder based upon microarray analysis of wild flounder and acute laboratory responses of flounders to prototypical stressors. © Crown copyright 2010: permission granted by Cefas.

Stressor Class	*Reference compound*	*Alde*	*Tyne*		*Elbe*		*North Sea*
			Howden	*Team*	*Cuxhaven*	*Brunsbuttel*	*Helgoland**
PAH	3-methylcholanthrene		0.06	0.06		0.06	0.25
PCB	AROCHLOR 1254		0.39	0.67	0.13	0.007	0.13
CH	LINDANE		0.14	0.67		0.13	
Peroxisomal proliferator	PFOA		0.09	0.06		0.005	
Heavy metal	Cd		0.005	0.32		0.2	0.13
Estrogenic	E2		0.38	0.07	0.38	0.07	
Infectious disease (viral)	Furunculosis		0.19	0.5	0.05	0.00004	0.01
Chemical Oxidative stress	tBHP			0.15		0.0003	

DIAGNOSIS	background	no response, control	*sampling 2 years after severe floods transported sediments from Elbe
	p>0.01<0.2	detectable response	
	p>0.01 cf. test treatment	significant response	
	p<0.01 cf. test treatment	severe impact	

Figure 3.54 CSEMP redesign sample stations. © Crown copyright 2010: permission granted by Cefas.

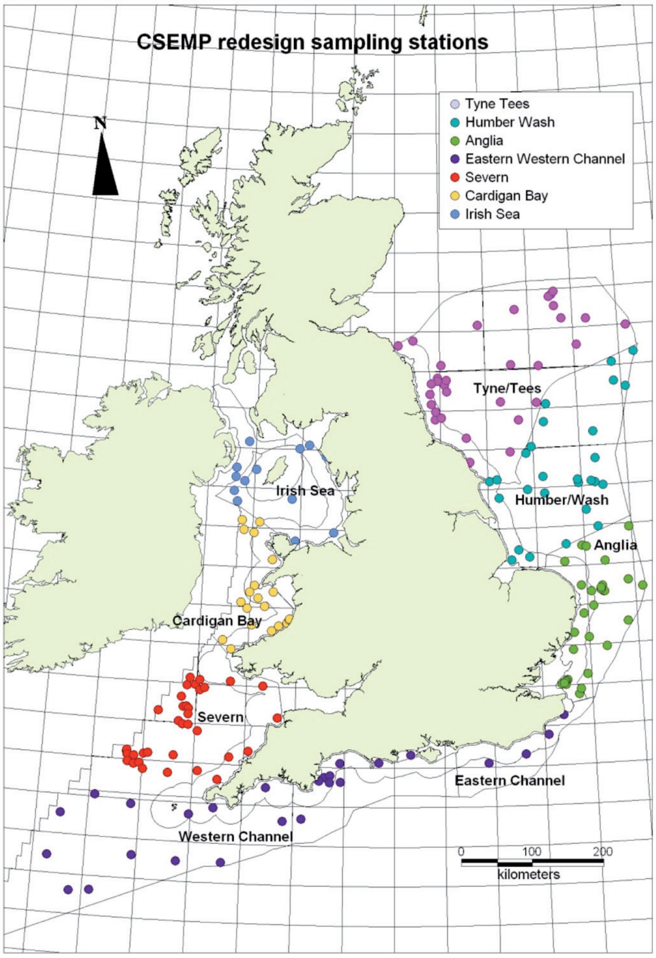
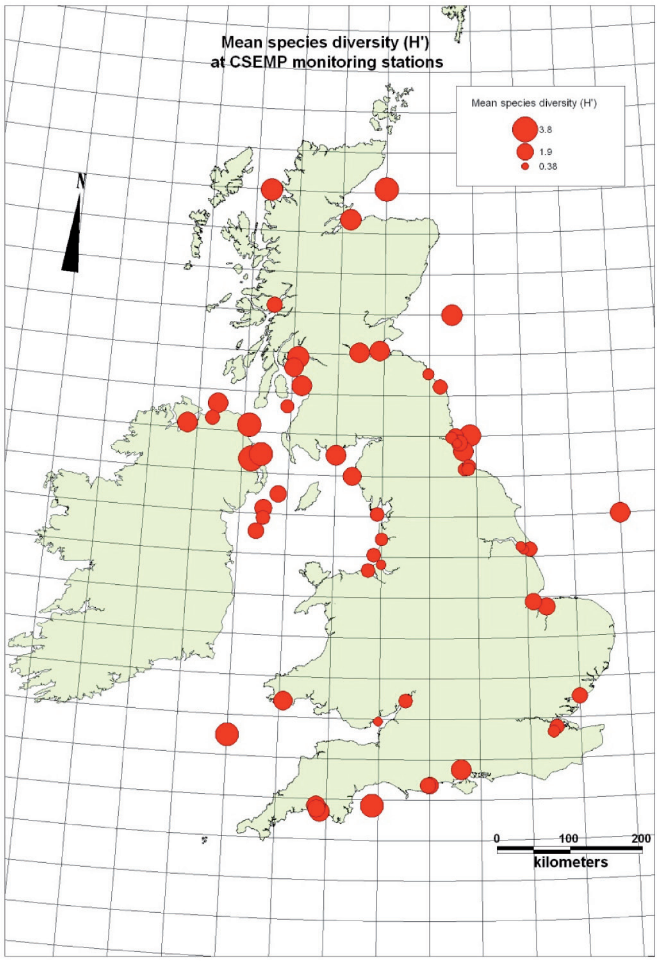


Figure 3.55 Mean species diversity over time at CSEMP temporal trend stations located around the UK. © Crown copyright 2010: permission granted by Cefas.



of change of biological and physical parameters within our regional seas to be monitored and assessed.

3.1.4.7.3 Presentation of the evidence

Results from the analysis of data collected from CSEMP monitoring stations around the UK (Figure 3.55 and Table 3.40) demonstrate the diversity (Shannon-Wiener H') and the temporal variability of benthic communities in soft sediments around the UK coast. Levels of species diversity observed are what would be expected from the soft sediment coastal and offshore ecosystems around the UK coast.

The results showed that the highest levels of diversity observed as expected were primarily at offshore stations and that inshore stations generally demonstrated lower diversity. Species diversity was also found to be variable over time, highlighting the importance of using time series data during the assessment of benthic communities as well as other biological and physical parameters.

A comparison of species diversity values derived from time series data spanning six years from CSEMP monitoring stations situated away from any known point source impacts and

Table 3.40 Mean benthic species diversity from CSEMP stations located around the UK.

Station	Time series	Latitude	Longitude	Mean species diversity	Standard deviation	Minimum species diversity	Maximum species diversity	Range of species diversity
TyneTees_TTInter_Se01	1999-2005	55.008	-1.133	2.888	0.137	2.648	3.010	0.363
HumWash_HWOpenSeaS_Se01	1999-2005	54.000	2.000	2.456	0.337	2.004	2.999	0.995
WestChan_WCInterE_SE01	1999-2005	50.430	-3.122	3.264	0.155	3.070	3.472	0.402
Severn_SeOpenSeaW_Se01	1999-2005	51.250	-6.000	3.266	0.236	3.040	3.675	0.635
IrishSea_BelfastLoughInnr_se01	1999-2007	54.668	-5.807	3.777	0.364	3.187	4.322	1.135
IrishSea_BelfastLoughOutr_se01	1999-2007	54.733	-5.600	3.184	0.226	2.611	3.933	1.322
MinchMalin_BannEstuary_se01	1999-2007	55.149	-6.685	1.455	0.405	1.229	1.735	0.506
MinchMalin_FoyleFaughanE_se01	1999-2007	55.067	-7.217	2.444	0.079	2.110	3.531	1.421
IrSGyreIntermed_se02	1999-2007	54.067	-5.500	2.010	0.717	1.198	3.019	1.821
IrSGyreIntermed_se03	1999-2007	54.250	-5.200	1.861	0.428	1.226	2.552	1.326
IrSGyreOpenSea_se01	1999-2007	53.950	-5.500	1.371	0.583	0.594	2.338	1.744
IrSIntermediateW_se01	1999-2007	55.083	-5.885	3.442	0.475	2.376	3.972	1.596
MMSIntermediate_se01	1999-2007	55.333	-6.583	2.498	0.267	2.096	2.806	0.710
SISOpenSea_se01	1999-2007	53.783	-5.633	1.679	0.488	1.024	2.402	1.378
TyneTees_Tweed_se01	1999-2004	55.770	-2.026	0.937	0.137	0.815	1.138	0.323
TyneTees_HolyBudl_se02	2001-2004	55.611	-1.768	1.456	0.266	1.164	1.792	0.628
TyneTees_Tyne_se02	1999-2005	54.985	-1.527	1.016	0.198	0.734	1.288	0.552
TyneTees_Tyne_se03	1999-2005	54.999	-1.441	1.736	0.548	1.157	2.401	1.244
TyneTees_Wear_se03	2002-2004	54.914	-1.407	0.670	0.301	0.322	0.856	0.533
TyneTees_TTInter_se03	1999-2005	54.816	-1.278	2.458	0.228	2.152	2.657	0.505
TyneTees_Wear_se04	2000-2005	54.917	-1.364	1.765	0.769	0.986	2.524	1.538
TyneTees_Tees_se01	1999-2004	54.592	-1.252	1.249	0.189	0.934	1.382	0.448
TyneTees_Tees_se03	1999-2005	54.595	-1.181	1.129	0.349	0.639	1.473	0.834
TyneTees_Tees_se02	1999-2005	54.630	-1.163	1.264	0.477	0.318	1.568	1.250
HumWash_Humber_se02	1999-2002	53.591	0.083	1.494	0.160	1.318	1.697	0.380
HumWash_Humber_se03	1999-2002	53.586	-0.043	0.725	0.481	0.317	1.407	1.089
HumWash_Humber_se04	1999-2002	53.626	-0.104	0.727	0.392	0.729	1.042	0.313
HumWash_Owash_se02	1999-2004	52.942	0.127	1.896	0.122	1.765	2.048	0.283
HumWash_Owash_se03	1999-2001	52.875	0.391	1.946	0.256	1.791	2.242	0.451
Anglia_Oblackwat_se01	1999-2004	51.761	0.997	1.653	0.412	1.059	2.180	1.121
EastChan_SouthanWa_se01	1999-2004	50.876	-1.380	2.598	0.227	2.225	2.836	0.611
Anglia_Medway_se01	1999-2004	51.334	0.455	0.984	0.356	0.392	1.345	0.953
Anglia_Medway_se02	1999-2003	51.389	0.521	1.484	0.138	1.332	1.672	0.340

Station	*Time series*	*Latitude*	*Longitude*	*Mean species diversity*	*Standard deviation*	*Minimum species diversity*	*Maximum species diversity*	*Range of species diversity*
WestChan_PlymSo_se02	1999-2004	50.423	-4.207	2.322	0.242	2.107	2.712	0.604
WestChan_PlymSo_se03	1999-2003	50.384	-4.197	2.045	0.198	1.823	2.254	0.430
EastChan_PooleHar_se01	1999-2004	50.686	-1.990	1.919	0.189	1.635	2.145	0.511
EastChan_PooleHar_se02	1999-2003	50.685	-2.030	1.571	0.182	1.362	1.779	0.417
WestChan_PlymSo_se04	1999-2004	50.349	-4.130	2.572	0.193	2.354	2.902	0.547
Severn_Severn_se01	1999-2001	51.728	-2.476	1.303	0.051	1.290	1.358	0.068
Severn_Severn_SE03	1999-2001	51.471	-3.024	0.651	0.453	0.164	1.060	0.896
Severn_MilfordHav_se01	1999-2004	51.702	-4.919	2.235	0.477	1.858	2.907	1.049
IrishSea_DeeNWales_se03	1999-2001	53.337	-3.275	1.341	0.110	1.214	1.416	0.202
IrishSea_Mersey_se02	1999-2001	53.410	-3.009	0.730	0.330	0.459	1.098	0.639
IrishSea_LiverpoolBay_se02	1999-2004	53.527	-3.161	1.391	0.537	0.614	1.954	1.340
IrishSea_Ribble_se01	1999-2004	53.727	-3.000	1.104	0.169	0.929	1.380	0.451
IrishSea_MorecambeBay_se02	1999-2004	54.033	-3.100	1.408	0.601	0.543	2.000	1.457
IrishSea_Cumbria_se01	2000-2004	54.500	-3.650	2.209	0.237	1.951	2.515	0.564
Clyde_FirthCInnerCumbraes_se01	1999-2004	55.822	-4.978	2.256	0.277	2.024	2.513	0.490
Clyde_FirthCInnerDunoon_se01	1999-2004	55.948	-4.894	2.982	0.243	2.773	3.344	0.571
Clyde_IrvineBay_se01	1999-2004	55.599	-4.790	2.545	0.195	2.238	2.864	0.626
Forth_FirthFInnerOffshore_se01	1999-2004	56.050	-3.100	2.466	0.500	1.763	2.905	1.142
MinchMalin_LochLinnheS_se01	1999-2004	56.580	-5.472	1.660	0.225	1.328	1.881	0.553
Forth_LowerForthEstuary_se01	2000-2004	56.025	-3.543	2.399	0.563	1.420	2.799	1.379
Clyde_FirthCOuterOffshore_se01	2001-2004	55.333	-5.083	1.187	0.409	0.895	1.346	0.451
EScotland_EScOpenSea_se01	2001-2005	56.500	-1.500	2.783	0.240	2.352	2.883	0.530
IrishSea_BalcaryPoint_se01	2001-2004	54.750	-4.000	2.611	0.249	2.048	2.881	0.833
MinchMalin_TheMinchNorth_se02	2001-2006	58.000	-5.667	2.993	0.307	2.396	3.423	1.027
MorayF_MoFOpenSea_se01	2001-2005	58.050	-3.000	3.391	0.226	2.929	3.740	0.811
MorayF_WhitenessHead_se01	2001-2005	57.667	-3.817	2.766	0.262	2.415	3.011	0.595

stations situated close to, and within disturbed areas such as dredge disposal sites, revealed disturbed stations within dredge disposal sites to be more diverse than monitoring stations (Figure 3.56). However, analysis of community data (Figure 3.57) revealed that CSEMP temporal monitoring stations displayed greater stability over time when compared with stations within and in the vicinity of disturbed areas. The greater diversity found within the disposal site may be attributed to the disturbance regime and also the increased niche availability from disposed sediments which may differ from the *in situ* seabed sediments.

Figure 3.56 Mean (± 95% Confidence Interval) species diversity (H') over time at CSEMP and impacted sites within CP2 sea areas. © Crown copyright 2010: permission granted by Cefas.

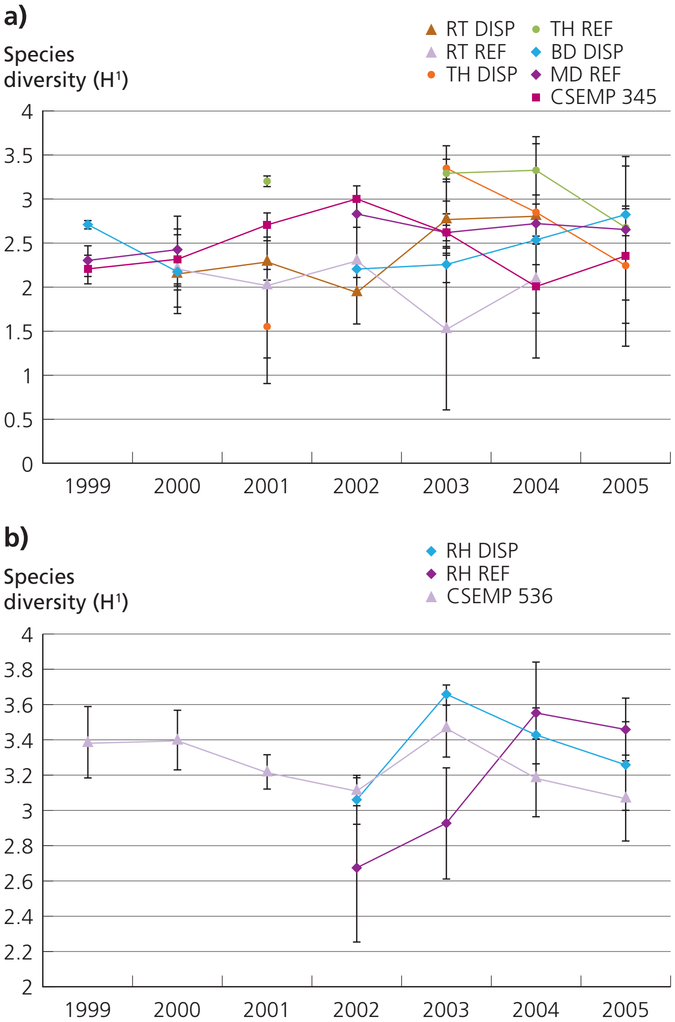
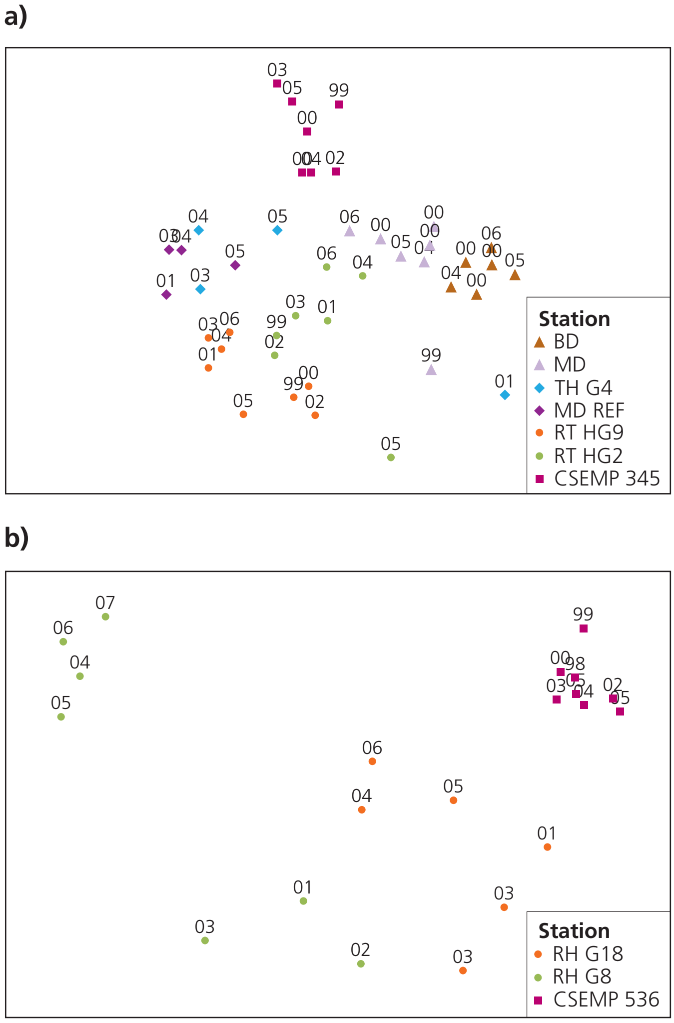


Figure 3.57 Multi-Dimensional Scaling ordinations of time-series data at CSEMP stations and impacted sites within Charting Progress 2 sea areas. © Crown copyright 2010: permission granted by Cefas.



3.1.4.7.4 Progress towards the vision of clean and safe seas

The ability to assess changes in diversity over time using temporal data sets collected as part of CSEMP and other monitoring programmes, such as the North Sea Benthos survey (Rees et al., 2007), permit levels of diversity to be observed over protracted time periods and also over large geographic areas. The results presented here demonstrate the difficulties in assigning a value to a parameter (species diversity) which is constantly changing. However, the opportunity to observe these changes over time could increase our ability to set levels which are accurate and sustainable. However, the ability to set accurate levels of diversity in terms of benthic communities are very dependent on the location of the sample stations (e.g. coastal or offshore) and also the habitat type (e.g. soft mud to rock and boulder reef). The fact that disturbed areas of the seabed may demonstrate higher diversity than natural levels must also be considered if realistic levels of diversity are to be set and monitored.

3.1.4.7.5. Need for further work

The continued analysis of samples collected under the auspices of the CSEMP redesign programme will permit a more extensive spatial assessment of marine soft sediment diversity within our regional seas to be carried out. It is clear from Figures 3.54 and 3.55 that a greater spatial assessment of our regional seas would be beneficial and allow the spatial variability of parameters such as species diversity to be studied. This increased spatial approach must also take into account other habitat types, which include gravels and rocky reef. This will enable a full assessment of the diversity of benthic communities in our regional seas to be carried out. The continued monitoring of CSEMP temporal stations is also needed to allow the temporal stability/fluctuations of benthic communities to be assessed. The results from both analyses will form important contributions to the classification of our regional seas under the EU Marine Strategy Framework Directive.

3.1.4.7.6 Overall conclusion and forward look

Results from the analysis of data collected from CSEMP monitoring stations around the UK demonstrate the diversity (Shannon-Wiener H’) and the temporal variability of soft sediment benthic communities around the UK coast. Levels of species diversity observed are what we would expect to observe from coastal and offshore soft-bottomed marine benthic habitats. Monitoring programmes such as CSEMP which assess benthic communities around the UK coast provide a unique opportunity to examine the environmental status of our regional seas at both a spatial and temporal scale. Without these and other monitoring programmes the assessment of benthic community variability within our regional seas would be far more limited. These basic analyses demonstrate a very simple method by which actual levels (minimum, maximum and ranges) of diversity could be documented and monitored. This kind of information and analyses will be essential during assessment of the environmental status of our regional seas required under the Marine Strategy Framework Directive. These and other data from marine surveillance programmes which monitor impacts in the marine environment also provide the opportunity to study other aspects of environmental change, such as climate change (IPCC, 2007), the introduction of alien species (Occhipinti-Ambogi, 2007) and anthropogenic impacts (Rees et al,. 2006).

Appendix 1: Legislation

European legislation and regulation

The EU Water Framework Directive (2000/60 EEC), came into force in 2000 and requires organic contaminants to be monitored as part of the chemical quality assessment for Priority Hazardous Substances and Priority Substances. These contaminants are those considered to be of most concern within the European Community. The requirement is for waters to achieve good environmental status (using a combination of ecological and chemical assessments). Good chemical status is defined in terms of compliance with all the quality standards established for chemical substances at European level. Thus, chemical status will either pass or fail the standards. Chemical status does not directly affect ecological status but a body of water cannot be classified as good unless both the ecology and chemistry are at least 'good'. We can say that 'No body of water can achieve good status if it does not pass these minimum water quality standards'.

Organics (and other contaminants) must be monitored in transitional and coastal waters (out to 3 nautical miles in Scotland, 1 nautical mile elsewhere in the UK). This surveillance programme establishes a network of waterbodies, where the data collected can be used to validate risk assessments (i.e. risk of waterbody status being affected by discharges) and provide long-term monitoring. This chemical monitoring programme is not fully established, since EQS values have yet to be confirmed for some contaminants and the list of contaminants has not been finalised.

There have been recent initiatives regarding mercury (the EU Mercury Strategy, EU restrictions on the sale of mercury in instruments, EU export ban, UNEP agreement on a Globally Legally Binding Instrument).

The Dangerous Substances Directive (76/464/EEC) and its daughter Directives control discharges into the environment that are liable to contain dangerous substances. The aim is to prevent adverse effects on the environment. Substances are divided into List 1 and List 2 substances.

List 1 (may also be known as Black and Red list) substances are those which are particularly toxic, persistent and which may tend to accumulate in the environment, such as mercury. The Directive instructs us to eliminate discharges of List 1 substances.

List 2 (may also be known as the Grey list) substances are those with the potential for a lower level of harm, such as copper. The Directive instructs us to reduce the input of List 2 substances into the water environment.

All waterbodies must be sampled that receive a direct discharge of a dangerous substance which has numeric limits in its consent/permit.

The Shellfish Waters Directive (2006/113/EEC) is concerned with the protection of coastal and estuarine water quality where shellfish and shellfish larvae grow. The Shellfish Directive lays down requirements for the quality of designated waters which support shellfish. Its purpose is to safeguard shellfish populations from the harmful consequences resulting from the discharge of polluting substances into the sea. We must sample all designated shellfish waters.

Although the Dangerous Substances Directive and the Shellfish Waters Directive fall under the remit of the Water Framework Directive, they will not actually be repealed until 2013, after which the Water Framework Directive will provide a minimum standard that meets the requirements of the old Directives.

Commission Regulation (EC) No. 208/2005 sets maximum levels for benzo[*a*]pyrene as a PAH marker compound in seafood. Maximum permitted levels are 2 µg/kg wet weight in fish muscle, 5 µg/kg wet weight in crustaceans and cephalopods, and 10 µg/kg wet weight in bivalve molluscs (EC, 2005). Within the UK, the Food Standards Agency has set pragmatic guideline values for two additional PAHs to be applied following emergency incidents. These guideline values are: 15 µg/kg wet weight for benz[*a*]anthracene and dibenz[*a,h*]anthracene.

The UK is a signatory to ASCOBANS (the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas) www.ascobans.org. ASCOBANS was concluded in 1991 as the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS) under the auspices of the Convention on Migratory Species (CMS or Bonn Convention) and entered into force in 1994. In February 2008, an extension of the agreement area came into force which changed the name to ‘Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas’. Signatory states make efforts to conserve the small cetacean species within the ASCOBANS area, managing threats to their existence, such as by-catch, habitat deterioration and other anthropogenic disturbance.

3.2 Radioactivity

3.2.1 Key points

i. Introduction

Radioactivity in the marine environment arises from both natural and man-made sources, and can be harmful to both humans and other animals. A UK strategy is in place, intended to meet an international commitment for environmental concentrations to be 'close to zero' by 2020. Changes in radioactivity concentrations in the environment continue to be closely monitored through national monitoring programmes and OSPAR periodic reports.

ii. How has the assessment been undertaken?

Environmental concentrations and levels have been used to assess the exposure of both humans and wildlife.

iii. Current status of radioactivity and past trends

- Discharges from Sellafield are at very low levels, relative to their 1970s peak and generally maintain a downward trend. Technetium-99 (^{99}Tc) discharges have been reduced to below 10 TBq per annum, two years ahead of schedule, and environmental concentrations of this nuclide are returning to pre-EARP (Enhanced Actinide Removal Plant) levels.
- Remobilisation of radionuclides from deeper sediment layers into surface sediments and overlying waters is now the principal source of caesium-137 (^{137}Cs) and plutonium in the Irish Sea. Increased concentrations of plutonium-239/240 (^{239,240}Pu) in certain areas of the Irish Sea and Solway Firth suggest redistribution is an emerging factor for this radionuclide.
- Concentrations of polonium-210 (^{210}Po) in seafood around Whitehaven are now generally within the range of natural variability. Polonium-210 is responsible for ~50% of the radiation dose to seafood consumers around Sellafield.
- Significant reductions have been achieved in tritium and carbon-14 (^{14}C) discharges from the radiopharmaceutical plant in Cardiff. Concentrations of these radionuclides in fish and molluscs are decreasing, as is the associated radiation dose, although tritium levels remain higher than elsewhere in coastal waters.
- The offshore oil and gas industry is responsible for a large proportion of the total α-radioactivity discharged into UK waters. Elevated levels of the naturally-occurring nuclides radium-226 (^{226}Ra), radium-228 (^{228}Ra) and lead-210 (^{210}Pb) are found in produced water, which arises with the oil and is discharged to sea.
- Radioactive particles have been detected on beaches around Sellafield and Dounreay. Monitoring programmes are in place at both of these sites to retrieve contaminated particles from the foreshores. At Dounreay, an offshore programme of particle recovery is underway.
- Radiation exposures from authorised releases to the most-exposed groups of people around nuclear licensed sites remain generally low, and are well within the UK and EU annual dose limit of 1 mSv. The highest annual doses from a single source of man-made radionuclides are around 0.2 mSv at Sellafield. Most of this dose is due to the legacy of earlier discharges.

- In a one-off assessment, dose rates in aquatic systems were found to be less than the threshold of 40 µGy/hr in all cases except near the Springfields nuclear fuel manufacturing site in Lancashire. In this case, new discharge limits were put in place to ensure that the dose rates would reduce to below the threshold.

iv. What has driven change?

An overall thrust to reduce environmental levels of radioactivity and the tightening of legislation has driven change.

v. What are the uncertainties?

There is a substantial programme of environmental monitoring for radioactivity which has been established for a number of years. The annual results of this programme, together with implications for human health and environmental health are published annually in the Radioactivity in Food and the Environment (RIFE) report series. These reports confirm that the environment and man are adequately protected and that the uncertainty in dose estimations is relatively low.

vi. Forward look

The aim is to comply with the OSPAR objective of environmental concentrations ‘close to zero’ by 2020. A revised Strategy for 2006–2030 was published in July 2009 and extends the scope of UK Government policy to include clear targets for radioactive discharges from decommissioning activities and from the non-nuclear sector.

3.2.2 Introduction

Radioactivity in the marine environment is the result of an array of natural and anthropogenic processes. Natural radiation, from the decay of primordial radionuclides in the earth’s crust and from cosmic radiation interacting with the atmosphere, is ubiquitous in the marine environment (e.g. ^{14}C; potassium-40, ^{40}K; and the uranium- and thorium-decay series). The anthropogenic input, including artificial radionuclides such as ^{137}Cs and plutonium isotopes, can be broadly classified as resulting from one of four processes: fallout from weapons testing; nuclear accidents; (historic) offshore dumping of solid and liquid waste; and direct discharges from industrial processes (e.g. nuclear power production, nuclear fuel production and reprocessing, naval operations, radiopharmaceutical production, and medical diagnostics and procedures). Concentrations of naturally-occurring radionuclides can also be enhanced by industrial practices (e.g. oil and gas production, phosphate and steel processing, uranium mining), and this is often referred to as TNORM (technologically-enhanced naturally occurring radioactive material).

The direct discharge of artificial radionuclides into UK coastal waters has historically been dominated by those made into the eastern Irish Sea from the fuel reprocessing plant at Sellafield. Although these discharges may mask the effects of those from other sources, some localised effects can become apparent, such as raised tritium concentration levels in the Bristol Channel from the GE Healthcare radiochemical plant in Cardiff. Elsewhere, environmental levels are typically difficult to distinguish from those associated with discharges from Sellafield and from global fallout.

Discharges of radioactive substances from nuclear licensed sites are subject to authorisation and regular monitoring to ensure compliance with UK and EU dose limits. The UK Strategy for Radioactive Discharges 2001 – 2020 (Defra, 2002) sets out a clear framework for progressive reductions in discharges from nuclear licensed

sites, which provided the basis for the UK meeting its OSPAR obligation that inputs to the marine environment should give rise to environmental concentrations of close to zero[1] (OSPAR, 1998b). A revised discharge strategy for the period 2006 – 2030 (DECC, 2009) has undergone consultation and was published in July 2009. The main UK sites discharging into the sea are shown in Figure 3.58.

3.2.3 Developments since *Charting Progress*

Since the publication of *Charting Progress* in 2005, authorised discharge limits at a number of nuclear sites have been reviewed and reduced where practicable. Significantly, in 2006, a variation to Sellafield's existing authorisation resulted in a reduction in the ^{99}Tc limit to 10 TBq per annum, comparable to historic discharge levels and significantly lower than the peak discharge of almost 200 TBq in 1995, following the introduction of the Enhanced Actinide Removal Plant (EARP). Also in 2006, the Cardiff laboratory of GE Healthcare, a pharmaceutical company manufacturing radioactive labelling compounds, announced that their tritium recycling initiative, Project Paragon, had been successful in developing a suitable process; a resulting plant is being commissioned. This project had been developed in response to reduced discharge limits imposed by the Environment Agency in support of the UK Discharge Strategy.

Nuclear power generation at two Magnox reactors ceased as planned at the end of 2006, bringing the number of these reactors in the decommissioning phase to nine. The remaining operational Magnox reactors are at Oldbury

Figure 3.58 Licensed nuclear sites discharging radioactive material into the marine environment. © Crown copyright 2010.

and Wylfa. Oldbury power station has recently been granted a lifetime extension, enabling it to continue operation beyond its scheduled closure date of December 2008. Wylfa power station is due to cease operation in 2010. The closure and decommissioning of Magnox nuclear power stations impacts the discharge of radioactivity in the local area and indirectly at Sellafield via the Magnox Reprocessing Plant. The target of ending Magnox fuel reprocessing operations by 2012 has been affected by technical problems at Sellafield, and the target date is now 2016 (Defra, 2008).

1 The ultimate aim of the OSPAR Strategy for radioactive substances is to achieve concentrations in the environment near background levels for naturally-occurring radioactive substances and close to zero for artificial radioactive substances.

3.2.4 Presentation of the evidence

3.2.4.1 Trends in discharges from Sellafield

Discharges to the marine environment from the nuclear fuel reprocessing plant at Sellafield have shown a marked downward trend from their peak in the 1970s. A succession of measures has seen the quantities of many discharged radionuclides reduced by an order of magnitude, and, in the case of some particularly significant nuclides such as ^{137}Cs and plutonium-α, by two orders of magnitude. Occasionally, the introduction of technologies to abate discharges or handle new waste streams has led to increases in discharges of other nuclides, as happened when the Thermal Oxide Reprocessing Plant (THORP), which allowed the reprocessing of fuel from Advanced Gas-cooled (AGR) and Light Water (LWR) reactors, came into operation in 1994. The increased discharge of cobalt-60 (^{60}Co) and strontium-90 (^{90}Sr), among others, was modest and has subsequently been reduced.

The introduction of EARP, however, gave rise to a dramatic increase in the discharge of the β-emitter ^{99}Tc. Although concentrations in the waste stream of ruthenium-106 (^{106}Ru) and the actinides were significantly reduced, the relatively low and uniform levels of ^{99}Tc discharge (2 to 7 TBq/yr) during the 1980s and early 1990s increased to almost 200 TBq in 1995 (Hunt et al., 1997) before stabilising at around 60 to 80 TBq/yr from 1997 to the early 2000s (Figure 3.59). Technetium is highly mobile in seawater (e.g. Aarkrog et al., 1987; Leonard et al., 1997; Brown et al., 2002; Lindahl et al., 2003; Kershaw et al., 2004), and is also liable to bioaccumulate (Knowles et al., 1998; Swift and Nicholson, 2001), resulting in predictable increases in concentrations in biota (Figure 3.59) and, consequently, radiation dose. In response to these increases in ^{99}Tc discharges (and concerns over the potential socio-economic effects on the fishing industry, including from other European States) suitable storage and treatment techniques were developed and

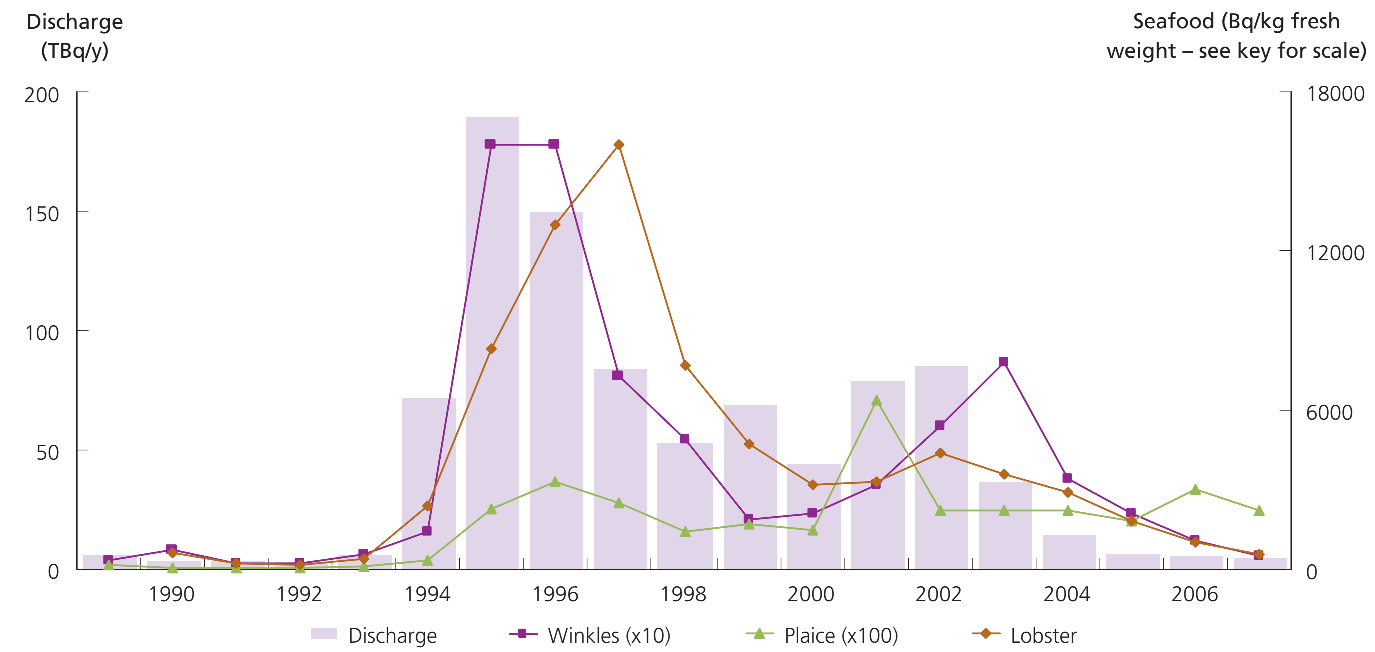


Figure 3.59 Annual ^{99}Tc liquid discharge from Sellafield and concentrations in winkles, plaice and lobster collected near Sellafield. Source: Environment Agency et al. (2008). © Crown copyright 2010.

the authorised annual discharge limit was reduced from 200 TBq to 90 TBq in 2000, and then to 20 TBq in 2004. Following the successful introduction of a novel treatment method using tetraphenylphosphonium bromide (TPP) and cognisant of the need under the OSPAR Convention agreement to ensure progressive decreases in discharges, further reductions in authorised limits were stipulated by the environment agencies, and in 2006 the authorised annual discharge limit was reduced to 10 TBq. The ^{99}Tc discharge in 2007 was less than 5 TBq.

DECC's (Department of Energy and Climate Change) revised Discharge Strategy (DECC, 2009) sets out a series of targets for future discharges from Sellafield. These targets are based on assumptions about the cessation of reprocessing operations as the UK's existing nuclear power stations reach the end of their operational lives. Projections beyond 2016 are subject to greater uncertainty, as the waste arising from decommissioning operations is more difficult to quantify. However, annual discharges of 0.05 TBq for total α-activity (compared to ~0.25 TBq currently), 10 TBq for total β-activity (~40 TBq currently), 10 TBq for tritium (~1000 TBq currently) and 0.1 TBq for ^{99}Tc (~5 TBq currently) are targeted for 2030[2], with intermediate targets set for 2020.

3.2.4.2 Environmental concentrations of radioactivity

3.2.4.2.1 Concentrations in the Irish Sea

Elevated concentrations of radioactivity are present in the Irish Sea, compared with other UK waters, as a result of discharges to the marine environment from Sellafield and other installations. Concentrations in many matrices have fallen since the 1970s as discharges have been reduced, although the behaviour of different radionuclides is strongly affected by their mobility in seawater, affinity for sediments, and tendency to accumulate in biota. There is a substantial programme of sampling and analysis to monitor the effects of discharges. This is undertaken by the environment agencies and the Food Standards Agency, and the results are published annually (Environment Agency et al., 2008). Monitoring data are also collated and assessed as part of the OSPAR Periodic Report series (e.g. OSPAR, 2007b).

Caesium-137 concentrations in seawater have declined considerably since the mid-1970s, in response to lower discharges. The most recent seawater survey, in June 2007 (Environment Agency et al., 2008), found concentrations of up to 0.2 Bq/l in surface seawater, with the higher levels being found in the eastern Irish Sea along an arc from the Solway Firth through the Isle of Man to Liverpool Bay. This compares to levels above 12 Bq/l in the mid-1970s. The correlation between concentration levels and proximity to Sellafield observed in previous surveys appears to be weakening, due to significant remobilisation of ^{137}Cs from subtidal sediments contaminated by historical discharges. This is now the dominant source term to the water column (Leonard et al., 1998) with an estimated inventory of almost 600 TBq (McCubbin et al., 2008). To the west of the Isle of Man, concentrations were below 0.1 Bq/l, and in the southern Irish Sea they were less than 0.05 Bq/l and decreased with distance from Sellafield. This broadly reflects the predominant northwards migration of water in the Irish Sea to the west of the Isle of Man.

The concentration of ^{137}Cs in fish and shellfish near Sellafield has remained fairly stable over the past five years (~10 Bq per kilogram

2 These target levels are subject to change in the final Discharge Strategy document in light of ongoing discussions with industry.

winkles, 4 to 6 Bq per kilogram lobster/plaice), and is low by 1970s standards. The dilution of concentrations with distance from Sellafield is not as marked in the eastern Irish Sea as it was when significant reductions in discharges were initially achieved, owing to the remobilisation of ^{137}Cs from sediments.

Levels of tritium in the Irish Sea are low, with maximum concentrations of 4 to 7 Bq/l found during the June 2007 seawater survey. These levels were confined to Morecambe Bay and the southern Scottish coastline, and compare to concentrations of 10 to 20 Bq/l in this area during the previous survey in 2005. The primary source of tritium is Sellafield, although the nuclear power stations at Heysham and Wylfa also discharge this radionuclide to the marine environment. The low levels observed off the Cumbrian coast in June 2007 are likely to be due to low discharges from Sellafield at that time due to reduced operation of THORP. Tritium is highly mobile in seawater, and so disperses readily and is transported out of the Irish Sea via the North Channel.

Technetium-99 released by Sellafield follows the same transport pathways as ^{137}Cs. The elevated concentrations observed in seawater and biota, particularly lobster and *Fucus vesiculosus*, in the eastern Irish Sea during the period of high ^{99}Tc discharges are declining steadily and proportionately in response to the reduced inputs (Figure 3.59). However, increased concentrations in *F. vesiculosus* have been recorded in recent years in areas remote from Sellafield, such as Fishguard and Carlingford Lough. Leonard et al. (2004) concluded that such effects are likely to be the result of complex hydrographic transport patterns in the Irish Sea dispersing different ^{99}Tc pulses to a variable degree. Recent studies (McCubbin et al., 2006, 2008) have indicated that a proportion of the Sellafield discharge has become associated with subtidal sediments in the Irish Sea, providing a potential mechanism for future remobilisation. The total inventory of ^{99}Tc in Irish Sea sediments has been estimated at 37 TBq (McCubbin et al., 2008), located predominantly on fine-grained sediments in the eastern Irish Sea (Figure 3.60).

Concentrations of plutonium and americium-241 (^{241}Am) in fish and shellfish have remained broadly level in recent years, despite a step-change reduction in discharges of these radionuclides following the commissioning of EARP. These radionuclides are readily adsorbed onto sediment, and this is now the dominant source in the Irish Sea. Levels of both radionuclides in mud samples from Ravenglass, 10 km south of Sellafield, have shown a slight upward trend (Environment Agency et al., 2008), although it is not clear whether this is due to redistribution of subtidal sediments or a process by which older sediments, contaminated by historical discharges, return to the surface; ^{241}Am will also grow as it is a decay product of plutonium-241 (^{241}Pu). Worsening wind and wave conditions, a potential result of climate change (Wang et al., 2004), could lead to these sediments being further disturbed, leading to their potential remobilisation into the water column.

Enhanced ^{210}Po concentrations are now very difficult to detect. This naturally-occurring radionuclide was discharged from a phosphate processing works at Whitehaven, 30 km north of Sellafield. Discharges decreased substantially in 1993, and ceased in 1995, since when concentrations have declined to stable levels. A detailed study of natural background levels of ^{210}Po (Young et al., 2003) has enabled the persisting concentrations, particularly in shellfish, to be placed in context.

Figure 3.60 *Distribution of ^{99}Tc in Irish Sea subtidal sediments based on 2005 and 2006 survey data (black triangles denote sampling locations). Source: McCubbin et al. (2008). © Crown copyright 2010.*

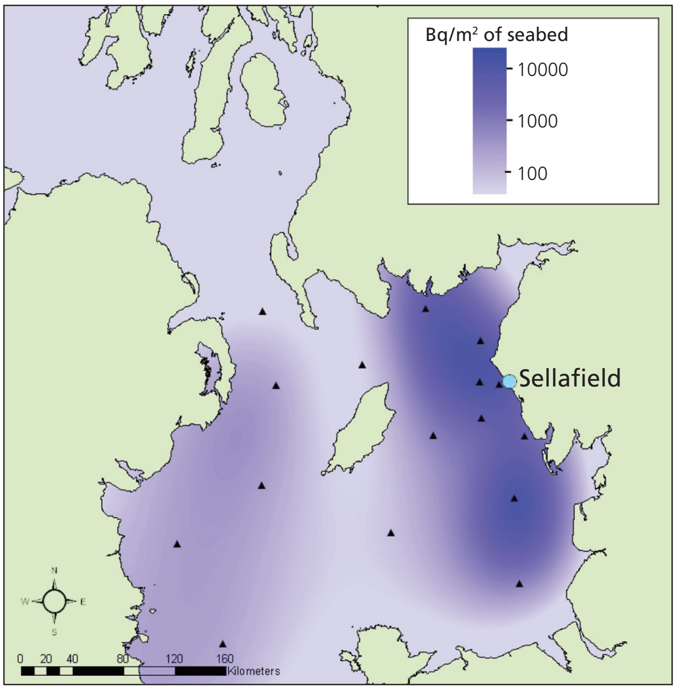
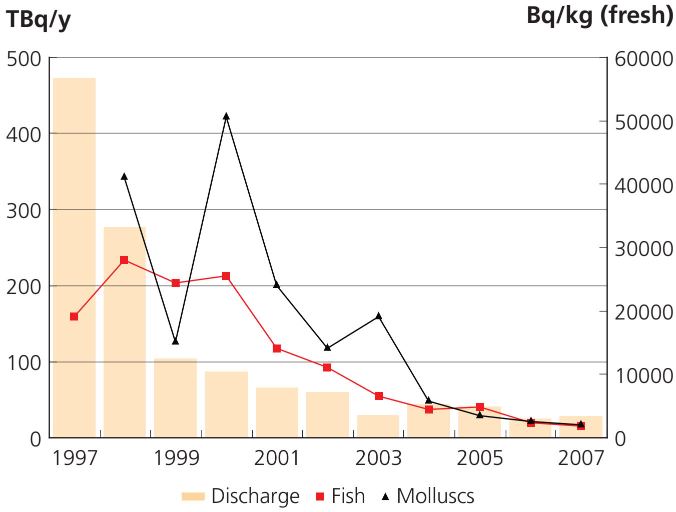


Figure 3.61 Annual tritium discharge (TBq, left-hand scale) from Cardiff and mean concentrations in fish and molluscs (Bq per kilogram, right-hand scale). Source: Environment Agency et al. (2008). © Environment Agency.



3.2.4.2.2 Concentrations elsewhere

Tritium concentrations in the Bristol Channel continue to decline (Figure 3.61), but remain elevated due to operations at and discharges from the GE Healthcare pharmaceutical plant at Cardiff. Discharges of tritium and ^{14}C from Cardiff have generally decreased since the late 1990s as the Environment Agency has sought to reduce authorised limits and improve overall waste management at the site. The introduction of a tritium recycling initiative (Project Paragon) should provide continued decreases in the discharge of tritium, particularly in its organically-bound form. Organically-bound tritium (OBT) has the potential to transfer through the food chain and to bioaccumulate in fish and shellfish, and this process led to high concentrations of OBT (above 50 kBq/kg) in fish and mollusc samples from the Bristol Channel being found in the late 1990s. Environmental concentrations have responded to lower inputs in recent years and in 2007 fish and shellfish samples contained tritium levels of around 2 kBq/kg.

In the North Sea, concentrations of tritium and ^{137}Cs are lower than in the Irish Sea, due to lower direct inputs and progressive dilution of Irish Sea sources. At many sampling stations, ^{137}Cs levels were difficult to distinguish from the background of fallout from weapons testing and the Chernobyl accident in April 1986. Tritium levels were close to the limit of detection. The effect of routine discharges from nuclear power stations on the east coast of the UK is generally imperceptible above background levels and the effects of historic discharges from Sellafield, which have been transported into the North Sea by the prevailing residual currents.

3.2.4.3 Offshore oil and gas industry

Concentrations of TNORM are discharged into UK waters from offshore oil and gas platforms as a constituent of produced water and insoluble scale. Radionuclides found in these discharges include ^{226}Ra and ^{228}Ra, and their decay products such as ^{210}Pb and ^{210}Po. Since the publication of the first UK Discharge Strategy (Defra, 2002) there has been an effort to compile information regarding the extent of non-nuclear discharges, and this is provided to OSPAR to further enable assessment of the effects of this source.

Estimates of the total radioactivity discharged per annum into UK waters from offshore platforms vary considerably. The MARINA II project (van Weers, 2003) calculated that as much as 4.6 TBq of both α- and β-emitting activity was discharged during the late 1990s, while more recent assessments (Defra, 2008), made with the benefit of the recently-compiled pollution inventories, put the annual discharge at around 0.75 TBq of ^{226}Ra (an α-emitter) and 0.25 TBq of ^{228}Ra (a β-emitter) over the period 2004–2005, with future projections of an increase in discharges due to operational and decommissioning sources.

The enhancement of radium in the local zone around an oil platform has been considered as part of the MARINA II project. Sazykina and Kryshev (2003) estimated that concentrations of each of the radionuclides ^{226}Ra and ^{228}Ra, due to the discharge of produced water, would be between 0.005 and 0.01 Bq/l. This compares with a typical concentration of ^{226}Ra in seawater of ~0.002 Bq/l.

3.2.4.4 Radioactive particles

Radioactively contaminated particles have been found in offshore and foreshore areas around Dounreay and Sellafield.

Fragments of irradiated fuel have been identified in the marine and coastal environment around Dounreay since 1979, and at nearby Sandside Bay since 1984. Following finds in 1997, SEPA (Scottish Environment Protection Agency) recommended that the Scottish Office considered imposing a FEPA (Food and Environment Protection Act) Order, banning the taking of seafood from an area covering a 2 km radius around the outfall pipe. This order remains in place, and is now administered by the Food Standards Agency (Scotland). Comprehensive overviews of the current state of knowledge concerning Dounreay particles can be found from the periodic reports of the Dounreay Particle Advisory Group (DPAG). An intensive monitoring programme at Sandside Bay and the Dounreay foreshore continues in order to ensure that particles are promptly detected and removed. The probability of a member of the public encountering a fuel particle at Sandside Bay has been assessed to be around one in 20 million (DPAG, 2008).

Enhanced monitoring on the beaches around Sellafield since 2006 has uncovered a number of radioactive items (small particles, pebbles and stones). The finds contain a range of radionuclides, but in terms of activity are principally ^{137}Cs, with some containing mainly ^{241}Am and associated plutonium isotopes. Identification of the origin of these finds is in progress, but it is highly likely that the vast majority, if not all, are related to past events and incidents at Sellafield. A large number of the finds are of an average dimension of 1 cm or less and some of these particles are the size of sand grains. The rest are stones and pebbles. The majority of the finds have been found buried at depths of up to 20 cm in sand close to the Sellafield site. Monitoring along the Cumbrian coast is set to continue as part of the routine environmental monitoring programme. The

health risk from contact with the majority of particles recovered is considered unlikely to be significant (Environment Agency, 2008).

3.2.5 Progress towards the vision of clean and safe seas

3.2.5.1 Radiation exposures to humans

The calculation of radiation exposures to consumers of seafood from around nuclear licensed sites forms part of a comprehensive monitoring programme overseen by the UK environment agencies and the Food Standards Agency. One of the aims of this programme is to ensure that the doses due to radioactive discharges do not exceed the annual UK dose limit of 1 mSv, and the results of these dose calculations are published annually (see, for example, Environment Agency et al., 2008). The assessed dose takes account of the changing habits of consumers of fish and seafood, and so trends in radiation exposure are not solely reflective of environmental levels of radiation.

3.2.5.1.1 Assessment criteria

Radiation dose estimates for humans are an important measure to use in establishing Good Environmental Status in respect of radioactivity since they provide a direct indication of health risk.

Table 3.41 presents our expert opinion on the status and trends for radioactivity within the UK seas. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems, based on radiation exposures at constituent nuclear sites. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (↔). The confidence rating is classified as low (I), medium (II) or high (III) basedon the number of indicators available in that region and the agreement between them. The forward look is classified as positive (↑), neutral (↔) or negative (↓).

The status at each site has been assessed in relation to the magnitude of risk as determined by the radiation exposure to humans in 2007 (Figure 3.62). The threshold between 'green' and 'amber' status is 0.02 mSv, a screening level for significant pathways (Allott, 2005; NDAWG, 2008) and close to the International Atomic Energy Agency level of *de minimis* dose (IAEA, 1999). The threshold between 'amber' and 'red' status is 0.2 mSv, ten times the lower boundary value and below the legal limit of 1 mSv. In its 2007 Recommendations (ICRP, 2007), the International Commission on Radiological Protection assessed the risk coefficient for cancer and serious genetic (heritable) effects to be $5.7\ 10^{-5}$ per mSv, a risk considered to be proportionate to dose for doses below about 100 mSv. Thus, the 0.20 mSv threshold between the allocation of an 'amber' and 'red' status is approximately equivalent to a 1 in 100 000 lifetime risk of cancer and heritable disease.

High-rate seafood consumers around Sellafield are typically subject to the highest radiation exposure due to liquid discharges in the UK. In 2007, this group was assessed to have received 0.24 mSv as a result of current and historic Sellafield discharges, with around 80% due to ^{239,240}Pu and ^{241}Am in molluscs. ^{99}Tc contributed about 2%. There has been a gradual increase in the dose to this group over the past five years (Figure 3.63), primarily due to increases in consumption rates of the local population, although the rise from 2006 to 2007 is largely attributable to an elevation of ^{241}Am concentrations in molluscs. Figure 3.63

Table 3.41 Regional assessment summary for radioactivity.

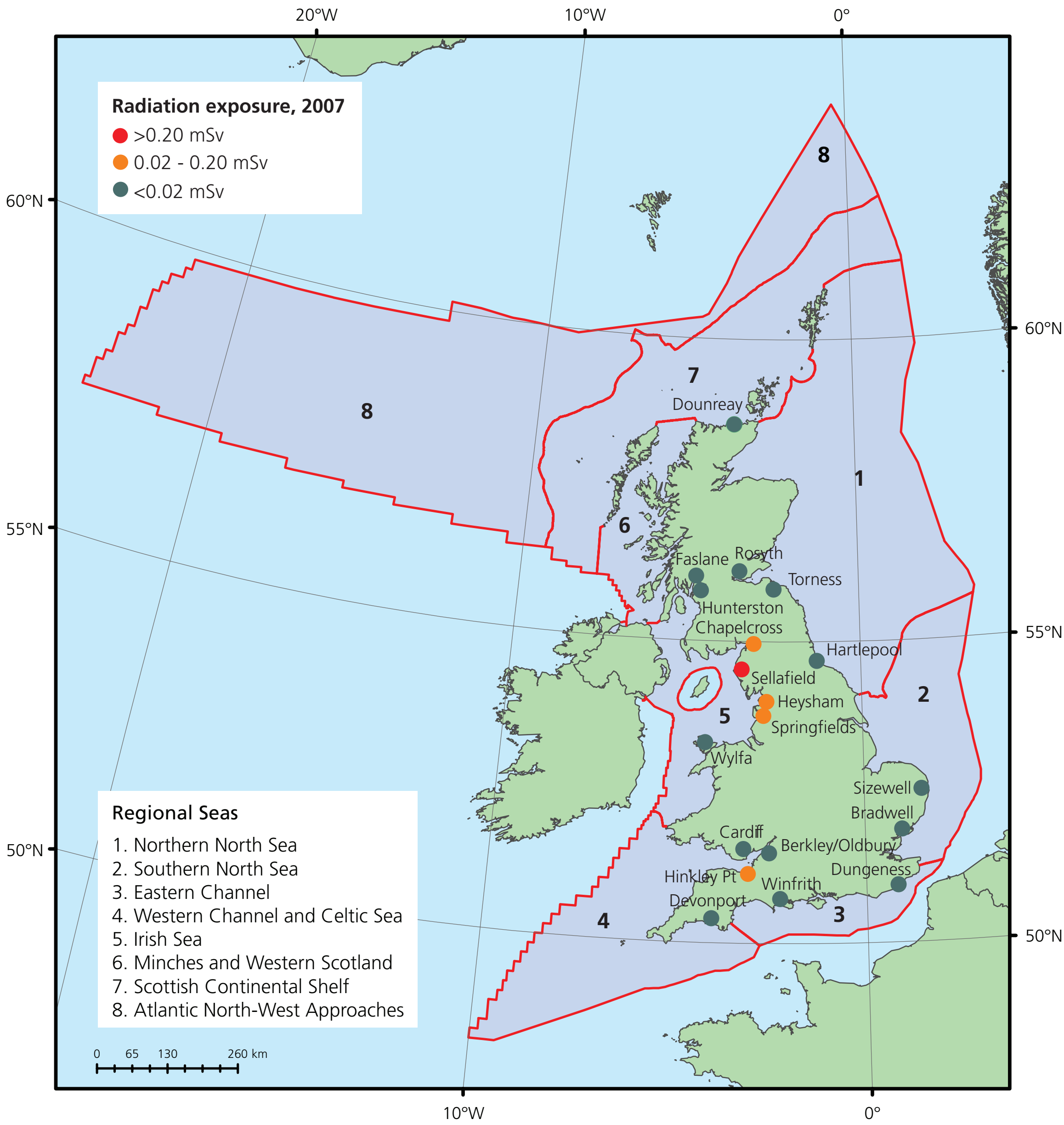
CP2 Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
5	Discharges from Sellafield and resuspension of historic Sellafield discharge from marine sediments. ^{210}Po concentrations in seafood at Whitehaven	Concentrations have reduced in response to lower inputs from Sellafield, although this effect is not as marked for certain radionuclides which have become associated with sediments in the Irish Sea. Radiation exposure remains stable and low by historic standards, but very sensitive to levels of TNORM (^{210}Po)	↑		II	↑
4	Discharges of organically bound tritium and ^{14}C from GE Healthcare plant, Cardiff.	Reduced discharges are leading to reduced concentrations in biota. New recycling plant for OBT expected to reduce inputs further	↑		III	↑
1 – 3, 6, 7	Far-field effects of Sellafield discharge. TNORM from oil and gas industry (Regions 1 & 2)	Levels close to Chernobyl/ weapons fallout background make Sellafield influence difficult to detect outside Irish Sea. Limited data on the effects of oil/gas TNORM discharge	↔		II	↔
8	No significant anthropogenic sources		↔		II	

shows a time-series of radiation exposures to a number of critical groups around the Irish Sea. The dilution of radionuclide concentrations away from the vicinity of Sellafield translates into significantly lower doses, although in some cases this may also be enhanced by lower seafood consumption rates (Environment Agency et al., 2008).

The dose attributable to TNORM (after subtracting background levels) is also calculated for the Sellafield group, in light of their proximity to the former phosphate processing works at Whitehaven. The dose from TNORM in 2007 was 0.28 mSv, primarily due to levels of ^{210}Po in molluscs which were above their median background concentration but within the bounds of natural variability (Young et al., 2003). The combined dose to the Sellafield high rate seafood consumers is therefore 0.52 mSv, below the annual dose limit of 1 mSv. Since 1999, the combined annual dose from artificial radionuclides and TNORM has ranged from 0.40 to 0.57 mSv, with no discernible trend besides a high sensitivity to small changes in ^{210}Po concentrations. This represents a significant reduction from individual annual doses of up to 1.9 mSv from radiocaesium in the late 1970s and 5 mSv from ^{210}Po and ^{210}Pb in the early 1980s.

Radiation exposures around other nuclear licensed sites in the Irish Sea are largely due to the effects of past and current Sellafield discharges. Annual doses from marine pathways at Chapelcross and Heysham nuclear power

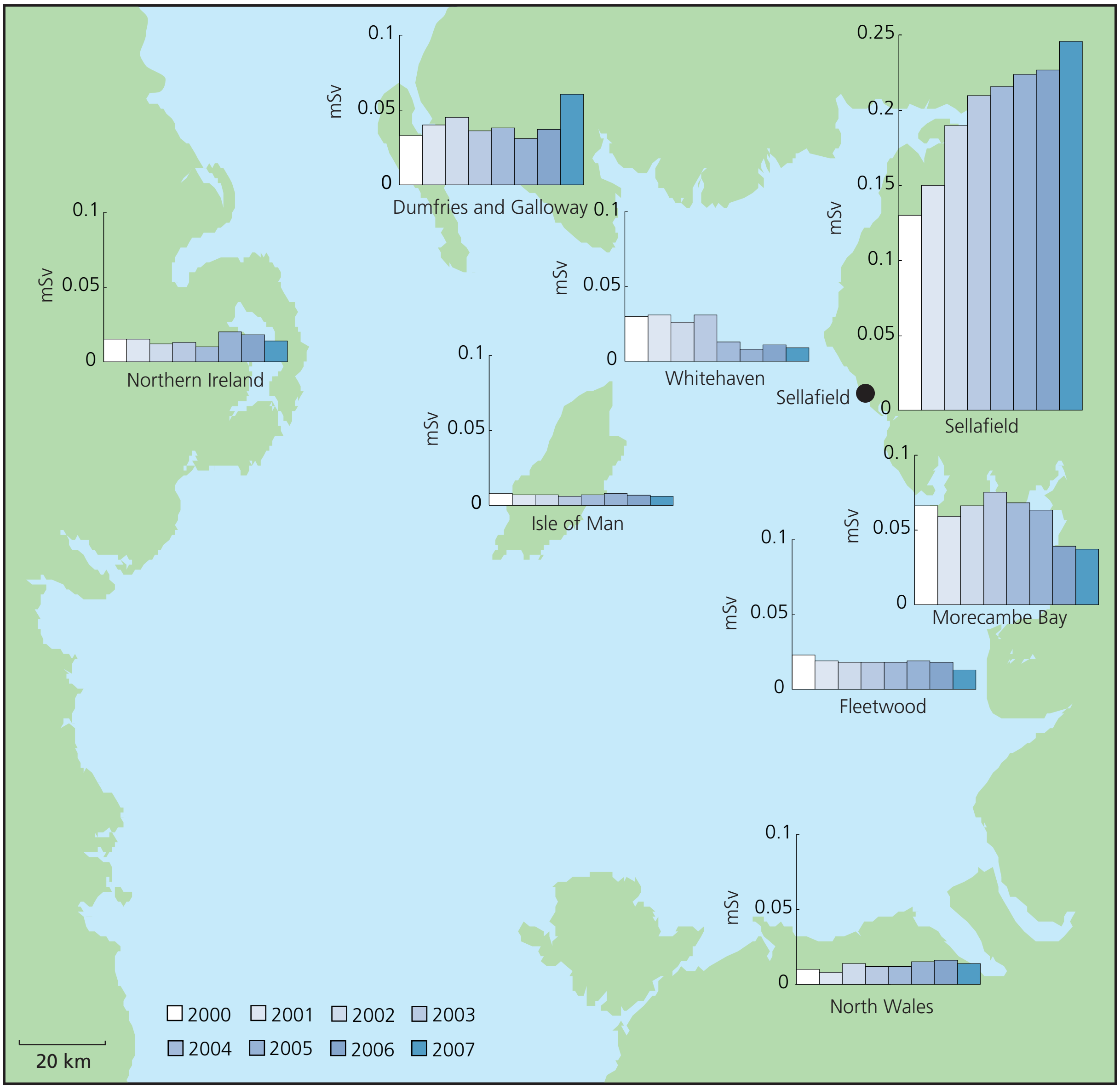
Figure 3.62 Regional assessment of radiation risk in UK waters in 2007. Source: Environment Agency et al. (2008). © Environment Agency.



stations and the Springfields nuclear fuel manufacturing plant are typically greater than 0.02 mSv, although a downward trend in these exposures is evident. Exposure to houseboat residents at Springfields is linked to the concentrations of radionuclides in local sediments, and so is strongly affected by the redistribution of contaminated sediments.

Away from the eastern Irish Sea, exposures to artificial radionuclides are generally less than 0.02 mSv, or 2% of the annual dose limit of 1 mSv. In the Bristol Channel, the dose to the most exposed group at Cardiff has decreased from above 0.1 mSv in 2000 to 0.014 mSv in 2007, due principally to the reduced discharges and concentrations of tritium and ^{14}C in local seafood. Doses at Hinkley have recently risen

Figure 3.63 Individual radiation exposures to seafood consumers from artificial radionuclides in the Irish Sea, 2000–2007. Source: Environment Agency et al. (2008). © Environment Agency.



above 0.02 mSv after a habits survey identified individuals with a large occupancy time over contaminated sediments, and in 2007 their exposure was assessed to be 0.029 mSv. These figures may be contrasted with the average annual dose to the UK population of 2.7 mSv (HPA, 2005), of which over 80% is from natural background radiation (mostly radon).

3.2.5.2 Radiation exposures to wildlife

The radiation exposure of wildlife at a number of Natura 2000 sites has been assessed by the Environment Agency (Allott and Copplestone, 2008a) and SEPA, and the dose rates compared to a threshold of 40 µGy/hr. This value represents the difference between exposure from natural sources (estimated at 60 µGy/hr) and the 100

μGy/hr level below which no adverse effects are expected to occur (Larsson et al., 2004). The highest dose rate to the worst affected organism in a Natura 2000 site was 520 μGy/hr in the Ribble and Alt Estuaries Special Protection Areas, affected by Springfields discharges. This was a conservative assessment based on the assumption that Springfields discharges were made at the site permit levels. The second highest dose rate, at 41 μGy/hr ,was the Drigg Coast Special Areas of Conservation, affected by Sellafield's discharges. Since this assessment was made, a new assessment methodology based on the ERICA (Beresford et al., 2007) assessment tool has become available. Using this more recent methodology, the worst affected organism for the Drigg Coast Special Areas of Conservation was reduced to 20 μGy/hr. The Drigg Coast Special Areas of Conservation was also considered in an ERICA project case-study (Wood et al., 2008) which concluded that there was no indication of significant impact from ionising radiation on the sand dune biota. This Natura 2000 site will be kept under review.

The Ribble and Alt Estuaries Special Protection Areas dose rate was significantly in excess of the agreed threshold and so this Natura 2000 site was included in the final stage (determination of permissions) of the Habitats Regulations implementation. A separate report is available for this determination process (Allott and Copplestone, 2008b), which concluded that new authorisation limits for the Springfields Fuels Ltd site (in effect from January 2008) would ensure that the dose rates to reference organisms and feature species will be less than 40 μGy/hr.

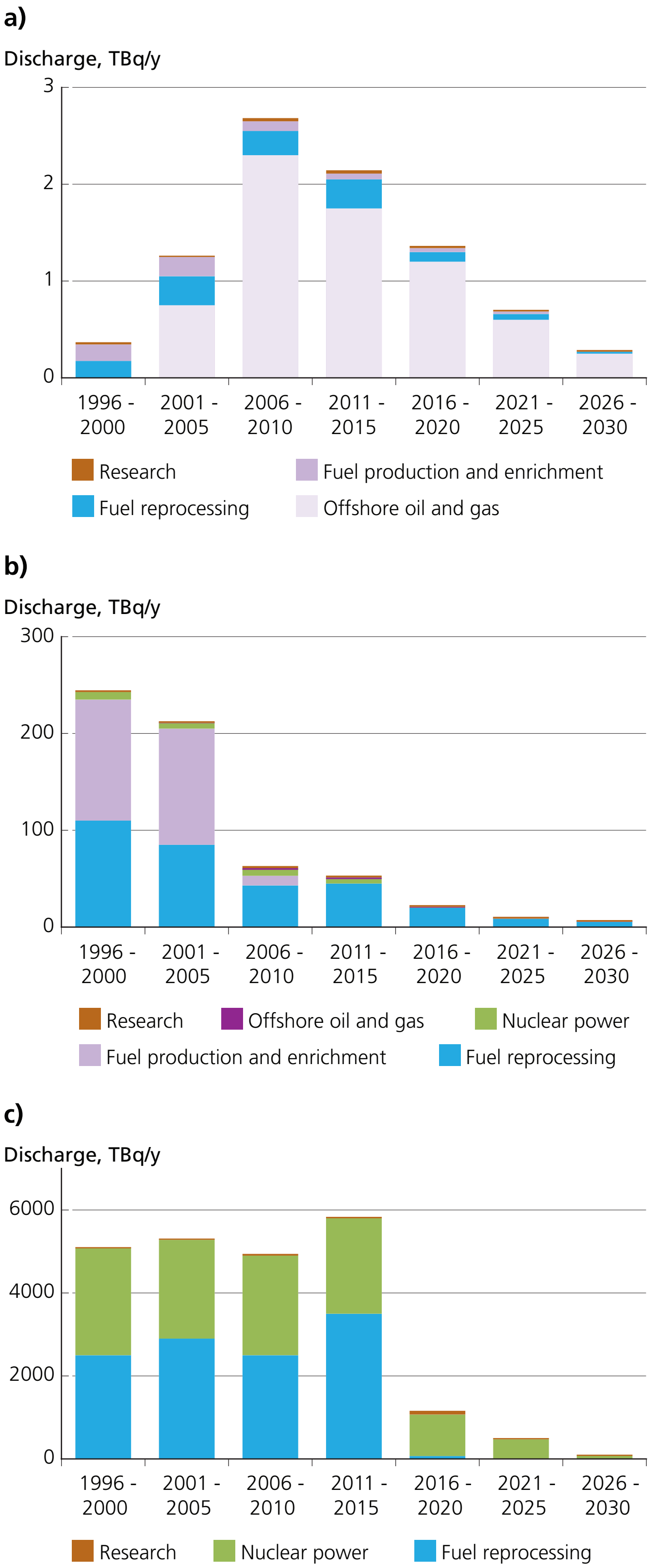
Radiation dose rates to marine biota around offshore oil platforms were calculated as part of the MARINA II project (Sazykina and Kryshev, 2003). Using estimates of the enhanced local radium concentrations, internal dose rates of around 1 to 3 μGy/hr for molluscs were obtained. Dose rates to fish and crustaceans were less than those for molluscs.

3.2.6 Forward look and need for further work

The revised UK Discharge Strategy (DECC, 2009) sets out discharge projections to 2030 for α- and β-emitting radioactivity and tritium from the nuclear and non-nuclear sectors (Figure 3.64), with additional ^{99}Tc targets for Sellafield. These include both operational and decommissioning discharges. Contributions from new nuclear-build and the possible extension of operations at current nuclear power stations have not been included in the projections.

Radiation exposure around the UK is currently dominated by the effects of past discharges from Sellafield and the phosphate processing works at Whitehaven. Discharges from Sellafield are forecast to drop significantly by 2020 as the reprocessing of Magnox fuel is completed, although some intermediate increases in the discharge of α-activity and tritium are expected. While significant reductions in Sellafield discharges have already been made, they still contribute a large proportion of the total α- and β-activity discharged by the nuclear industry. Sellafield and the nuclear power sector are forecast to discharge comparable amounts of tritium, although the low radiotoxicity of this nuclide and its ready dispersion in the environment render it less important from the perspective of radiation exposure. Stringent regulation of new nuclear-build and improvements in waste treatment technologies are likely to mean that future nuclear power generation will produce lower discharges than those in use to date.

Figure 3.64 Actual and projected discharges of liquid radioactive waste from the UK nuclear and non-nuclear industry, 1996–2030. (a) α-emitting radionuclides; (b) β-emitting radionuclides; (c) tritium. Source: Defra (2008). © Crown copyright 2010.



The higher risks identified in Region 5 (Irish Sea) are caused by radioactivity already in the environment, particularly in sediments. Further constraints on discharges, while necessary, will have a minor effect on these risks in the short term. Other mitigation such as environmental remediation is prohibitively expensive and operationally challenging. It is not considered necessary to constrain food pathways to reduce risks.

An upward trend in concentrations of plutonium and ^{241}Am in some coastal Irish Sea sediments could, if continued, cause the radiation exposure of seafood consumers on the Cumbrian coastline to increase. A lack of recent data on the distribution and total inventory of these nuclides makes any prediction of their future remobilisation difficult.

The offshore oil and gas industry is forecast to continue as the dominant source of α–emitting radioactivity (Figure 3.64), with annual ^{226}Ra discharges projected to exceed 2 TBq until 2010. Monitoring of environmental concentrations around oil platforms is limited; assessment of radiation exposures in pathways potentially contaminated by this discharge deserves further study.

3.3 Eutrophication

3.3.1 Key points

i. Introduction

Eutrophication has been identified as one of the major threats to the health of estuarine, coastal and marine ecosystems around the world. Eutrophication is 'the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned' (definition according to the Urban Waste Water Treatment Directive (UWWTD), 91/271/EEC). Assessment tools have been developed which focus both on the risk posed by nutrient enrichment, now and in the near future, and on the undesirable disturbance to biology and water quality. Interest is therefore both in terms of ecological health and interference of human use of marine waters.

ii. How has the assessment been undertaken?

The OSPAR Comprehensive Procedure methodology was applied to coastal and offshore waters that had been identified as being at possible risk due to nutrient enrichment. Results of similar assessments carried out in support of the UWWTD and the EU Nitrates Directive were used for estuarine waters. Field measurements were assessed against a checklist of parameters including concentrations of nutrients, chlorophyll and dissolved oxygen, phytoplankton indicator species, macrophytes and toxin-producing algae. The assessments used a 'weight of evidence' approach which identified non-problem areas, potential problem areas (those at risk of eutrophication), and problem areas on the basis of evidence of undesirable disturbance to the ecosystem.

iii. Current status of eutrophication and past trends

UK coastal and offshore waters in each of the eight UK regions were shown to be non-problem areas. The coastal waters include five areas that had previously been assessed as areas of ongoing concern (East England, East Anglia, Liverpool Bay, the Solent and the Firth of Clyde). These areas were shown to be nutrient enriched and in some there was evidence of accelerated growth of algae and higher forms of plant life, but there was no evidence for undesirable disturbance, and the trend in nutrient loading pressure indicated that the risk was not increasing. However, 17 estuarine waters were identified as problem areas and five are classified as potential problem areas. These water bodies are also designated as Sensitive Areas and Nitrate Polluted Waters (eutrophic) and are subject to management measures required by the relevant EU Directives.

iv. What has driven change?

The major pressures occur in the east, south and north-west of England where input of nutrients of anthropogenic origin (notably nitrate and phosphate from agriculture and urban waste water sources) has resulted in nutrient enrichment of coastal waters. Assessments indicate no significant changes in eutrophication status over the past decade. Coastal and offshore waters remain non-problem areas, while a small number of waters (e.g. estuaries and harbours) continue to be problem areas or potential problem areas. The designation of Nitrate Vulnerable zones covering 69% of the land in England, 14% of Scotland, 4% of Wales and the whole of Northern Ireland is likely

to lead to a reduction in nutrient inputs from agriculture, as is the effective implementation of the UWWTD which will reduce nutrient inputs from waste water. Since 1998, total inputs of phosphorous have declined by approximately 6% to 9% per year in all regions except the Atlantic (Region 8), while total inputs of dissolved inorganic nitrogen have decreased by 2% per year in the Northern North Sea (Region 1) and the Irish Sea (Region 5).

v. What are the uncertainties?

Confidence in the assessments of eutrophication is high in the majority of areas due to the availability of extensive datasets, and enhanced monitoring in regions previously indicated as areas of ongoing concern.

v. Forward look

Nutrient reduction programmes (through measures under the relevant EU Directives and OSPAR) are in place to protect the small water bodies which have been identified as problem areas or potential problem areas. However, the environmental response to the reduction in nutrient inputs is expected to be relatively slow.

Key findings

- Recent assessments of eutrophication status indicate that each of the *Charting Progress 2* assessment areas as a whole do not suffer from eutrophication problems.
- Some estuarine areas are nutrient enriched and are at risk from or currently affected by eutrophication. Seventeen of these areas have been identified as sensitive or polluted waters, or conservation areas. These have been classified as problem areas. Five others have been classified as potential problem areas.
- Inputs of nutrients (nitrogen and phosphorus) to problem areas are being managed. However, the extent to which these protective measures will lead to ecological recovery is uncertain, due to the complexity of the eutrophication process which may not be simply reversible.

3.3.1.1 Regional assessment

Table 3.42 presents our expert opinion on the status and trends for eutrophication within the UK seas. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. Trend arrows provided based on the evidence available, showing whether the state or condition of the component is improving (↑) or deteriorating (↓), or where there is no overall trend discernable (↔). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them. The forward look is classified as positive (↑), neutral (↔) or negative (↓).

3.3.1.2 Contribution towards GES descriptors

The EU Marine Strategy Framework Directive (MSFD) provides broad qualitative descriptors for determining 'Good Environmental Status' (GES). For eutrophication, the emphasis is on identification of any negative impacts of anthropogenic nutrient inputs. The descriptor requires that human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algal blooms and oxygen deficiency in bottom waters. The potential indicators or criteria for assessing GES for eutrophication are under discussion, but are likely to be broadly similar to those under

Table 3.42 Regional assessment summary for eutrophication.

CP2 Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
1	Nutrient input (nitrogen and phosphorus) from agriculture and urban waste water treatment in limited areas. The overall pressure is low	The region as a whole does not suffer from eutrophication problems although there are some small estuarine and coastal areas that are assessed as problem areas	↑		III	↑
2	There is significant pressure due to nutrient input from agriculture which covers large areas of eastern England and from urban waste water due to the high population density	The coastal and offshore areas of the region are assessed as non-problem areas even though there is nutrient enrichment and some evidence of accelerated growth, particularly in coastal waters. The physical nature of this region (tidal hydrodynamics, inorganic turbidity restricting light), especially in coastal areas, makes it resistant to developing eutrophication problems	↔		III	↔
3	There is some pressure due to nutrient input from agriculture and from urban waste water	The offshore waters are non-problem areas, being influenced predominantly by the inflow of water from the Atlantic. There are several areas at the coast which have very restricted exchange and are problem areas. The pressures associated with the specific problem areas are being managed	↑		III	↑
4	Relatively low population density though this varies seasonally and is high in summer. Significant intensive agriculture but confined to a few areas. Overall pressure is low	There is no nutrient enrichment in this area with the exception of small estuaries and lagoons which are problem areas and the specific sources of the enrichment are being managed	↑		III	↑
5	There is significant pressure in the eastern part of the region due to nutrient input from agriculture and from urban waste water due to the high population density	The coastal and offshore areas of the region are assessed as non-problem areas even though there is nutrient enrichment and some evidence of accelerated growth, particularly in coastal waters. The physical nature (hydrodynamics, inorganic turbidity restricting light) of this region, especially in coastal areas, makes it resistant to developing eutrophication problems	↑		III	↑

6	The overall pressure from nutrient input is low due to the low population density. The pressure is higher in the southern part of the region due to agriculture	The region as a whole does not suffer from eutrophication problems	↑		III	↑
7	The overall pressure from nutrient input is very low due to the low population density	The region as a whole does not suffer from eutrophication problems	↑		III	↑
8	There are no nutrient pressures in this region	The region as a whole is unlikely to suffer from eutrophication problems	*		III	

development by the EU Water Framework Directive (WFD) and OSPAR, which have already adopted this approach. Indicative lists of characteristics, pressures and impacts may require the development of additional criteria – for example, enrichment by organic matter, and other elements characterising the marine food web. In terms of the latter, inclusion of grazers (e.g. zooplankton and filter-feeders such as mussels and oysters) may contribute towards improved understanding of the biological response to nutrients and towards improved management of eutrophication. High grazing impacts will result in under-estimates of phytoplankton biomass and/or growth (as indicated by chlorophyll concentrations), and egestion of faecal pellets may contribute towards oxygen depletion due to the accumulation of pellets in boundary layers, especially near or within the seabed.

3.3.1.3 Future risks

Improved understanding of eutrophication has shown that there is seldom a simple dose-response relationship between nutrient input and ecosystem responses. Responses are complex and may be direct or indirect. Also, various factors (such as light availability and advective losses) may moderate the response or influence the susceptibility of an ecosystem to nutrient enrichment. Future changes in our understanding of eutrophication are likely to indicate even greater complexity in the biological response to nutrient enrichment, with multiple stressors/pressures, multiple factors determining sensitivity, and complex feedbacks between the different biological responses in an ecosystem. Apart from nutrient inputs, other stressors include climate change, fish harvesting, toxic contaminants and aquaculture. While future scenarios for climate change remain unclear, more extreme environmental conditions such as drier summers, episodic downpours and increased storminess will impact upon anthropogenic nutrient loads from all terrestrial and atmospheric sources into the marine environment, particularly in terms of timing and magnitude of inputs. The biological responses to altered nutrient loads, and any potential negative impacts, will be affected by other factors responding simultaneously to climate change, such as changes in water temperature and changes in physical and/or chemical oceanographic processes.

3.3.1.4 Cross-cutting issues

The overall conclusion of the assessment of the status of plankton (see the relevant section in the HBDSEG Feeder Report) is in line with the conclusions of this assessment of eutrophication status where major oceanographic and climatic changes are responsible for changes in phytoplankton biomass (seen as changes in the Continuous Plankton Recorder greenness index), composition and distribution in the North Sea and wider Atlantic. The influence of nutrient enrichment on plankton productivity is restricted to some coastal areas. These issues are addressed here.

Moderate levels of nutrient enrichment (particularly from sewerage or other outfalls) may provide improved feeding opportunities for waterbirds. High levels of nutrient input may, however, result in de-oxygenation and a decline in food resources (see the relevant section in the HBDSEG Feeder Report). Dense mats of algae, for example, may grow in shallow areas with high nutrients and enriched organic matter, potentially contributing to de-oxygenation of intertidal sediments and negative impacts on the prey for waterbirds.

3.3.2 Introduction

Availability of nutrients in the marine environment is vital for the growth of primary producers, such as phytoplankton and other algae, which sustain marine food webs. The natural processes of nutrient recycling are essential for ecosystem health and productivity; however, nutrient enrichment due to human activities can have negative impacts in susceptible ecosystems. Symptoms of these negative impacts include changes in the species composition of phytoplankton communities, blooms of harmful algae, the development of hypoxic (and anoxic) conditions due to decomposition of the accumulated biomass, loss of submerged aquatic vegetation due to shading, and changes in the community structure of benthic animals.

Management of eutrophication depends on a clear understanding of cause and effect. It is often assumed that there is a simple dose-response relationship between nutrient input and ecosystem response but our conceptual understanding of eutrophication is more complex. For example, Cloern (2001) described complex responses to nutrient inputs, including both direct and indirect effects, and the role of environmental ‘filters’ in moderating the response or determining the sensitivity to undesirable disturbance. These filters include environmental factors such as the light climate and advective losses. Further research on eutrophication is likely to reveal even greater complexity in the biological response to nutrient enrichment, multiple factors determining sensitivity, and complex feedbacks between the different biological processes in an ecosystem (Cloern, 2001). The effect of multiple stressors (e.g. nutrient input, climate change, fish harvesting, and toxic contaminants) also needs to be taken into account as these pressures can have similar ecosystem impacts.

The key policy measures that can be used to manage the risks of eutrophication are shown in Table 3.43. The UWWTD deals with the treatment that urban waste water (sewage and industrial) must receive before discharge in order to prevent eutrophication. The UWWTD requires that sewage being discharged to designated Sensitive Areas (SAs) should be subjected to more than secondary treatment and to specified standards. Appraisals of waste water treatment plants discharging into the SAs and the installation of remedial measures where

Table 3.43 Summary of legislative and regulatory measures to assess and mitigate risks of eutrophication.

Measure	*Overall aim*
EU Habitats Directive (CEC, 1992)	To contribute to ensuring biodiversity through the conservation of natural habitats and of wild fauna and flora. Measures are designed to maintain or restore, at favourable conservation status, natural habitats and species of wild fauna and flora of Community interest and take account of economic, social and cultural requirements and regional and local characteristics.
EU Nitrates Directive (CEC, 1991a)	To protect waters against pollution caused by nitrates from agricultural sources. If a water body is found to be polluted (eutrophic), then it is identified as a Nitrate Vulnerable Zone (NVZ).
EU Urban Wastewater Treatment Directive (CEC, 1991b)	Lays down minimum standards for sewerage systems and treatment, which vary according to the size of the population served and the nature of the receiving water. If a water body is found to be eutrophic, or is identified to have potential to become eutrophic in the near future if protective action is not taken, then the area is designated as a 'sensitive area'.
EU Water Framework Directive (CEC, 2000)	To achieve good ecological status in water bodies by 2015. The WFD does not specifically mention eutrophication but the differences between 'good' and 'moderate' ecological status for plant quality elements in coastal waters are linked to 'accelerated growth' and 'undesirable disturbance'. In water bodies that fail to meet good ecological status, a programme of measures must be implemented.
OSPAR Strategy to combat eutrophication	To combat eutrophication in the OSPAR maritime area, in order to achieve and maintain by 2010 a healthy marine environment where eutrophication does not occur. To assist Contracting Parties in identifying areas in a consistent way where nutrient inputs may cause pollution, and to periodically assess the eutrophication status of the OSPAR maritime area and progress made towards the Strategy's objective, OSPAR developed a common assessment framework: the Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area. Five assessment parameters and their assessment levels as defined by the Common Procedure have been developed to form an integrated set of Ecological Quality Objectives (EcoQOs, Table 3.50) for eutrophication for the North Sea with the overarching objective that all parts of the North Sea should have the status of non-problem areas with regard to eutrophication by 2010, as assessed under the Common Procedure. The implementation of the Eutrophication Strategy takes place within the framework of the obligations and commitments of the various Contracting Parties under other international agreements. This includes the EU Nitrates Directive (91/676/EEC), Urban Waste Water Treatment Directive (91/271/EEC), the Water Framework Directive (2000/60/EEC) and the Marine Strategy Framework Directive (2008/56/EC).
Marine Strategy Framework Directive 2008/56/EC	To establish a framework within which Member States shall take the necessary measures to achieve or maintain good environmental status in the marine environment by the year 2020 at the latest.

appropriate are either completed or underway, but a full report on how the various measures have been implemented is under preparation but not yet available. Belfast Lough provides an example of environmental improvement following the introduction of nitrogen removal (see Case Study 4 on Page 224).

The EU Nitrates Directive (91/676/EEC) seeks to prevent (and reduce) pollution of waters by nitrates from agricultural sources by placing restrictions on agricultural practices which contribute to pollution. Lands draining into polluted waters have to be designated as 'Nitrate Vulnerable Zones' (NVZs). NVZs were designated in 1996 and again in 2002, bringing the total coverage to approximately 69% of England and 14% of Scotland; there has been a (small) increase in Wales, and Northern Ireland has a whole territory designation. An Action Programme of measures has been implemented by farmers within these NVZs to reduce losses of nitrate from agricultural land. The Action Programme promotes best practice in the use

and storage of fertiliser and manure, and builds on the guidelines set out in the Code for Good Agricultural Practice for the Protection of Water (www.defra.gov.uk/farm/environment/cogap/). In Scotland, the Action Programme has been amended by the Action Programme for Nitrate Vulnerable Zones (Scotland) Regulations 2008 with effect from 1 January 2009.

The Water Framework Directive (2000/60/EEC) seeks to protect surface waters and ground waters from human impacts, and provides for the management of water status. The Directive requires that almost all surface waters are to achieve Good Ecological Status by 2015. Good Ecological Status is attributed to a region where the biological community shows a low level of distortion resulting from human activity, but deviates only slightly from the community in undisturbed surface waters. The WFD extends to at least 1 nm beyond the coastal baseline for biological quality elements, and 12 nm for physico-chemical quality elements. The first round of River Basin Management Plans, including programmes of measures, is currently being implemented so it will be some years before any effect on eutrophication status is observed.

The eutrophication objective of the OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic is to achieve a healthy marine environment where eutrophication does not occur, by 2010. Where areas are identified as eutrophication problem areas (see Box 3.3), countries are committed to reducing the relevant nutrient inputs by 50% (relative to a baseline which was set as 1985) and are encouraged to achieve this through the application of measures taken under the relevant EU Directives.

The EU Habitats Directive (92/43/EEC) implemented in the UK in 1994 aims to contribute to protecting biodiversity through the conservation of natural habitats, wild plants and animals. This includes the creation of a network of marine and terrestrial protected areas ('Natura 2000 sites') which support significant numbers of wild birds and their habitats or which support rare, endangered or vulnerable natural habitats and species of plants or animals other than birds. Where nutrients are identified as a threat to 'favourable conservation status' specific action may be taken.

The EU Marine Strategy Framework Directive (2008/56/EC) provides broad qualitative descriptors for determining Good Environmental Status (GES). For eutrophication, the emphasis is on minimising human induced eutrophication, especially the adverse effects it creates including losses in biodiversity, ecosystem degradation, harmful algal blooms and oxygen deficiency. The potential indicators and criteria for assessing GES for eutrophication are under development but will need to build on the work already done by, for example, OSPAR and in support of the Water Framework Directive.

Additional information on toxic algae and toxins in shellfish flesh are collected in fulfilment of the EU Shellfish Hygiene Directive (91/492/EEC). However, it should be noted that the link between nutrient enrichment and incidence of toxic algae is still under investigation. The most recent work indicates that the abundance of the harmful algal bloom species that occur in UK and Irish coastal waters is not related to anthropogenic nutrient enrichment. There is a history of toxic algal problems occurring in some areas, particularly on the west coasts of the UK, where there are low or no significant nutrient inputs.

Box 3.3: Assessment parameters and thresholds used in recent assessments of eutrophication status

For the second application of the OSPAR Comprehensive Procedure, in 2007, the UK's marine coastal and offshore waters were divided into 41 assessment areas designed to encompass the range that might be subject to eutrophication as a result of land-based anthropogenic nutrient input, and where there was a risk that an undesirable disturbance may occur. These were assessed against OSPAR's harmonised assessment criteria (Table 3.44): nutrient concentration and ratios, chlorophyll concentrations, phytoplankton indicator species, macrophytes, dissolved oxygen levels, incidence of fish kills, changes in the zoobenthos and other possible effects of nutrient enrichment such as toxins in bivalve shellfish (Table 3.45). All measured indicators were compared against thresholds for each criterion to produce a + or – score, with + scores indicating exceedance of the threshold. For each area, an overall score was assigned for each category (I: Causative factors; II: Direct Effects; III: Indirect Effects; IV: Other Possible Effects); scores for all categories were combined to produce an initial classification of the status of the area. Where all scores were +, they were classified as problem areas (and assigned 'traffic light' red status) showing evidence of an undesirable disturbance to the marine ecosystem due to anthropogenic

Table 3.44 Harmonised assessment criteria within the OSPAR Comprehensive Procedure.

Assessment parameter	
Category I: Degree of nutrient enrichment	*1. Riverine total N and total P inputs and direct discharges (RID)* Elevated inputs and/or increased trends (compared with previous years) *2. Winter DIN- and/or DIP concentrations* Elevated level(s) (defined as concentration >50% above salinity-related and/or region-specific background concentration) *3. Increased winter N/P ratio (Redfield N/P = 16)* Elevated *cf* Redfield (>25)
Category II: Direct effects of nutrient enrichment (during the growing season)	*1. Maximum and mean chlorophyll a concentration* Elevated level (defined as concentration >50% above spatial (offshore) / historical background concentrations) *2. Region/area specific phytoplankton indicator species* Elevated levels (and increased duration) *3. Macrophytes including macroalgae (region specific)* Shift from long-lived to short-lived nuisance species (e.g. *Ulva*)
Category III: Indirect effects of nutrient enrichment (during the growing season)	*1. Degree of oxygen deficiency* Decreased levels (<2 mg/l: acute toxicity; 2–6 mg/l: deficiency *2. Changes/kills in zoobenthos and fish kills* Kills (in relation to oxygen deficiency and/or toxic algae) Long-term changes in zoobenthos biomass and species composition *3. Organic carbon/organic matter* Elevated levels (in relation to III. Relevant in sedimentation areas)
Category IV: Other possible effects of nutrient enrichment (during the growing season)	*1. Algal toxins (DSP/PSP mussel infection events)* Incidence (related to II.2)

Table 3.45 Summary of assessment parameters, reference conditions and thresholds used in recent assessments of eutrophication status in UK marine waters. Reference conditions are given as means, ranges or background values. Thresholds used for assessment are shown in grey shaded boxes. Green shading indicates where OSPAR thresholds are not exceeded; red shading indicates where thresholds are exceeded. For some parameters (e.g. dissolved oxygen), more detail is required by Directives such as the EU Water Framework Directive.

1. Nutrient concentration (Gowen et al., 2002)								
Nitrate + Nitrite (µM)			*Phosphate (µM)*			*Silicate (µM)*		
Mean	*Range*	*+ 50 %*	*Mean*	*Range*	*+ 50 %*	*Mean*	*Range*	*+ 50 %*
7.20	5.25–9.90	10.80	0.45	0.34–0.65	0.68	3.27	2.30–5.15	4.91
Mean Winter DIN (November–February)						Threshold (Ref value + 50 %)		
Coastal waters: normalised to a salinity of 32, reference value = 13 µM							20 µM	
Offshore waters: normalised to a salinity of 34.5, reference value = 10 µM							15 µM	
2. Nutrient ratios								
			Standard background ratio value			*Ratio with + 50 %*		
N:P			16:1			24:1		
N:Si			2.2:1			3.3:1		
3. Chlorophyll (March–September)								
			Standard background ratio value			*Chlorophyll standard + 50 %*		
Offshore waters			6.7 µg/l and C:Chl factor of 0.012			10 µg/l		
Coastal waters			10 µg/l and C:Chl factor of 0.02			15 µg/l		
4. Phytoplankton indicator species (Devlin et al., 2007)								
Total cell count – assessment of occurrences over 10^7						The incidence of counts exceeding these values should be <25% of all sampling times over 5 years.		
Phaeocystis cell count – occurrences over 10^6								
Any single taxon – occurrences over 10^6								
Counts of chlorophyll – occurrences exceeding 10 µg/l								
5. Macroalgae (Scanlan et al., 2007)								
Macroalgal biomass			*Macroalgal cover*			*Macroalgae assessment*		
< 500g/m² wet weight			<15% of intertidal area			OSPAR threshold not exceeded		
> 500g/m² wet weight			>15% of intertidal area			OSPAR threshold exceeded		
6. Dissolved oxygen (Best et al., 2007)								
≥5.7 mg/l		All life-stages of salmonids and estuarine fish				OSPAR threshold not exceeded		
≥4.0 < 5.7 mg/l		Presence of salmonids and estuarine fish						
≥2.4 < 4.0 mg/l		Most life-stages of non-salmonid adults				OSPAR threshold exceeded		
≥1.6 < 2.4 mg/l		Presence of non-salmonids, poor survival of salmonids						
<1.6 mg/l		No salmonids present marginal survival of resident species						
7. Zoobenthos and fish kills								
Incidence of fish kills or documented changes in zoobenthos to assess disturbance, related to eutrophication								
8. Toxin levels in bivalve mollusc tissue (Shellfish Hygiene Directive 91/492/EEC)								
Toxin	Maximum permitted levels		Proportion of failed tissue samples (whichever was most for ASP/PSP/DSP)					
PSP	80 µg per 100 g flesh		<10% = no undesirable disturbance			>1 % = undesirable disturbance		
DSP	Presence in flesh							
ASP	20 µg per g flesh							

enrichment by nutrients. In areas classified as non-problem areas (assigned 'traffic light' green status) there were no grounds for concern that anthropogenic nutrients posed risks to the marine ecosystem now or in the near future. Potential problem areas (assigned 'traffic light' amber status) were identified where there were reasonable grounds for concern that the nutrient enrichment may be causing or may lead in time to an undesirable disturbance to the marine ecosystem (see www.ospar.org/ Work Area Eutrophication). The final status classification is derived by taking all relevant information into account.

Estuaries and embayments previously assessed under the EU Urban Waste Water Treatment Directive, the Nitrates Directive or the Habitats Directive as Sensitive Waters (eutrophic) or identified as Nitrate Vulnerable Zone polluted waters (eutrophic), were deemed to be either problem areas or potential problem areas with respect to eutrophication.

Assessments of the risks of eutrophication require adequate information about the background and reference values to be applied in the case of each criterion. Where good historical data exist for a given area this is a simple process. However, in most UK waters the historical record is limited and either proxy evidence or derived values have to be used to set assessment standards and thresholds. A fundamental assumption applied in the OSPAR Comprehensive Procedure assessments was that Atlantic water, which enters the shelf seas of northern Europe, provides a suitable background condition from which to derive standards for the initial assessment. This assumption was used to set standards for nutrient concentrations and to derive standards for nutrient ratios, chlorophyll concentration and the potential level of primary production.

3.3.2.1 Definition of eutrophication

The applicable definition of eutrophication is *...the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned* (taken from the UWWTD, 91/271/ EEC), and is further qualified as referring to *...the undesirable effects resulting from anthropogenic enrichment by nutrients as described in the OSPAR Common Procedure* (OSPAR, 2003b). 'Anthropogenic' includes any human activities which can result in, or contribute to, eutrophication in the marine environment and can be managed, and/or whose contribution to eutrophication can be prevented, reduced or eliminated.

The definition of eutrophication indicates that there are four criteria and three causal links to be considered in an assessment of eutrophication. The first criterion is enrichment by nutrients, the second is accelerated growth of algae and higher forms of plant life, and the third and fourth are disturbance to the balance of organisms present in the water, and to the deterioration of the quality of that water. This formulation has been confirmed in European case law, for example, in the judgement of the European Court in Case C-280/02. One of the key issues to be assessed is that of 'disturbance', which is defined as a perturbation of a marine ecosystem that appreciably degrades the health or threatens the sustainable human use of that ecosystem (Tett et al., 2007b).

3.3.2.2 Environmental impacts

Examples of the negative environmental impacts that may be linked to nutrient enrichment include:

- Changes in the species composition of phytoplankton communities due to changes in relative availability of different nutrients
- Blooms of harmful toxin-producing algae, which accumulate in filter-feeding shellfish, and may affect marine predators through food chain effects or humans through consumption of contaminated shellfish
- Loss of submerged vegetation due to shading
- Development of hypoxic (and anoxic) conditions due to decomposition of excess plant biomass
- Changes in the community structure of benthic animals due to oxygen deficiency or toxic algae
- Fatalities of benthic fauna or fish due to oxygen deficiency
- Scum formation on beaches due to excessive algal growth.

It is not easy to demonstrate the cause and effect relationships between anthropogenic nutrient inputs and undesirable disturbance. In specific cases, it may be easy to demonstrate localised cause and effect relationships between, for example, nuisance/toxic species and human activity. However, on a broader scale this is seldom feasible. During a recent study in the UK sector of the North Sea (north of the Dogger Bank), high frequency results from a fixed mooring show evidence of reduced oxygen concentration (<6 mg/l dissolved oxygen) in bottom waters. This was attributed to thermal stratification restricting the supply of atmospheric oxygen during the summer months. This is likely to be exacerbated by enhanced stratification in response to climate change.

3.3.2.3 Socio-economic impacts

Costing the impact of an undesirable disturbance in the marine ecosystem is difficult, due to the complex relationship between the different ecosystem components and the multifaceted ecosystem services provided by the seas. Direct socio-economic consequences of the negative environmental impacts of eutrophication include loss to fisheries and fisheries-dependent activities, either through reduced catches in the case of direct kills (e.g. fish) or the loss of product in the case of shellfisheries when accumulated toxins have rendered the harvested shellfish unfit for human consumption. Shellfisheries may, at times, be forced to close because of the presence of toxin-producing algae, leading to further economic losses. If poisoned shellfish are consumed, either because of a screening failure or unregulated harvesting, the human consequences can be severe, ranging from diarrhoea, to memory loss, paralysis and death.

The indirect socio-economic impacts of undesirable disturbances in the marine environment are diverse and also difficult to quantify. Rapid growth of phytoplankton and/or opportunistic macroalgae, loss of salt marshes due to increased shading, and hypoxic conditions due to decomposition of accumulated biomass may result in loss of freshwater, estuarine and coastal habitats. Loss of habitats may result in decreased diversity and abundance of fish, birds and mammals, thus leading to economic losses for the associated tourist and recreational industries which are significant. Similarly, unsightly scum or foam formation on beaches may affect the tourism and recreational

industries. Loss of tourist days and damage to tourist infrastructure can lead to economic impact.

Costs associated with reducing inputs of anthropogenic nutrients from point-sources (sewage, industrial) or diffuse discharges (agriculture) into estuarine and coastal waters are significant and need to be weighed against the environmental and socio-economic benefits that can be achieved as well as ensuring compliance with relevant measures.

3.3.3 Developments since *Charting Progress*

Areas were assessed according to a classification scheme, and categorised as non-problem areas (roughly equivalent to 'good' and 'high' ecological status under the Water Framework Directive, see Section 3.3.2), potential problem areas, and problem areas (Box 3.3). Assessment criteria were based on the degree of nutrient enrichment, and the subsequent direct, indirect, and other possible effects in the marine environment. The approach adopted followed that of the OSPAR Strategy to Combat Eutrophication, using the Common Procedure for the Identification of the Eutrophication Status of the Maritime Area adopted by OSPAR. The Common Procedure has two components: a Screening Procedure for the identification of obvious non-problem areas, and a Comprehensive Procedure to determine the eutrophic status of all other areas. The Comprehensive Procedure was applied to areas that had previously been screened as being at risk. Assessments of trophic status were based on a holistic checklist of parameters, known as a set of harmonised assessment criteria (Table 3.44).

3.3.3.1 Developments in key pressures

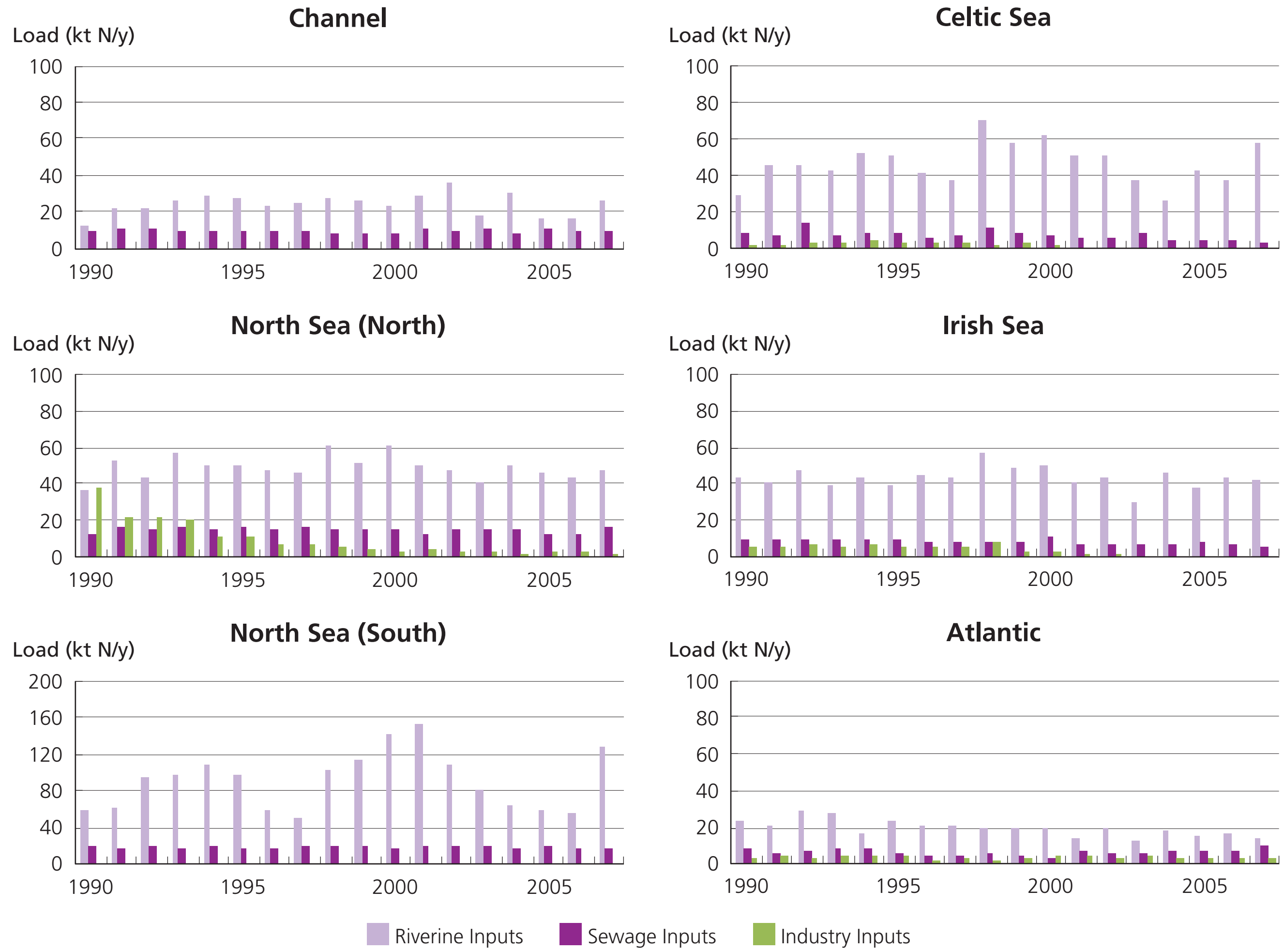
Sources of anthropogenic nutrient enrichment include human and animal waste, industrial processes such as the manufacture and use of fertilisers, run-off from agricultural land or diffuse drainage, the burning of fossil fuels for transport (e.g. shipping) and energy production, and aquaculture. These activities result in nutrients, notably inorganic nitrogen and phosphorus, being released into the aquatic environment and to the atmosphere. Nutrients from land-based sources may enter the sea directly or via rivers. Atmospheric emissions of nitrogen may be deposited into marine waters through precipitation or deposition processes.

Data on nutrient inputs from land-based sources were obtained from the OSPAR Riverine Inputs and Direct Discharges (RID) programme (see also Section 3.1: Hazardous Substances). For reporting to OSPAR, indicative loads are calculated for six main catchment areas, which are broadly similar to the regions adopted for *Charting Progress 2* (Figure 3.65). These loads are calculated from representative subsets of rivers in each region which are monitored routinely, and provide information on relative trends in available data.

Estimates of annual loads of dissolved inorganic nitrogen (Figure 3.66) obtained from the OSPAR RID programme from 1990 to 2007 indicate that the highest loads in all regions were from riverine sources. The lowest values were observed in the Atlantic region, while the highest values were observed in the Southern North Sea. Analyses of significant trends over a 10-year period (1998–2007, Table 3.46) indicate that the main decreases in load have been observed from industrial sources. Sewage loads have decreased significantly in two regions: the Irish Sea and Celtic Sea. Riverine loads (flow

Figure 3.65 UK areas for reporting input of nutrients to transitional and coastal waters to OSPAR. Inputs to small sampling regions (N1&2, E1–E30 and SC1–SC5) are combined into larger areas representing the North Sea (North), the North Sea (South), the Channel, Celtic Sea and Atlantic. These regions are broadly similar to the regions adopted for Charting Progress 2 and calculated loads provide information on relative trends. © Crown copyright 2010: permission granted by Cefas.

Figure 3.66 Annual loads of dissolved inorganic nitrogen to the OSPAR reporting areas from riverine (including natural, industrial, sewage and agricultural sources in freshwaters), industrial and sewage inputs to transitional and coastal waters. Riverine inputs have not been corrected to take account of flow rates. © Crown copyright 2010: permission granted by Cefas.

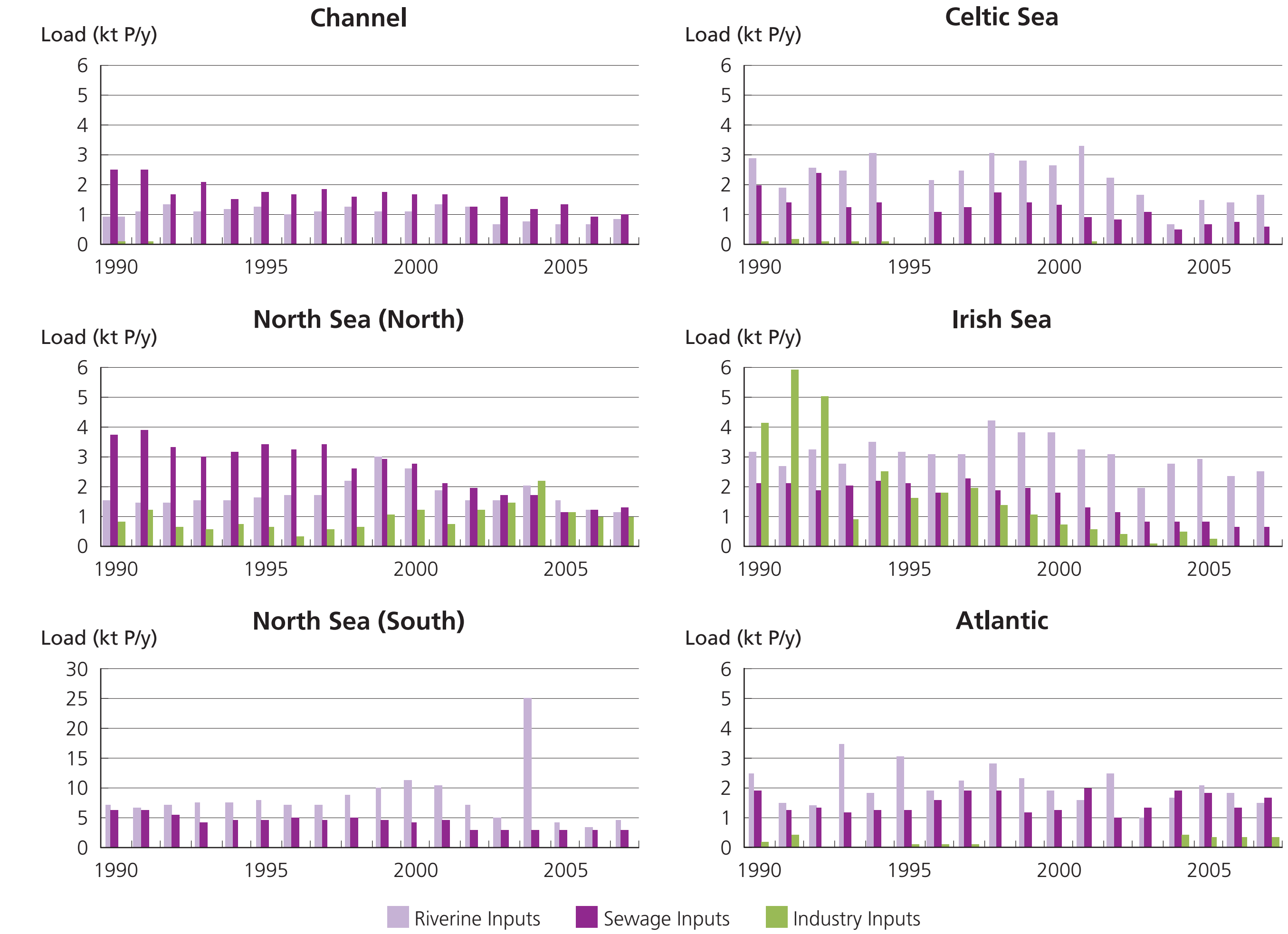


corrected) have shown significant decreases in only two regions: the Channel and the Atlantic. Sewage loads have shown a significant increase in the Atlantic region.

Estimates of annual loads of phosphorus from different sources (Figure 3.67) indicate that the main inputs are region-specific. Sewage inputs appear to be the main source in the Channel and Northern North Sea, whereas riverine and sewage inputs are high in the Southern North Sea and the Atlantic. In the Irish Sea, relative loads indicate that riverine inputs now provide the dominant inputs, whereas industrial loads were the main input in the early 1990s. Analyses of significant trends (1998–2007) indicate that sewage and riverine inputs of phosphorus have decreased in all regions except the Atlantic, while industry inputs have decreased in the Irish Sea and the Celtic Sea (Table 3.46).

Estimates of the total annual loads of dissolved inorganic nitrogen and phosphorus into each of the reporting areas for the OSPAR RID programme from 1990 to 2007 are shown in Figure 3.68. Analyses of significant trends

Figure 3.67 Annual loads of phosphorus to the OSPAR reporting areas from riverine (including natural, industrial, sewage and agricultural sources in freshwaters), industrial and sewage inputs to transitional and coastal waters. Riverine inputs have not been corrected to take account of flow rates. © Crown copyright 2010: permission granted by Cefas.



(1998–2007, Table 3.46) indicate that total inputs of phosphorus have declined by around 6% to 9% per year in all regions except the Atlantic, while total inputs of dissolved inorganic nitrogen have decreased by 2% per year in the Northern North Sea and Irish Sea.

The nutrient loads and observed trends reported here provide strong evidence of the key pressures which may elicit a response in terms of eutrophication. The data are, however, strongly influenced by the spatial and temporal extent of the nutrient monitoring programme and should therefore be considered as being only indicative of overall inputs and trends. Large changes observed in some regions may be attributable to factors such as changes in human activity, enhanced or reduced monitoring, or changes in precipitation. For example, the increase in industrial phosphorus inputs to the Atlantic (Table 3.46) is attributable to a sudden increase observed in the loads into the Clyde. Also, the large decrease in industry load to the Irish Sea (Table 3.46) may be due to the absence of data for the River Eden (the main contributor) from 2004 onwards.

Table 3.46 Estimated percentage annual change in loadings of dissolved inorganic nitrogen and phosphorus into the OSPAR reporting regions (Figure 3.65) over a ten-year period (1998 to 2007). Negative numbers indicate downward trends; positive numbers indicate upward trends. Shading indicates that the change is significant at the 5% level. Riverine loads are shown with and without corrections for flow rates.

Region	*Dissolved Inorganic Nitrogren*					*Dissolved Inorganic Phosphate*				
	Total Load	*Industry Load*	*Sewage Load*	*Riverine Load – no flow correction*	*Rivers – flow corrected*	*Total Load*	*Industry Load*	*Sewage Load*	*Riverine Load – no flow correction*	*Rivers – flow corrected*
North Sea North	-2	-10	-1	-2	-2	-6	4	-10	-7	-7
North Sea South	-1	-14	-1	-1	0	-8	-1	-6	-9	-8
Channel	-2	9	1	-3	-4	-6	-4	-7	-5	-6
Irish Sea	-2	-24	-3	-1	-2	-9	-34	-12	-4	-5
Celtic Sea	-2	-36	-12	-1	-2	-8	-29	-9	-7	-8
Atlantic	0	0	7	-3	-3	-1	38	0	-3	-4

Recent work on inputs of nutrients to the broader North Sea region, using data from 1995 and 1996, has shown that the total input of dissolved nitrogen from UK riverine sources is about 18% of the total North Sea load, and that UK nutrients tend to stay in UK waters.

Atmospheric deposition of nitrogen-containing compounds on UK marine waters results principally from emissions from sources in all the surrounding land areas, not just the UK, and from a range of sea-based sources, such as shipping and offshore activities. Given this mix of UK and non-UK sources, it is convenient to consider UK emissions to ascertain progress made by the UK in reducing this 'atmospheric pathway' pressure.

Emissions of nitrogen can be usefully divided into reduced nitrogen and oxidised nitrogen. The former, which is released principally as ammonia from agricultural sources, is the smaller component. In transport the ammonia, which is very water soluble will tend to deposit and hence decreases significantly in both concentration and deposition rate with increasing distance from sources. The ammonia may also react with other species in the atmosphere to form ammonium salts which make up the majority of deposition to the seas surrounding the UK. The oxidized nitrogen species are emitted principally as nitrogen monoxide (NO) with smaller fractions of nitrogen dioxide (NO_2) often grouped together as NO_X. NO normally makes up 70% to 90% of the total NO_X emissions. In the atmosphere NO may be oxidized to NO_2 and then nitric acid. The latter may then react to form nitrate salts which dominate deposition at a distance from sources. Of less importance is nitrous oxide (N_2O) – this is a very unreactive gas and so while some is oxidised on aerosol surfaces and becomes available to deposit, it is of greatest importance as a potent greenhouse gas.

The time trend of emissions of nitrogen oxides and ammonia based on Defra's UK National Atmospheric Emission Inventory for 2006 are shown in Figure 3.69. Emissions of nitrogen oxides decreased by 78% between 1990 and 2006 and have decreased by 6.4 % since

Figure 3.68 Total annual loads of dissolved inorganic nitrogen and phosphorus to the OSPAR reporting areas. Loads were calculated from riverine, sewage and industrial inputs (see Figures 3.66 and 3.67). © Crown copyright 2010: permission granted by Cefas.

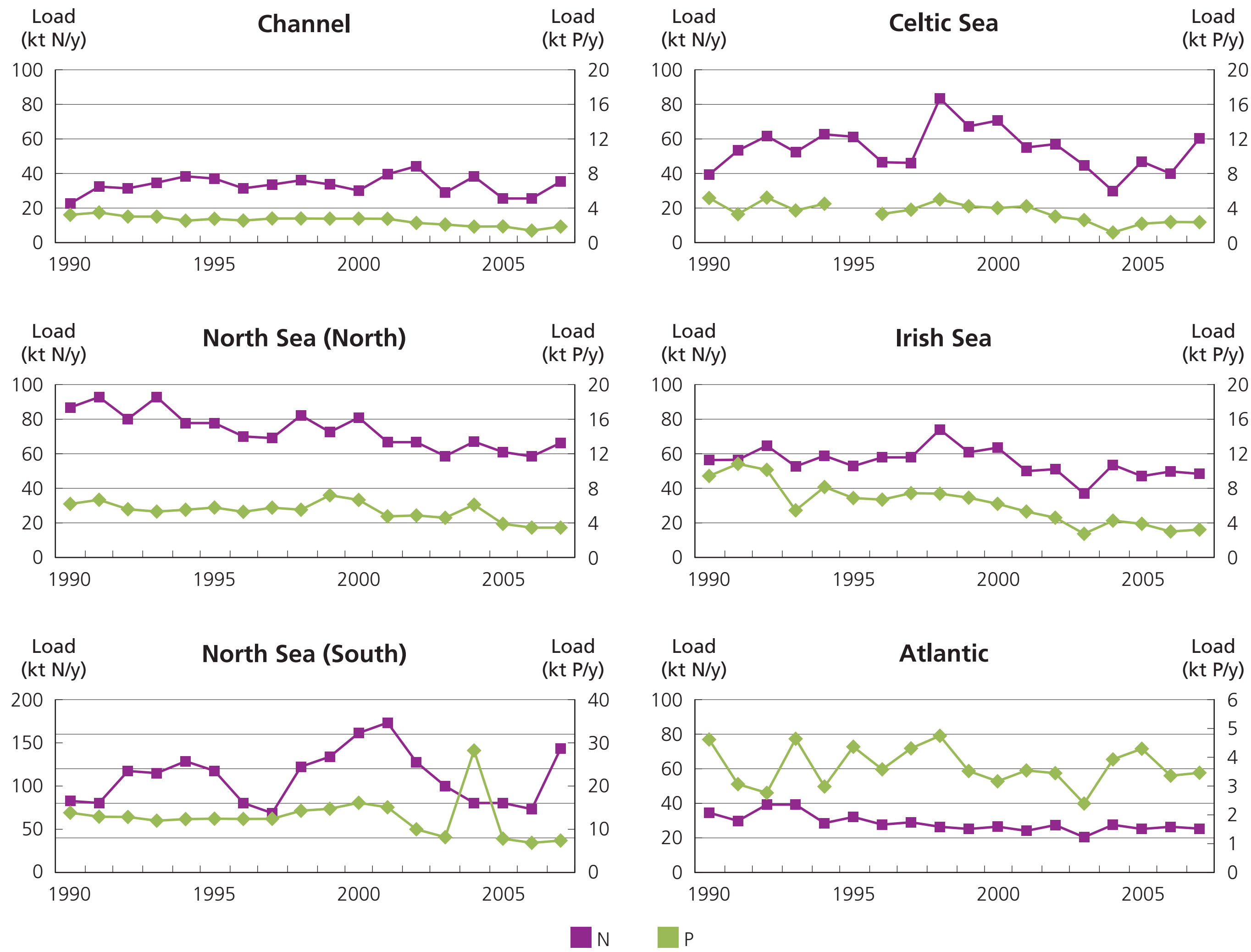
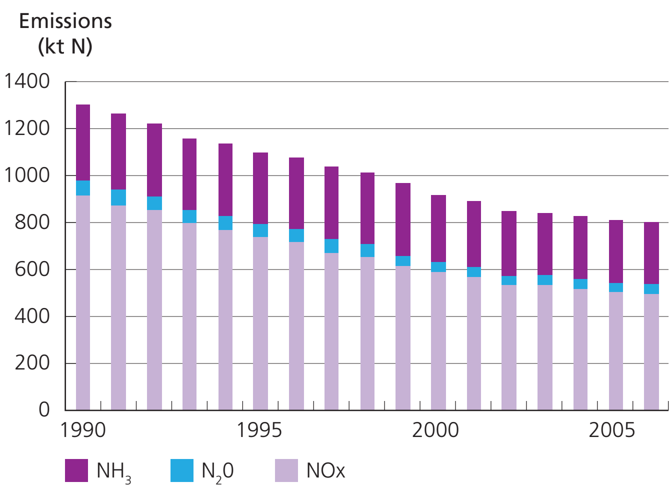


Figure 3.69 Emissions of nitrogen species to the atmosphere between 1990 and 2006. Data are shown for ammonia (NH_3), nitrogen oxides (N_2O) and oxidised nitrogen species (NO_x, i.e. nitrogen monoxide and nitrogen dioxide). © AEA Technology.



the 2002 data reported in *Charting Progress*. The recent reductions are principally due to reductions in emissions from traffic as more modern cars with significantly lower emissions have replaced older vehicles. Reductions have also been due to cleaner emissions from power stations fitted with low NO_X burners, and closures of older coal-fired capacity. For aircraft, emissions attributed to the UK are from all domestic flights, and from takeoff and landing of international flights. The emissions of ammonia have decreased by 20% since 1990 and by 3.5% since 2002 as a result of changes in the numbers of agricultural animals in the UK.

The UK has a range of measurement networks which address nutrient deposition. Some of these sites are coastal and the long-term measurement trends may be taken to indicate the trend in deposition of nutrients at the coast. The measurement results from coastal sites reported to OSPAR are shown in Figures 3.70 and 3.71. A recent OSPAR assessment has suggested that there is no significant downward trend in deposition of nutrients at UK measurement stations (OSPAR, 2009a). This confirms trends observed across the UK as a whole over the period of measurements. However, concentrations of NO_2 have decreased significantly across the UK and there appears to be a non-linearity between emissions and deposition. No significant decrease in deposition has occurred since *Charting Progress*.

The principal inputs of nitrogen are either the nitrate ion or the ammonium ion. Different marine species have different rates of uptake of nitrate and ammonium hence the deposition of each is shown separately for the oxidized and reduced forms in Table 3.47.

Figure 3.70 Trend in ammonium concentrations in rainwater at coastal sites in the UK. © AEA Technology.

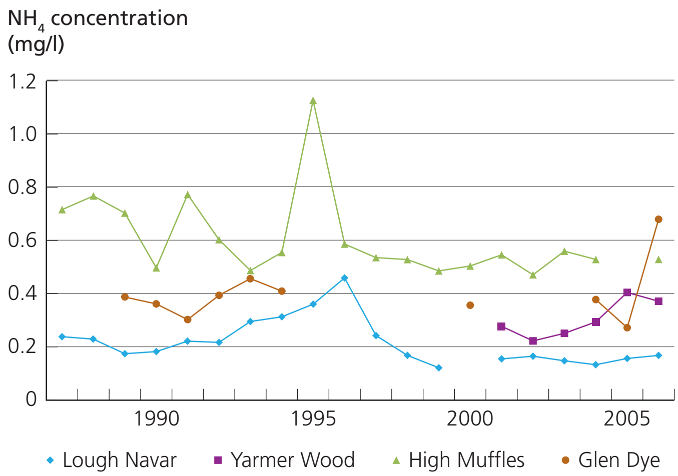
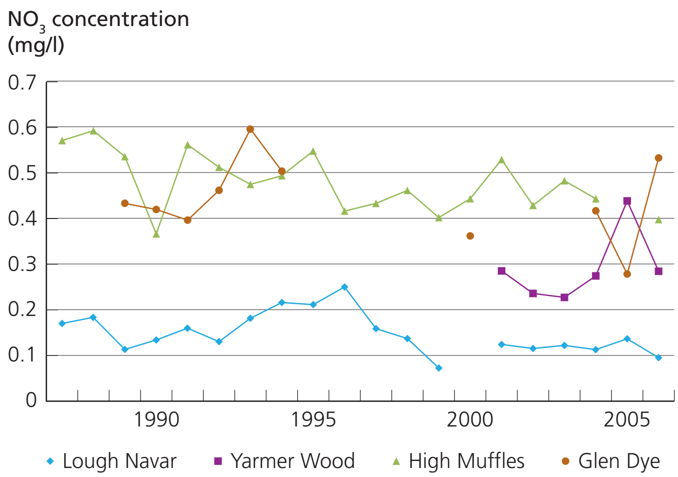


Figure 3.71 Tend in nitrate concentrations in rainwater at coastal sites in the UK. © AEA Technology.



Modelling of the atmospheric transport of nitrogen species has been carried out by the UNECE (United Nations Economic Commission for Europe) European Monitoring and Evaluation Programme's Meteorological Synthesising Centre – West (MSC-W) at the Norwegian Meteorological Institute on behalf of contracting parties to the UNECE Convention on Long-range Transboundary Air Pollution. The model calculates the deposition of oxidized and

Table 3.47 Comparison of land-based nitrogen sources (2006, from the RID programme, Figure 3.68) and atmospheric inputs to the seas surrounding the UK.

CP2 Region		Atmospheric input (kt N)			% of total input of nitrogen from the atmosphere
	UK land-based input (kt nitrogen)	Oxidised nitrogen species	Reduced nitrogen species	% of atmospheric input from oxidised nitrogen species	
1	59	44	34	56	57
2	74	20	20	50	35
3	17	7.0	9.0	44	49
4	45	20	18	52	46
5	57	9.1	17	36	35
6	26	4.8	5.1	48	28
7	0	17	10	62	100
8	0	18	10	65	100

reduced nitrogen species to Europe and the North-West Atlantic. The data outputs provided by MSC-W were used to assess the inputs of reduced and oxidized nitrogen to each of the nine Regional Reporting areas by interpolating from the 50 km grid to a 1 km grid and then summing the deposition in each area. The results are shown in Figure 3.72 as the deposition rate to each sea area and in Figure 3.73 as the total deposition to each sea area. Figure 3.72 shows that the sea areas nearest to the major land masses have the highest deposition rates. This indicates that the more remote the sea area, the lower the deposition rate. However Figure 3.73 shows that the Northern North Sea has a greater total deposition, being moderately polluted. This region is larger in area than the Southern North Sea and the two Channel boxes, which have higher deposition rates.

The results are compared in Table 3.47 with the inputs of nitrogen from the coastline and rivers into the relevant sea area. It can be seen that the atmosphere contributes 28% to 57% of the nitrogen inputs to each of the regional areas with a significant coastline. It can also be seen that the sea areas in which the atmosphere is a more important source of nutrients are those in which a higher proportion of the atmospheric input is from oxidised species. This may reflect the relatively short travel distance of ammonia from coastal agriculture or the relatively low input of land-based sources in less populated catchments.

Existing measures have been developed to address damage to terrestrial and freshwater ecosystems through eutrophication and also to protect human health. As a result, emissions of nitrogen oxides from transport and major combustion sources will reduce up to 2020 and changes in the UK agricultural industry are likely to continue to decrease the emissions of ammonia.

Future trends in emission of oxidized nitrogen species are dependent on both the application of policies to continue to reduce emissions from combustion sources and by the move to lower carbon emission sources for both electricity

Figure 3.72 Atmospheric deposition rates for nitrogen species to the UK seas. © AEA Technology.

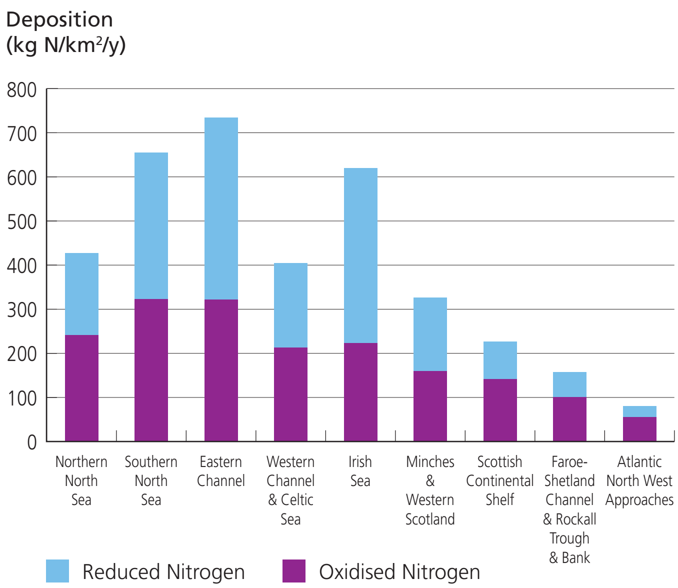
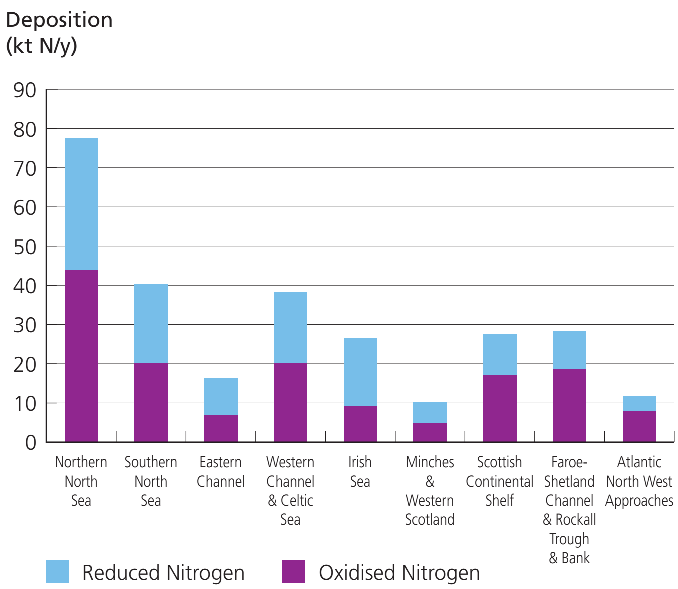


Figure 3.73 Atmospheric deposition of nitrogen species to the UK seas. © AEA Technology.



generation and transport. Trends in ammonia releases depend on how UK agriculture responds to a number of specific policy measures aimed at reducing releases of nutrients to water courses and the atmosphere, and the challenges of change in the UK food market and agricultural support system.

3.3.3.2 Developments in approach, and indicators used

UK waters considered to be at risk from eutrophication were, until relatively recently, identified mainly by measurements of winter concentrations of nutrients as drivers of eutrophication, and their impacts on summer concentrations of phytoplankton chlorophyll. Assessments were made using thresholds set by the Comprehensive Studies Task Team (CSTT, 1994, 1997). However, these measurements provided little information on the disturbance to the balance of organisms and could not adequately identify harmful consequences of nutrient enrichment (Tett et al., 2007b). Since then, considerable effort has been directed towards the development of improved tools for assessing the risks and impacts of eutrophication. Assessment criteria under development by OSPAR, and in relation to the EU WFD and the EU MSFD include descriptors of the ecological responses to nutrient inputs and the identification of any undesirable disturbance to the biology. The OSPAR Common Procedure for the Identification of Eutrophication Status (OSPAR, 2001b, 2003b), for example, uses exceedance of nutrient concentrations above background-related thresholds as one step in a series of diagnostic steps for evidence of eutrophication. The presence of high nutrient concentrations is regarded as a potential cause for concern, and invokes application of the Comprehensive Procedure, which requires assessments of eutrophication status to be based on the agreed harmonised criteria (Table 3.44) and a holistic checklist of parameters. Assessment tools developed under

the WFD provide additional approaches which consider susceptibility of different water bodies to eutrophication.

The most recent assessment of the risks and impacts of eutrophication in UK waters with high nutrient concentrations was carried out under the OSPAR Comprehensive Procedure in 2007 (see Boxes 3.3 and 3.4). In total, 41 areas were assessed. These included coastal and offshore marine waters where salinity was above 30 (see Figure 3.74), and smaller estuaries (transitional waters) and embayments (see Figure 3.75) assessed previously under EU Directives and designated as Sensitive Areas (under the UWWTD), Nitrate Vulnerable Zones (under the Nitrates Directive), or High Priority (under the Habitats Directive). The results of the assessment under the OSPAR Comprehensive Procedure (2007) form the basis of this report on the eutrophication status of UK waters (for details and national and regional reports see www.cefas.co.uk/ospardocs). The coastal and offshore assessment areas used in the assessment do not entirely match the areas adopted for *Charting Progress 2*, but are similar enough to be considered representative of these regions.

After the first application of the OSPAR Comprehensive Procedure, in 2002, the overall UK eutrophication monitoring programme was modified to provide additional surveillance in particular areas of concern. The improved availability of data has resulted in improved confidence in our assessments of the risks of eutrophication, and in the science which underpins our understanding. In addition, a thorough review was undertaken of the thresholds applied for each of the harmonised criteria, taking account of regional differences where this was required, lessons learned from the first application of the OSPAR Comprehensive Procedure and national developments in the field of eutrophication assessment, including work with respect to EU Directives. For simplicity and transparency and where it was scientifically justified, similar thresholds were used across the wide variety of water types in the UK area. As a result of this review, some of the harmonised assessment criteria were developed further to incorporate developments in understanding and to improve confidence in the assessments. Chlorophyll was assessed using the 90th percentile for the March to September growing season, to include high spring-bloom chlorophyll values. The mean and maximum levels were also reported for comparison. For phytoplankton indicator species, an index was used instead of a set of individual species, to provide a better assessment of disturbance to the community. The index includes measures of *Phaeocystis* spp. and any phytoplankton taxa with abundance over a defined threshold. Similarly, for areas where macrophytes are significant, an index was developed to include the area covered by macrophytes and the biomass of opportunistic species. This national development is being considered for adoption by the Intercalibration Process for the EU WFD.

The second application of the OSPAR Comprehensive Procedure, in 2007, has therefore helped to contribute towards an improved understanding of the eutrophication status of UK waters, as well as the assessment methods for each of the harmonised criteria.

3.3.3.3 Developments in research

Coastal observatories have been established in Liverpool Bay and the North Sea (see www.emecogroup.org), to improve the spatial coverage, quality and frequency of *in situ* monitoring of environmental conditions

Box 3.4:
Data sources for assessments of risks and impacts of eutrophication

All sources of data used for recent assessments of eutrophication are summarised on Table 3.48. Data were available from four main sources: the Clean Safe Seas Environmental Monitoring Programme (CSEMP, previously the National Marine Monitoring Programme, NMMP) held at the Centre for Environment, Fisheries and Aquaculture Science (Cefas); the Water Framework Directive (WFD) databases on UK nutrients, chlorophyll, phytoplankton and dissolved oxygen held by the Environment Agency (EA) and the Scottish Environment Protection Agency (SEPA); the SmartBuoy monitoring network (Cefas, see www.cefas.co.uk/monitoring); and the OSPAR Riverine Inputs and Direct Discharges (RID) programme (EA and SEPA). High frequency data were obtained from SmartBuoy monitoring systems deployed in surface waters at two sites in the Thames embayment, at one site in Liverpool Bay (north-eastern Irish Sea) and at one site in the southern North Sea (see Figure 3.65). For the north-eastern Irish Sea data sources were supplemented with data from Kennington et al. (2003, 2004, 2005).

The WFD databases were compiled to meet the requirements of the Directive (2000/60/EEC). Data from five regions (England, Wales, Scotland, Northern Ireland and the Republic of Ireland) were geographically linked to WFD coastal (salinity 30–34) and transitional water bodies (salinity <30), and did not cover the entire geographic regions of all assessment areas used here, particularly for offshore waters.

Opportunistic green macroalgae, for example, are monitored at sites known to have had macroalgal blooms in the recent past or present; such as the southern estuaries of Langstone and Chichester (EA). In Wales, the Countryside Council for Wales (CCW) intertidal monitoring Phase 1 project mapped opportunistic macroalgae.

There is no national monitoring programme specifically investigating the effects of eutrophication on the zoobenthos. Regular monitoring has, however, been undertaken by Cefas to assess the effects of disposal of dredged material, under the Food and Environment Protection Act, the effects of aggregate extraction on benthic communities, and the ecological status of former (pre-1998) sewage sludge disposal sites. Through CSEMP, sampling occurs at a suite of stations in estuaries, coastal areas and offshore areas.

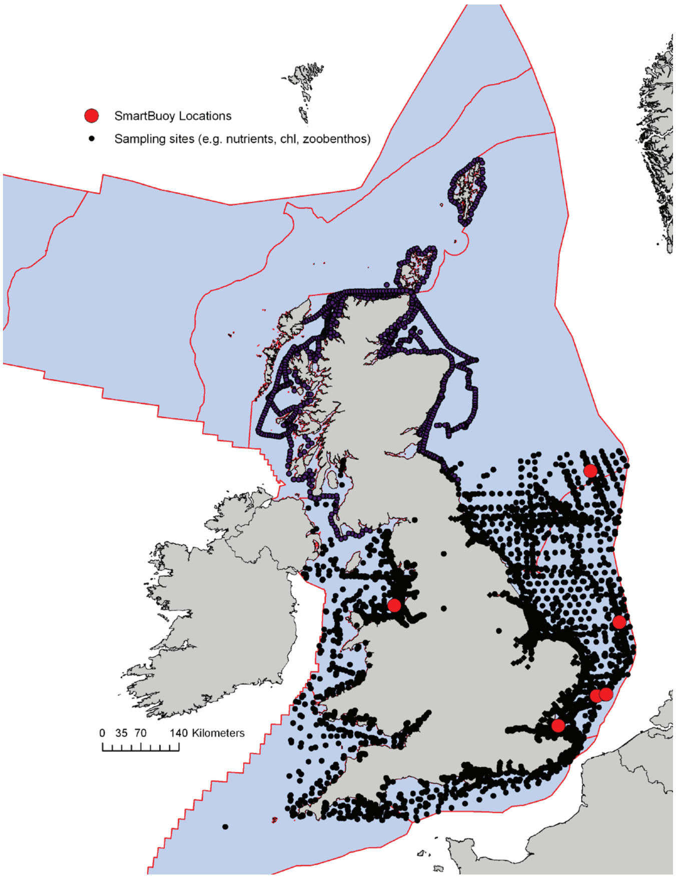
The availability of data from smaller transitional and coastal areas was highly variable. Where possible, data were obtained from the sources outlined above. All data sources are described in the reports for these regions under the OSPAR Comprehensive Procedure (see www.cefas.co.uk/ospardocs).

Table 3.48 Summary of data sources for recent assessments of eutrophication in UK waters.

Assessment parameter	*Type of data collection*	*Data (start/ end dates)*	*Number of samples*	*Data (organisations)*
Category I - Degree of nutrient enrichment				
1. Riverine inputs and direct discharges (area specific)	RID Programme	1992–2007	N/A in North Sea (north and south). In other regions, annual loads were calculated from the OSPAR Riverine Inputs and Direct Discharges (RID) programme.	Environment Agency (EA), (SEPA)
2. Nutrient concentrations (area specific) Elevated level(s) of winter DIN and/or DIP	EA, CSEMP (Cefas) SmartBuoy	1999–2004/5/6 2001-2006	45 in North Sea (north) 577 in North Sea (south) 2502 on Eastern English Coast 2656 East Anglia + 456 SmartBuoy 247 in East English Channel 1122 in Solent and southern estuaries 213 in Bristol Channel 1361 in Liverpool Bay + 384 SmartBuoy 849 in North-Eastern Irish Sea	CSEMP agencies, EA, Cefas Cefas
3. N/P ratio (area specific) Elevated winter N/P ratio (Redfield N/P = 16)	EA, CSEMP (Cefas)	1999–2004/5/6	45 in North Sea (north) 579 in North Sea (south) 3567 in East Anglia 291 in East English Channel 1425 in Solent and southern estuaries 231 in Bristol Channel 1362 in Liverpool Bay 839 in North-Eastern Irish Sea	CSEMP agencies, EA, Cefas
Category II - Direct effects of nutrient enrichment (during growing season)				
1. Chlorophyll a concentration (area specific)	CSEMP (Cefas), EA SmartBuoy	1999–2003/4/5/6 2001–2005/6	442 North Sea (north) 612 North Sea (south) 2032 in East Anglia 530 in East English Channel 844 in Solent and southern estuaries 639 in Bristol Channel 1689 in Liverpool Bay + 507 SmartBuoy 1021 in North-Eastern Irish Sea + 102 SmartBuoy	CSEMP agencies, Cefas, EA, University of Liverpool (NEIS), Cefas
Elevated maximum and mean level			753 daily means in North Sea (south) + 855 SmartBuoy	
2. Phytoplankton indicator species (area specific) Elevated levels of nuisance/toxic phytoplankton indicator species (and increased duration of blooms)	UK Phytoplankton database	1999–2004/5	No data in offshore areas 492 samples in WFD waterbodies in East Anglia ~150 samples in WFD water bodies 107 samples in WFD water bodies in Solent and southern estuaries 28 samples in WFD water bodies in Bristol Channel 201 samples in WFD water bodies in Liverpool Bay 876 samples in WFD water bodies in North East Irish Sea	EA, SEPA

3. Macrophytes including macroalgae (area specific)	EA, SEPA, CCW Phase I intertidal mapping		No data in offshore areas Surveys in 4 estuaries inn East Anglia No recorded problems in East English Channel 15 surveys in Solent and southern estuaries Whole Welsh coastline mapped No recorded problems in Liverpool Bay 3 coastal surveys in WFD water bodies in North East Irish Sea	EA, SEPA, Countryside Council for Wales (CCW)
Category III – Indirect effects of nutrient enrichment (during growing season)				
1. Oxygen deficiency Decreased levels (<2 mg/l: acute toxicity; 2–6 mg/l: deficiency) and lowered % oxygen saturation	EA & CSEMP (Cefas) SmartBuoy	2001–2004/6 1999–2006 2002–2006	123 North Sea (north) 30 North Sea (south) 581 in East Anglia 373 in East English Channel 212 in Solent and southern estuaries 1104 in Bristol Channel 383 in Liverpool Bay + 55 SmartBuoy 138 in North-Eastern Irish Sea	EA & CSEMP (Cefas) Cefas
2. Zoobenthos and fish Kills (in relation to oxygen deficiency and/ or toxic algae) Long-term area-specific changes in zoobenthos biomass and species composition	Fisheries & benthic data	Fish: 1998–2006 Benthos: 2000	1 benthic site & a spatial study in North Sea (north). Sea Fisheries do not cover offshore waters 1 benthic station & a spatial study in North Sea (south) 2 Thames benthic sites 1 benthic site in East English Channel 4 benthic sites in Bristol Channel 4 benthic sites in Liverpool Bay 1 benthic site in North East Irish Sea	Northeast Fisheries, Kent & Essex Fisheries Committee, Southern Fisheries Committee, South-Wales Fisheries Committee, Northwest Fisheries Committee, Cefas
3. Organic Carbon/ Organic Matter	N/A			
Category IV – Other possible effects of nutrient enrichment (during growing season)				
1. Shellfish flesh samples Incidence of DSP/PSP/ ASP mussel infection events (related to II.2)	Biotoxin monitoring programme for FSA	1999–2006	No data in offshore areas 197 in Eastern English Coast 310 in East Anglia 88 in East English Channel 271 in Solent and southern estuaries 248 in Bristol Channel 255 in Liverpool Bay 82 in North-Eastern Irish Sea	FSA, Cefas

Figure 3.74 Monitoring locations for data used to support the CSSEG eutrophication assessment. Locations of in situ *monitoring buoys (SmartBuoys, see Pearce et al., 2002) are also shown. © Crown copyright 2010: permission granted by Cefas.*



relevant to eutrophication. High-frequency *in situ* monitoring systems (SmartBuoys) deployed in these regions provide valuable data for improving present and future nutrient monitoring programmes and therefore for improving confidence in assessments of eutrophication (Case Study 1; see Heffernan et al., 2010). Ship-based studies continue to be combined with moorings to further improve our understanding (Weston et al., 2004, 2008a,b).

The evidence base on the availability of light in UK waters has been improved due to the establishment of robust relationships between turbidity (as measured by concentrations of

Figure 3.75 Smaller estuaries and embayments assessed using the OSPAR Comprehensive Procedure due to previous designations under EU Directives (UWWTD, Nitrates Directive, and Habitats Directive). (a) Scotland, Northern Ireland and northern England, and (b) English south coast and Wales. Red areas are those deemed problem areas and green areas are those deemed non-problem areas (see Figure 3.82). © Crown copyright 2010: permission granted by Cefas.

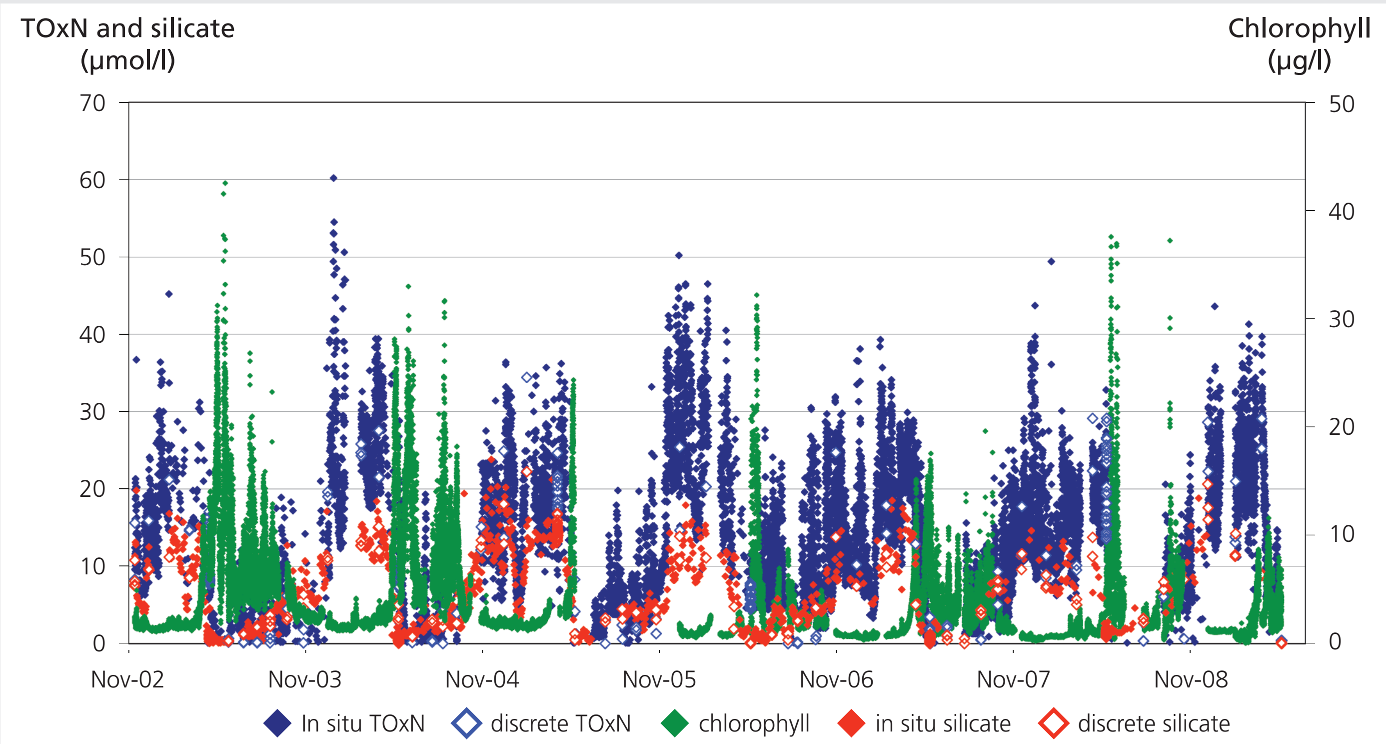
suspended particulate matter or Secchi depth) and K_d , the vertical extinction coefficient for irradiance (Devlin et al., 2008; Foden et al., 2008). New and improved data have supported the development of risk assessment tools which take into account the susceptibility of different

Case Study 1: Liverpool Bay

Liverpool Bay is a coastal region of freshwater influence (salinity 30–34) that receives waters from the Mersey, Ribble and Dee estuaries. There is clear evidence of nutrient enrichment in Liverpool Bay but there is also robust evidence that neither accelerated growth of phytoplankton nor undesirable disturbance to the balance of organisms occurred during the assessment period (2001 and 2005, see Gowen et al., 2002; Tett et al., 2008).

Since November 2002, high frequency fixed point observations obtained by deployment of a Cefas SmartBuoy have formed a key part of the monitoring strategy in Liverpool Bay. This has resulted in a large volume of data leading to more robust conclusions about eutrophication status. Figure 3.76 shows a time series of daily TOxN (nitrate + nitrite) concentrations measured at 2-hourly intervals using an *in situ* nutrient analyser and in the laboratory from water samples collected 3 to 4 times each week and stored automatically on the SmartBuoy. These data are used to calculate the over-winter DIN

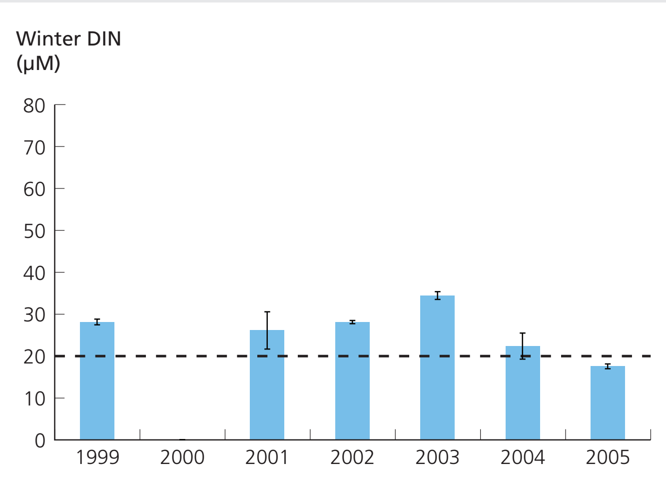
*Figure 3.76 A time series of daily nitrate + nitrite (TOxN), silicate and chlorophyll concentrations (*in situ* data) from the Cefas SmartBuoy in Liverpool Bay, together with the results from periodic ship-based surveys (discrete). Measurements are made every 2 hours using a NAS-2E nutrient analyser for TOxN and several times each week with the waters sampler for TOxN and silicate. Chlorophyll is based on data from a fluorometer that measures twice each hour for 10 minute periods at 1 Hz. The analyser carries an onboard standard and is also compared with ship-based measurements carried out alongside the buoy. Preserved water samples collected on the buoy are returned to the laboratory for subsequent analysis. All daytime measurements of chlorophyll fluorescence are removed prior to calibration against extracted chlorophyll concentration measured in water samples collected alongside the buoy by ship. © Crown copyright 2010: permission granted by Cefas.*



(dissolved inorganic nitrogen) concentration and determine whether Liverpool Bay is enriched with nutrients (Figure 3.77).

Evidence of accelerated growth is provided by the level of chlorophyll during the growing season (March to September). Assessment of this parameter has been improved by using measurements from SmartBuoy. A high frequency time series of SmartBuoy chlorophyll concentration is shown in Figure 3.76. SmartBuoy chlorophyll data augment the dataset that is assessed against an appropriate threshold (Figures 3.78). For the assessment period (2001 to 2005) there was judged to be no evidence of accelerated growth.

Figure 3.77 Winter (November to March) nutrients in waters of coastal salinity (30–34) in Liverpool Bay, with mean dissolved inorganic nitrogen concentrations normalised to a salinity of 32. Error bars indicate 95% confidence limits, the dashed line indicates the threshold value for data normalised to a salinity of 32 (there are too few data for coastal ratios in 2000). The SmartBuoy measurements began in November 2002. Winter nutrient concentrations above the threshold are regarded as evidence of enrichment. © Crown copyright 2010: permission granted by Cefas.



The use of SmartBuoy high frequency time series measurements has not only improved the quality of the evidence base used in the assessment of the eutrophication status of Liverpool Bay but has also provided insights into the dynamic nature of the region. Figure 3.76 shows the large degree of variability of nutrients (TOxN and silicate) and chlorophyll on a variety of time scales. Short-scale (hours to days) variability in nutrients is strongly driven by tides and episodic inputs of freshwater due to increased runoff. The winter build-up of nutrient concentrations is followed by a rapid drawdown in the spring associated with the timing of the spring bloom that is a prominent feature in Liverpool Bay. Significant inter-annual variability is also apparent in all of these time series demonstrating the challenge faced by traditional monitoring approaches in resolving the variability of key ecosystem variables that are used as indicators of eutrophication status.

As well as spatial information from ships, satellite remote sensing is beginning to provide more robust information on chlorophyll in turbid coastal waters. Figure 3.79 shows a map of chlorophyll in Liverpool Bay derived from satellite data together with surface concentrations of chlorophyll measured along a track sailed by a research vessel. There is good agreement between the different datasets and this provides a way of establishing good confidence in the information used to derive status assessment.

Figure 3.78 Chlorophyll concentrations during the growing season in coastal waters (salinity 30–34) in Liverpool Bay, from spatial (ship-based) and SmartBuoy observations. (a) 90th percentiles and (b) means and 95% confidence limits.

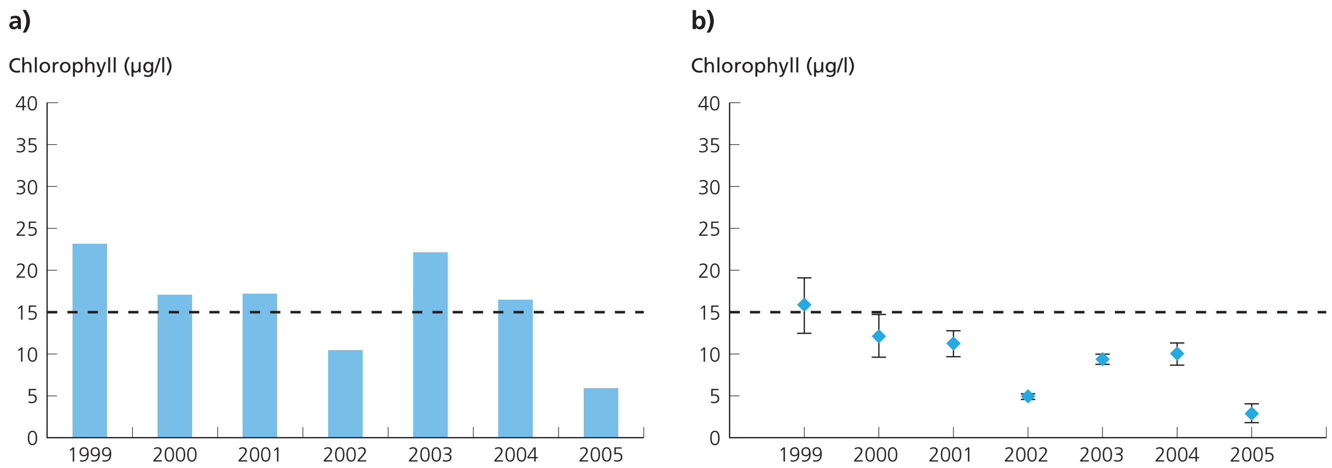
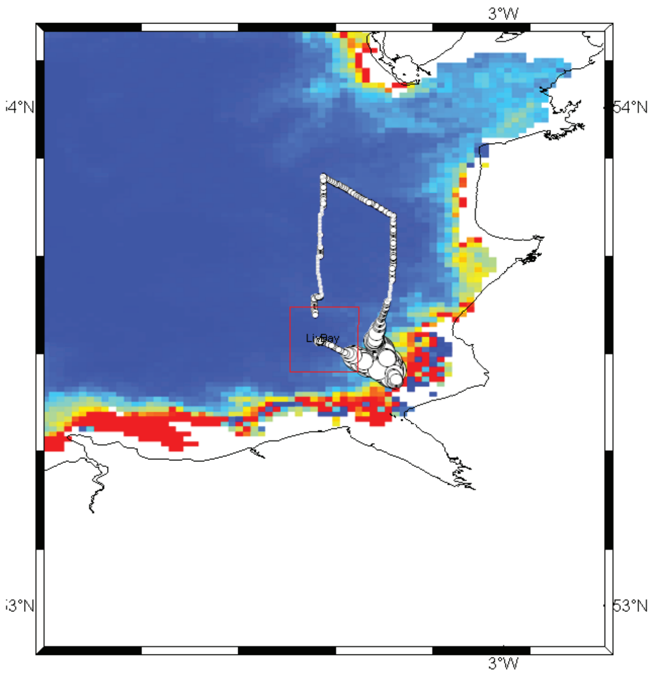


Figure 3.79 Satellite remote sensing of chlorophyll derived from ocean colour measurements in Liverpool Bay on 10 May 2006 derived from MODIS radiance measurements using the OC5 algorithm for Case-II waters. Blue represents low and red high values of chlorophyll. The track of the research vessel and the relative surface concentration of fluorescence are also plotted. In this image higher surface concentrations are visible inshore by remote sensing and from underway surface chlorophyll fluorescence measurements compared to those offshore and in the region of the SmartBuoy indicated by the red box.



water bodies, particularly estuaries and coastal waters, to eutrophication (Painting et al., 2007; see Case Study 2).

In Scotland, six long-term monitoring sites on the east and west coast coasts, and the Western and Northern Isles are regularly sampled by scientists and local volunteers which, combined with targeted ship-based monitoring activities, provide the monitoring base for assessing nutrient levels and their possible eutrophication effects. An ecosystem model for Scottish sea lochs has been developed which aims to predict the susceptibility of these environments to increases in nutrient inputs from fish farms (Tett et al., 2007a; Gubbins et al., 2008). Monitoring is being undertaken to gather data on underwater light to better define the relationships between light attenuation, suspended particulate matter, chlorophyll and salinity in these systems to improve the models. Fish farm developments in sensitive sea loch environments are carefully managed to avoid the risk of eutrophication.

The development of tools for assessing the risks and impacts of eutrophication includes the development of Ecological Quality elements and objectives for monitoring nutrient enrichment and potential eutrophication effects (OSPAR, 2005). Five Ecological Quality elements were originally identified (Table 3.50 on Page 220); for each element, desired levels (Ecological Quality Objectives, EcoQOs) were defined in order to assess potentially negative impacts of eutrophication. These EcoQOs, reviewed by the International Council for the Exploration of the Sea (ICES), have been tested and developed further in a number of studies, including a pilot project for the North Sea. For the ICES review, criteria for good indicators were that they should respond to anthropogenic influences, be generally present in coastal waters, be measurable with high accuracy and precision, have well-defined reference conditions, be cost-effective, and be easy to communicate to the public. Under its Safeguarding our Seas strategy, the UK is committed to testing and reviewing these EcoQOs for the coastal waters of England and Wales.

There has been considerable progress in the development of numerical ecosystem models for assessing risks and impacts of nutrient enrichment. Recent work with ecosystem models has shown that subsurface oxygen minima observed at sites in the stratified North Sea (e.g. at the Oyster Grounds) occur naturally and do not require an external supply of anthropogenic nutrients. The development of a one-dimensional model to simulate the concentration of suspended inorganic material in coastal waters provides an important step towards improved modelling of the submarine light regime and will lead to improved models to simulate eutrophication.

An initial application of the PCI (Phytoplankton Community Index) in the eastern Irish Sea has been carried out with no evidence of Undesirable Disturbance in Liverpool Bay (see Tett et al., 2008).

3.3.4 Presentation of the evidence

3.3.4.1 Evidence and changes in evidence

The results of the recent assessment of eutrophication status of UK seas generally confirm those of the first application in 2002. The evidence clearly shows, with a good degree of confidence, that most coastal and marine waters around the UK are non-problem areas with respect to eutrophication and show no signs of undesirable disturbance. However, the evidence confirms that there are a number of small estuaries, loughs and harbours which are problem areas with respect to eutrophication,

Case Study 2: Estuaries in the east of England

A simple box model was used to predict the magnitude of growth by the phytoplankton and macroalgal communities in five east coast estuaries (the Humber, Wash, Thames, Colne/Blackwater and Deben) in response to nutrient input. This Combined Phytoplankton Macroalgal (CPM) model was developed by combining two earlier models, one for phytoplankton (see Painting et al., 2007), based on the CSTT approach, and one for macroalgae (Aldridge et al. unpubl, Cefas). In the model, growth is strongly influenced by a number of factors, including the retention time of water, relative light regimes, estuary depth, and the extent of the intertidal area suitable for macroalgal growth (see Table 3.49).

The model was run using average (2000–2005) catchment-corrected loads of dissolved inorganic nitrogen (Figure 3.80) and precautionary site-specific data per estuary (e.g. slowest water exchange rates, zero losses of nitrogen due to denitrification). Model results (Figure 3.80a) showed no phytoplankton production in the Humber, the Wash or the Thames. This was attributed to light limitation, exacerbated by the mean depth of the estuaries. Observed phytoplankton biomass (chlorophyll) was considered to have been advected in from adjacent coastal water. In the absence of phytoplankton growth in the Humber, Wash and Thames, the model predicted growth by macroalgae, with highest production rates (245 g C / m^2 per year) in the Humber. These results were sensitive to estimates of the intertidal area available for macroalgal growth, which are influenced by factors such as substratum type and exposure.

For the Colne, Blackwater and Deben, model results indicated relatively high levels of production by phytoplankton, largely attributable to more favourable light conditions in these estuaries (see Table 3.49, smaller values for K_d). Relatively high nutrient concentrations in these estuaries (Figure 3.80b) were attributable to their relatively small volumes compared with the Humber, Thames and Wash where higher nitrogen loads were diluted by larger estuary volumes.

Table 3.49 Site-specific data for assessing the susceptibility of five east coast estuaries to nutrient enrichment using a simple box model. WFD data were used to estimate the total area, and the intertidal area available for macroalgal growth. Average depths were obtained from mean sea level and used to calculate estuary volumes and exchange rates using the tidal prism method. The values for average light attenuation were obtained from previous work (see Devlin et al., 2008).

Estuary	*Area, km^2*	*Average depth, m*	*Available intertidal area, %*	*Exchange rate, E/d*	*Mean K_d/m*
Humber	326.5	7.4	11.3	0.08	8.52
Wash	611.6	6.9	8.8	0.07	4.62
Thames	453.2	6.9	5.9	0.07	4.3
Colne	7.0	2.9	28.9	0.13	2.69
Blackwater/Colne	52.3	3.4	28.9	0.13	2.3
Deben	7.8	3.1	29.0	0.09	3.84

The CPM model results were compared with the scale proposed by Nixon (1995) for assessing trophic status from net annual primary production estimates: oligotrophic: 0–100 g C / m^2; mesotrophic: 101–300 g C / m^2; eutrophic: 301–500 g C / m^2; hypertrophic: >500 g C / m^2. Estuaries identified as being eutrophic according to this scale may not show any signs of eutrophication as defined by the EU Urban Waste Water Treatment Directive (CEC, 1991b). Additional attributes need to be considered to assess negative impacts. None of the predicted levels of production exceeded Nixon's threshold for eutrophic.

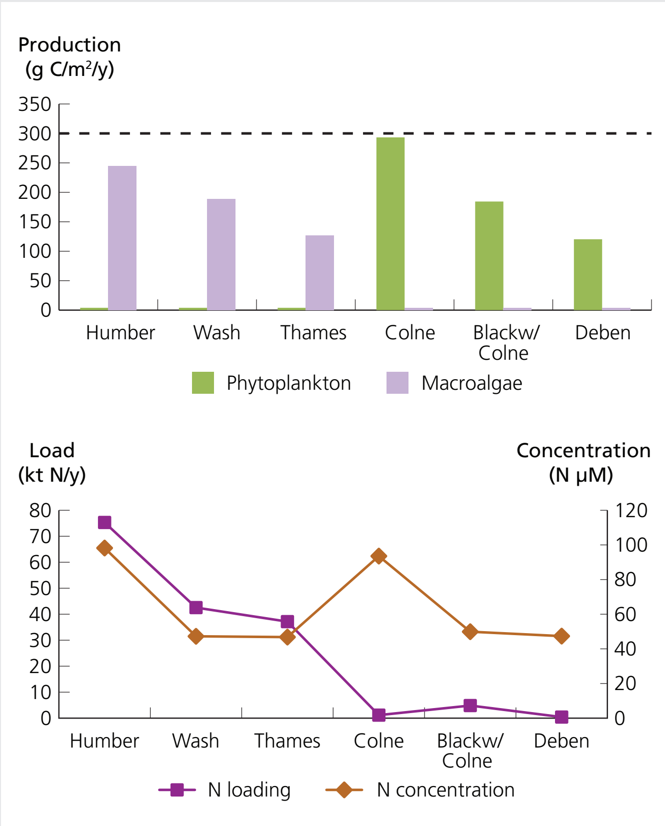


Figure 3.80 (a) Summary of results from the Combined Phytoplankton Macroalgal (CPM) model used to assess susceptibility of east coast estuaries to the input of dissolved inorganic nitrogen. Average loads (b) were calculated using catchment corrected data from 2000 to 2005. The model calculates in situ nutrient concentrations from data on estuary volume, and predicts primary production (g C / m^2 / y) by phytoplankton and macroalgae. The dashed line in (a) indicates a threshold of 300 g C / m^2 / y, proposed by Nixon (1995) for assessing eutrophic status, but not necessarily eutrophication as defined by the Urban Waste Water Treatment Directive. © Crown copyright 2010: permission granted by Cefas.

Table 3.50 Ecological Quality Elements and Objectives for monitoring and assessing the biological response to nutrient enrichment (Bergen Declaration, 2002: Annex 3, Table B).

Ecological quality element	*Ecological quality objective*
(m) Changes/kills in zoobenthos in relation to eutrophication	There should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species
(q) Phytoplankton chlorophyll a	Maximum and mean chlorophyll a concentrations during the growing season should remain below elevated levels, defined as concentrations > 50% above the spatial (offshore) and/or historical background concentration
(r) Phytoplankton indicator species for eutrophication	Region/area - specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration)
(t) Winter nutrient concentrations (dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphate (DIP))	Winter DIN and/or DIP should remain below elevated levels, defined as concentrations > 50% above salinity-related and/or region-specific natural background concentrations
(u) Oxygen	Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region-specific oxygen deficiency levels, ranging from 4–6 mg oxygen per liter

Note: The Scheveningen workshop (Skjoldal et al. 1999) defined Ecological Quality (EcoQ) as *...an overall expression of the structure and function of the aquatic systems*, and Ecological Quality Objectives (EcoQOs) as *...the desired level of the EcoQ relative to the reference level. Reference level was defined as the level of the EcoQ where the anthropogenic influence on the ecological system is minimal.*

or are at risk due to factors such as restricted circulation. Examples of assessments from the Ythan, Belfast Lough and Scottish sea lochs supporting aquaculture are given in Case Studies 3, 4 and 5, respectively.

The overall results (see Figures 3.81 and 3.82) for the UK were: 19 areas classified as non-problem areas (green shading), 17 areas classified as problem areas (red shading) and 5 areas classified as potential problem areas (amber shading), with respect to eutrophication.

The areas assessed fall into two categories.

- Coastal and offshore marine waters (salinity >30) – which were identified as non-problem areas in 2002 remain non-problem areas. There is more confidence in the results of the current assessment, especially in the coastal areas identified in 2002 as areas of particular ongoing interest, due to enhanced monitoring and research programmes that were designed to detect any adverse anthropogenic related

Figure 3.81 Final classification for UK assessment areas. © Crown copyright 2010: permission granted by Cefas.

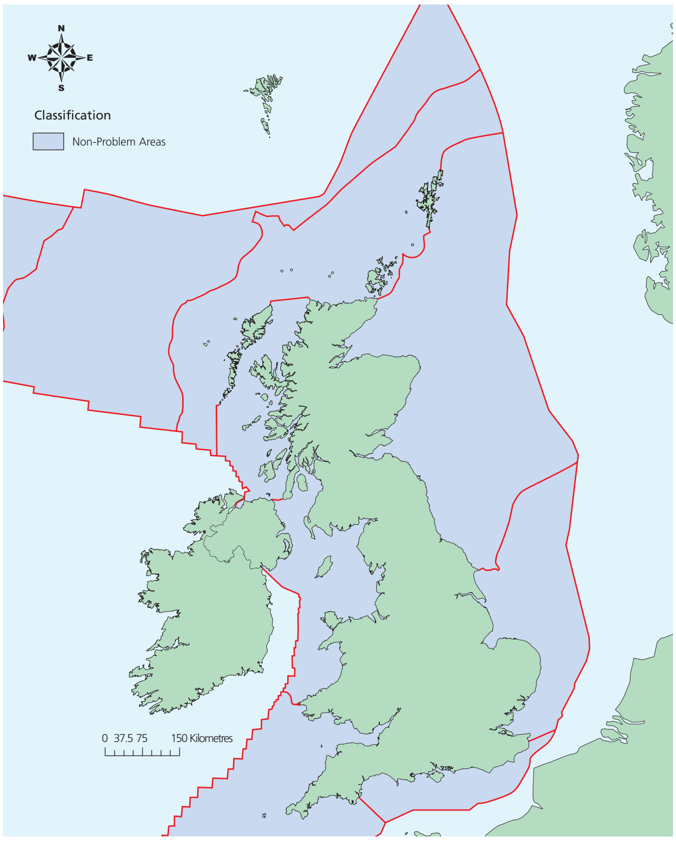
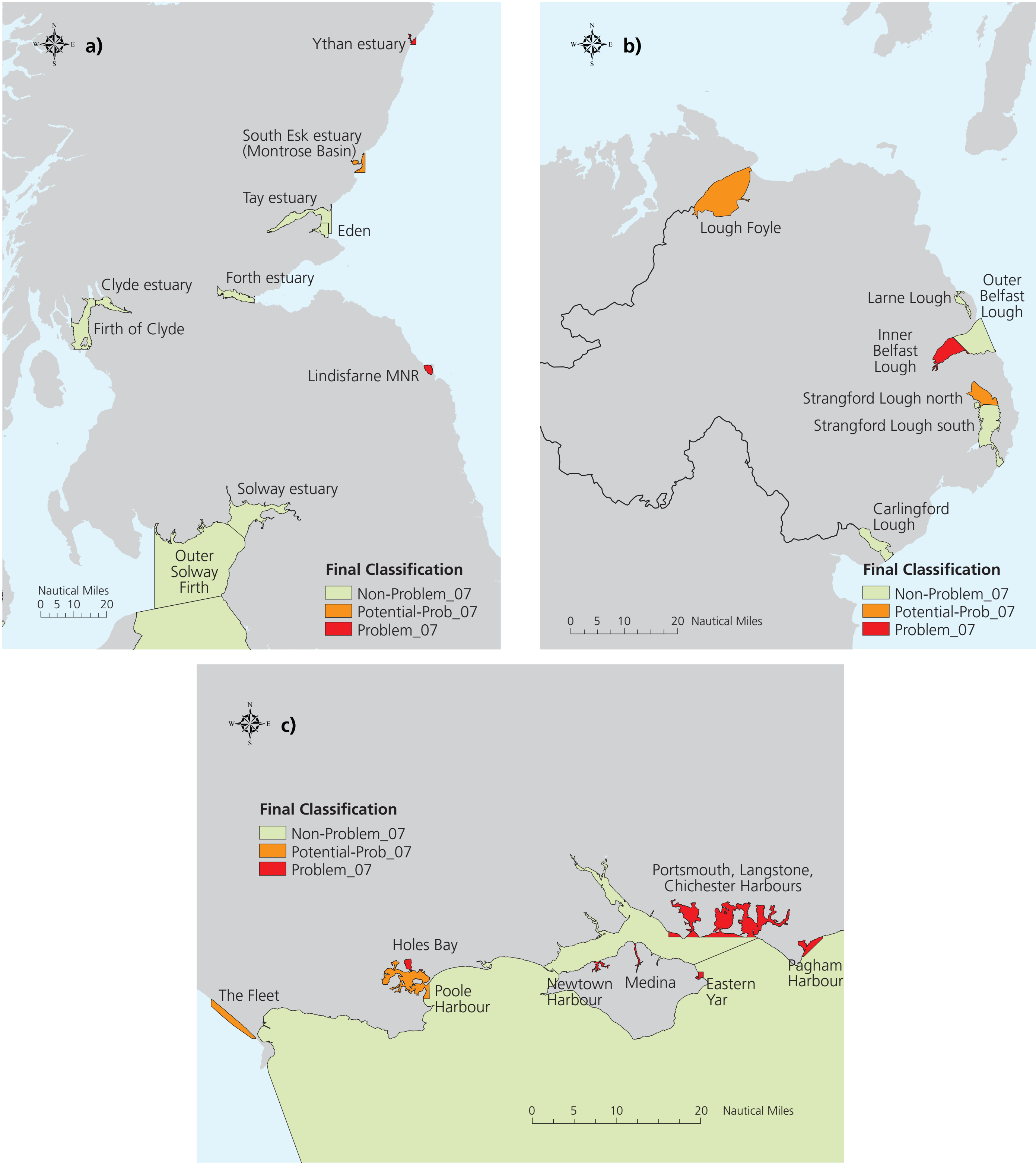


Figure 3.82 Assessment areas in (a) Scotland, (b) Northern Ireland, and (c) English south coast showing final classifications from the second application of the OSPAR Comprehensive Procedure. Note: in smaller estuaries (Montrose Basin and Ythan) shading indicates the estuary and not the coastal waters. © Crown copyright 2010: permission granted by Cefas.



Case Study 3: The Ythan

The Ythan is a small (8 km) estuary approximately 20 km north of Aberdeen in NE Scotland (Figure 3.83). The catchment of the estuary was designated as a Nitrate Vulnerable Zone (NVZ) under the Nitrates Directive in 2000 due to high nitrate concentrations in the ground water. The estuary was designated as a Sensitive Area under the Urban Waste Water Treatment Directive because of the presence of opportunistic green algae on the intertidal zone (Figure 3.84). It was also designated as an OSPAR problem area in 2002 following the first application of the Comprehensive Procedure.

Measures to reduce diffuse inputs of nitrogen from agriculture were initiated as a result of the NVZ designation. Nutrient inputs from waste water were reduced as a consequence of the Sensitive Area designation by the installation of phosphate stripping at Ellon waste water treatment works and the removal of the smaller discharge from Newburgh from the estuary.

The trophic status of the estuary was reviewed for the second application of the OSPAR Comprehensive Procedure (Table 3.51) using monitoring data collected up to and including 2005 (for details see under Scotland Reports at www.cefas.co.uk/ospardocs). High inter-annual variability was observed in freshwater loads of dissolved inorganic nitrogen (DIN), observed concentrations of DIN, and the growth of opportunistic green algae in the intertidal zone. This variability results from changes in climatic conditions which influence river flow, water temperature and insolation. Although data indicate a decrease in DIN loading after 2000, concentrations of DIN in the estuary continue to exceed the assessment criterion, and extensive growths of opportunistic green algae continue to be observed in intertidal areas. Occasional observations of relatively high concentrations of chlorophyll at low salinities in the estuary are attributed to the presence of freshwater rather than estuarine algae as the high flushing rates of the estuary inhibit phytoplankton growth. There was no evidence that nutrient inputs from the estuary to the adjacent coastal waters promoted

Figure 3.83 Location of the Ythan Estuary and monitoring sites. © SEPA.

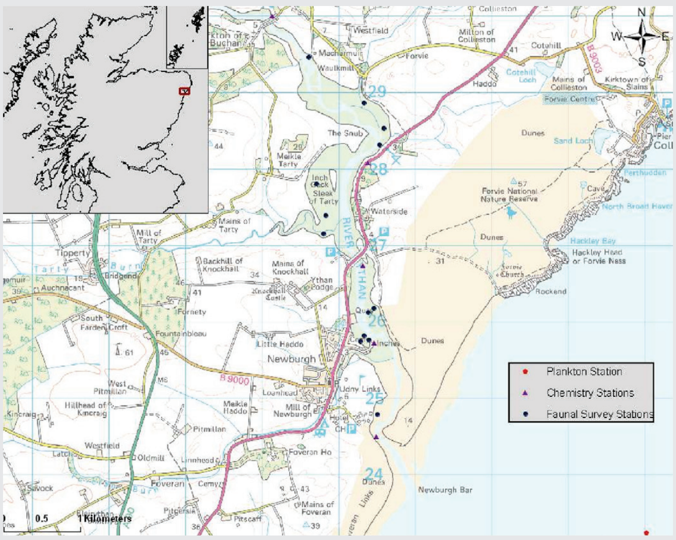


Figure 3.84 Opportunistic green algae in the Ythan Estuary. © SEPA.

Table 3.51 Summary of results of the second application of the OSPAR Comprehensive Procedure to the Ythan estuary using OSPAR's Harmonised Assessment Criteria.

Category	*Assessment parameter*	*Summary of results*	*Score*
Degree of Nutrient Enrichment (I)	Riverine inputs and direct discharges of total N and total P	Riverine inputs of N are high and have increased since 1980 but there is evidence that concentrations have started to decrease since 2000	–
	Winter DIN and/or DIP concentrations	Winter DIN concentrations exceed 30μM at a salinity of 25	+
	Winter N:P ratio (Redfield N:P = 16)	The N:P ratio is greater than 24	+
Direct Effects (II)	Maximum and mean chlorophyll a concentration	Chlorophyll a concentrations greater than 15 μg/l have been recorded in the estuary	+
	Area-specific phytoplankton indicator species	There is no evidence of modification of phytoplankton in adjacent coastal waters	–
	Macrophytes including macroalgae	The Ythan estuary was designated as an NVZ in 2000 on the basis of extensive growth of opportunistic green macroalgae. There is no evidence that the situation has significantly improved	+
Indirect Effects (III)	Oxygen deficiency	Dissolved oxygen concentrations do not fall below the assessment criteria	–
	Changes/kills in zoobenthos and fish kills	There is no evidence for modification of the benthos or records of kills	–
	Organic carbon/organic matter	There is no evidence of organic enrichment of the sediments	–
Other Possible Effects (IV)	Toxin-producing algae (DSP/PSP mussel infection events)	There are no records of mussel infection events in the estuary	–

Key to the Score
+ Upward trends, elevated levels, shifts or changes in the respective assessment parameters
– Neither upward trends nor elevated levels nor shifts nor changes in the respective assessment parameters
? Not enough data to perform an assessment or the data available are not fit for the purpose

phytoplankton growth, as the nutrient rich waters are rapidly diluted and dispersed by local currents.

From the review of the trophic status for the second application of the OSPAR Comprehensive Procedure, it was concluded that the Ythan continues to be eutrophic. Further monitoring is required to assess any trends which are currently being masked by the large inter-annual variation in the data.

Case Study 4: Belfast Lough

Belfast Lough is a shallow semi-enclosed marine bay in which the principal watercourse is the River Lagan which enters at Stranmillis. The total catchment of Belfast Lough is 900 km^2 and freshwater input from the River Lagan is augmented by several streams along its shores. The seabed of the Lough slopes gradually from Belfast where there are extensive mudflats, to a depth of about 20 m at the outer limit. The total area of the Lough has a flushing time of 1.44 days with higher flushing rates in the Outer Lough. Tidal currents are weak and oscillatory in the Inner Lough resulting in a predominantly sheltered area where the currents are dominated by tides. The Outer Lough is exposed and water exchange with the North Channel is rapid. A clockwise rotatory current has been documented in the Outer Lough (Parker et al., 1988) and these physical conditions result in less potential for eutrophication. The physical oceanography of the Inner and Outer Lough results in significant chemical and biological differences.

The UK originally issued guidance for identifying sensitive areas (eutrophic) under the UWWTD in March 1993. This guidance has been used alongside the Comprehensive Studies Task Team guidance issued by the UK authorities in 1997 (MPMMG, 1997). In 1993, there was insufficient evidence to warrant identification of Inner Belfast Lough as a sensitive area according to the definition of eutrophication (Article 2, 11) and the criteria in Annex IIA (a) of Directive 91/271/EEC.

A four year study (1992–1996) was completed by Service and Durrant (1996) and concluded that Inner Belfast Lough was eutrophic. However, the area was not designated as sensitive under the UWWTD in 1997 primarily because the main source of nutrients to the Inner Lough was not from waste water treatment works but from an industrial discharge. The Environment and Heritage Service (EHS), the predecessor organisation to the Northern Ireland Environment Agency (NIEA), delayed identification, awaiting the outcome of a further trophic status study and modelling. The purpose of these further studies was to determine whether management of the industrial nutrient discharge would be sufficient to return Inner Belfast Lough to a more normal trophic status.

The Inner Lough was designated as a sensitive area under the UWWTD in 2001 following a recommendation from a study by Charlesworth and Service (1999). Supplementary guidance under the UWWTD was issued in May 2002, this was closely aligned with the OSPAR Common Assessment Criteria for Eutrophication. A report in 2003 confirmed the trophic status of Inner and Outer Belfast Lough using this methodology (issued in 1993 and supplemented in May 2002) for identifying eutrophic waters. All data sources are described in the detailed report, which is included as part of the second application of the OSPAR Comprehensive Procedure (see www.cefas.co.uk/ospardocs). The report documented a number of changes in Inner Belfast Lough from 1998 to 2002 which resulted in a reduction of nutrient inputs, concentrations and an overall improvement in trophic status. These include: the introduction of full secondary treatment with nutrient (nitrogen) removal at Belfast (Duncrue. see Figure 3.85) waste water treatment works from December 1998; secondary treatment at Kinnegar waste water treatment works operational from December 2000, with nitrogen reduction operational from June 2001; a progressive tightening of the discharge consent of a major industrial discharger (fertilizer plant) and the eventual closure of the plant in December 2002; and the rapid development of a shellfishery (largely bottom culture mussels)

Figure 3.85 Belfast Lough showing the Northern Ireland Environment Agency (NIEA) monitoring locations. © Northern Ireland Environment Agency.

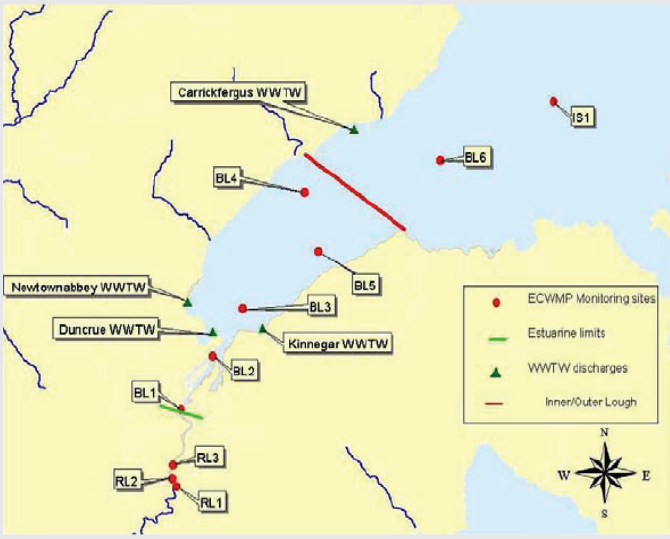
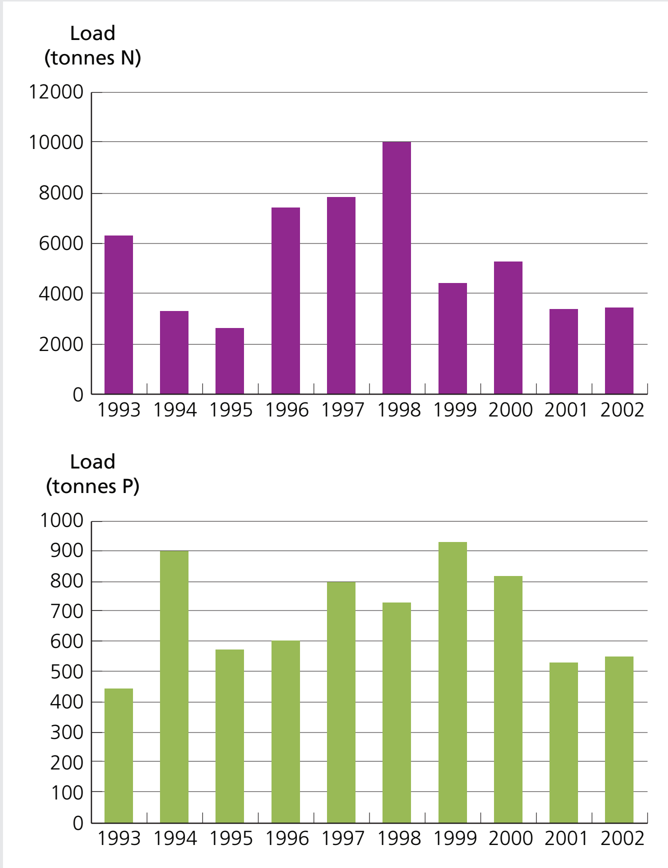


Figure 3.86 Total loadings of nitrogen and phosphorus to Belfast Lough. Data source: NIEA, OSPAR RID Programme. © Northern Ireland Environment Agency.



in Inner Belfast Lough since 2000. Total loads of nitrogen and phosphorus to Belfast Lough (Figure 3.86) show considerable variability between years. For nitrogen, total loads appeared to decrease after 1998. Similar trends were not observed for phosphorus.

The conclusions from the updated assessment in 2003 were that the Inner Belfast Lough was eutrophic although there was evidence of reduced nutrient inputs which resulted in improvements in this area. The Inner Lough was still enriched in nutrient concentrations; there was still evidence of accelerated algal growth on occasions; and there was still evidence of an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned.

Physical conditions in the Outer Lough do not favour eutrophication because of the rapid exchange of waters with the waters of the North Channel. The waters were nutrient enriched, and showed signs of accelerated growth of algae during the 1990s. However, there was no evidence of an undesirable disturbance to the balance of organisms present in the water and water quality during this period. Therefore the Lough was not designated as sensitive. The conclusions from the updated assessment in 2003 were that the Outer Lough was not enriched with nutrients, there was no evidence of accelerated algal growth or higher plant forms, and there was no evidence of an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned. The Outer Lough was therefore assessed to be not eutrophic or likely to become eutrophic in the near future if protective action was not taken within the terms of Annex IIA(a) of the Directive.

Case Study 5: Monitoring of Scottish sea lochs for the effects of nutrients from fish farms

The majority of UK marine fish farming takes place in the sheltered waters of sea lochs, and voes, of the west coast, and the Western and Northern Isles of Scotland. Nutrient discharges from fish farms in these semi-enclosed waters have the potential to result in nutrient enhancement. In order to assess the potential eutrophication status of these regions the UK undertook an extensive programme of monitoring and assessment between 2002 and 2006 covering some 38 water bodies supporting fish farms (Figure 3.87). Hotspot areas were targeted for the assessment, where according to simple models relating nitrogen discharge rates from fish farms to flushing rates of sea lochs, nitrogen enhancement was predicted to be highest.

Surveys of these regions were conducted at key times of the year by research vessel and parameters monitored according to the OSPAR Comprehensive Procedure and assessed against the Harmonised Assessment Criteria. A summary of results is shown in Table 3.52. An overall assessment of the parameters according to the Comprehensive Procedure was undertaken and resulted in non-problem area classifications for all the sea lochs assessed with respect to nutrient inputs from aquaculture.

Figure 3.87 Sea lochs, voes, sounds and bays monitored by Fisheries Research Services during 2002–2006 and assessed as non-problem areas with respect to the effects of nutrients from fish farms by applying the OSPAR Comprehensive Procedure for eutrophication assessment. © Marine Scotland Science.

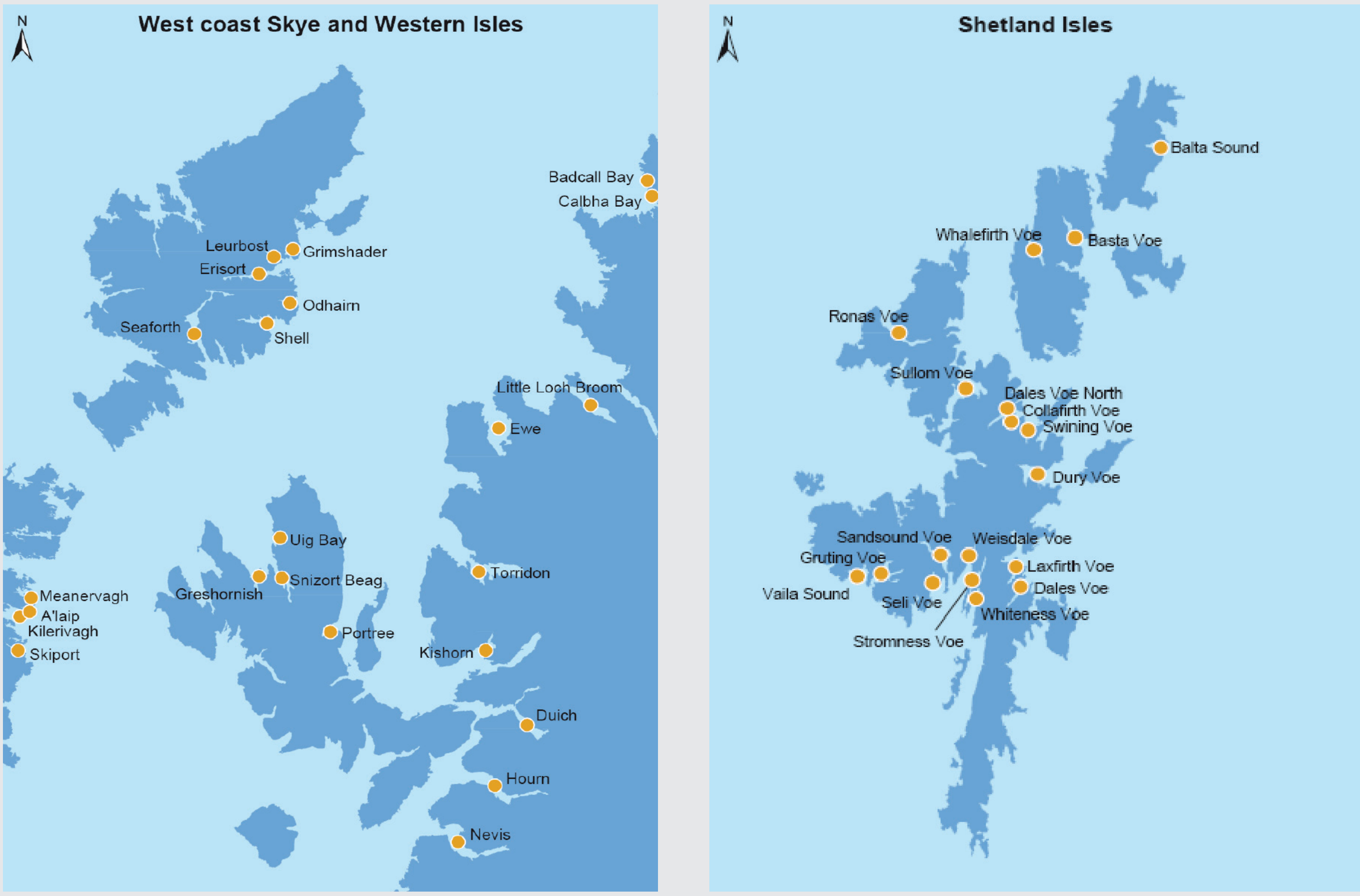


Table 3.52 Summary of results of the application of the OSPAR Comprehensive Procedure, Harmonised Assessment Criteria to 38 Scottish sea lochs supporting aquaculture.

Assessment parameter	*Summary of findings for Scottish sea lochs*
Category I: Degree of nutrient enrichment	
1. Nutrient inputs	In general, nutrient inputs from aquaculture decreased from 2003–2005 across Scotland as feeding efficiency improved and production declined over this period. Regulatory restrictions will prevent future increases in nutrient discharges from aquaculture in most hotspot areas
2. Nutrient concentrations	Winter nutrient (dissolved available inorganic nitrogen) concentrations did not exceed the criteria of 50% above background concentrations of coastal waters for any lochs
3. Nutrient ratios	Winter N:P ratios did not exceed 50% above background values (16:1) for any sea lochs.
Category II: Direct effects of nutrient enrichment	
1. Chlorophyll a concentrations	The 90th percentile of measured values did not exceed 50% above background values for coastal waters at any of the sea lochs surveyed
2. Phytoplankton indicator species	Potentially toxic and nuisance species were recorded at several sea lochs at densities typical for Scottish waters. The occurrence of these species is not thought to be related to nutrient inputs from aquaculture
3. Macrophytes and macroalgae	Percentage area coverage of 'nuisance' green macroalgae in the inter-tidal zone did not exceed the assessment level of 15% at any of the sea lochs surveyed
Category III: Indirect effects of nutrient enrichment	
1. Oxygen deficiency	The 5th percentile of measured values never fell below the assessment level of 4 mg/l. Some bottom waters in sea loch basins showed lower values which are a result of the natural hydrography of the lochs and are not caused by nutrients from aquaculture
2. Fish kills	There are occasional kills of farmed fish caused by jellyfish and harmful phytoplankton in sea lochs. These are not related to eutrophic conditions
3. Organic carbon	Organic carbon levels in sediments vary naturally with hydrography and are high close to fish farms. Levels are not of concern with respect to eutrophication assessment
Category IV: Other effects of nutrient enrichment	
1. Algal toxins	Extensive monitoring reported the presence of amnesic, paralytic and diarrhetic shellfish toxins in water, plankton and shellfish from several sea lochs. The occurrence of these toxins is typical for Scottish waters and not thought to be related to nutrient inputs from aquaculture.

changes that could threaten the non-problem area status. These areas are East England, East Anglia, Liverpool Bay and the Solent and the Clyde.

- Restricted regions including estuaries, loughs and harbours – some of which were identified as problem areas or potential problem areas in 2002. Through the ongoing assessment programme related to the implementation of the relevant EU Directives a further five problem areas and three potential problem areas have been identified. Many of these are small water bodies.

The initial assessments were subject to an international peer-review panel and a public workshop, the outcomes of which were taken into consideration in the final assessment.

3.3.4.2 Confidence in observed changes (adequacy and confidence of the data)

Data for the period 2001 to 2005 were used for most of the areas assessed. There were some areas where adequate data (in terms of quantity or quality) were not available for some parameters and this resulted in assigning a lower confidence to the parameter assessment. For smaller transitional and coastal areas assessed under previous EU Directives, monitoring data from this period (2001–2005) were also used, where possible. In many areas, this was not possible, and details of the monitoring and assessment periods can be found in the original reports for these regions (see www.cefas.co.uk/ospardocs).

The data sets used were considered to be generally fit for purpose, providing adequate spatial coverage and temporal resolution to carry out a confident assessment. There were, however, variations in data coverage between areas. This reflects the current level of perceived risk and the practicalities of monitoring. Also, sampling variations existed between the different parameters where, for example, there were more data available for winter nutrient concentrations and chlorophyll than for phytoplankton species. The adequacy of the data was reflected in the confidence with which conclusions were reached about the status of each area. Levels of confidence were reported in detail in individual area reports and the UK National Report (www.cefas.co.uk/ospardocs).

Few attempts have been made to include estimates of primary production in assessments. Rodhe (1969) proposed net annual primary production thresholds for assessing eutrophic status of ‘naturally eutrophic’ and ‘polluted’ freshwater (>75 and >350 g C m^2/ y, respectively). For coastal marine waters, Nixon (1995) proposed a similar scale for assessing trophic status. However, these descriptors of eutrophic status have no direct bearing on the term ‘eutrophication’, as defined by the EU UWWTD (1991). Water bodies identified as being eutrophic under Rodhe or Nixon’s scales may not show any signs of the negative impacts of nutrient enrichment or, therefore, eutrophication.

There are numerous gaps in our knowledge and understanding of the susceptibility of ecosystems to nutrient enrichment. Future improvements are likely to indicate great complexity in the biological response to nutrient enrichment, with multiple stressors/pressures, multiple factors determining sensitivity, and complex feedbacks between the different biological responses in an ecosystem. Apart from nutrient inputs, other stressors include climate change, fish harvesting, toxic contaminants and aquaculture. While future scenarios for climate change remain unclear, more extreme environmental

conditions such as drier summers, episodic downpours and increased storminess will impact upon anthropogenic nutrient loads from all terrestrial and atmospheric sources into the marine environment, particularly in terms of timing and magnitude. The biological responses to altered nutrient loads, and any potential negative impacts, will be affected by other factors responding simultaneously to climate change, such as changes in water temperature and changes in physical and/or chemical oceanographic processes.

It is not easy to demonstrate unequivocal links between anthropogenic nutrient inputs and undesirable disturbance. In specific cases, it may be easy to demonstrate localised cause and effect relationships – for example, between nuisance/toxic species and human activity. However, on a broader scale this is seldom feasible. In offshore waters of the North Sea, for example, recent research has shown that hypoxic conditions can develop naturally in bottom waters in some regions during periods of stratification. Improved confidence in assessments of eutrophication therefore requires improved scientific understanding of how ecosystems function, and improved monitoring and assessment of undesirable disturbance in relation to different types of ecosystems.

3.3.4.3 Contribution to the GES descriptor

The MSFD provides broad qualitative descriptors for determining Good Environmental Status (GES). For eutrophication, the emphasis is identification of any negative impacts of anthropogenic nutrient inputs. The descriptor requires that human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algal blooms and oxygen deficiency in bottom waters. The potential indicators or criteria for assessing GES for eutrophication are under discussion, but are likely to be broadly similar to those under development by OSPAR, and in relation to the WFD, which have already adopted this approach. Indicative lists of characteristics, pressures and impacts may require the development of additional criteria – such as to include enrichment by organic matter, and other elements characterising the marine food web. In terms of the latter, inclusion of grazers (e.g. zooplankton and filter-feeders such as mussels and oysters) may contribute towards improved understanding of the biological response to nutrients and towards improved management of eutrophication. High grazing impacts will result in underestimates of phytoplankton biomass and/or growth (as indicated by chlorophyll concentrations), and egestion of faecal pellets may contribute towards hypoxia due to the accumulation of pellets in boundary layers, especially near or within the seabed.

3.3.5 Progress towards the vision of clean and safe seas

3.3.5.1 Status

Recent assessments of eutrophication status indicate that each of the CP2 assessment areas do not, as a whole, suffer from eutrophication problems. They were therefore classified as non-problem areas (Figure 3.81). However, a number of estuarine and coastal areas are nutrient enriched and are at risk from or currently affected by eutrophication. Seventeen water bodies have been identified as sensitive or polluted waters, or conservation areas. These have been classified as problem areas. Five water bodies have been classified as potential problem areas.

Region 1: Pressures from nutrient inputs are present due to agriculture and urban waste water treatment in limited areas. The overall pressure is low. The region as a whole does not suffer from eutrophication problems although there are some small estuarine and coastal areas that are assessed as problem areas.

Region 2: There is significant pressure due to nutrient input from agriculture which covers large areas of eastern England and from urban waste water due to the high population density. The coastal and offshore areas of the region are assessed as non-problem areas even though there is nutrient enrichment and some evidence of accelerated growth of algae and higher forms of plant life, particularly in coastal waters. The physical nature of this region (tidal hydrodynamics, inorganic turbidity restricting light), especially in coastal areas, makes it resistant to developing eutrophication problems.

Region 3: There is some pressure due to nutrient input from agriculture which covers large areas of eastern England and from urban waste water due to the high population density. The offshore waters are non-problem areas, being influenced predominantly by the inflow of water from the Atlantic. There are several areas at the coast which have very restricted exchange and are problem areas. The pressures associated with the specific problem areas are being managed.

Region 4: Relatively low population density, although this varies seasonally and is high in summer. Significant intensive agriculture but confined to a few areas. Overall pressure is low. There is no nutrient enrichment in this area with the exception of small estuaries and lagoons which are problem areas and the specific sources of the enrichment are being managed.

Region 5: There is significant pressure in the eastern part of the region due to nutrient input from agriculture and from urban waste water due to the high population density. The coastal and offshore areas of the region are assessed as non-problem areas even though there is nutrient enrichment and some evidence of accelerated growth of algae and higher forms of plant life, particularly in coastal waters. The physical nature (hydrodynamics, inorganic turbidity restricting light) of this region, especially in coastal areas, makes it resistant to developing eutrophication problems.

Region 6: The overall pressure from nutrient input is low due to the low population density. The pressure is higher in the southern part of the region due to agriculture. The region as a whole does not suffer from eutrophication problems.

Region 7: The overall pressure from nutrient input is very low due to the low population density. The region as a whole does not suffer from eutrophication problems.

Region 8: There are no nutrient pressures in this region. The region as a whole is unlikely to suffer from eutrophication problems.

3.3.6 Forward look and need for further work

Future changes in our understanding of eutrophication are likely to indicate even greater complexity in the biological response to nutrient enrichment, with multiple stressors/pressures, multiple factors determining sensitivity, and complex feedbacks between the different biological responses in an ecosystem. Apart from nutrient inputs, other stressors include climate change, fish harvesting, toxic contaminants and aquaculture.

With our current understanding, it is difficult to forecast the direct and indirect responses of ecosystems to multiple stressors or pressures with confidence. It is likely that it will become increasingly difficult to distinguish between effects of human inputs and those that may be due to climate change. In the future, monitoring and assessment work will need to be closely integrated with predictive modelling based on the use of susceptibility and ecosystem models of varying complexity. Ongoing validation of these models will be essential, to improve our confidence in these models as management tools.

Coastal observatories which have been established in Liverpool Bay and the North Sea will provide valuable data for improving present and future nutrient monitoring programmes and therefore for improving confidence in assessments of eutrophication. Future expansion of the SmartBuoy monitoring programme to include additional sites in the Celtic Sea/English Channel and in the North Sea will provide an invaluable contribution to these improvements. It is vital that ship-based studies continue to be combined with moorings to further improve our understanding.

The potential indicators or criteria for assessing GES for eutrophication are under discussion, but are likely to be broadly similar to those under development by OSPAR, and in relation to the EU Water Framework Directive which have already adopted this approach. Indicative lists of characteristics, pressures and impacts may require the development of additional criteria, such as the inclusion of enrichment by organic matter, and other elements characterising the marine food web.

3.4 Microbiological Contamination

3.4.1 Key points

i. Introduction

Microbiological monitoring of the marine environment is currently focussed on identifying faecal pollution of bathing waters and shellfish harvesting areas. There are three monitoring programmes undertaken nationally covering bathing waters, shellfish waters and shellfish hygiene.

ii. How has the assessment been undertaken?

Microbiological data are assessed against standards set within the EU Bathing Waters Directive, the EU Shellfish Waters Directive and the Shellfish Hygiene Standards within the EU Food Hygiene Regulations. Current standards assess bacterial contamination as indicative of levels of faecal pollution.

iii. Current status of microbiological contamination and past trends

Bathing Waters. In 2007, 96% of bathing waters met at least the imperative (compulsory) standard and 76% met the guideline (desirable) standard under the EU Bathing Waters Directive.

Shellfish Waters. In 2007, 40% of sampled shellfish waters met the guideline value under the EU Shellfish Waters Directive. This value is significantly more stringent than the guideline standard in the EU Bathing Waters Directive. Data for faecal coliforms in shellfish flesh were not available for many sites and this prevented an assessment of compliance for those sites. In England and Wales, the lack of data was primarily due to there being insufficient stock to sample. Measures are in place to further improve sampling compliance.

Shellfish Hygiene. In 2007, 21% of areas were assigned Class A status, 75% Class B, 3% Class C and fewer than 1% were assigned a Prohibited status for harvesting on the grounds of microbiological contamination. This represents a significant increase in the percentage of areas assigned Class B status since *Charting Progress*, primarily as a result of the reduction in contamination in several areas previously assigned Class C status.

iv. What has driven change?

The levels of compliance reflect the significant investment in sewage treatment and infrastructure driven by the EU Bathing Waters and Shellfish Waters Directives.

v. What are the uncertainties?

Data are quality controlled and reliable, but relate only to those areas required to meet legislation and do not represent the status of microbiological quality in UK waters overall.

vi. Forward look

Overall, reductions in the faecal contamination of the marine environment, as indicated by the outcome of monitoring programmes, should be associated with a reduction in the occurrence and concentration of faecal pathogens.

Further improvements in microbiological quality would require measures to reduce the impact of land run-off. This includes reducing misconnections in piping, developing sustainable drainage systems, and in implementing changes to land management, such as establishing buffer zones that exclude grazing animals from the vicinity of water courses and extending good

practice/controls on the storage and spreading of slurry and washing off of waste from farm hard standings.

Limited ability to measure viral loads in the environment and a lack of understanding of the dose/response relationship in humans means that viral standards have yet to be developed. Such issues will continue to be investigated with a view to developing a viable approach to the management of viruses.

3.4.1.1 Regional summary

The results of the regional assessment are shown in Table 3.53. The summary table presents our expert opinion on a single comparable assessment of status and trends for microbiological contamination within the UK seas. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. Trend arrows are based on the evidence available, showing whether the state or condition of the component is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them. The forward look is classified as positive (⬆), neutral (⬌) or negative (⬇).

3.4.2 Introduction

There are three national microbiological monitoring programmes undertaken in the UK marine environment. These are related to the requirements of the Bathing Waters Directive (76/160/EEC), Shellfish Waters Directive (2006/113/EC [which replaced 79/923/EC] and European food hygiene legislation (for shellfish hygiene; primarily Regulation (EC) No 854/2004, as amended). Each programme is organised separately for England and Wales, Scotland and Northern Ireland. The monitoring for the Bathing Waters and Shellfish Waters Directives is undertaken by the Environment Agency in England and Wales, the Scottish Environment Protection Agency in Scotland and the Northern Ireland Environment Agency in Northern Ireland. The shellfish hygiene monitoring under Regulation (EC) 854/2004 is the responsibility of the Food Standards Agency.

Faecal material from human or animal sources may contribute human pathogens (bacteria, viruses or parasites) to the aquatic environment and ingestion of water, or shellfish grown in such waters, may cause infection (Wittman and Flick, 1995; WHO, 2001). The microbiological aspects of the three programmes focus on determining the presence and concentration of faecal indicator bacteria to establish whether the area has been subject to such contamination (from human and/or animal sources) and thus whether there is a risk that pathogens may be present. A specific pathogen will not necessarily be present even if test results indicate significant contamination with faecal material – it will depend on whether that pathogen is present in the source of the contamination.

Standards are based on the concentration of faecal indicator bacteria in seawater or shellfish. These indicator bacteria have long been used to show the extent of contamination with faecal material, derived from human or animal sources.

The principal sources of faecal contamination reflected by the monitoring programmes are:

- Public sewage discharges
- Private sewage discharges
- Land run-off (contaminated by animal waste, sewage sludge applied to land, etc)

Table 3.53 Regional assessment summary for microbiological contamination.

CP2 Region	*Key factors and pressure*[a]	*What the evidence shows*	*Trend*	*Current status – bathing waters*	*Current status – shellfish growing waters*	*Confidence in assessment*	*Forward look*
1	Parts of the region are densely populated and subject to significant continuous and intermittent sewage inputs	Shellfish waters and hygiene quality poor at several sites	↔			III	↔
2	Apart from the densely populated Thames Estuary, the region is relatively lightly populated and subject to low levels of sewage pollution	Generally very good quality	↔			III	↔
3	Parts of the region are densely populated and subject to significant continuous and intermittent sewage inputs	Shellfish waters and hygiene quality poor at several sites	↔			III	↔
4	Parts of the region are densely populated and subject to significant continuous and intermittent sewage inputs	Shellfish waters and hygiene quality poor at several sites	↔			III	↔
5	Parts of the region are densely populated and subject to significant continuous and intermittent sewage inputs	Bathing water guideline compliance lower than other regions. Shellfish waters and hygiene quality poor at several sites	↔			III	↔
6	The region is lightly populated and subject to low levels of sewage pollution	Generally very good quality	↔			III	↔
7	The region is relatively lightly populated and subject to low levels of sewage pollution	Generally very good quality	↔			III	↔
8	No current monitoring			**NA**	**NA**		

[a] The microbiological quality of parts of coastal and estuarine waters in Regions 1 to 7 will be subject to localised small sewage discharges (e.g. small community discharges and/or septic tanks) and run-off from agricultural land.
NA: Indicates no activity in this region.

- Direct inputs to estuarine and coastal environments from farm animals, wild animals (including cetaceans and seals) and birds.

Historically, the principal sources of contamination demonstrated by the programmes have been continuous sewage discharges, but, as these have been improved, contamination due to intermittent discharges such as combined sewer overflows and land run-off (diffuse sources) has become relatively more important. In some parts of the UK, for example rural Scotland, sources of human sewage contamination tend to be localised (e.g. small septic tanks). Where the contributions from sewage inputs are small, the faecal

indicator bacteria detected in the monitoring programmes may derive largely from agricultural and other animal sources. Due to the filter-feeding behaviour of bivalve molluscan shellfish, microbiological and other contaminants may be found in concentrations many times greater than those in the surrounding seawater. It is now well documented that the consumption of shellfish exposed to sewage contamination may result in illness in humans. The small numbers of shellfish-related illnesses reported annually in the UK are due to noroviruses which cause acute diarrhoea and vomiting (Lees, 2000). Such outbreaks usually arise from areas that are assigned Class B (and subject to purification) or Class A (either consumed directly or following purification) status.

In a randomised trial undertaken in the UK, it was demonstrated that crude rates of gastroenteritis were significantly higher in people exposed to bathing waters meeting the imperative bacteriological limit (14.8 per 100) than those who were not exposed (Kay et al., 1994). Respiratory illness, ear and wound infections may also occur following bathing (Fleisher et al., 1996; Oliver, 2005). These problems may be due to introduced microbes or to bacteria that naturally occur in the marine environment, predominantly certain types of *Vibrio*. The latter may also cause gastro-intestinal infection or septicaemia following consumption of shellfish – this is not currently perceived as a problem in the UK although some types of *Vibrio* that may cause such infections have been found in UK seawater and shellfish. The current microbiological monitoring programmes will not determine the risk of contamination due to these other pathogens.

Overall, reductions in the faecal contamination of the marine environment, as indicated by microbiological status, will yield a reduced risk of infections arising from recreational water use or shellfish consumption. Good quality bathing waters also provide additional potential for growth in tourism and thus have a beneficial effect on local and regional economies. Good quality shellfisheries reduce the likelihood of illness resulting from shellfish consumption and are essential to the maintenance and growth of the shellfish industry. The commercial use of Class C areas for most shellfish species is limited, although cockles (and to some extent mussels) from such areas can be sold following appropriate heat treatment. There has been a trend towards supermarkets and other large purchasers of shellfish preferring to source shellfish that have both originated from Class A areas and that have been depurated (i.e. exposed in a clean area or placed in UV-treated recirculating water in order to reduce microbiological contamination). There are a number of benefits arising from having bathing waters and shellfish waters that have a low level of microbiological contamination.

3.4.2.1 Assessment

Assessments for bathing waters and shellfish hygiene were undertaken by determining the proportion of designated waters and classified harvesting areas complying with the criteria specified in the EU Bathing Water Directive and Regulation (EC) No 854/2004 (as amended), respectively (see also Sections 3.4.4.1 and 3.4.4.3). Assessments for shellfish waters were undertaken by determining the proportion of monitored designated waters that complied with the guideline value described in Section 3.4.4.2. Compliance was determined by year for the whole of the UK and also by year for each reporting region. All of the criteria are based on faecal indicator bacteria and yield an estimate of risk of contamination with faecal material in general rather than an assessment of the likely presence of specific pathogenic micro-organisms.

3.4.3 Developments since *Charting Progress*

A number of sewage improvements have been targeted at both bathing and shellfish waters and have reduced the extent of faecal pollution. In addition, a number of investigations have taken place to identify the key sources of pollution in several shellfish waters. These have been followed by the preparation of pollution reduction plans. The data show that the major reductions in contamination took place during the period covered by *Charting Progress*, and that any further improvements are masked by year–to-year fluctuations. It is anticipated that further progress, particularly with regard to compliance with the guideline value for shellfish waters, and Class A status for shellfish hygiene, will require the implementation of measures to reduce land run-off together with further reduction in the spill frequency of storm-related discharges. In specific areas, additional improvements may be required for continuous sewage discharges. A programme of investigations and sewage improvement schemes is currently underway in the UK and is expected to continue during the next water company investment period (2010 to 2015).

With respect to bathing waters, a pilot scheme was run at five bathing waters in the south-west of England that had displayed signs indicating potential poor water quality following significant rainfall events (using rainfall triggers of 10 mm of rainfall in 24 hours or 15 mm in 48 hours). The project was a precursor to the type of management measures that may be implemented under a new Bathing Water Directive (2006/7/EC). The requirements of the new Directive will need to be implemented by 2015.

In 2005, the Food Standards Agency introduced a system of long-term classifications for Class B areas in England and Wales showing consistent long-term compliance with Class B requirements. The system included the establishment of Local Action Groups and Local Action Plans, with local investigations being undertaken, and short-term control measures being considered, when trigger levels were breached in routine samples taken from areas subject to such classifications. The application of the short-term control measures should have benefits for public health and reduce the need to revise classifications unless a longer-term trend became apparent. The latter should provide more stability for the shellfish industry. Some elements, such as the Local Action Groups, are being considered for other parts of the UK.

The only development in indicators affecting the current assessment period (2002–2007) relate to the long-term classification system for shellfish hygiene. This will have the effect of reducing year–to-year change in classification status of beds assigned a long-term Class B status.

A number of studies have shown that diffuse pollution sources, such as land run-off from agricultural areas, contribute significantly to the contamination of both bathing waters and shellfish waters, especially during and after periods of heavy rainfall (Kay et al., 2009a). However, such contamination may contain pathogens, such as *Salmonella* spp. and it has been shown in the UK that classification of shellfish harvesting areas based on the microbiological monitoring programme is a good predictor of the risk of contamination with that pathogen (Lee and Younger, 2003).

3.4.4 Presentation of the evidence

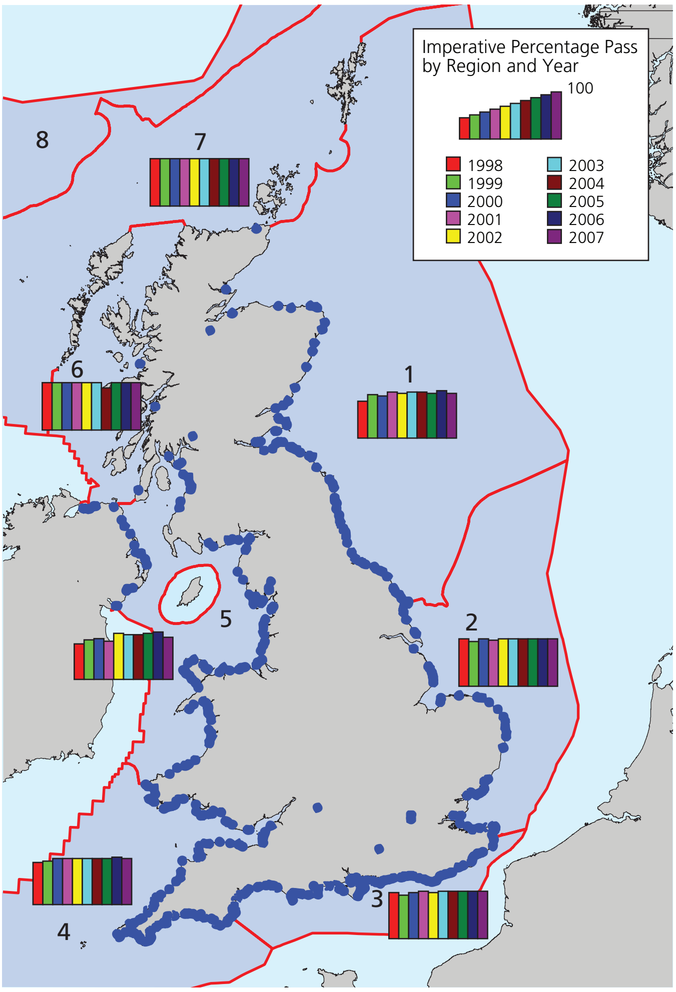
3.4.4.1 Bathing waters

Monitoring of both marine and freshwater bathing waters is required under the Bathing Waters Directive. A total of 578 marine bathing waters were identified under the Directive in the UK in 2007 (Figure 3.88). Twenty samples of water are taken during the bathing season from nominated sampling points in these waters and tested for total coliforms, faecal coliforms and faecal streptococci. The standards given in the legislation are shown in Table 3.54.

There was a marked increase in the percentage of bathing waters meeting the guideline standard over the assessment period, coincident with a marked decrease in the percentage of waters either only meeting the imperative standard or failing the requirements altogether (Figure 3.89). The main improvements took place between 1999 and 2003. There is a year-to-year variation in the percentage of marine bathing waters either meeting only the imperative standard or failing the requirements altogether, probably due to a combination of compliance assessment based on the required number of samples and fluctuation in environmental variables affecting the microbiological quality of the bathing waters. Data up to and including 2003 were included in the assessment for *Charting Progress*. The percentage of areas in each category has, in general, stayed broadly the same since then, with only year-to-year variation evident.

Regional compliance with the guideline bathing waters standard and the imperative bathing waters standard for the period 1998 to 2007 inclusive is shown in Figures 3.90 and 3.88, respectively. It should be noted that there is only one identified bathing water in Region 7 and so it is not valid to compare this region with

Figure 3.88 Location of UK identified bathing waters in 2007 (●) and regional compliance with the imperative bathing waters standard in UK bathing waters, 1998–2007. Sampling areas include a small number of inland bathing waters. © Crown copyright 2010: permission granted by Cefas.



others. With the exception of that region, the maps show that the overall improvement in compliance noted above was also seen in each of the regions, with the possible exception of Region 6. Currently, the rate of compliance is lowest in Region 5.

3.4.4.2 Shellfish waters

The Shellfish Waters Directive (2006/113/EC) *...concerns the quality of shellfish waters and applies to those coastal and brackish waters*

Table 3.54 Microbiological standards under the EU Bathing Waters Directive.

Parameter	*Guideline standard*	*Statistic*	*Imperative standard*	*Statistic*	*Minimum sampling frequency*
Total coliforms per 100 ml	500	80% of samples	10 000	95% of samples	Fortnightly (1)
Faecal coliforms per 100 ml	100	80% of samples	2000	95% of samples	Fortnightly (1)
Faecal streptococci per 100 ml	100	90% of samples	-		(2)
Salmonella per litre	-		0		(2)
Entero viruses PFU per 10 litres	-		0		(2)

(1) When a sample taken in previous years gave results which are appreciably better than those in the Annex to the Directive and when no new factor likely to lower the quality of the water has appeared, the competent authorities may reduce the sampling frequency by a factor of 2.
(2) Concentration to be checked by the competent authorities when an inspection in the bathing area shows that the substance may be present or that the quality of the water has deteriorated.

Figure 3.89 Percentage of UK identified marine bathing waters meeting the imperative and guideline standards, 1998-2007. © Crown copyright 2010: permission granted by Cefas.

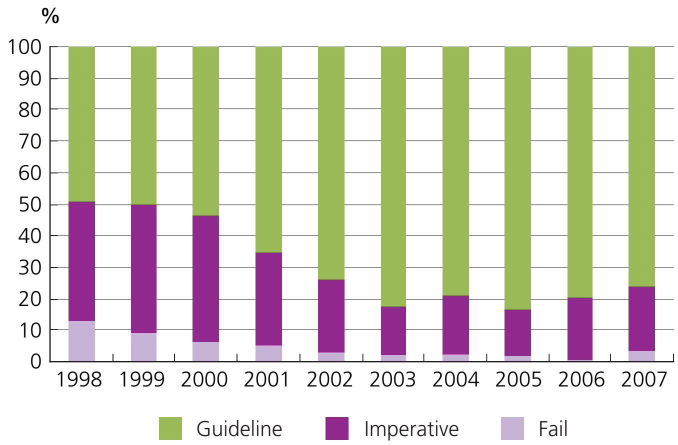
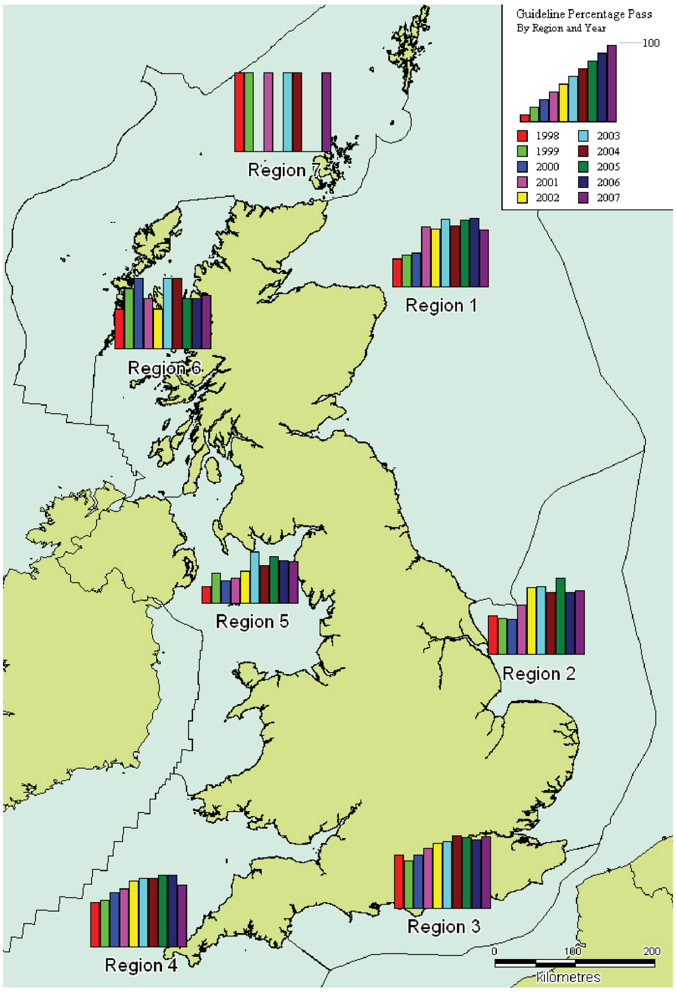


Figure 3.90 Regional compliance with the guideline bathing waters standard, 1998-2007. © Crown copyright 2010: permission granted by Cefas.



designated by the Member States as needing protection or improvement in order to support shellfish (bivalve and gastropod molluscs) life and growth and thus to contribute to the high quality of shellfish products directly edible by man. From a microbiological perspective, it requires quarterly sampling of shellfish and includes a guideline value of 300 faecal coliforms per 300 ml of flesh and intervalvular fluid. For compliance, this value should not be exceeded in 75% of samples. Whereas the shellfish hygiene legislation (see Section 3.4.4.3)

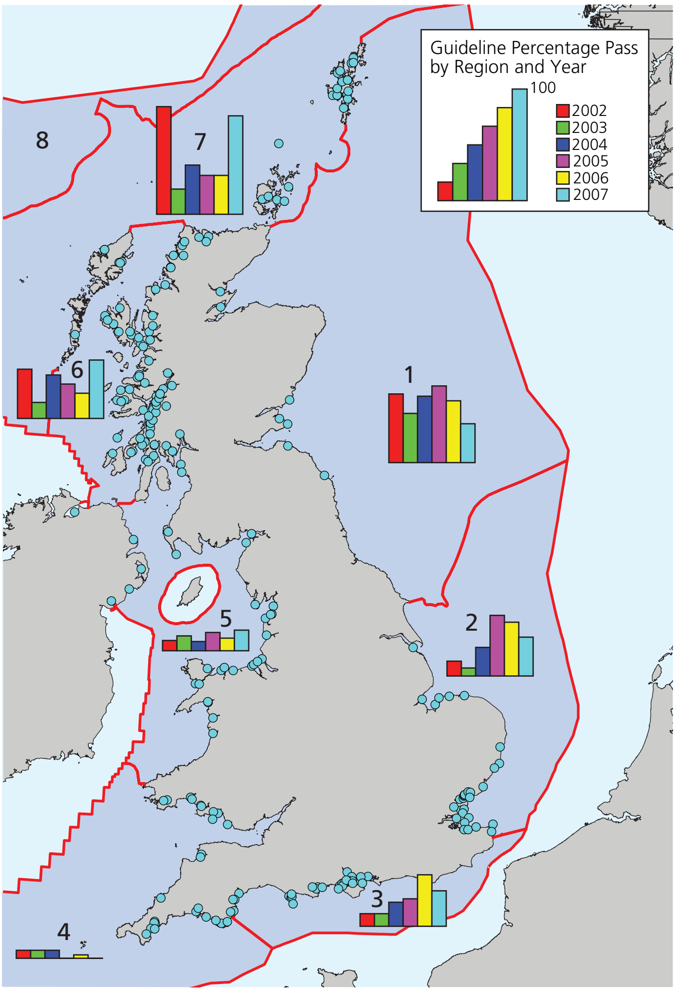
assigns areas into different classes based on an assessment of risk, and does not require that the extent of the pollution of a harvesting area is reduced, non-compliance with the requirements of the Shellfish Waters Directive requires that *...Member States shall establish programmes in order to reduce pollution and to ensure that designated waters conform*.

In 2007, there were 241 designated shellfish waters in the UK (see Figure 3.91). Due to practical difficulties in obtaining shellfish samples from some of these sites, data on faecal coliforms in shellfish flesh have not been obtained for all of these. Figure 3.92 shows the percentage of sampled sites in the UK that met the guideline standard for the years 2002 to 2007 inclusive.

There was a general trend of an increasing percentage of sampled waters meeting the guideline value over this period, although year-to-year variation is superimposed on this (Figure 3.92). Improvements are due to sewage improvement schemes that have either been specifically targeted at shellfish waters or at nearby bathing waters. The year-to-year variation is due partly to only a small number of results contributing to the assessment (four, taken quarterly) and partly to variability in environmental factors, principally rainfall, which causes the operation of intermittent discharges (such as combined sewer outfalls) and land run-off.

Regional compliance with the guideline value for the period 2002 to 2007 inclusive is also shown in Figure 3.91. For the reasons stated previously, the data presented do not include all designated waters. The map shows that, as with the overall compliance (see Figure 3.92) there is fluctuation from year to year. Overall, the percentage compliance in 2007 was greatest in

Figure 3.91 Location of UK designated shellfish waters in 2007 (●) and regional compliance with the guideline shellfish waters value, 2002–2007. © Crown copyright 2010: permission granted by Cefas.



All bacteria concentrations per 100 ml of seawater; Bathing Waters Directive: Imperative standard 10 000 total coliforms, 2000 faecal coliforms in 95% of samples; guideline standard 500 total coliforms and 100 faecal coliforms, in 80% of the samples.

Regions 6 and 7 and least in Region 4. Marked improvements in compliance were seen during the assessment period in Regions 2 and 3. The increase in compliance evident between 2006 and 2007 in Regions 6 and 7 was contributed to by a change in the approach to the monitoring programme, which markedly increased the amount of shellfish waters for which data were available for assessment.

Figure 3.92 Percentage of UK sampled shellfish waters meeting the faecal coliform guideline value, 2002–2007. © Crown copyright 2010: permission granted by Cefas.

3.4.4.3 Shellfish hygiene

Monitoring of *E. coli* in shellfish flesh is undertaken under Regulation (EC) No 854/2004 (as amended). In the UK, shellfish samples are normally taken on a monthly basis from representative points in each area. The samples are tested for *E. coli* by a reference method specified in the legislation (ISO TS 16649-3). The results of the monitoring are used to assign areas as having a Class A, Class B or Class C status. The criteria and resulting post-harvest requirements are shown in Table 3.55. Shellfish harvesting areas that do not conform to the requirements of Class A, Class B or Class C status are designated as Prohibited. Harvesting can also be prohibited if, regardless of the monitoring results, the competent authority deems that there is a risk to human health.

The locations of the classified harvesting areas in the UK are shown in Figure 3.93. In any one classified harvesting area, the commercially important species are usually monitored separately. In some instances, an indicator species may be used, although this approach is only used if that species has been shown to accumulate *E. coli* to at least the same extent as the commercial species it is representing. There may be more than one monitoring point per species per classified area if this is deemed necessary to adequately reflect different sources of contamination. A different approach has been taken to the definition of harvesting areas/ classification zones in the different constituent countries of the UK and this will influence comparisons across the whole UK and reporting regions. A different approach has also been taken to the use of seasonal classifications, whereby relatively consistent differences in the seasonal level of monitoring results are reflected in differential classifications across the year. In general, a larger use of seasonal classifications has been made in Scotland due to clearer differences in the seasonal pattern of contamination, and the greater proportion of aquaculture shellfisheries enabling greater use to be made of such classifications. In order to assist comparison across the UK, areas subject to seasonal classifications have been included in the present assessment as the worse of the two classifications, this being the class that would be given if no seasonal differentiation was to be made.

The proportion of UK harvesting areas that fell into each of the classes for the period 2001 to 2007 inclusive is shown in Figure 3.94. The graph shows that the percentage of Class B harvesting areas has increased over the period, largely as a result of a decrease in the proportion of Class C areas. There has also been a decrease in the number of areas designated as Prohibited. This is largely due to the effect of sewage improvement schemes. The percentage of areas assigned Class A status has remained relatively constant over the assessment period, albeit with fluctuations from year to year. Data up to 2003 were included in the assessment for

Table 3.55 Classification categories under Regulation (EC) No. 854/2004.

Classification category	*Microbiological criteria*	*Processing requirements*
Class A	≤ 230 *E. coli*/100 g of flesh and intravalvular liquid	May be consumed without treatment
Class B	≤ 4600 *E. coli*/100 g of flesh and intravalvular liquid in 90% of samples	Purification or relaying for 2 months(1) (or heat treatment by approved process)
Class C	≤ 46 000 *E. coli*/100 g of flesh and intravalvular liquid	Relaying for 2 months(1) or heat treatment by approved process

(1) The competent authority may agree a shorter period on the basis of a risk analysis undertaken by the food business operator

Figure 3.93 Location of UK classified shellfish production areas, 2007. © Crown copyright 2010: permission granted by Cefas.

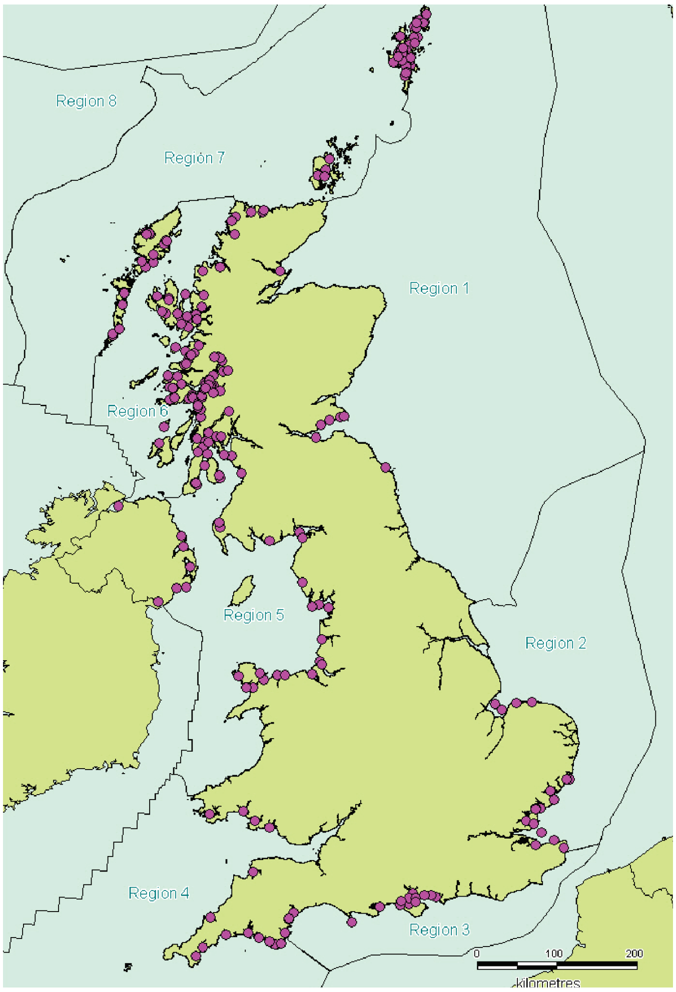
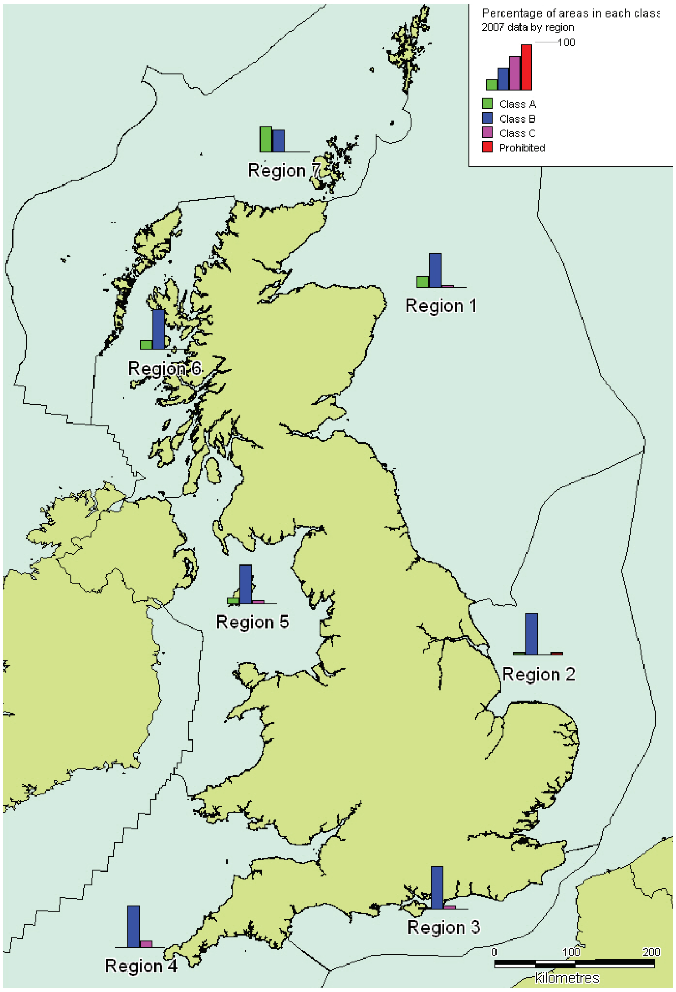


Figure 3.94 Percentage of UK commercial shellfish harvesting areas in each class, 2001–2007. © Crown copyright 2010: permission granted by Cefas.

Charting Progress – there has been an additional increase in the percentage of Class B areas since then.

The percentage of areas in each region assigned Class A, Class B, or Class C status or designated as Prohibited in 2007 is shown in Figure 3.95. The figure shows that the percentage of Class A areas is highest in Region 7 and lowest in Region 4. In contrast, the percentage of Class C areas is highest in Region 4 and lowest in Region 7. Many of the shellfisheries in Regions 1, 2, 3 and 4 are impacted by a combination of large public sewage discharges and land run-off from intensively farmed agricultural land, while

Figure 3.95 Percentage of areas in each classification category, 2007. © Crown copyright 2010: permission granted by Cefas.



those in Regions 6 and 7 are impacted by small community discharges, private septic tanks and land run-off from widespread agriculture.

3.4.5 Progress towards the vision of clean and safe seas

The microbiological status of the marine environment is a key factor in establishing a healthy marine environment for recreation and the production of seafood. The current national microbiological monitoring programmes are based on determination of risk of faecal contamination rather than the presence of specific pathogens. In general, all three monitoring programmes show evidence of a reduction in the microbiological contamination of the marine environment from faecal sources. This should be associated with a reduction in risk to human health, especially given that reductions to date have largely been due to sewage treatment and sewerage improvements and that sewage will contain both human viral and bacterial pathogens. Most of this reduction occurred during the period up to 2002 and further improvement has, in general, been less than the underlying year–to-year fluctuation in compliance due to other factors. However, investment in sewage/sewerage improvements and associated investigation work has continued. In England and Wales, a number of investigations have already been undertaken for shellfish waters with 15 more investigations and monitoring schemes undertaken in 2009. Further investigations are planned for the 2010 – 2015 spending period. A total of £2 billion has been spent on waste water treatment for bathing waters in past decade. In Scotland, Scottish Water is spending £26 million in the period 2006 – 2010 to improve sewage discharges to bathing waters and £7 million to improve shellfish waters (internal analysis undertaken during June 2008 by Scottish Water to inform the Impact Assessment of the River Basin Management Plan for Scotland River Basin District).

Between 2010 and 2014, Scottish Water is planning to spend £16 million on sewage improvements for Shellfish Waters and £7 million to investigate improvements required under the revised Bathing Waters Directive. A total of £87 million has been spent on completed bathing water-related improvements in Northern Ireland in the past three years, and £24.5 million on completed shellfish water-

related improvements. A programme covering two additional schemes costing £6.5 million has just been initiated.

There are a number of concerns with regard to the impact of climate change on the microbiological status of the UK seas and the associated potential health effects:

- More frequent heavy rainfall events increasing faecal contamination due to the operation of storm overflows and land run-off
- Higher average seawater temperatures increasing the extent of uptake of pathogens (and the indicator bacteria) by shellfish
- Higher average seawater temperatures causing an increase in the occurrence and concentration of *Vibrio*s which are pathogenic for humans.

3.4.6 Forward look and need for further work

A number of management measures are required.

Management measures relating to water quality and public health, with respect to faecal pollution, have two separate elements. One is the management of the potential pollution itself, either by eliminating or reducing the pollution at source, or by interrupting the route from the source to the receiving water. The other is to stop people coming into contact with the polluted water or shellfish.

Prior to instituting management measures for pollution sources, it is necessary to determine those sources primarily responsible for the contamination of the receiving waters and/or shellfish. With regard to the quality of shellfish growing waters, there is increasing interest in the application of both source tracing (such as through dosing potential contaminating sources with dyes, spores (e.g. *Bacillus globigii*) or bacteriophages) and source tracking (to differentiate human and animal pollution) (Scott et al., 2002; Drury and Wheeler, 2008). Presently, both human and animal faecal pollution are regarded as of public health significance and, therefore, differentiation of source is used to target investment of remediation procedures rather than differential treatment of harvested shellfish.

Management of the pollution sources themselves has tended to be concentrated mainly on continuous sewage discharges, with larger community discharges receiving at least secondary treatment. The frequency of discharges from storm tanks and combined sewer overflows has also been limited where these have impacted directly on bathing or shellfish waters – the impact of such intermittent discharges is still the subject of investigation and assessment (Kay et al., 2009c). Tertiary treatment of continuous discharges is often achieved using ultraviolet disinfection, although membrane treatment processes have also been successfully applied. Efficient future investment in sewage treatment and sewerage improvements intended to benefit bathing and shellfish waters would benefit from a structured review of the outcome of past and present improvement programmes: the Department for Environment, Food and Rural Affairs (Defra) has recently commissioned a study that will review the outcomes for a number of shellfish waters in England. Further improvements in water quality in a number of areas will need associated reductions in diffuse inputs of faecal contamination and initiatives such as the Catchment Sensitive Farming Programme will contribute to this. Incorporation of microbiological monitoring under the EU

Water Framework Directive and Marine Strategy Framework Directive programmes would allow a better assessment of progress.

With regard to the prevention of contact with contaminated waters or shellfish, further development of management measures needs to take into account the behaviour and persistence of the pathogens, rather than just the indicator organisms on which the monitoring programmes are based (Murray and Lee, 2009). For example, viruses persist for much longer in seawater, and take much longer to naturally depurate from shellfish, than do the bacterial indicators (Lees, 2000). Concerns have been raised about the detection of *Giardia duodenalis* and *Cryptosporium parvum*, protozoal parasites causing gastroenteritis, in samples of shellfish taken from harvesting areas (Gomez-Bautista et al., 2000; Schets et al., 2007). As with enteric viruses, current commercial purification practices will not completely remove cryptosporidia from shellfish (Sunnotel et al., 2007). While there have been a small number of cases of shellfish-related giardiasis reported in the United States, there is presently no epidemiological evidence to show that cryptosporidiosis is associated with shellfish consumption. Other viruses are also of concern, particularly *Norovirus* which is one of the most common bivalve-associated viral pathogens. Our limited ability to measure viral loads in the environment in relation to a dose-response relationship in humans means that environmental viral standards have yet to be established. Such issues continue to be investigated with a view to developing a viable approach to the management of viruses.

The new EU Bathing Water Directive includes a revision to the microbiological indicators, with compliance being based on both intestinal enterococci and *E. coli*. The standards will be tighter than those in the present Directive. The monitoring will be supplemented by profiling of bathing waters (examining the contaminating influences) and the possibility of discounting a proportion of failures deemed to be due to short-term pollution events, with management action being applied during the period of risk. The profiling and risk management approaches have parallels with developments in the shellfish hygiene field where sanitary surveys are required before new harvesting areas are classified and control actions are taken following high results or identified pollution events (Kay et al., 2009b).

Microbiological monitoring of the marine environment has concentrated on estuarine and near-shore coastal monitoring of indicator bacteria of faecal origin. Further information is required on background levels that occur farther from shore and on the occurrence of both faecally-derived and naturally-occurring pathogens. An EU standard method is being developed for the detection and enumeration of viruses in foods. It is anticipated that some form of viral monitoring of bivalve shellfish will be introduced into EU food hygiene legislation once the method has been properly validated (European Communities, 2005).

3.5 Oil and Chemical Spills

3.5.1 Key points

i. Introduction

This section provides information on accidental spillages of oil and chemicals into the sea, operational discharges of oil in produced water released during offshore oil and gas recovery, and atmospheric emissions of polycyclic aromatic hydrocarbons (PAHs). Oil and chemical spills are important as they can have significant, although generally short-term and localised, impacts on the environment.

ii. How has the assessment been undertaken?

Changes in the volume of oil discharged by the offshore industry and overall PAH emissions have been assessed over time. Accidental spillages of oil and chemicals cannot be assessed regionally, due to their episodic and random nature. Also, the cumulative impact of numerous small spills is difficult to assess as the volumes lost are generally not known.

iii. Current status of oil and chemical spills and past trends

- Since 2002, the volume of oil discharged in produced water from offshore installations has reduced by about 50%.
- Atmospheric emissions of PAHs have decreased by 80% since 1990, according to data compiled within the UK National Atmospheric Emissions Inventory.
- In 2007, the most recent year for which data are available, there were 654 accidental discharges of oil from ships and offshore platforms. This represented increases of 29% in discharges from ships and 13% in discharges from platforms relative to 2006. Most spills were small, with only 47 incidents involving spills greater than 2 tonnes.
- Compliance with an ecological quality objective (EcoQO) recommended by the OSPAR Commission relating to the proportion of oiled guillemots found on beaches around the North Sea could not be assessed, as no systematic monitoring programme currently exists. Major UK activity is at present limited to the Orkney and Shetland islands, with patchy coverage on the east coasts of Scotland and England. In the Orkney and Shetland islands, the EcoQO was met and the proportion of oiled guillemots is decreasing.

iv. What has driven change?

Discharges of oil in produced water have reduced as a result of the UK Government's commitment to an international agreement to reduce discharge volumes. Reductions in releases of PAH to the air have resulted mainly from improved discharge controls and changes in industrial practice.

v. What are the uncertainties?

Confidence in the estimates of oil discharged from offshore platforms is high as there is a mandatory reporting requirement to the UK Government. Similarly, confidence in the estimates of PAH emissions is high owing to a UK monitoring programme. The occurrence of oil and chemical spills cannot be assessed regionally, other than to say that most happen in the areas through which major shipping traffic passes.

vi. Forward look

There are increased risks from accidental spillages of oil and chemicals from the increasing traffic in heavy fuel oils from the former Soviet

Union past UK coasts, and due to the increasing size of container vessels which would make dealing with a casualty very challenging.

3.5.2 Introduction

Oil and oil-based contaminants reach the marine environment from a variety of sources including: rivers and run-off from land, atmospheric fall-out following discharge to atmosphere, the offshore oil and gas industry, and accidental spills. There is no comprehensive monitoring for oil in the marine environment, although selected PAHs are monitored in both mussels and sediments within the UK Clean Seas Environmental Monitoring Programme (CSEMP – see Section 1: Introduction). The UK Food Standards Agency (FSA) periodically monitors PAH concentrations in seafood, although it can be difficult or impossible to relate these samples back to their source locations. Information regarding oil spills from offshore production facilities and shipping is gathered through mandatory reporting procedures. However, many illegal discharges from ships may go unreported. Impacts of spilled oil are many and varied, including the smothering of marine animals in shallow waters, narcotic effects on shellfish (which often lead to them being predated before recovery), oiling of seabirds diving through the sea surface, and tainting in, and potential carcinogenic impacts from, contaminated shellfish. Mandatory guideline levels for some PAHs have been set by the EU and the UK Food Standards Agency in order to control this latter possibility. The likely impacts of an oil spill depend upon the type of oil, the quantity of oil spilled, the prevailing weather conditions, the location of the spill and the locally sensitive species present, and the response options selected. Oil may be rapidly dispersed or linger in the environment, and all levels of the food chain may be affected.

Key indicators related to oil and oil-based contaminants are:

- The number of oil spills from offshore installations and shipping each year
- The extent to which the OSPAR EcoQO on the impacts of oil spilt in the marine environment is met. The EcoQO states that the average proportion of oiled common guillemots found dead or dying on beaches in all winter months should be 10% or less of the total found dead or dying in 15 areas of the North Sea over a period of at least five years.

Within the UK, there are insufficient data gathered to allow the assessment in relation to the EcoQO to be made regionally.

This section includes information on trends in discharges of oil from offshore production facilities and atmospheric inputs of PAHs, as well as information on oil and chemical spills.

Through OSPAR, the UK committed the offshore oil and gas industry to a 15% reduction in the quantity of oil discharged in produced water by 2006, relative to the level of discharge in 2000. By 2006, the reduction made was 24%, rising to 45% by 2008.

3.5.3 Developments since *Charting Progress*

Annual discharges of oil in produced formation water reduced by about 25% during the period 2002 to 2006 (Figure 3.96), largely due to reductions in the volume of water discharged. The oil content of the discharged water remained constant at about 20 ppm until 2006, but reduced to about 15 ppm in 2007 due to improved produced water management. This brought the overall reduction in the amount of oil discharged each year between 2002 and

2007 to close to 50%. (https://www.og.berr.gov.uk/information/bb_updates/chapters/Table3_2.htm)

Under OSPAR Recommendation 2001/1, the UK was committed to reduce the volume of oil discharged in 2006 by 15% relative to 2000 for all offshore installations under its jurisdiction at that time. This requirement was met. The discharged oil is well dispersed and has little or no impact on surface sediment dwellers. No oil is now discharged with drill cuttings and the contribution from flaring at offshore installations is minor, at about 10 to 15 tonnes per annum across the UK continental shelf.

Emissions of PAHs (quantified as the sum of the US Environmental Protection Agency's 16 parent PAH priority pollutants) to the atmosphere have decreased by 84% since 1990. During 2006, the largest source of PAHs was road transport combustion, followed by domestic combustion (Figure 3.97). Ten years earlier, the major source was the aluminium smelting industry, which contributed about 50% of the total. Since then, emissions have declined due to improved practice and this industry is now responsible for only 1% of total PAH emissions.

3.5.4 Presentation of the evidence

3.5.4.1 Discharges from ships and offshore installations

Deliberate discharges of oil or oil/water mixtures from ships are prohibited within the North West European Waters Special Area, established by the International Maritime Organization under MARPOL Annex I in 1999. This includes all waters around the UK and its approaches. Information on accidental discharges of oil from ships and offshore platforms is compiled by the Advisory Committee on Protection of the Sea (ACOPS) on behalf of the Maritime

Figure 3.96 Oil discharged in produced water by the offshore oil and gas industry, 2002–2008. © Crown copyright 2010: permission granted by Cefas.

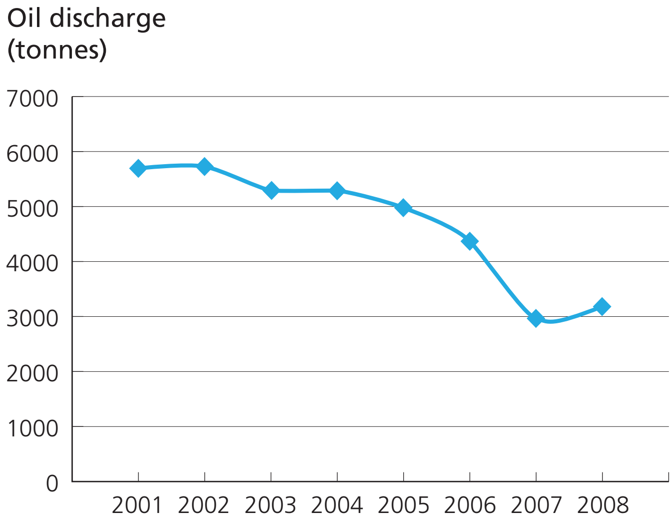
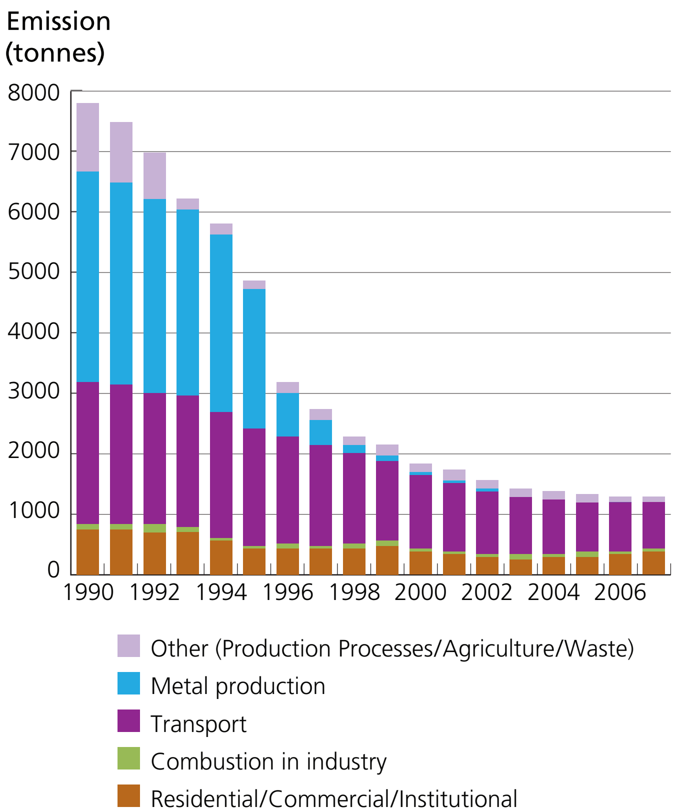


Figure 3.97 Emissions of PAH to the UK atmosphere, 1990–2007 (as the sum of the US EPA 16 parent PAH priority pollutants). © AEA Technology.



and Coastguard Agency each year. In 2007, 654 separate discharges from these sources were identified (ACOPS, 2008). The number of discharges from vessels increased by 29% during 2007 compared to 2006, reversing the continuing downward trend observed each year during the period 2000 to 2006 (see Figure 3.98). The annual increase was attributed to more frequent discharges from vessels both in port and at sea, including 15 additional spills caused by vessel casualties (Figure 3.99). A smaller increase of 13% was apparent in the number of accidental discharges from offshore oil and gas installations during 2007, the fourth consecutive annual increase. Around 80% of all reported discharges were in the open sea, 17% were in ports and harbours and 3% in other marine areas.

Figure 3.98 Annual totals for reported discharges attributed to ships and offshore oil and gas installations, 2000–2007. © Trevor Dixon, ACOPS.

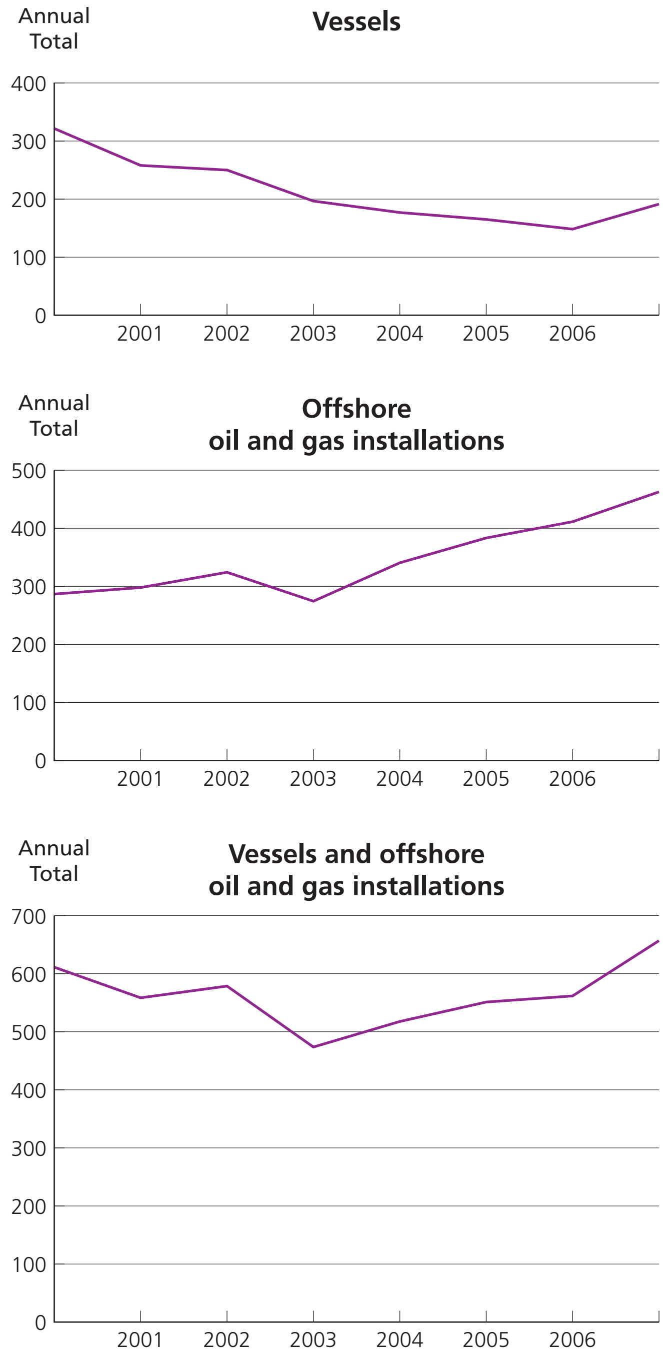
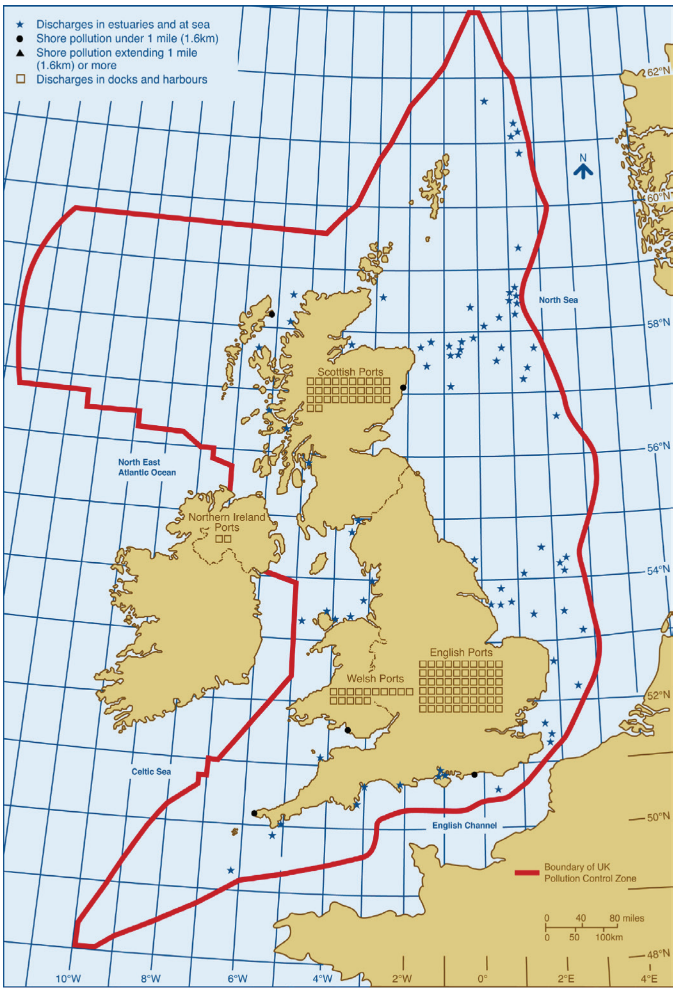


Figure 3.99 Locations of reported oil discharges attributed to ships. © Trevor Dixon, ACOPS.



3.5.4.2 Incidents and accidents involving vessels

Although the majority of incidents are minor, a number have led to the actual or potential release of significant amounts of oil and/or chemicals into the sea from ships during the period 2002 to 2008.

- The car-carrier *Tricolor* sank off Zeebrugge in the southern North Sea on 14 December 2002. Oil released from the sunken vessel caused some limited pollution in the south-east of England, with more in Belgium. Removal of the wreck was completed in 2004.
- On 31 January 2006, the chemical tanker *Ece* sank in the central English Channel following a collision, together with her cargo of 10 000 tonnes of phosphoric acid intended for fertiliser manufacture. Subsequent study showed elevated phosphate concentrations (about four times above background) and enhanced phytoplankton growth close to the wreck site (Kelly-Gerreyn et al., 2007), but no widespread impact.
- On 9 January 2007, 157 tonnes of sodium bicarbonate were released from the *Dunlin alpha* oil platform in the northern North Sea when a hose sheared during pumping from a supply vessel.
- On 20 January 2007, the container ship *MSC Napoli* was deliberately grounded in Lyme Bay to prevent her sinking offshore (Figure 3.100). Most of the 4000 tonnes of oil aboard were removed safely to another vessel, but around 150 tonnes were lost into Lyme Bay resulting in roughly 3000 seabird casualties (Law, 2008).

Figure 3.100 The container ship MSC Napoli *grounded off Branscombe, Devon, in Lyme Bay in January 2007.*
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Around 150 tonnes of oil had earlier been lost off the Brittany coast of France when the vessel first got into difficulties. The cargo was held in 2300 containers, about 150 of which held hazardous goods. Around 100 containers were lost overboard after the ship grounded, but none of these contained hazardous materials. The remaining containers were transferred to Portland Port and, once empty, the *MSC Napoli* began to be dismantled for recycling. Monitoring studies undertaken in Lyme Bay in the vicinity of the grounded vessel indicated that the impact of the incident was less than had initially been feared on the basis of the hazardous goods manifest listing the hazardous chemicals and the quantities carried on board (Law, 2008). Removal of the wreck was completed in August 2009.

- On 3 August 2007, the cargo vessel MV *Jork* struck an unmanned satellite gas platform off Norfolk and sank the following day, with the loss of 4 tonnes of gasoil.
- On 1 February 2008, the Roll-on/Roll-off ferry *Riverdance* grounded on Cleveleys Beach, Blackpool, and could not be refloated. All the oil and lorries aboard were removed, and the vessel was cut up for recycling. This process was completed by November 2008 with no pollution.
- Also on 1 February 2008, the fishing vessel *Spinningdale* ran onto rocks on the island of St Kilda, a World Heritage Site. Concern was expressed about the possibility of there being rats on the vessel and them getting ashore, but this concern was unfounded.
- On 15 January 2008, the timber carrier *Ice Prince* sank 26 miles off Portland Bill in the English Channel. 2000 tonnes of wood carried as deck cargo were released and came ashore around Worthing in East Sussex. The sunken vessel contained 313 tonnes of intermediate fuel oil, which presented a future pollution risk. The removal of this oil was completed on 6 May 2009.
- On 19 January 2009, another timber carrier, the *Sinegorsk*, lost 1500 tonnes of sawn timber off Newhaven, East Sussex, during a storm. On this occasion, the wood came ashore in Kent and, after passing through the Dover Strait, Suffolk.

3.5.5 Progress towards the vision of clean and safe seas

The largest inputs of oil and chemicals are still thought to come from land-based sources via riverine and atmospheric inputs. This is despite large reductions of inputs of oil from refineries to rivers (around 20-fold since 1981) and of emissions of PAHs to the atmosphere (around 4-fold since 1990).

Inputs to the different regions around the UK vary. Due to prevailing winds, inputs from the atmosphere mostly affect the North Sea (Regions 1 and 2). Inputs from refineries occur where these facilities are located (Regions 1 to 5) but the effects are probably now more localised than was the case in the past.

Oil and chemical spill incidents are, by their very nature, episodic and can occur almost anywhere. The UK (as for other European countries such as France) is in a particularly vulnerable situation as a number of extremely busy shipping channels pass our coast, with high levels of passing traffic. The degree of risk increases for certain cargoes and as the size of vessels increases in line with operational economics. Of particular concern are two developments since *Charting Progress*:

- The increasing transport of heavy and residual fuel oils from the former Soviet Union states through the Baltic and Barents Seas and past the UK. This is now at a level of over

50 million tonnes per annum, and presents a serious risk of severe pollution if these vessels were to run ground or sink in UK waters. These heavy fuel oils are not generally amenable to treatment using chemical oil dispersants. Trans-shipping of heavy oils from small to large vessels may also present a risk.

- The economies of scale have been driving container ships to become larger – whether this will change as a result of the global recession remains to be seen. However, the largest container ships using the Port of Felixstowe are now 150 000 tonne vessels, with a cargo capacity of 14 500 20-foot container equivalent units, or 2.5 times as big as *MSC Napoli*. Dealing with an incident involving the loss of one of these vessels would severely strain the response capability of the UK, well-prepared though it is for incidents.

Mitigation measures are in place to reduce the risk of an incident occurring and to reduce the impact of any oil or chemical spill. These measures include:

- Vessel traffic monitoring from shore stations
- Routeing measures, including traffic separation schemes
- Oil spill contingency planning ensures plans are in place to mitigate the effects of any oil spill
- Emergency towing vessels located around the UK coastline, which are able to tow or escort vessels in distress
- Domestic and international Regulations to control ship-to-ship transfers of oil carried as cargo, which are currently in development.

3.5.6 Forward look and need for further work

As the UK Continental Shelf oil and gas industry moves to the decommissioning phase for many installations, oil discharges in produced water are likely to continue to decline as fields are closed down. Monitoring needs to be undertaken on impacts relating to decommissioning activities and legacy pollution (e.g. from drill cuttings piles).

Following the Defra public consultation on the UK Oil Spill Treatment Products Approval Scheme by the Marine Fisheries Agency (the licensing body at the time which has since been incorporated into the Marine Management Organisation), the scheme is being revised to allow the approval of dispersants which will disperse heavy fuel oils effectively, but which may exert slightly more toxicity. Research is currently being undertaken to develop new testing protocols so that new dispersants can be tested and approved for use on heavy fuel oils in the seas around the UK.

3.6 Litter

3.6.1 Key points

i. Introduction

Beach litter. The amount of litter in our seas and on our beaches continues to be a cause for concern. Litter is unsightly and can cause harm to marine wildlife through entanglement and ingestion, through smothering of the seabed and as a route for the introduction of invasive species.

Offshore litter. Seabed litter has been surveyed at only a limited number of sites and data are sparse.

ii. How has the assessment been undertaken?

The data used in the assessment arise from two sources. For beach litter, this is the annual series of surveys undertaken by volunteers for the Marine Conservation Society. Offshore litter data have been collected during cruises associated with the UK Clean Seas Environmental Monitoring Programme (CSEMP) and other research cruises.

iii. Current status of litter and past trends

Beach litter. Beach litter surveys indicate that, in general, quantities of litter on UK beaches have shown no appreciable decrease over the period 2003 to 2007. Average litter densities on UK beaches remain high, at over 2000 litter items/km surveyed, compared to around 1000 items/km surveyed when monitoring began in 1994.

Offshore litter. There is wide variability between sites and at some sites in successive years. Data indicate that there is generally a low abundance of litter on the seabed. However, significantly higher densities of litter found at Carmarthen Bay, North Cardigan Bay, Celtic Deep and Rye Bay, would suggest that these are areas of accumulation, i.e. litter sinks. Polythene, rope, polypropylene twine and hard plastics were the most common forms of litter found.

iv. What has driven change?

There has been little change in the density of beach litter since *Charting Progress*. Litter remains high but relatively steady at around 2000 items /km surveyed.

Increased participation in recycling schemes by the general public and implementation of relatively new legislation may take time to show effect. Implementation of the EU Marine Strategy Framework Directive at the European level will also help to drive change. For offshore litter, there are too few data to assess temporal or regional trends.

v. What are the uncertainties?

For beach litter, data are comparable on an annual basis as protocols are followed during monitoring surveys. For offshore litter, data are too sparse to allow meaningful assessments regarding temporal or regional trends to be made.

vi. Forward look

While marine litter would appear to be a largely preventable issue, the wide range of sources of litter, the number of pathways by which it enters the marine environment, and the fact that litter can be easily transported by winds and currents, makes it a complex issue to address.

Both beach and offshore litter surveys have found that the majority of litter is made of plastic. Plastic litter can take hundreds, if not

thousands, of years to break down, fragmenting into small pieces, and it may never truly biodegrade.

Standardised methods of beach litter monitoring should be used throughout the UK. The methodology currently used by the Marine Conservation Society is comparable to that recommended by OSPAR and that in the recently published UNEP guidelines (Cheshire and Adler et al., 2009).

At present, responsibility for marine litter falls to several UK bodies. For example, local authorities, the Environment Agency and the Maritime and Coastguard Agency. Further effort must be given to helping all bodies responsible for litter to coordinate their efforts and so enable them to fully enforce existing legislation.

- More research is required on the environmental and economic impacts of marine litter
- Future action must be focussed on achieving zero input of litter to the marine environment, i.e. stopping litter at source
- Increased harmonisation and enforcement of existing laws is required between EU countries
- Better education regarding the problems and effects of marine litter is required in schools, for the general public and for all seafarers
- More data are required on the extent and spread of offshore litter – both floating and on the seabed
- Action needs to be taken to reduce the problems caused by derelict and abandoned fishing gear.

3.6.1.1 Regional assessment

3.6.1.1.1 Beach litter

During the period 2003 to 2007, 17 780 volunteers took part in beach litter surveys on 1557 UK beaches, together covering 807 km of coastline and collecting 1614 739 items of litter. The surveys took place on the third weekend of September, in line with Ocean Conservancy's International Coastal Cleanup and the beaches were self-selected by volunteer groups.

The main sources of beach litter in all regions were public litter, fishing litter, sewage-related debris and shipping litter. Plastic items made up the bulk of the material found, accounting for around 70% of the total litter. Every year up to half of all litter cannot be confidently sourced. This principally consists of small unidentifiable fragments or items that could have come from a number of sources. The beach litter findings show regional differences.

Region 1 has the lowest levels of beach litter. Public litter is the highest category with densities of about 500 items/km surveyed. Litter density rose by about 30% between 2004 and 2007.

Region 2 has the second lowest density of beach litter with an average of 1500 items/km surveyed. Public litter and fishing litter comprise the majority of sourced litter. Litter density rose by about 75% between 2004 and 2007.

Region 3 has had a fall in litter density, from around 3500 items/km surveyed in 2003 to just under 1500 items/km surveyed in 2007. This decrease appears to have been driven largely by a drop in the amount of public litter found on beaches in the south and south-west of England.

Region 4 has the highest litter density for the whole of the UK; at just under 5000 items/km surveyed. The region also has the highest levels of public, fishing and shipping litter. The reason for the high litter load on these beaches may be due to the high rate of tourism and fishing in the region, and increased input of litter from the English Channel through prevailing winds and currents.

Region 5 shows very little variation in litter density between 2003 and 2007. It has the highest levels of sewage-related debris of all UK regions. This is mainly due to the influence of two beaches in the area with extremely high levels; high numbers of cotton bud sticks are found at East Bay, Helensburgh and Saltings to Bowlings every year.

Regions 6 to 8 have insufficient data to draw any firm conclusions about trends in litter.

3.6.1.2 Contributions towards GES descriptors

The importance of tackling marine litter has been highlighted in the EU Marine Strategy Framework Directive (MSFD), which includes marine litter as one of eleven high level descriptors of good environmental status (GES). All Member States, including the UK, must put in place a programme of measures by 2016 to ensure that ...*Properties and quantities of marine litter do not cause harm to the coastal and marine environment*. This will require a better understanding of the environmental impacts of marine litter, and if necessary, action will be taken to address the issue.

Knowledge gaps currently exist in the area of marine litter, for example the extent of the implications for marine wildlife. Work has begun to explore possible research opportunities to further expand our knowledge of the harm that litter causes in the marine environment, and to consider what future monitoring may be required to assess litter levels.

3.6.1.3 Future risks

As public and other land-based litter are the greatest sources of beach litter, increased development of coastal areas will have implications for litter levels as will increased use of the coast for recreation.

If no action is taken litter will continue to accumulate and increase in the marine environment and on our beaches with environmental and economic implications.

3.6.1.4 Cross-cutting issues

The effects of litter on wildlife and local communities involve some cross-over between the work of CCSEG and the work of the Healthy and Biologically Diverse Seas Evidence Group (HBDSEG). The effects of litter on fishing catch indicate a possible cross-over with the Productive Seas Evidence Group (PSEG).

3.6.2 Introduction

The United Nations Environment Programme (UNEP) Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA) defines marine litter as ...*any persistent, manufactured or processed solid material discarded, disposed of, or abandoned in the marine and coastal environment*. The Marine Conservation Society would argue that the term persistent should not be used, as a man-made item in an aquatic environment is litter irrespective of how long it will take to degrade (MCS, 2007).

Litter, ranging from plastic bottles to sanitary towels, crisp packets to strapping bands and fishing nets to tyres, is an all too persistent and common form of pollution in the marine and coastal environment, causing harm to wildlife and human interests.

Litter enters the marine environment and is deposited on beaches from a variety of sources, including direct littering by beach visitors, discarded or lost fishing gear from fishing vessels, illegal dumping by ships and small marine craft, sewage discharges via combined sewer overflows (CSOs) and fly-tipping. Litter is also carried by rivers and streams into coastal waters, so littering in urban areas can make a significant contribution to marine litter. Much of the litter accumulates along the strandline, deposited by the incoming tides, while sand dunes, groynes, rocky areas and promenades also act as sinks for wind-blown litter. Litter may be transported over long distances by currents and wind.

At present, the only long-term dataset concerning beach litter in the UK is held by the Marine Conservation Society, a charity that runs two beach litter survey projects. There are limited data available for submerged litter at coastal and offshore locations. However, over the past six years observations have been made during trawling activities associated with the UK Clean Seas Environmental Monitoring Programme.

There are many different types of litter that, accidentally or intentionally, enter our seas and are deposited on our beaches. These can be divided into categories according to their likely material or source.

3.6.2.1 Material

Surveys indicate that plastics constitute the majority of debris found on beaches in the UK. Common litter items on UK beaches include plastic pieces, rope, bottle lids, crisp and sweet wrappers, cotton bud sticks, cigarette stubs and pieces of fishing net. All of these are now entirely or partly composed of plastic (MCS, 2007).

As plastics slowly break down, smaller and smaller fragments and fibres are created. Plastic particles can contain additives, and may also adsorb chemicals from their surrounding environment. If the particles are ingested by sediment dwellers these contaminants could enter the food chain, although the potential environmental impact of this process is currently unknown.

Microscopic plastic particles have been found in plankton samples dating back to the 1960s, but a significant increase in abundance has been recorded since then (Thompson et al., 2004). Other forms of plastic that are of concern are the plastic preproduction pellets that form the feedstock of all plastic items and the small plastic beads used as scrubbers in cleaning products.

3.6.2.2 Sources of marine and coastal litter

Identification of the sources of litter is vital to ensure that preventative measures are directed effectively to reduce the quantity of litter in the environment. There are four main sources of litter: the public, shipping, fishing vessels and sewage outfalls. The proportion of litter on an individual beach that can be attributed to each of these sources varies according to local inputs. However, even beaches that are remote from direct inputs can be affected by large levels of litter carried by prevailing winds and currents.

Each year up to 40% of all litter cannot be confidently sourced, either because the litter items are too small to identify or because they could have come from a number of sources. The litter that can be sourced can be divided into two broad categories: items from offshore sources and items from land-based sources.

3.6.2.2.1 Offshore sources

Commercial and recreational fishing and the shipping industry make up about 15% of all sourced beach litter in the UK. Discarded fishing gear, including fishing nets, rope, hooks, lines and weights represents a major source of litter on many beaches, especially in areas where commercial or recreational fishing is intensive. Lost or discarded fishing nets and rope pose a particular threat to marine wildlife through entanglement and ingestion as well as to fishermen and other seafarers through fouling of active fishing gear and ship propellers.

Shipping litter includes pallets, strapping bans and drums, as well as litter derived from containers lost at sea. Cargo may be washed overboard during stormy weather and contribute to coastal litter levels. The lost cargo may also pose a hazard to shipping.

3.6.2.2.2 Land-based sources

Land-based sources contribute over 40% of all sourced litter on UK beaches (MCS, 2007). Litter may be carried out to sea or deposited on beaches from land-based sources via winds, drains, rivers, and storm-water systems. Public litter either dropped directly onto the beach or dropped inland and then blown or swept out to sea contributes significant amounts of litter in coastal areas.

Other coastal sources include illegal fly tipping, and unprotected waste disposal sites, as lightweight rubbish from uncovered sites may blow into the sea or adjacent waterways. Local businesses, including fast food outlets, camp sites, building sites and fish farms can also be a source of litter. Industrial plants synthesising plastic are the main source of raw plastic pellets, which are discharged in effluent to rivers and coastal waters or lost during packing and transport.

Sewage-related debris (SRD) including items such as cotton bud sticks, sanitary towels, plastic tampon applicators, tampons, nappies, condoms, toilet fresheners and wet wipes can enter the marine environment via the sewage system, either via inadequately treated sewage, illegal connections or combined sewer overflows after heavy rainfall.

3.6.3 Impacts of marine litter

The disposal of litter at sea or dropping of litter on beaches has a wide range of environmental and economic impacts. It can cause harm to wildlife and ecosystems, a loss or reduction in the aesthetic quality of beaches and coastal areas, and potentially a loss of tourism in seaside areas with high levels of beach litter. Lost or discarded fishing gear can also catch or entangle a wide range of marine species, including seabirds and marine mammals.

3.6.3.1 Environmental impacts

3.6.3.1.1 Impacts on wildlife

The full impact of litter on marine species is difficult to assess because of limited research in this area and the fact that sampling is largely constrained to land-based observations of beached animals. Research so far has primarily measured frequency of interaction rather than

assessing the proportion of a population that is affected. Marine litter can directly harm wildlife as a result of entanglement and ingestion, and a wide range of species can be affected.

Sub-lethal effects (such as difficulties in feeding following litter ingestion, or increased energy needed for swimming following entanglement in litter, which can lead to a decreased ability to survive and/or reproduce) caused by plastic ingestion are difficult to estimate, but are probably more common than lethal effects (Ryan, 1990; Pemberton et al., 1992).

3.6.3.1.2 Entanglement

A variety of litter items are known to cause entanglement to marine species, including seals and dolphins. Items such as fishing nets, fishing line, plastic bags, strapping bands and four/six-pack yokes can reduce movement and potentially result in serious injury, death by starvation, drowning or suffocation. Discarded fishing nets particularly can continue 'ghost fishing', i.e. catching and killing fish and marine animals after the nets have been lost or discarded at sea or on the seabed.

It is difficult to assess the full rate of entanglement of marine animals by litter for several reasons. For example, animals that die as a result of entanglement may sink to the seafloor, or be consumed by predators, and entanglement in litter may happen anywhere at sea whereas most observations of entangled marine animals are land-based observations. However, at least 144 marine species are known to have become entangled in litter (Laist, 1997).

3.6.3.1.3 Ingestion

Ingestion of litter can cause physical damage and mechanical blockage of the oesophagus and digestive system, resulting in a false sensation of fullness or satiation, as the litter may remain in the stomach. This can lead to internal infections, starvation and even death. Ingestion of marine litter has been reported in 177 marine species (Laist, 1997).

Plastic items, such as plastic bags and balloons have been mistakenly identified as food and eaten by some marine mammal, turtle and shark species. Turtles, particularly leatherback turtles; the most commonly seen turtles in UK waters, are especially at risk from plastic bag ingestion, as these bags closely resemble jellyfish, their primary prey, when suspended in the water column. Plastic bags along with sheeting and plastic pieces are the predominant synthetic items found in the stomachs of turtles.

A recent study of leatherback turtles noted a marked increase in ingested plastic from the late 1960s. 34% of leatherback turtles found washed up dead contained plastic, mainly plastic bags. Of those that contained plastic, 8.7% contained sufficient quantities to cause death by starvation although the authors suggested that ingestion of any plastic may compromise turtle survival (Mrosovsky et al., 2009).

Bird species thought to be the most susceptible to ingestion of plastic particles are surface-feeders (albatrosses, fulmars, shearwaters, petrels, gulls) and plankton-feeders (auklets, puffins). Research into the stomach contents of dead fulmars from the North Sea showed that 94% of stomachs contained pieces of plastic (Van Franeker et al., 2008).

3.6.3.1.4 Transport of marine species

Floating items of plastic debris are a potential method of transport for colonisation by non-native species. Plastics provide a good surface for fouling organisms to attach to, as they remain buoyant and are extremely durable. The

invasive barnacle species *Elminius modestus* was found on plastic on the shoreline of the Shetland Islands (Barnes and Milner, 2005). In the North Sea, modifications of the coastline and increased levels of marine litter may provide more favourable habitats for the settlement of the benthic life stages of jellyfish.

3.6.3.1.5 Beach cleaning operations

An indirect effect of litter on the coastal environment is the ecological impact of mechanical cleaning, which is carried out by many local authorities to keep popular beaches free of litter, predominantly through the summer season.

Although mechanical cleaning clears the majority of litter, it fails to remove smaller objects such as cotton bud sticks (Somerville et al., 2003), and also strips the beach of its natural strandline, an important part of the beach food web. Removing organic matter from the beach can cause significant reduction in diversity at microbiological and endofaunal levels (Kerckhof, 2004). Many invertebrates, such as amphipods (small crustaceans including sand hoppers), form an important component of the diet for some bird species and beaches with the highest amphipod counts also support a large local bird population.

3.6.3.1.6 Socio-economic impacts

There is no doubt that the general public and local authorities are concerned about, and object to, the quantity of litter on streets and beaches. A Clean Coast Scotland Community Council Questionnaire (Roberts, 2002) sent to 72 coastal Community Councils reported that 60% of respondents cited beach litter as their major concern, followed by sewage pollution and shipping waste. An Encams survey of beach users found that a clean beach was the biggest factor influencing a visit to the beach. Beach litter, and in particular broken glass, followed by sanitary items were found to be the biggest cause of offence to beach users (Encams, 2003).

Marine litter such as syringes, broken glass and cans cause injuries and may have impacts on human health (WHO, 2003). Recreation and tourism are particularly affected by the presence of sewage related debris. SRD on a beach suggests that adjacent waters are contaminated by sewage, which constitutes a health risk to bathers, divers, sailors, surfers and other water users (Sheavely and Register, 2007). Many more people are becoming aware of these risks and may avoid beaches where SRD is a recurring problem.

Local communities can also suffer economic loss through clean-up costs, fouling of marine equipment and fishing gear, direct competition with fisheries (ghost fishing) and reduced value of catches.

3.6.3.1.7 Clean-up costs

Repeated beach clean-up efforts reduce the amount of debris on the shore in the short term, but these reactive efforts can be expensive and time consuming, and do not directly address the source of the problem. Local authorities, and ultimately local taxpayers, bear the financial burden of clearing litter on UK beaches. In a survey of 56 local authorities in the UK, the annual expenditure on beach cleaning ranged from £15 per kilometre in West Dunbartonshire to £50 000 per kilometre in Wyre and came to a total of £2197 138 (KIMO, 2000). A study of a 6 km stretch of beach at Studland in Dorset found that one million visitors per year resulted in 12 to 13 tonnes of litter being collected each week during the summer, at a cost of £36 000 per year (Environment Agency, 2001). The direct

and indirect cost of litter on the Kent coastline has been estimated at over £11 million per annum (Gilbert, 1996).

Harbour authorities also have to pay for the costs of keeping navigational channels clear of litter. A survey of 42 harbour authorities reported that up to £26 100 is spent per year in some ports, to clear fouled propellers and remove debris from the water (KIMO, 2000).

3.6.3.1.8 Fishing interests

Marine litter can result in lost revenue for fisheries, due to the time and effort involved in sorting debris from the catch, while larger items may damage fishing gear. A survey of fishermen in Shetland reported that 92% had accumulated marine debris in their nets; 69% had had their catch contaminated and 92% had snagged their nets on debris on the seabed (KIMO, 2000). Costs associated with the time spent to clear and repair nets and from lost catch due to contamination amounted to between £6000 and £30 000 each year (KIMO, 2000). Litter can also cause problems such as fouled propellers and blocked seawater intakes and evaporators causing engine failure, costly repairs, and delays.

3.6.3.1.9 Damage to vessels

Millions of pounds of insurance claims are made every year worldwide as a direct result of damage to vessels caused by floating litter. Cargo pallets and shipping containers may be particularly hazardous if they damage other vessels and can cause further incidents if they disable a vessel in a shipping lane.

3.6.4 Developments since *Charting Progress*

Since *Charting Progress*, litter levels on UK beaches have remained high, and with the exception of Region 3, there has been no significant decrease or increase in the amounts and types of litter found. In *Charting Progress* litter was reported to be a largely preventable problem and it was assigned a 'red' status (unacceptable) in some areas. For *Charting Progress 2* the overall traffic light assessment for beach litter is amber (some problems) in Regions 1 to 5, with insufficient evidence available to make an assessment for the other regions.

Unfortunately, there has been little success in decreasing the amount of litter in our seas and on our beaches. This is due to a number of factors including the myriad of sources that contribute to marine litter, how to correctly target the main sources of litter and the problem of how to deal with the litter already present in the marine environment.

There is however better public awareness of waste issues in general with greatly increased levels of recycling, and this has resulted in more public participation in community clean-up schemes including beach clean-up schemes. The Fishing for Litter scheme (see Section 3.6.5.3.4) is also highlighting the problems of marine litter to the fishing community.

Legislation implemented since *Charting Progress* such as The Merchant Shipping (Port Waste Reception Facilities) Regulations and the Updated Code of Practice on Litter and Refuse should help to improve litter levels at sea and on beaches. However these need more time to show any positive effect. The EU Marine Strategy Framework Directive should also help to increase understanding of the problems of marine litter and to propose actions to achieve Good Environmental Status.

3.6.4.1 Developments in approach and indicators used

To date, the focus has been on collecting data on litter amounts, material types and sources. Work under OSPAR (a commission which guides international cooperation on the protection of the marine environment of the North-East Atlantic by fifteen Governments of the western coasts and catchments of Europe, together with the European Community) is now being carried out to develop indicators for sources of beach litter and on developing ecological quality objectives for marine litter in the North-East Atlantic region. Litter is also one of the 11 descriptors chosen to assess Good Environmental Status under the EU Marine Strategy Framework Directive, and criteria and methodologies are being developed to enable this assessment to be made, at a regional or sub-regional level. Gathering data to allow a robust assessment will be essential, and the United Kingdom Marine Monitoring and Assessment Strategy (UKMMAS) community may need to develop a more comprehensive monitoring programme to do this. UNEP and OSPAR are developing worldwide protocols for monitoring marine litter, which would meet the need for harmonisation of these activities.

Ongoing research in the Netherlands on the stomach contents of northern fulmars has confirmed the continued presence of litter particularly plastic in the Northern North Sea. This research has been used to develop a proposed ecological quality objective (EcoQO) on plastic particles for the OSPAR area.

3.6.4.2 Relevant research/activities

Recent research has focused on the breakdown of larger plastic items to microplastic particles (Thompson et al., 2004) and on plastic pellets in the oceans. There has been a particular focus on the ability of these particles and pellets to adsorb toxins onto their surfaces (Mato et al., 2001; Takada et al., 2006), and on the possibility of these toxins being ingested and the toxins being transferred to organisms at the base of the food web (Teuten et al., 2007). Research is also being carried out on the role of marine litter as a vector for non-native species transport (Barnes, 2002; Barnes and Milner, 2005).

Research on the contents of fulmar stomachs around the North Sea is showing a decrease in the amount of industrial plastic (pellets) and an increase in the amount of user plastics in fulmar stomachs. The birds with the highest loads of litter were found in areas closest to the busiest shipping lanes (Van Franeker et al., 2008).

3.6.5 Presentation of the evidence

3.6.5.1 Links between European related policies

The MARPOL Convention (as enacted by The Merchant Shipping (Prevention of Pollution by Garbage) Regulations and the Merchant Shipping (Port Waste Reception Facilities) Regulations in the UK) is the main legislative link between EU countries. However, there needs to be better coordination between ports and countries on reporting facilities for waste amounts/deliveries and a harmonization of the charging system.

3.6.5.2 Beach litter

3.6.5.2.1 Beach litter surveys

The Marine Conservation Society (MCS) has carried out an annual UK-wide beach litter survey, through its Beachwatch project, every year since 1993. The surveys are compatible with the OSPAR methodologies and are used as the UK contribution to the International Coastal Cleanup report.

3.6.5.2.2 Data sources

The litter surveys are carried out by volunteer groups, with each group recording all the litter found on a minimum 100 metre stretch of beach using data sheets provided by MCS. The data sheets are divided up according to material types, for example, plastic, metal and wood for ease of recording. These are then returned to MCS for validation and entry onto a UK beach litter database. The information collected is then analysed to determine average beach litter levels, and to examine material types and sources. Volunteer groups are giving clear guidance on how to carry out surveys to ensure that results are comparable across the UK.

This project provides the only long-term data set on beach litter levels, material types and sources on beaches in the UK. The range of data available is generally good although certain regions, notably Regions 6, 7 and 8, have inadequate data coverage to draw any firm conclusions about trends.

3.6.5.2.3 Confidence in the data

Some of the limitations of the data set stem from the programme being an entirely volunteer led scheme and include the following:

- Some volunteer groups survey areas of more than 100 m, limiting the survey area to 100 m would standardize the data better
- Although initial training workshops were given to many groups, these need to be repeated/ refreshed so that all volunteers are briefed in terms of accuracy. However all groups do receive an initial pack explaining the methodology required
- Some misidentification probably does occur although each data sheet is examined before entry onto the database to pick up any unusual or unlikely entries
- Although every attempt is made to source items as accurately as possible, some grey areas do occur, for example, plastic bottles are currently sourced to public litter although many may come from shipping or fishing or be used as bailers or oil containers. Sourcing items as accurately as possible is a task that is under constant review and may never be wholly accurate
- Owing to the nature of volunteering work, the same beaches may not be consistently surveyed each year and this also needs to be addressed.

Despite these limitations the litter surveys are carried out each year in a consistent manner by each group of volunteers and so results are comparable year on year and do give a good indication of trends in litter levels and sources.

3.6.5.2.4 Trends

There does not seem to have been a significant decrease or increase in the amount of litter found on UK beaches since 2003 (see Figure 3.101) despite legislation governing marine litter and increased public awareness about waste and recycling. In general, litter levels are high throughout all regions – an average of over 2000 items of litter per kilometre surveyed was found over the period 2003 to 2007. However, if data collected since monitoring started are considered then there has been a considerable rise in the amount of litter on UK beaches. In 1994, 1045 items/km surveyed were recorded and by 2007 this had risen to 2054 items/km surveyed. This equates to an increase of 96.5%. The majority of this increase occurred between 1994 and 2003, since then litter levels have been relatively steady but still high.

Throughout all regions most sourced litter continues to come from the general public either through direct littering on beaches or through land-based litter being swept or blown onto beaches.

Over the whole of the UK, fishing is consistently the second major identified source of beach litter, followed by sewage-related debris, shipping, fly tipping and medical uses. Every year up to half of all litter cannot be confidently sourced. This component consists of small unidentifiable fragments or items that could have come from a number of sources.

Plastic litter items remain the highest material source of litter in all regions (see Figure 3.102) with on average over 1500 plastic litter items found per kilometre of beach surveyed.

Figure 3.101 Beach litter items (all types) per kilometre surveyed in all UK regions, 2003–2007. © Marine Conservation Society.

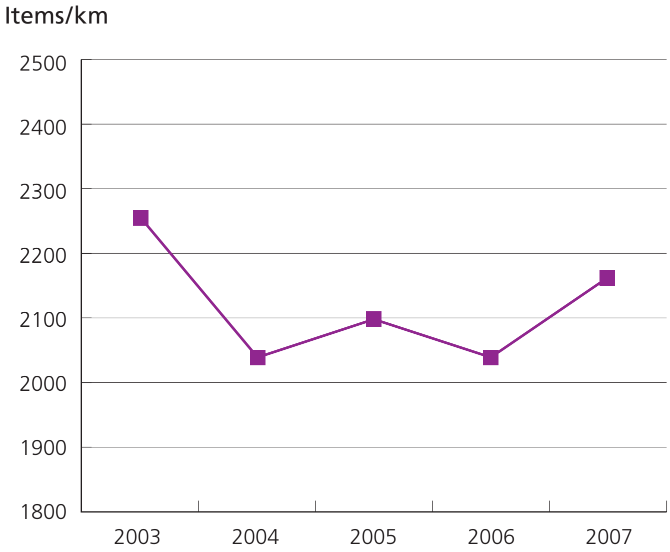
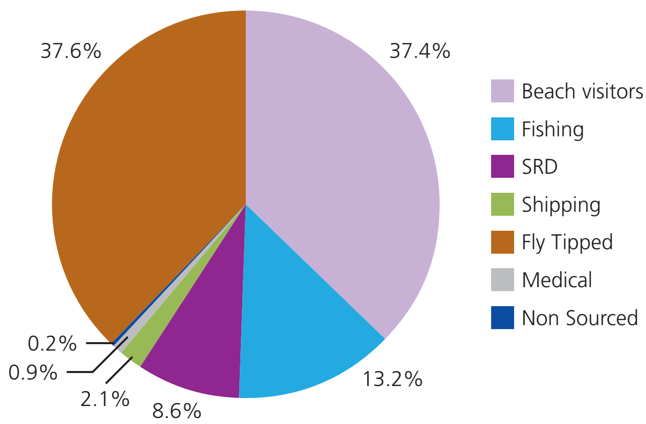


Figure 3.102 Percentages for each litter source 2003–2007. © Marine Conservation Society.



The numbers of beaches and the lengths of beaches surveyed for each region are given in Appendix 1, while a breakdown of source items per kilometre surveyed for each region is given in Appendix 2.

3.6.5.2.5 Gaps

There are noticeable gaps in the beach litter data for Regions 6, 7 and 8. For floating and seabed litter a number of ad hoc surveys have been carried out in the past. A more comprehensive programme for monitoring seabed litter is needed across all regions.

3.6.5.2.6 Regional seas beach litter data

A comparison the total number of litter items collected in Regions 1 to 6 each year since 2003 is given in Figure 3.103.

Region 1. The Northern North Sea has the lowest levels of beach litter around the whole of the UK coast with an average of just over 1300 items/km (Figure 3.104). Public litter makes the greatest contribution with densities of around 500 items/km each year. Fishing litter makes the second most important contribution but is still only represents about half the UK average for each year. Litter density rose by about 30% from 2004 to 2007.

Figure 3.103 Total litter items per kilometre surveyed for Regions 1 to 6, 2003–2007. © Marine Conservation Society.

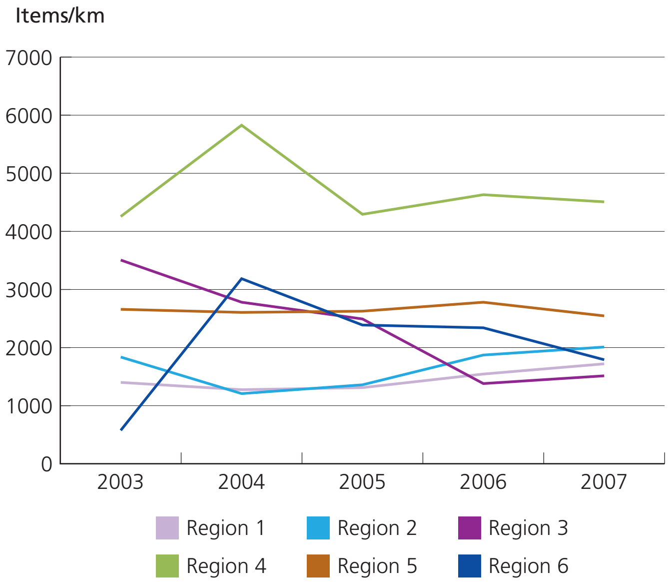
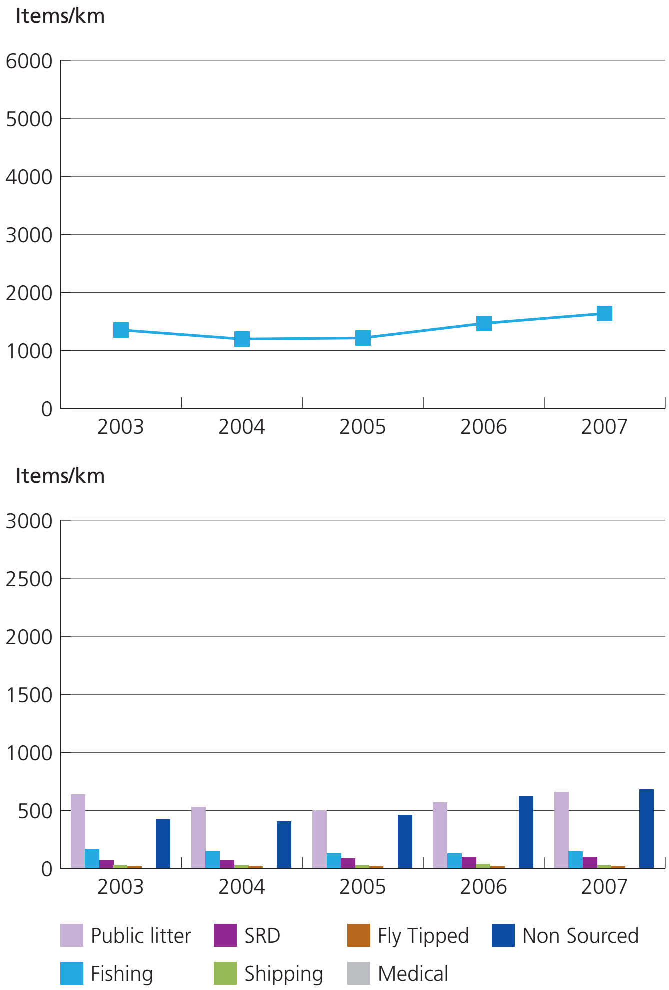


Figure 3.104 Total litter items per kilometre surveyed in Region 1 and litter items by source, 2003–2007. © Marine Conservation Society.



Region 2. The Southern North Sea has the second lowest density of beach litter with an average of 1500 items/km surveyed (Figure 3.105). Public litter makes the greatest contribution with average densities of 680 items/km. Fishing litter again makes the second most important contribution but with densities well below that of public litter; an average of around 155 items/km were found. As with Region 1, litter density rose between 2004 and 2007, but in this case by roughly 75%.

Region 3. The Eastern Channel unlike the other regions shows a clear downward trend in litter density, from almost 3500 items/km in 2003 to just under 1500 items/km in 2007 (Figure 3.106). Although public litter again provides the main contribution with an average density of 927 items/km, the downward trend seems to have been driven largely by a drop in the amount of public litter. The precise reason for this is unclear and needs to be investigated further.

Figure 3.105 Total litter items per kilometre surveyed in Region 2 and litter items by source, 2003–2007. © Marine Conservation Society.

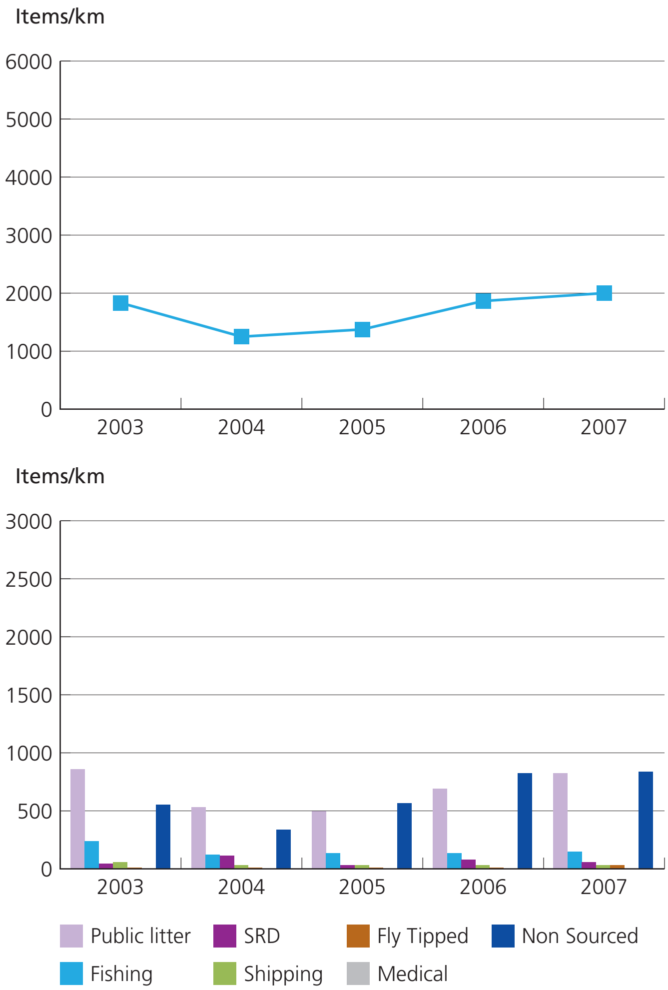
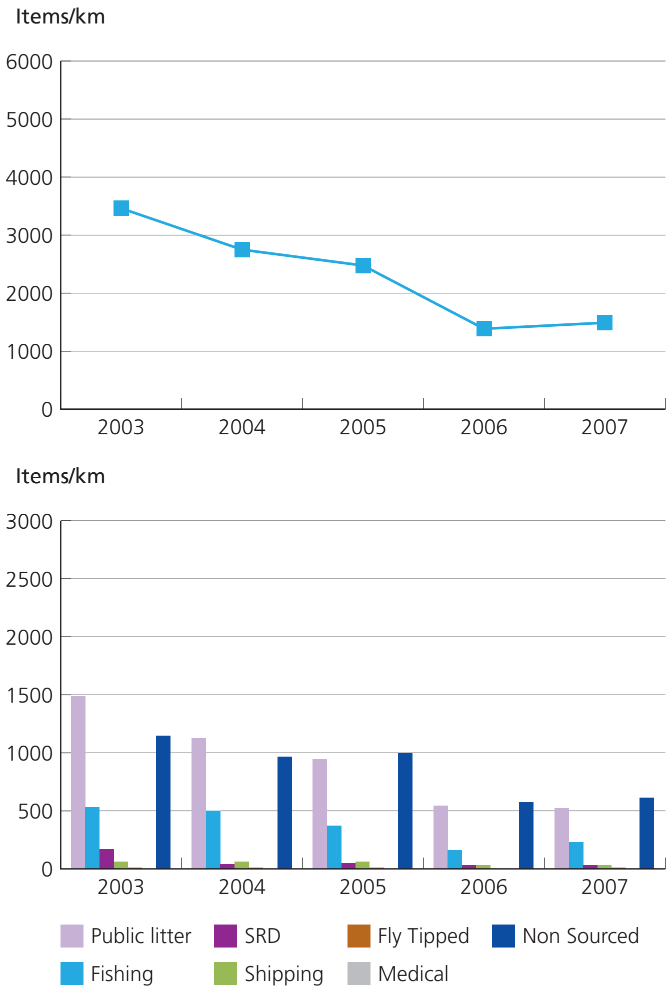


Figure 3.106 Total litter items per kilometre surveyed in Region 3 and litter items by source, 2003–2007. © Marine Conservation Society.



Region 4. The Western Channel and Celtic Sea region shows the highest litter densities for the whole of the UK; with an average of just over 4600 items/km, well above the UK average of just over 2000 items/km (Figure 3.107). As well as high levels of public litter (average 1506 items/km) the region has extremely high levels of fishing and shipping litter (averages of 885 and 79 items/km, respectively). The reason for the high litter load on these beaches may be due to the high rates of tourism and fishing in Region 4, and increased input of shipping litter from the Channel through prevailing winds and currents.

Region 5. The Irish Sea shows very little variation in litter density between 2003 and 2007 (Figure 3.108). Public litter again makes the greatest contribution with an average of 857 items/km. Region 5 also has the highest levels of sewage-related debris of all UK regions. However, this is mainly due to the influence of two beaches in the area with extremely high levels; high numbers of cotton bud sticks have been found every year at East Bay, Helensburgh and Saltings to Bowlings (see Table 3.56).

Figure 3.107 Total litter items per kilometre surveyed in Region 4 and litter items by source, 2003–2007. © Marine Conservation Society.

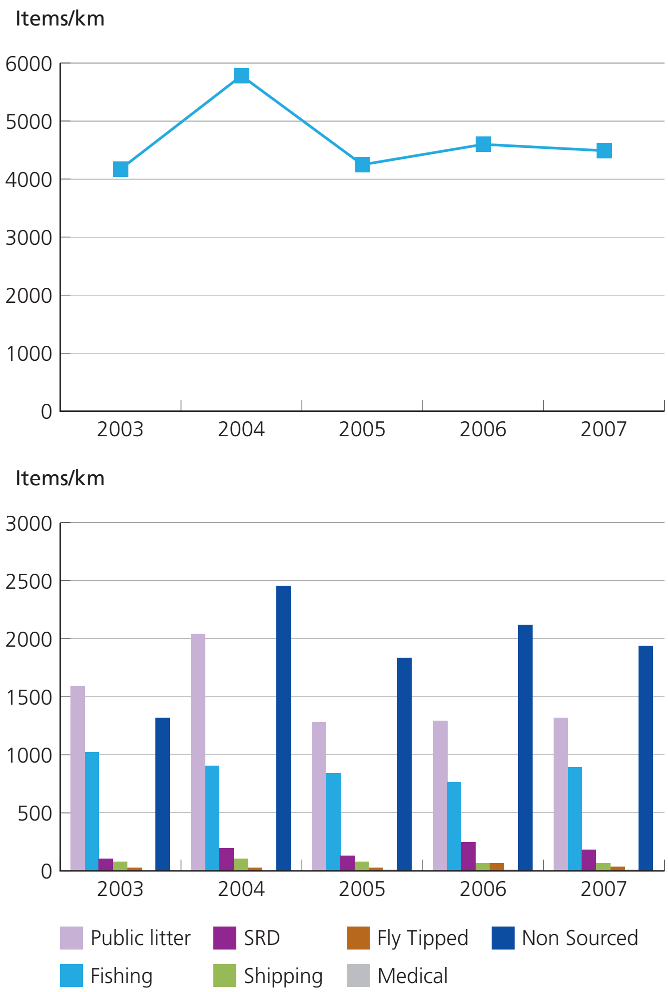


Figure 3.108 Total litter items per kilometre surveyed in Region 5 and litter items by source, 2003–2007. © Marine Conservation Society.

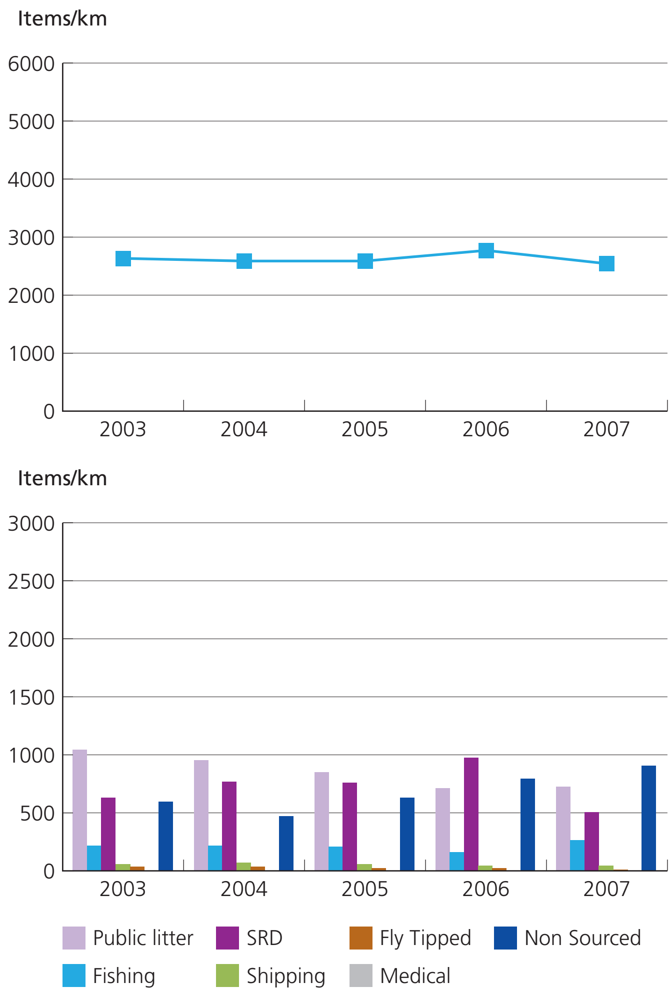


Table 3.56. Number of cotton bud sticks at two Scottish beaches, 2003–2007.

Beachwatch Year	*Saltings to Bowling*	*East Bay*
2003	10 000	353
2004	2 170	3127
2005	5 900	5180
2006	13 500	7416
2007	no survey	8525

Region 6. Only a few data are available for Region 6, Minches and Western Scotland, ranging from 2 to 5 beaches and 0.5 and 1.45 km surveyed. Therefore it is difficult to draw any firm conclusions. However the main sources of litter were public litter and fishing litter.

Region 7. The Scottish Continental Shelf was only surveyed once in 2006. The main sources of litter were public litter and fishing litter (983 and 883 items/km, respectively). Again these numbers need to be treated with caution, as no further data are available.

Region 8. The Atlantic North-West Approaches contains no appreciable land masses and has never been surveyed for beach litter.

3.6.5.2.7 Beach litter summary

The Marine Conservation Society Beachwatch litter surveys provide a snapshot in time of litter over one weekend in the year. The data indicate that litter levels are still high throughout the UK, with an average of over 2000 items of litter per kilometre of beach surveyed. There are regional variations, and overall litter levels are highest in Region 4. The major contributions are from public litter and fishing litter. The only departure from the overall pattern is in Region 3 where there seems to be a definite downward trend in litter levels.

Table 3.57 presents our expert opinion on the status and trends for beach litter. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. Trend arrows are based on the evidence available, showing whether the state is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). The confidence rating of each assessment is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

3.6.5.2.8 Plastics

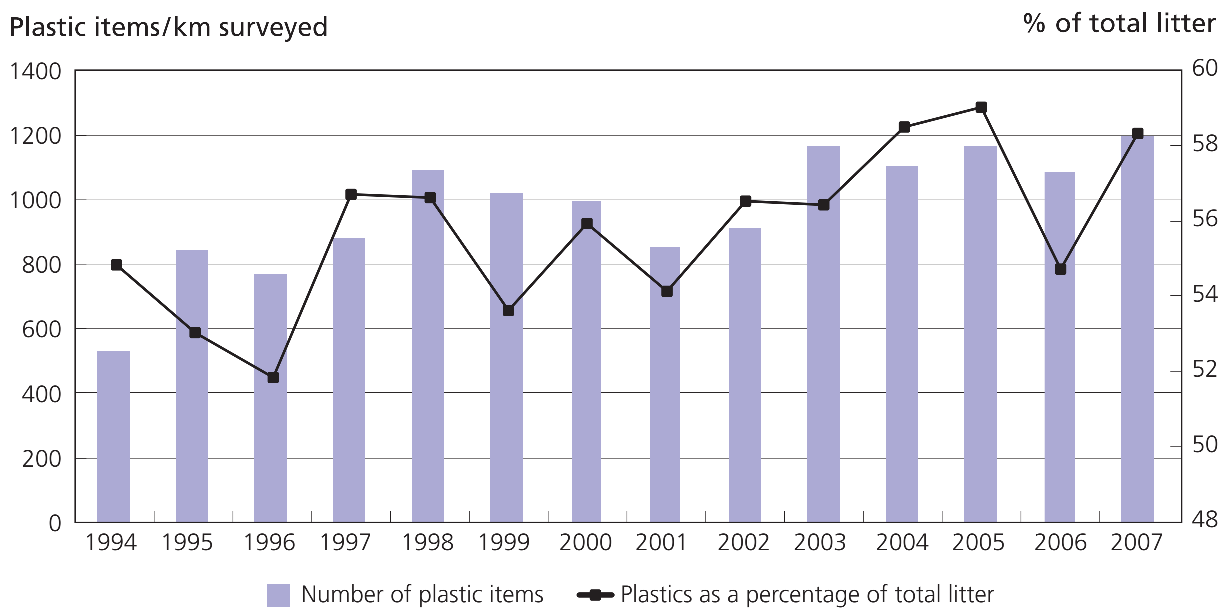
Over the past 15 years, plastic litter has consistently accounted for over 50% of all litter found during Marine Conservation Society Beachwatch surveys (Figure 3.109). Most items of sewage-related debris are also entirely or partially made of plastic. This means that the overall percentage of beach litter caused by persistent plastics is over 70%. The density of plastic found on UK beaches has increased by 126% since 1994 and small plastic pieces have been the number one item found in every Beachwatch survey since 1998.

The increase in plastic usage, the replacement of many 'traditional' packaging materials such as glass by plastic and increasing use of synthetic fishing nets has exacerbated the problem. Some of plastic's main qualities which make it such a useful material such as durability and light weight also make it a menace when disposed of in the marine environment and on beaches.

Table 3.57 Regional assessment summary for beach litter.

CP2 Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
1	Public litter (500 items/km)	Lowest levels of beach litter, but 30% rise from 2004 to 2007.	↔		III	?
2	Public litter and fishing litter (1500 items/km)	Second lowest density of beach litter, but 75% rise from 2004 to 2007	↔		III	?
3	Public litter	Downward trend in density of litter	↑		III	?
4	Highest levels of public, fishing and shipping litter	The highest litter densities in the UK; just under 5000 items/km	↔		III	?
5	Highest levels of sewage-related debris of all UK regions	Very little variation over the period 2003 to 2007	↔		III	?
6		Wide variability between sites and at some sites in successive years	?		0	
7			?		0	
8			?		0	

Figure 3.109 Plastic litter items per kilometre of UK beaches surveyed and as a percentage of total beach litter, 1994–2007. © Marine Conservation Society.



3.6.5.3 Offshore litter data

3.6.5.3.1 Fishing surveys

Litter was collected during fishing surveys, which were undertaken at intermediate and offshore sites around England and Wales in June/July between 2003 and 2008 using a Granton trawl – a large bottom fishing net. At each station, the trawl was towed for 30 minutes at a speed of 2 knots covering a distance of 1 nautical mile (1848 m). In general, two or more tows were conducted at each site, depending on the numbers of fish caught. After each tow, litter was removed from the net, either from the catch or from the net itself if the litter was entangled. The number of litter items, type of litter and size were recorded from each tow, however, for comparative purposes, only data from the first two tows were assessed.

Data were classified into 12 categories: polythene sheeting/bags (e.g. food wrapping, bin liners), plastic (e.g. hard plastic drink bottles, fish baskets, guttering), rope and twine (e.g. polypropylene rope or strands), metal (e.g. crisp packets, food cans, flex), glass (e.g. wine bottles and jam jars), rubber (e.g. gloves, boots, tyres), cloth (e.g. rags and clothing), paper/cardboard (e.g. food wrapping and cellophane), polystyrene (e.g. food, drink containers), wood (e.g. planks and boxes), sanitary/sewage (e.g. tampons) related debris and other (e.g. shoes, television set).

The density of litter (number of items per hectare), was then determined for each station by dividing the total number of litter items per station by the area trawled:

Area trawled per tow
= Spread of net × distance travelled
= 25.91 m × 1848 m
= 4.79 ha

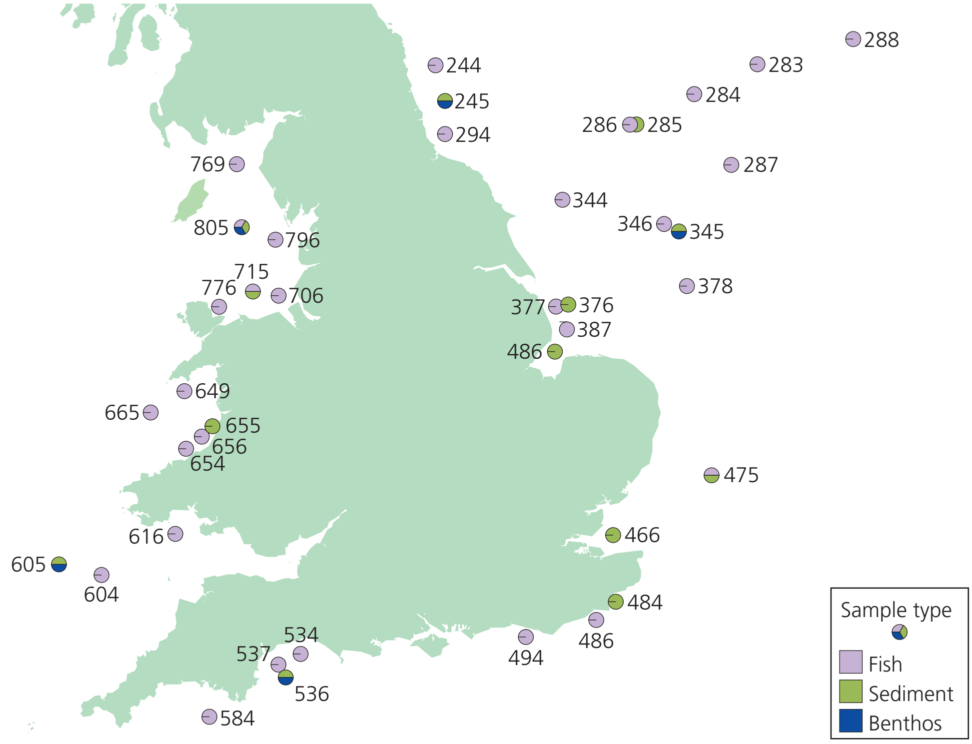
3.6.5.3.2 Litter density

Litter was collected from 31, 26, 28, 24, 24 and 26 locations in 2003, 2004, 2005, 2006, 2007 and 2008, respectively. Figure 3.110 shows the position of the sampling sites, referenced by number, with site names and positions given in Table 3.58.

The density of litter ranged from 0 – 16.81 items/ha (Figure 3.111 and Table 3.58). Litter densities of <1 items/ha were found at the majority of stations, i.e. 122 out of the 152 stations sampled over the six-year period. However, significantly higher densities occurred (maximum values) in Carmarthen Bay, 19 (16.81 items/ha), Rye Bay, 13 (11.17 items/ha), Celtic Deep, 18 (6.58 items/ha) and North Cardigan Bay, 20 (11.06 items/ha). Over the sampling period the most impacted sites for litter were Carmarthen Bay and North Cardigan Bay when for five of the six sampling years the density was in excess of 1 item/ha and at Off Morecambe when for three of the six sampling years the density was greater than 1 item/ha (see Figure 3.111 and Table 3.58).

At many sites, there was generally little year-to-year change in the density of litter, but in some instances there were large differences. For example, at Carmarthen Bay a maximum of 16.81 items/ha was recorded in 2003, with a minimum of 0.42 items/ha in 2007. The reason for this may be the prevailing weather conditions before and during the sampling period. In 2007 and 2008 the weather prior to and during the cruise was poor with frequent gales, and litter density was very low; the maximum value recorded at any site over the two years was 1.15 items/ha. In 2003 and 2004, the weather conditions were calm, prior to and during the cruise, when maximum litter density was recorded; eight sites and six sites recorded

Figure 3.110 Positions of trawling stations used for the offshore litter surveys. © Crown copyright 2010: permission granted by Cefas.



litter densities of above 1 item/ha, respectively. Similarly, weather conditions were good to fair in 2005 and 2006 when high litter density values were recorded (see Table 3.58). This coincides with beach litter observations when maximum litter density is usually recorded after strong winds and gales, indicating that seabed litter is moved and washed ashore.

3.6.5.3.3 Typological analysis

The total number of litter items collected per station ranged from 0 to 161, the latter recorded at Carmarthen Bay in 2003. In four of the six years the most frequent category of litter observed was polythene sheet and bags, ranging from 28.1% to 62.7% of the total litter items recorded. In two of the six years rope and polypropylene twine was the most frequent category observed with a range of 12.8% to 45.4% of the total litter items recorded. The third most frequent category recorded was hard plastic, ranging from 7.1% to 17.6% of the total items collected. Metal items were the fourth most frequent category collected. Table 3.59 shows the data for all categories.

3.6.5.3.4 'Fishing for Litter'

Coordinated by the UK arm of KIMO (Local Authorities International Environmental Organisation) the 'Fishing for Litter' scheme encourages fishermen to bring back to port for proper disposal litter brought up in fishing nets and gear. The litter is monitored on the quayside. The project provides bags and covers waste disposal costs. Schemes have been successfully run in Shetland, Scotland, Sweden, Netherlands and Denmark. In the UK new schemes are beginning in the south-west. In Scotland from 2005 to 2008, 15 harbours and 117 vessels took part in the scheme collecting just over 117

Table 3.58 Litter density recorded at coastal and offshore locations around England and Wales between 2003 and 2008.

Location	*Station number (see Figure 3.110)*	*Lat/Long*	*Litter density (no items / hectare / trawl site / year)*					
			2003	*2004*	*2005*	*2006*	*2007*	*2008*
Amble	244	55.16.01N 01.15.26W	0.31	0.52	0.1	0.1	0.21	0.1
NE Dogger	288	55.18.05N 02.53.82E	1.26	0.63	0	0.1	0.31	0
North Dogger	283	55.04.08N 02.05.40E	0.21	0.10	0.42	NS	0	0
West Dogger	286	54.46.76N 01.17.69E	0.73	0.00	0.42	0.31	0.31	0.31
Central Dogger	287	54.30.00N 02.42.53E	0.21	0.84	0.73	0.31	0	0.31
Tees Bay	294	54.45.25N 01.08.31W	0.42	0.73	0	0.42	0.31	0.21
Off Flamborough	344	54.14.72N 00.29.91E	0.42	0.00	0.42	0	0	0.1
Off Humber	346	54.03.92N 01.47.46E	0.52	1.36	0.42	0.42	0.21	0.21
Humber	376	53.19.37N 00.25.47E	0.42	0.21	0	0.21	NS	0
Indefatigable Bank	378	53.33.40N 02.04.92E	0.52	1.26	0	1.04	0.42	0.52
Inner Wash	387	53.08.50N 00.33.30E	NS	0.10	NS	NS	NS	NS
Outer Gabbard	475	52.01.86N 02.06.57E	0.31	NS	0.1	1.57	0.1	1.04
Rye Bay (Outer)	486	50.46.74N 00.46.83E	0.52	11.17	1.46	0.63	0.52	0.73
Off Newhaven	494	50 45.59N 00 00.00E	NS	NS	0.21	0.31	0	0.42
Inner Lyme Bay	534	50 36.86N 02 55.82W	NS	NS	0.31	0.31	NS	0.31
South Eddystone	584	50 06.44N 04 06.06W	NS	NS	0.21	1.86	0.84	0.63
West Lundy	604	51 09.79N 05 26.67W	NS	NS	0.31	NS	0.42	0.84
Celtic Deep	605	51 15.00N 06 00.00W	6.58	NS	NS	NS	NS	NS
Carmarthen Bay	616	51.32.82N 04.35.13W	16.81	16.71	1.77	11.06	0.42	1.15
North Cardigan	649	52.42.44N 04.32.29W	4.28	4.81	3.13	7.62	1.04	0.21
South Cardigan Bay	654	52.10.90N 04.29.87W	0.21	NS	0.42	3.55	0.31	NS
Cardigan Bay	655	52.27.78N 04.14.82W	0.84	NS	NS	NS	NS	NS
Inner Cardigan	656	52.18.00N 04.16.35W	0.00	0.42	1.14	0.52	0.63	0.94
Outer Cardigan	665	52.23.76N 04.53.72W	0.63	0.63	0	NS	NS	0
Burbo Bight	705	53.28.24N 03.20.47W	1.25	0.42	0.42	0.21	0.21	0.31
Liverpool Bay	715	53.28.32N 03.41.91W	0.63	0.10	0.1	0.31	0.21	0.21
St Bees	769	54.30.71N 03.47.63W	0.84	0.42	0.1	0.1	0	0.31
Red Wharf Bay	776	52.33.46N 04.12.84W	0.00	0.84	0.52	0	0.21	1.04
Off Morecambe	795	53.55.31N 03.23.23W	1.88	0.10	1.35	0.73	0.21	1.15
SE Isle of Man	805	54.03.36N 03.52.47W	1.04	0.00	0.1	0.31	0.21	0.1
Outer Dundrum Bay	816	54.04.81N 05.37.29W	1.04	1.26	0.31	0.63	NS	NS
			8	**6**	**5**	**6**	**1**	**4**

Density of litter greater than 1 item / hectare NS = Not Sampled

Figure 3.111 Litter density (items/hectare) recorded at coastal and offshore locations around England and Wales between 2003 and 2008. For station positions see Table 3.58 and Figure 3.110. © Crown copyright 2010: permission granted by Cefas.

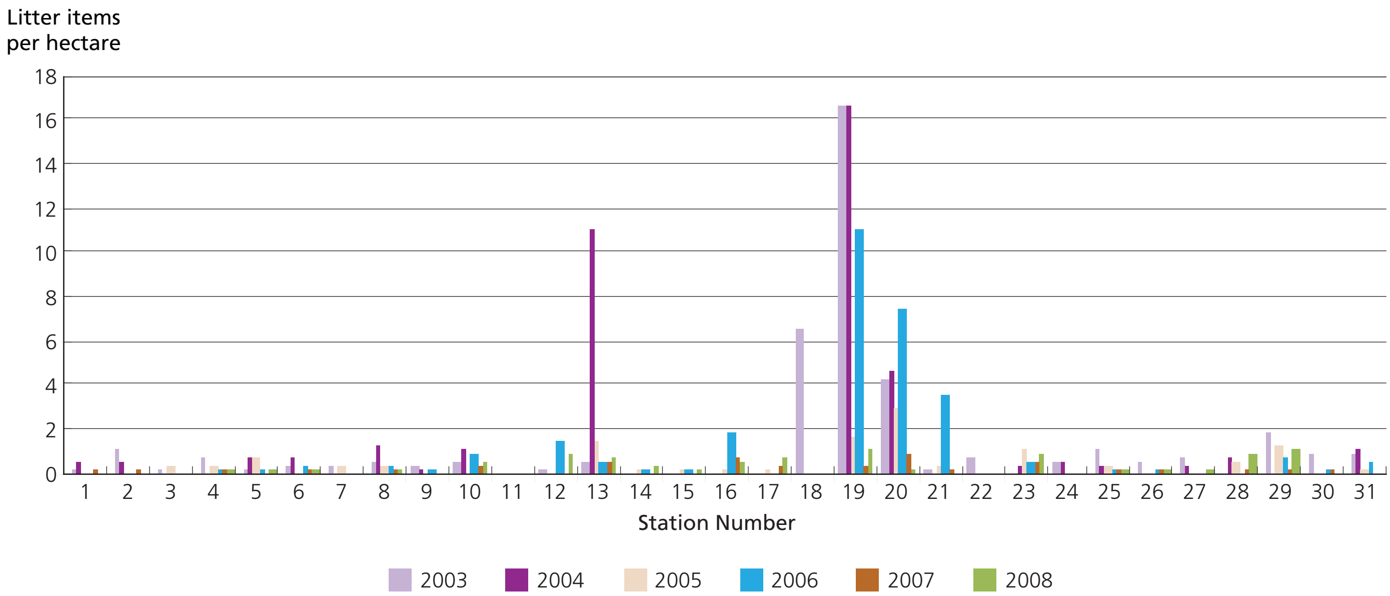


Table 3.59 Quantities of offshore litter by category, collected around the UK between 2003 and 2008.

Year	*No. sites trawled*	*Poly-thene sheet/ bag*	*Plastic*	*Rope/ twine*	*Metal/ foil*	*Glass*	*Sanitary*	*Rubber*	*Cloth*	*Wood*	*Card/ Paper*	*Poly-styrene*	*Other e.g. shoe*
No of items													
2003	31	254	37	52	34	0	1	13	11	2	3	1	2
2004	26	211	39	64	54	12	9	8	0	0	0	5	4
2005	28	66	10	32	13	3	2	2	0	1	0	0	10
2006	24	88	30	142	12	1	0	11	3	1	7	0	18
2007	24	20	9	21	4	2	0	7	0	0	1	0	1
2008	26	46	19	20	7	0	0	8	2	2	2	0	2
Percentage													
2003	31	62.7	9.1	12.8	8.4	0.0	0.3	3.2	2.7	0.5	0.7	0.3	0.5
2004	26	52.5	9.5	15.9	13.4	3	2	2	0	0	0	0.7	1
2005	28	47.5	7.1	23	9.3	2.1	1.4	1.4	0	0.7	0	0	7.1
2006	24	28.1	9.6	45.4	3.8	0.3	0	3.5	1	0.3	2.2	0	5.7
2007	24	30.8	13.8	32.3	6.1	3.1	0	10.8	0	0	1.5	0	1.5
2008	26	42.6	17.6	18.5	6.5	0	0	7.4	1.9	1.9	1.9	0	1.9

tonnes of litter. Plastic and polystyrene were the main types of litter found (KIMO 2008).

3.6.5.3.5 Offshore litter summary

The offshore litter data, while limited, provide some insight into the distribution and type of litter found on the seabed in coastal and offshore locations around England and Wales. The method of collection used during the fish surveys provides both qualitative and quantitative information. The results indicate that litter is ubiquitous in UK waters, but at generally low levels. The significantly higher densities of litter found at Carmarthen Bay, North Cardigan Bay, Celtic Deep and Rye Bay, would suggest that these are areas of accumulation, i.e. litter sinks.

The data also suggest that weather conditions may play an important role in the accumulation of litter, particularly at the litter sinks, where the sea is shallow and rough weather can scour out the accumulated litter. In this respect it is difficult to identify trends.

However, work at Hydraulics Research (Wallingford, Oxford) on coastal management, suggests that the coastline of England and Wales can be divided into 11 major sediment cells and 47 sub-cells. These cells are defined as lengths of coastline that are relatively self-contained as far as the movement of sand and shingle is concerned. In 2002, The Marine Pollution Monitoring Management Group suggested that litter may tend to circulate within these cells. The sites at Carmarthen Bay, North Cardigan Bay and Rye Bay are all located within sediment sub-cells, which would seem to support this hypothesis.

In addition, the bulk of the litter items found were plastic, polythene and polypropylene which only degrade slowly and therefore a decrease in seabed litter may only become evident after many years. The results of the 'Fishing for Litter' initiative support the findings that plastics form the majority of litter items in the marine environment.

It is important to continue to monitor seabed litter and sampling should continue and be increased both in terms of sampling frequency and geographic coverage. This could be achieved by taking advantage of the fishing surveys conducted around the UK waters by scientific institutes and in developing agreed protocols for collecting and recording litter data.

Table 3.60 presents our expert opinion on the status and trends for offshore litter. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. Trend arrows are based on the evidence available, showing whether the state is improving (⬆) or deteriorating (⬇), or where there is no overall trend discernable (⬌). The confidence rating of each assessment is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them. 0 indicates no information so no confidence.

3.6.6 Progress towards the vision of clean and safe seas

The evidence shows that marine debris, and plastic debris in particular, continues to accumulate on UK beaches and in our seas, and that this is having a negative impact on achieving the vision of clean and safe seas. While litter levels on beaches remain high, the situation is less certain in offshore areas. The limited evidence available indicates that litter is present on the seabed, although at a generally low abundance, but that hot spots do exist.

Table 3.60 Regional assessment summary for offshore litter.

CP2 Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
1	Shore and fishing litter	Limited information, but low levels	?		**I**	?
2	Shore and fishing litter	Limited information, but low levels	?		**I**	?
3	Shore and fishing litter	Limited information, but low levels	?		**I**	?
4	Shore and fishing litter	Limited information, but some high levels	?		**I**	?
5	Shore and fishing litter	Limited information, but some high levels	?		**I**	?
6	Shore and fishing litter	Limited information, but low levels	?		**I**	?
7	No information				**0**	
8	No information				**0**	

3.6.6.1 Climate change

Increased rainfall over the past few summers has already adversely affected water quality around the UK coasts and storms regularly drive more litter onto beaches. If, as anticipated, climate change brings increased storm activity this, together with rising sea levels could bring increased debris onto beaches.

3.6.7 Forward look and need for further work

In order to prevent items from becoming litter, it is important to tackle the sources of litter.

Reactive measures, such as beach cleaning, are useful in the short-term but do not provide long-term solutions to the problem and are only economically viable on amenity beaches where tourist revenue is important. It is also important to realize that marine litter is not simply an aesthetic problem but has environmental, ecological and economic impacts.

While local authorities, water authorities, industry and the Government must all play their part to reduce and clean up litter, every member of the public must also accept individual responsibility to minimise their impact on the marine and coastal environment and to support national legislative measures and educational initiatives to reduce litter at source. Only then will coastal communities, wildlife and other beach users realise the full benefits of a clean, litter-free coast.

The problems of marine litter are complex not least because litter comes from so many sources and knows no boundaries. Although data on beach litter levels is generally good, monitoring needs to be continued and there needs to be increased monitoring of seabed litter both in terms of frequency and geographical coverage. More importantly, at present there is no consensus as to what assessment criteria should be used on land or at sea, and no agreement as to what constitutes harm to the marine environment, and this needs to be resolved as soon as possible.

The problems that litter can cause to wildlife have been well documented around the world. However the extent and significance of impacts on the ecosystem and on individual species in the UK have not yet been fully determined. There is a clear need for further monitoring

and research to assess the level of harm caused by litter in the marine environment. A good starting point could be to bring together all the data held in marine mammal stranding groups, beached bird surveys, seal sanctuaries and other similar organisations and combine this with the necropsy reports of the Zoological Society of London to provide base line statistics on the impacts of litter to wildlife.

Short-term gains to tackle litter in general could be achieved by focussing public education campaigns on public litter, sewage-related debris, further environmental education of all seafarers both commercial and recreational, and integration into the National Curriculum. Longer term goals could include (i) better implementation of existing legislation, together with better coordination of legislation between the EU countries, (ii) more coordination between the bodies responsible for litter in the UK, such as local authorities, the Environment Agency and the Maritime and Coastguard Agency, (iii) possible designation of all UK waters as a Special Area under MARPOL Annex V and (v) targeted campaigns to decrease the amount of fishing and shipping litter entering the environment.

Appendix 1: Regional yearly summary of beach litter data

Year / Region	*Region 1*	*Region 2*	*Region 3*	*Region 4*	*Region 5*	*Region 6*	*Region 7*	*Total*	*UK average*
2003 Items/km	**1342.96**	**1766.30**	**3429.75**	**4145.80**	**2583.48**	**442.20**	**0**		**2263.25**
No. beaches	41	32	44	46	40	3	0	206	
km surveyed	31.44	25.78	20.32	10.71	21.79	5.00	0	115.04	
2004 Items/km	**1203.55**	**1155.94**	**2709.49**	**5744.78**	**2520.65**	**3130.00**	**0**		**2041.42**
No. beaches	51	34	53	49	42	3	0	232	
km surveyed	41.77	30.14	24.58	9.48	20.07	0.50	0	126.54	
2005 Items/km	**1222.17**	**1288.13**	**2443.14**	**4198.89**	**2537.73**	**2306.21**	**0**		**2100.96**
No. beaches	66	43	67	64	43	5	0	288	
km surveyed	44.82	33.70	26.20	23.36	19.47	1.45	0	149.00	
2006 Items/km	**1470.04**	**1783.97**	**1343.62**	**4557.19**	**2706.41**	**2272.09**	**2677.14**		**2045.31**
No. beaches	68	51	75	63	58	2	1	317	
km surveyed	42.67	41.29	41.91	18.91	25.46	0.86	0.35	171.11	
2007 Items/km	**1637.32**	**1940.62**	**1439.62**	**4442.89**	**2479.31**	**1731.15**	**0**		**2166.97**
No. beaches	61	55	87	59	47	4	0	313	
km surveyed	25.83	43.00	36.10	18.95	22.67	1.22	0	147.76	

Appendix 2: Litter items per kilometre surveyed by source and region

BW03 Region	*Region 1*	*Region 2*	*Region 3*	*Region 4*	*Region 5*	*Region 6*	*UK average*
Public litter	637.41	859.49	1494.36	1585.74	1043.69	145.40	982.42
Fishing litter	165.88	232.88	538.26	1018.86	217.94	91.20	332.72
Sewage-related debris	65.18	47.83	174.12	102.77	628.32	11.00	188.35
Shipping litter	29.68	52.99	65.06	79.53	55.81	54.80	51.84
Fly tipping	16.79	10.63	10.68	29.50	39.28	2.60	19.16
Medical litter	1.21	2.29	3.30	4.39	2.98	0.40	2.42
Non-sourced litter	426.81	560.19	1143.95	1325.03	595.46	136.80	686.35

BW04 Region	Region 1	Region 2	Region 3	Region 4	Region 5	Region 6	UK average
Public litter	535.27	536.33	1131.61	2049.17	953.21	928.00	832.60
Fishing litter	143.53	125.61	501.53	903.36	218.75	1312.00	282.27
Sewage-related debris	67.20	117.09	38.93	189.39	768.90	92.00	194.13
Shipping litter	33.09	32.28	60.74	106.77	65.32	248.00	49.75
Fly tipping	15.87	8.36	10.78	32.18	36.77	4.00	17.58
Medical litter	1.92	2.02	2.44	4.75	2.54	8.00	2.38
Non-sourced litter	406.67	334.24	963.47	2459.17	475.16	538.00	662.70

BW05 Region	Region 1	Region 2	Region 3	Region 4	Region 5	Region 6	UK average
Public litter	503.94	502.36	948.44	1284.67	855.82	592.41	751.00
Fishing litter	124.71	138.62	370.10	844.61	201.96	522.07	297.84
Sewage-related debris	85.10	33.62	47.13	134.67	762.49	53.79	162.76
Shipping litter	27.69	32.32	60.83	72.82	54.70	208.97	46.93
Fly tipping	17.11	8.04	11.18	21.40	23.78	57.93	15.96
Medical litter	2.01	4.51	4.31	7.96	4.52	6.90	4.29
Non-sourced litter	461.61	568.66	1001.14	1832.75	634.44	864.14	822.18

BW06 Region	Region 1	Region 2	Region 3	Region 4	Region 5	Region 6	UK average
Public litter	571.79	686.42	541.16	1289.44	707.00	666.28	692.45
Fishing litter	131.91	130.78	159.93	767.78	155.51	641.86	216.21
SRD	94.11	84.74	31.35	241.84	980.33	131.40	224.44
S	38.60	32.82	27.70	64.94	47.32	236.05	40.00
FT	16.92	16.59	4.72	61.61	20.85	3.49	19.31
Medical litter	1.78	2.52	2.08	7.77	5.18	4.65	3.21
Non-sourced litter	614.93	830.11	576.68	2123.80	790.21	588.37	849.69

BW07 Region	Region 1	Region 2	Region 3	Region 4	Region 5	Region 6	UK average
Public litter	657.49	823.44	525.39	1323.87	726.95	382.79	767.34
Fishing litter	151.67	149.74	233.52	892.11	269.12	455.74	286.57
Sewage-related debris	102.00	58.17	27.15	178.09	507.19	1.64	142.06
Shipping litter	28.96	34.42	30.78	69.36	43.32	146.72	39.35
Fly tipping	17.46	31.42	8.26	35.73	16.54	22.13	21.51
Medical litter	3.21	3.40	2.83	8.29	4.15	3.28	3.97
Non-sourced litter	676.54	840.03	611.69	1935.45	912.04	718.85	906.17

Appendix 3: Legislation and regulation

A.1 Main legislation affecting ocean-based sources

The main piece of legislation controlling the prevention of pollution and litter into the marine environment from ocean-based sources is the International Convention for the Prevention of Pollution from Ships 1973, modified by the protocol of 1978, generally known as MARPOL 73/78.

MARPOL 73/78 has six annexes covering different types of pollution. Annex V sets minimum specific distances from land within which certain wastes, such as glass, food and metals can and cannot be disposed of. Under Annex V, as with other MARPOL annexes, certain sea areas are designated as special areas. The disposal of waste in special areas is much more strictly controlled than outside a special area. Each special area may have different requirements. The disposal of all plastics is prohibited throughout the world's oceans. The UK ratified Annex V in December 1988. Annex IV contains requirements to control pollution of the sea by sewage.

The UK's interpretation of MARPOL Annex V and IV is via The Merchant Shipping (Prevention of Pollution by Sewage and Garbage) Regulations 2008 and the Merchant Shipping (Port Waste Reception Facilities) (Amendment) Regulations 2009.

A.1.1 The Merchant Shipping (Prevention of Pollution by Sewage and Garbage) Regulations 2008

These regulations replaced the Merchant Shipping (Prevention of Pollution by Garbage) Regulations 1998 and introduce minor amendments made to MARPOL Annex V, as well as implementing MARPOL Annex IV - Sewage.

The sewage elements of the regulations apply to all UK ships engaged in international voyages and all foreign ships in UK controlled waters over 400 gross tonnage or above and ships certified to carry 15 or more passengers. Ships which fall into this category are required by law to have installed either a type approved sewage treatment plant, comminuting and disinfecting system or a holding tank. Discharge from these systems is also controlled and varies according to the certified system used.

The North Sea is designated a Special Areas under MARPOL, and dumping of any waste is prohibited within 12 nautical miles of land. Beyond this limit, it is illegal to dispose of any waste other than food wastes.

The main provisions of the regulations are:

- The garbage elements of the regulations apply to all UK ships and all foreign ships in UK controlled waters
- Ships engaged in international voyages over 400 gross tonnage or above and ships certified to carry 15 or more passengers must carry and complete a garbage record book
- Ships over 400 gross tonnage or above and ships certified to carry 15 or more passengers must carry a garbage management plan

- All ships over 12 m in length must display placards to inform the crew and passengers of the relevant prohibitions on the disposal of garbage.

A.1.2 The Merchant Shipping (Port Waste Reception Facilities) (Amendment) Regulations 2009

These Regulations replace The Merchant Shipping (Reception Facilities for Garbage) Regulations 1997 and the Merchant Shipping (Port Waste Reception Facilities Regulations 2003.

The 1997 regulations required all ports and terminals in the UK to:

- Provide adequate reception facilities for ship-generated waste
- Prepare a waste management plan to be approved by the Secretary of State.

The 2003 Port Waste Reception Facilities Regulations brought into force three significant changes:

- All ships must provide notification before entering into the port or terminal of the waste they will discharge, including information on types and quantities
- All ships must deliver their waste to the port reception facilities before leaving the port or terminal, unless they have sufficient dedicated storage capacity to store the waste until the next port of call
- All ships must pay a mandatory charge to make a significant contribution to the cost of the port reception facilities for ship generated waste, whether they use them or not.

Additionally:

Recreational craft authorised or designed to carry no more than 12 passengers and fishing vessels are exempt from the requirements to notify posts and from the mandatory charge, but must deliver their wastes to the port making their own arrangements and pay on a commercial basis.

Member States must ensure proper monitoring of compliance with the Directive, by means of spot checks and the exchange of information between ports. Ships that do not deliver waste in one port should be reported to the next port of call for a more detailed inspection.

Ports are required to cover the costs of the post waste reception facilities including costs of treatment and disposal. How the port recovers its costs was left open so systems vary from port to port with some charging on a ship by ship basis and others incorporating the cost into harbour/port dues. However the charge is calculated, the port must make clear the amount of the charge, and the way in which it has been calculated.

The 2009 amendment brought into force new sections relating to Annex IV of MARPOL on the treatment, storage and disposal of sewage. It includes sewage in the definition of ship-generated wastes. As sewage can legally be discharged, Ports are only required to have sufficient sewage disposal facilities for ships, if there is a significant demand for the use of such a facility.

A.2 Main legislation affecting land-based sources

A.2.1 Environmental Protection Act (1990)

Under the Environmental Protection Act (EPA) 1990, it is an offence to drop litter in any public place, including beaches. Local authorities and authorised persons are able to prosecute those found littering, or to issue a fixed penalty notice if appropriate. The EPA also places duties on, and gives powers to, local authorities to keep their beaches clear of litter according to the Code of Practice. In 2000, a revised Code of Practice extended the requirements from amenity beaches only to all beaches. Individual local authorities must decide the level of cleanliness that they are able to give to any non-amenity beaches in their area. Duty bodies are advised that they may find it helpful to encourage voluntary groups to assist in cleaning up beaches. The Code of Practice was updated again in 2006. The code now carries a description of aquatic litter, and guidance that suggests that between May and September beaches should be subject to a frequent monitoring routine, and cleansed to as practicable a standard as possible. The code also recommends that duty bodies carry out regular monitoring and appropriate cleansing of beaches that are used outside of the traditional bathing season (Defra, 2006).

A.2.2 EU Water Framework Directive (200/60/EC)

Under Annex VI Part A of the EU Water Framework Directive, measures are required to be introduced by 2015 to ensure compliance with the revised Bathing Water Directive (2006/7/EC). The revised Bathing Water Directive itself was transposed into UK domestic legislation under Water Resources Statutory Instrument No. 1097 The Bathing Water Regulations (England & Wales) 2008, Scottish Statutory Instruments No. 170 The Bathing Waters (Scotland) Regulations 2008 and the Statutory Rules of Northern Ireland No. 231 The Quality of Bathing Water Regulations (Northern Ireland) 2008.

There is a clear statutory obligation on competent authorities to introduce measures by 2015 to control and manage waste at designated bathing sites which poses a risk to water quality and bathers' health. This will presumably require the development of appropriate waste and debris standards.

3.7 Algal Toxins

3.7.1 Key points

i. Introduction

Algal toxins (also known as biotoxins) are natural chemicals produced by certain species of marine algae. They are of concern as they can be accumulated by shellfish (e.g., mussels, cockles, oysters) when they feed, and so can affect human consumers. Concerns have been expressed that nutrient enrichment can increase the occurrence of harmful algal blooms in which the biotoxins are produced. However, a recent study concluded that the abundance of harmful algal bloom species in UK and Irish coastal waters is not related to nutrient enrichment from human sources.

ii. How has the assessment been undertaken?

Comprehensive biotoxin monitoring programmes compliant with EU legislation are undertaken in all regions of the United Kingdom, and the data derived from these programmes has been used in this assessment.

iii. Current status of algal toxins and past trends

- Comprehensive biotoxin monitoring programmes compliant with EU legislation are undertaken in all regions of the United Kingdom
- The testing methods employed by all official monitoring laboratories on behalf of the competent authority follow United Kingdom National Reference Laboratory (UKNRL) protocols (where these exist) and are accredited to ISO17025 standards where possible
- Marine biotoxins and toxic algae were found to be present in samples from monitored areas during the three years of data examined
- Within the dataset, the occurrence of marine biotoxins and toxic algae did not increase either temporally or spatially
- The number of shellfish samples in which toxins were detected was small, generally fewer than 5% and less than 1% in some years
- Based on these data, the current monitoring regime within the UK (which monitors the three classes of toxins responsible for amnesic shellfish poisoning, ASP; diarrhetic shellfish poisoning, DSP; and paralytic shellfish poisoning, PSP) provides sufficient human health protection for shellfish consumers in respect of ASP, DSP and PSP toxins.

Algal toxins were not assessed in the first UK assessment, reported in *Charting Progress* and so no trend can be determined.

iv. What has driven change?

Within the dataset examined for this report, the occurrence of marine biotoxins and toxic algae did not increase either temporally or spatially.

v. What are the uncertainties?

The testing methods employed by all official monitoring laboratories on behalf of the competent authority follow UKNRL protocols (where these exist) and are accredited to ISO17025 standards where possible. There is high confidence in the shellfish toxin data. Phytoplankton monitoring is undertaken on a more limited scale.

vi. Forward look

Based on the data assessed in this section, the current monitoring regime within the UK provides sufficient human health protection in relation to algal toxins in seafood.

Other algae occur in UK waters which generate additional toxins; notably the azaspiracids and spirolides and the toxins responsible for Neurotoxic Shellfish Poisoning. Current monitoring arrangements cannot detect these, so further research is needed to assess the risk they pose and to determine whether they should be monitored in the future.

3.7.2 Introduction

At certain times of the year naturally occurring marine algae can occur which may produce potent biotoxins. These biotoxins can accumulate in filter-feeding bivalve molluscs, and in certain circumstances in other shellfish such as grazing gastropods.

Human consumption of shellfish contaminated with biotoxins may pose a serious health risk. In the EU (including the UK), there are currently three major biotoxin groups which may be detected in shellfish and which are subject to statutory testing to protect human health. These are the Paralytic Shellfish Poisoning (PSP) toxins, the lipophilic toxins, responsible for Diarrhetic Shellfish Poisoning (DSP) and the Amnesic Shellfish Poisoning (ASP) toxins.

PSP toxins, produced by the dinoflagellate genus *Alexandrium* in UK waters, are absorbed quickly from the gastrointestinal tract, with high concentrations leading to symptoms occurring within a few minutes. These symptoms include a tingling/burning sensation in the lips, tongue and face with gradual progression to extremities (paraethesis) progressing to numbness. Symptoms usually develop within 30 minutes. Paralysis of the upper and lower limbs may follow, ataxia, constrictive sensation in the throat and incoherent speech. In high doses paralysis can extend to the respiratory system leading to respiratory arrest followed by death. In a few cases gastrointestinal disturbances occur with nausea, vomiting and abdominal pain (van Egmond, 1993).

DSP symptoms include diarrhoea (92% of cases), nausea (80%), vomiting (79%), abdominal pain (53%) and chills (10%). Initial onset of these symptoms is within 30 minutes to several hours, but rarely exceeding 12 hours. Symptoms can persist for a few days (van Egmond, 1993). The occurrence of these lipophilic toxins in shellfish in the UK has been associated with the genus *Dinophysis* and *Prorocentrum lima*.

ASP caused by Domoic acid is produced by the diatom genus *Pseudo-nitzschia*. Domoic acid has only been recorded in one outbreak in Canada in 1987, in which 250 individuals became ill. Symptoms observed were gastrointestinal distress (after 24 hours) and central nervous system disorders (48 to 72 hours), of those affected 4 died (Perl et al., 1990).

Owing to the health risk to consumers posed by contaminated shellfish, legal controls are placed on the production and marketing of fishery products worldwide. In the EU and until 01 January 2006, controls were prescribed in Council Directive 91/492/EEC, as amended and enabled in the UK by the Food Safety (Fishery Products and Live Shellfish) (Hygiene) Regulations 1998. These were replaced in January 2006 by Regulation (EC) 854/2004. This Regulation prescribes the requirements for the Central Competent Authority in relation to biotoxin monitoring in live bivalve molluscs. Regulation (EC) 853/2004 lays down specific hygiene rules for the hygiene of foodstuffs and prescribes the statutory maximum levels of biotoxins permitted in live bivalve molluscs. The Regulations are further supported by Regulation (EC) 2074/2005 which lays down the implementing measures for certain products, including live bivalve molluscs and for the organisation of Official Controls (OC). Regulation (EC) 882/2004 prescribes the requirements for analytical methods used for analysis of OC samples and the validation status of these methods. The above package of EU Regulations is directly applicable across all member states and is intended to ensure a uniform approach to feed and food law across Europe.

The legal requirements essentially require EU Member States to have in place an OC monitoring system. This system checks for the presence of marine biotoxins (PSP, DSP and ASP) in shellfish production and relaying areas, and in products placed on the market. It also checks for the possible presence of toxin producing phytoplankton in all active production and relaying areas. Under the above EU legislation, the prescribed competent authority is required to take action to close the production or relaying area and prevent harvesting or sale of products found to contain levels of biotoxins above the limits prescribed in the legislation.

3.7.3 The UK biotoxin monitoring programmes

3.7.3.1 England and Wales

In England and Wales, the central competent authority is the Food Standards Agency (FSA) which delegates stated OC functions through local Food Authorities, such as local enforcement and sampling activities. The Centre for Environment, Fisheries and Aquaculture Science (Cefas) has been contracted by the FSA since 2001 to coordinate the delivery of the biotoxin programme, undertake all shellfish and potentially toxic phytoplankton laboratory analysis and provide scientific advisory duties to the FSA for the OC programme for marine biotoxins. Monitoring for algal biotoxins is divided into two elements: the flesh monitoring element where samples of shellfish from designated shellfish harvesting and relaying areas are tested; and the water monitoring element where water samples are collected from sites within harvesting and relaying areas. Flesh samples are usually collected and submitted to the testing laboratory on a monthly basis, except in areas with a history of shellfish biotoxins or toxic phytoplankton, where samples may be collected on a more frequent basis (weekly or fortnightly) dependant on the level of risk. Water samples are collected monthly from October to March and fortnightly from April to September. The more frequent sampling schedule is imposed as a higher incidence of potentially toxic algae is expected during these months. At present, 68 English and Welsh classified shellfish production areas are being monitored.

3.7.3.2 Scotland

In Scotland the competent authority is the Food Standards Agency Scotland (FSAS), which delegates stated OC functions through local Food Authorities. Cefas is contracted by the FSAS to undertake laboratory analysis for PSP, DSP and ASP toxins, co-ordinate and provide scientific advisory duties for the OC programme for marine biotoxins in Scotland. Under the scope of its contract with FSAS, Cefas has been responsible for the testing of shellfish samples for PSP toxins from April 2005, DSP toxins from September 2005 and ASP toxins from April 2008. Prior to this, PSP, DSP and ASP testing was conducted by Fisheries Research Services (FRS), Aberdeen and ASP testing by Integrin Advanced

Biosystems Marine Resource Centre, Oban. The phytoplankton monitoring programme is separate from the flesh monitoring programme and delivered by the Scottish Association for Marine Science (SAMS), Oban. The scope of the Scottish biotoxin monitoring programme changed in 2006, as a result of an inshore biotoxin risk assessment undertaken by the FSAS in 2004. In the revised inshore programme, shellfish production areas are sampled through representative monitoring points (RMPs) and associated harvesting areas (AHAs). An assessment of suitable RMPs and AHAs was undertaken and the programme rolled out to all local authorities throughout 2006 and early 2007. This resulted in a reshaping of the programme, with revised sampling points and new sampling frequencies for PSP, DSP and ASP monitoring. In the revised programme, RMPs are currently sampled weekly for PSP and at variable frequency for ASP and DSP, depending on seasonal risk. At present there are more than 80 RMPs and 170 AHAs covering 174 classified production areas. Water samples for phytoplankton monitoring are collected from between 35 to 48 coastal locations on a weekly basis between April and September, with fortnightly sampling conducted in October and monthly sampling from November to February.

3.7.3.3 Northern Ireland

In Northern Ireland, the competent authority is the Food Standards Agency Northern Ireland (FSANI) which delegates OC functions through District Councils. A regular monitoring programme was established in 1993. The Agri-Food and Biosciences Institute (AFBI) Northern Ireland is contracted by FSANI to undertake analysis for biotoxins and potentially toxic phytoplankton as well as to provide scientific advice on these topics. Phytoplankton sampling is carried out on a fortnightly basis from all classified shellfish harvesting areas. Flesh samples for biotoxin analysis were taken on a monthly basis from each site up until the end of 2007. Revisions to the programme were then made on the basis of a risk assessment undertaken by AFBI and FSANI. As a result shellfish production areas within designated water bodies are divided into two groups with one group sampled once a fortnight and the remaining group sampled the following fortnight. At present there are 43 classified shellfish beds which are monitored.

3.7.3.4. Testing methods and reporting

The methods currently employed for the biotoxin monitoring programme are as follows:

- PSP: the AOAC 959.08 official method (AOAC, 1990) mouse bioassay (MBA) in combination with qualitative screening and quantitative by HPLC (certain shellfish species only) based on the AOAC 2005.06 official method for PSP (AOAC, 2005)
- DSP: an MBA based on a modification of the Yasumoto et al. (1984) method
- ASP: an HPLC method (Quilliam et al., 1995). In Northern Ireland an immunobiosensor is employed as a screen for low risk species – oyster, mussel, cockle and clam (Traynor et al., 2006)
- Phytoplankton analysis using a 5 to 50 ml settlement (Utermöhl) chamber and inverted microscope.

Laboratory methodology follows UKNRL protocols, where these exist, and are accredited to ISO17025 standards where possible. The action limits for shellfish biotoxin levels used in the national programmes are those defined by the EU regulations: 80 µg saxitoxin (STX) equivalents/100 g shellfish flesh for PSP, 20 µg domoic acid/g shellfish flesh for ASP and

presence of lipophilic toxins (DSP) as shown by MBA. The action/threshold levels set for potentially toxic algae are:

- *Alexandrium* spp. presence (this equates to 20 cells/l in Scotland and Northern Ireland and 40 cells/l in England and Wales)
- *Dinophysis* spp. 100 cells/l
- *Prorocentrum lima* presence in Scotland (this equates to 20 cells/l) and 100 cells/l in England, Wales and Northern Ireland
- *Pseudo-nitzschia* spp. 50 000 cells/l (Scotland) or 150 000 cells/l (England, Wales and Northern Ireland).

Results are reported to the relevant FSA offices usually within 24 to 30 hours of sample receipt and communicated to stakeholders by the FSA. The FSA is immediately alerted when PSP and ASP toxins are detected, when signs indicative of DSP are observed, or when potentially toxic algae are recorded above the action limit.

3.7.3.5 Closures

In the UK, where the action limit for marine biotoxins in flesh samples is exceeded or a positive result occurs in the flesh monitoring programme, the affected sites are closed and continue to be tested until two consecutive negative or below maximum permitted level results are obtained. Once obtained any harvesting restrictions at the sites are lifted. Samples for closed sites are usually collected at weekly intervals. An earlier re-testing scheme is currently being assessed in Scotland.

Within the phytoplankton monitoring programme when cell concentrations increase above action levels (set by the competent authority), shellfish samples and another water sample are requested for testing the following week. Shellfish samples are tested for the associated shellfish biotoxin only. No closures would result solely from a water sample exceeding the action limit for potentially toxic algae.

By implementing harvest restrictions and monitoring production areas and RMPs, the UK is complying with EU legislation while enhancing knowledge of shellfish biotoxin occurrence and profiles. This monitoring supports the shellfish industries own end product testing regimes and ensures that the public are aware of any shellfish safety concerns.

3.7.4 Developments since *Charting Progress*

Algal toxins were not assessed for *Charting Progress* and as a consequence it is not possible to report developments since the last assessment.

3.7.5 Evidence of monitoring effectiveness

3.7.5.1 England

PSP intoxication in England is rare, four incidents have been reported since 1828 (Ayres, 1975). The most recent English occurrence was in 1968 when 78 people suffered illness after consuming mussels from sites in the north-east of England (Ayres and Cullum, 1978). Since this outbreak monitoring has been ongoing in English waters. Initially restricted to the north-east coast from April to September for PSP, the biotoxin monitoring programme has gradually been increased to its current level which monitors all three shellfish biotoxins and potentially toxic phytoplankton in all commercial shellfish production areas. Closures due to PSP and DSP detection by the programme occur most years

but are not common and found in only a small number of areas. There have been no closures due to ASP.

During the monitoring period there has been no recorded incident of illness from shellfish poisons from commercially harvested stock.

3.7.5.2 Wales

One incident of PSP occurred in Barry, Wales in 1909 when 19 people became ill with PSP-like symptoms (Ayres, 1975). Sampling of two sites within Wales was initiated in 1993 for PSP and DSP from March until September. This programme as with the English monitoring programme was gradually increased to cover all three shellfish biotoxins and potentially toxic phytoplankton in all commercial Welsh shellfish production areas. Although PSP and DSP have been detected in shellfish during monitoring of this region, at present there has been no recorded shellfish poisoning outbreak attributed to Welsh shellfish during the monitoring period.

3.7.5.3 Scotland

Within Scotland, PSP outbreaks were recorded in 1827 and 1958 (Ayres, 1975). A small-scale monitoring programme for PSP was in operation from 1968. In 1991, after a large-scale occurrence of PSP, the programme was revised and the number of sampling stations increased to 65, with testing for both PSP and DSP. In 1998, ASP was included in the monitoring programme and in 1999 a large-scale closure of the king scallop industry resulted from the presence of ASP. Closures for all three shellfish poisons occur in most years in Scottish shellfish.

In recent years (post 2001), there has been one recorded incidence of biotoxin poisoning in shellfish consumers from a monitored area. DSP toxins were confirmed to be the cause of a case of intoxication of 171 individuals after they ate contaminated mussels harvested from a Scottish site in June 2006. Although the site involved was routinely monitored for the presence of toxins in the flesh of shellfish and harmful phytoplankton in the water, on this occasion the sampling programme missed a rapid increase in toxicity which occurred in the area over the space of a single week.

When sampling resumed, both the water and flesh data confirmed the occurrence of a lipophilic toxin episode due to *Dinophysis* spp. It is worth noting that this DSP outbreak occurred prior to the implementation of the new Scottish monitoring regime, which is based on risk assessment as required by EU Regulations.

These results emphasise the need for weekly sampling of classified shellfish production areas to monitor for the presence of toxins and harmful phytoplankton during high risk periods. More importantly, it underlines the importance for the Industry to take account of biotoxin risks when developing harvesting plans and to ensure suitable control of marketed products through end-product testing (EPT), in order to protect public health.

3.7.5.4 Northern Ireland

Historic records on shellfish toxicity in Northern Ireland are few. The first well documented case was in 1990 when concentrations of PSP toxins peaked at 5496 mouse units in mussels from Belfast Lough and resulted in a ban on commercial harvesting as well as the issuing of public health warnings. PSP toxicity in mussels was not detected again until 1996 when a large bloom of *Alexandrium tamarense* within the Lough was recorded.

Dinophysis spp. are regularly recorded in Northern Ireland coastal waters but rarely reach densities greater than a few hundred cells per litre. The exception was a bloom of *D. acuminata* in 1994 which resulted in MBA positive results for DSP toxins. No reports of human illness were recorded during this incident.

In Northern Ireland coastal waters, ASP toxicity has only been recorded in king scallops from a classified harvesting area in Strangford Lough. Low level toxicity in whole scallop tissue samples has routinely been recorded from this site since ASP testing began in 1999.

3.7.6 The UK Biotoxin Monitoring Programme 2005–2008

The UK biotoxin monitoring programme is split into three regions: England and Wales, Scotland, and Northern Ireland. Biotoxin findings from the Official Control Biotoxin Monitoring Programme for the three years until March 2008 are outlined in this section for each of the three areas (see also the summary in Figure 3.112). Occurrences of toxic shellfish and phytoplankton are described for each region and a map for each year is provided to illustrate the distribution of these occurrences (Figures 3.113 to 3.118).

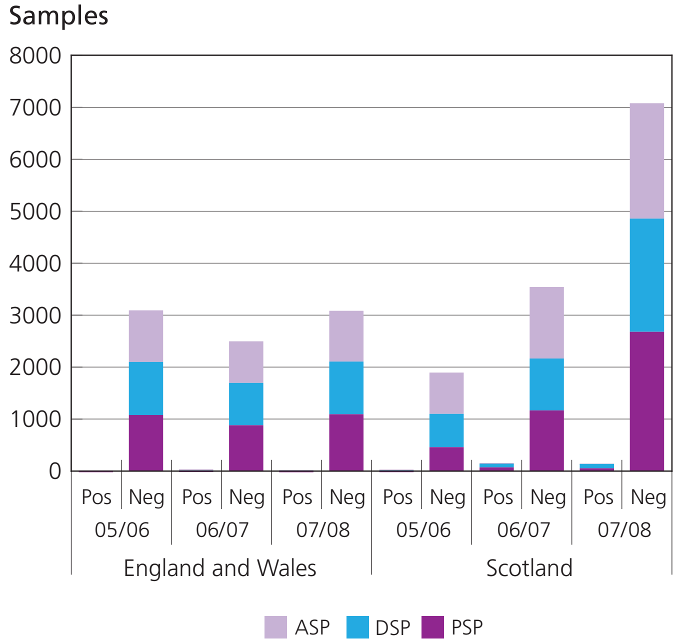
3.7.6.1 England and Wales

3.7.6.1.1 April 2005 – May 2006

Source: *PSP and DSP Biotoxin Monitoring Programme for England and Wales, 1st April 2005 to 31st May 2006, Cefas contract report C2333.*

A total of 1141 shellfish samples from 116 inshore sampling locations representing (directly or indirectly) 64 of the classified English and Welsh shellfish harvesting or relaying production areas (coverage rate of 92.7% of active classified

Figure 3.112 Numbers of samples testing positive or negative for the presence of ASP, DSP and PSP toxins in samples from England, Wales and Scotland 2005-2008. Note low frequency of samples testing positive. © Crown copyright 2010: permission granted by Cefas.



areas during the reporting period) were tested. A total of 737 water samples were also collected from 61 of the classified production areas and were submitted to Cefas for phytoplankton analysis.

PSP. 1085 flesh samples were tested for PSP toxins. Toxins were detected in four samples from two separate areas, Fowey and Holy Island. However, toxin concentrations only exceeded the action level of 80 µg STX eq. per 100 g of flesh for PSP toxins in two consecutive samples, collected from Fowey on 4 July and 11 July 2005. The affected site was subject to temporary harvesting restrictions.

DSP. 1023 flesh samples were tested for DSP/lipophilic toxins. No samples were found to be positive for DSP.

Figure 3.113 The occurrence of PSP toxin concentrations exceeding the Action Level (>80 µg STX eq. per 100 g shellfish tissue) in the UK. (a) April 2005–March 2006; (b) April 2006–March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.

Figure 3.114 The occurrence of positive samples for DSP in the UK. (a) April 2005–March 2006; (b) April 2006–March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.

ASP. 1004 flesh samples were tested for ASP toxins. Toxins were detected in nine samples from two classified production areas: Taw/Torridge and Milford Haven, between May and June 2005. None of the samples recorded concentrations higher than 20 µg [domoic+*epi*-domoic acid]/g [shellfish tissue].

***Alexandrium* spp**. This genus was detected in 86 samples from 17 production areas, in particular the Fal (present May to September, maximum concentration 8800 cells/l), Milford Haven (present April to September, maximum concentration 1200 cells/l), Poole (present July to September, maximum concentration 1240 cells/l), Salcombe (present May to August, maximum concentration 5400 cells/l) and Portland (present April to October, maximum concentration 107 000 cells/l). No correlation was established between the presence of *Alexandrium* spp. in water and the presence of detectable levels of PSP toxins in shellfish flesh.

***Dinophysis* spp**. This genus was detected in 30 samples from 11 production areas, in particular the Camel (present May to August, maximum concentration 80 cells/l), Fal (present May to July, maximum concentration 280 cells/l) and Fowey (present April to July, maximum concentration 40 cells/l). The action level was only breached in seven samples. No correlation was found between the presence of *Dinophysis* spp. and DSP toxin levels in shellfish.

Prorocentrum lima. This species occurred in three samples. Concentrations of *P. lima* were greater than the action level (set at 100 cells/l) on one occasion, which was in a sample from Portland taken in August. No correlation was established between presence of *P. lima* in water and presence of DSP toxins in shellfish flesh.

***Pseudo-nitzschia* spp**. This genus was found in 277 samples from 38 production areas. Cell concentrations were variable and in 96% of cases did not exceed 50 000 cells/l. Three samples from Burry Inlet, Dee and Portland exceeded the action level set at 150 000 cells/l during April to May; no correlation between the high phytoplankton counts and the presence of ASP toxins in the flesh of shellfish from the same areas was established.

The detection of ASP toxins at Milford Haven in May to June 2005 was associated with the presence of a known domoic acid producing *Pseudo-nitzschia* species, *P. australis*, at concentrations below 55 000 cells/l. This is the first recorded observation of this species in the coastal waters of England and Wales.

Five *Pseudo-nitzschia* species: *P. australis, P. delicatissima, P. multiseries, P. fraudulenta* and *P. pungens* speciated in English and Welsh coastal waters. However, only *P. australis* and *P. multiseries* have been directly linked to toxicity in UK shellfish.

3.7.6.1.2 June 2006 – March 2007

Source: *PSP and DSP Biotoxin Monitoring Programme for England and Wales, 1st June 2006 to 31st March 2007, Cefas contract report C2333.*

A total of 941 shellfish samples were submitted from 103 inshore sampling locations (coverage rate of 92% of the active classified areas during the reported period) for testing. A total of 879 water samples collected from 54 of the classified production areas were submitted to Cefas for phytoplankton analysis.

PSP. 892 flesh samples were tested for PSP toxins. Toxins were detected in eight mussel samples from three separate production areas:

Figure 3.115 The occurrence of ASP toxin concentrations exceeding the Action Level (>20 mg [domoic+epi-domoic acid]/g [shellfish tissue]) in the UK. (a) April 2005–March 2006; (b) April 2006–March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.

Fal, Fowey and Holy Island, during June to July. Toxin concentrations exceeded 80 µg STX eq. per 100 g shellfish tissue in two samples collected from two sampling locations in the Fal Estuary on 27 June and 2 July 2006. Affected sites were subject to temporary harvesting restrictions.

DSP. 821 shellfish samples were tested for DSP/lipophilic toxins. Five samples (including four mussel samples and one native oyster sample) from three separate sampling locations collected in September 2006 (two in the Camel and one in the Fal estuary) were found to be positive by the official control monitoring bioassay method. Affected sites were subject to temporary harvesting restrictions. Toxin profiles determined by LC-MS indicated that the Camel mussel samples contained okadaic acid, dinophysistoxins (DTXs), and azaspiracids (AZA1, 2 and on occasion 3), as well as trace levels of pectenotoxin2 in one sample and low levels of spirolides in another sample. The Fal native oyster sample contained low levels of spirolides, DTX2 and PTX2 as well as trace levels of OA/DTX esters.

ASP. 823 shellfish samples were tested for ASP toxins. Toxins were detected in 21 samples (including cockles, mussels, and both Pacific and native oysters) from seven separate classified production areas: Blackwater, Burry Inlet, Colne, Fal, Milford Haven, Three Rivers and West Mersea. None of these samples recorded results above the maximum permitted level of 20 µg [domoic+epi-domoic acid]/g [shellfish tissue]. A cockle sample collected from Three Rivers in June 2006 recorded, at 12.09 µg [domoic+*epi*-domoic acid]/g [shellfish tissue], the highest ASP toxin content detected in England and Wales since at least 2001.

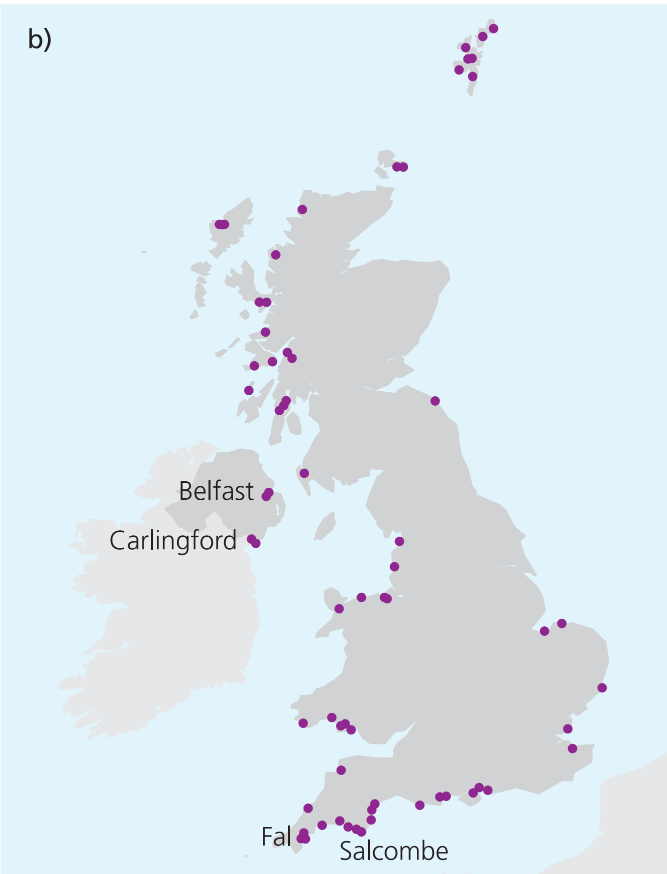
***Alexandrium* spp**. This genus was detected in 150 samples from 31 production areas from June to November 2006 and March 2007, with maximum cell concentrations reaching 17 000 000 cells/l at Salcombe (collected late June 2006). A correlation was established between the presence of *Alexandrium* spp. in water and the presence of detectable levels of PSP toxins in shellfish flesh, for the sites in the Fal Estuary that had been subject to harvesting restrictions.

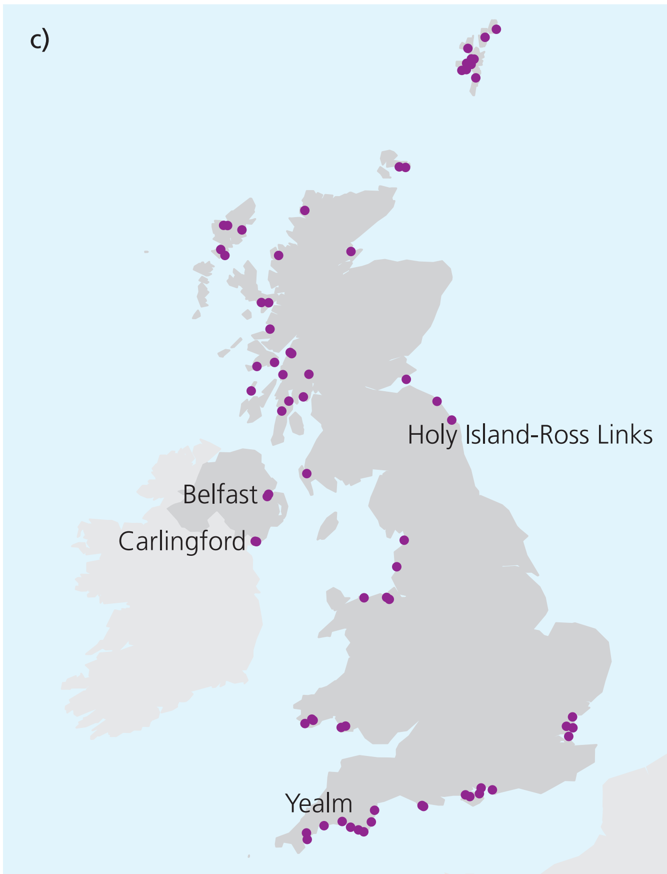
***Dinophysis* spp**. This genus was found in 83 water samples from 16 production areas (detected June to November 2006) that were mainly in the south-west of England. *Dinophysis* levels were above action limits (set at 100 cells/l) on 31 occasions. A correlation was established between the presence of *Dinophysis* spp. in water and the presence of lipophilic toxins in shellfish flesh, for sites that had been subject to harvesting restrictions in the Fal and Camel estuaries. These sites were also those at which the highest densities of *Dinophysis* spp. (max. 1040 cells/l) were recorded.

Prorocentrum lima. This species was found in 10 samples (detected July to August and October to November) and the action levels were exceeded on 5 occasions. No correlation was established between the presence of *Prorocentrum lima* in water and the presence of DSP toxins in shellfish.

***Pseudo-nitzschia* spp**. This genus was detected in 374 samples from 48 production areas (detected June to August and October 2006). Cell concentrations were variable and in 96% of cases did not exceed 50 000 cells/l. Thirty-two samples exceeded the action level of 150 000 cells/l. A correlation between the presence of *Pseudo-nitzschia* spp. and the occurrence of ASP toxins in shellfish flesh could only be established

Figure 3.116 The presence of Alexandrium spp. in the UK. (a) April 2005–March 2006; (b) April 2006– March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.





for some of the sites where ASP toxins were detected, although the cell concentrations involved were often well below the action level for these organisms.

Three *Pseudo-nitzschia* species: *P. delicatissima*, *P. cf. fraudulenta* and *P. pseudodelicatissima* were speciated in English coastal waters. None were linked to toxicity in UK shellfish.

3.7.6.1.3 April 2007 – March 2008

Source: *PSP and DSP Biotoxin Monitoring Programme for England and Wales, 1st April 2007 to 31st March 2008, Cefas contract report C2333.*

A total of 1163 shellfish samples were submitted from 111 inshore sampling locations, representing 64 of the classified English and Welsh shellfish harvesting or relaying production areas (coverage rate of 94% of the active classified areas during the reporting period) for testing. A total of 1122 water samples collected from 55 of the classified production areas were submitted to Cefas for phytoplankton analysis.

PSP. 1099 flesh samples were tested for PSP toxins. Toxins were detected by MBA in three mussel samples from Holy Island during May to June 2007. Toxin concentrations did not exceed 80 µg STX eq./100 g [shellfish tissue] in any of these samples, although one sample collected on 15 May gave an average concentration of 76 µg STX eq./100 g.

DSP. 1024 shellfish samples were tested for DSP/lipophilic toxins. Nine recorded positive results using the reference method. Affected sites were subject to temporary harvesting restrictions. Toxin profiles determined by LC-MS indicated that samples from Holy Island, where 5 of the positives mussel samples were collected during June to September 2007, contained okadaic acid, dinophysistoxins (DTXs) and azaspiracids. One sample was also found to contain yessotoxins. The Salcombe Pacific oyster sample (collected early July) contained yessotoxins. None of the lipophilic toxins screened for were detected in the Southampton Water native oyster sample (collected mid-June), the Fal mussel sample (collected late June) or the Three Rivers cockle sample (collected early October).

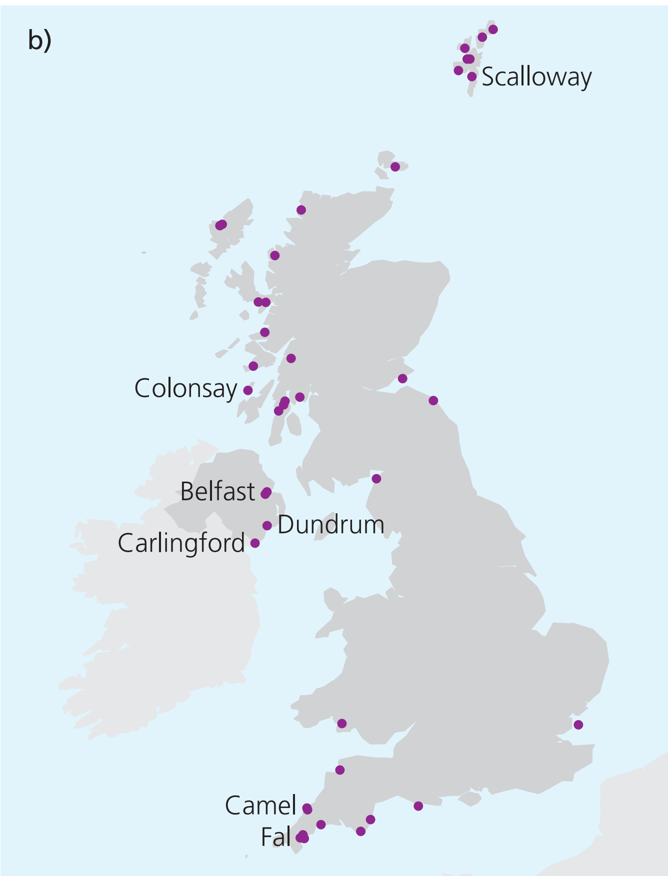
ASP. 985 shellfish samples were tested for ASP toxins. Toxins were detected in 6 samples (including cockles, mussels and Pacific oysters) from five classified production areas: Burry Inlet (collected July), Colne (collected November), Holy Island (collected May), Taw/Torridge (collected July) and Three Rivers (collected July), all at concentrations below 8 µg [domoic+epi-domoic acid]/g [shellfish tissue].

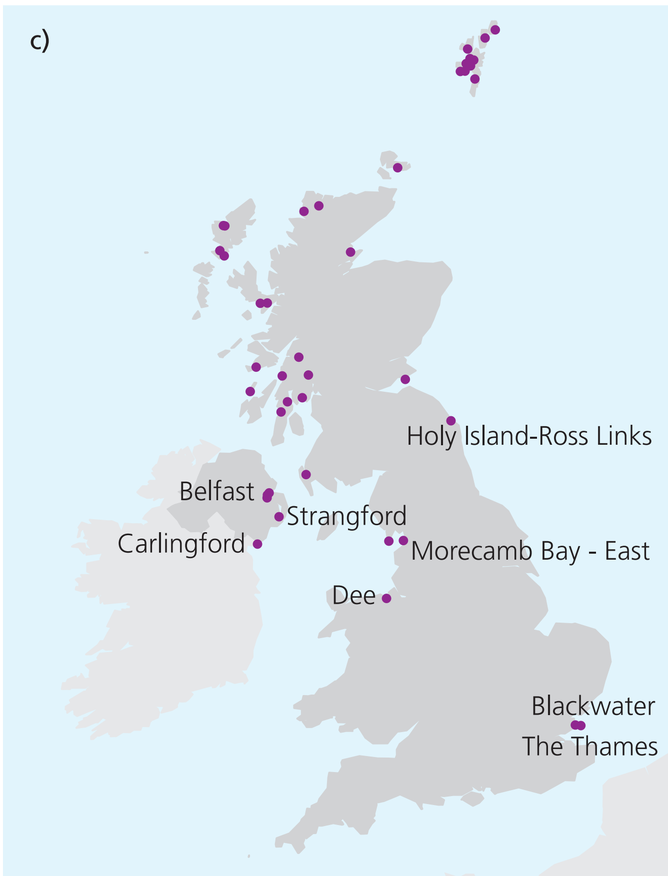
***Alexandrium* spp**. This genus was detected from April to October 2007 in 139 samples from 24 production areas, with maximum cell concentrations above 2 million cells/l at Yealm in July. A correlation was established between the presence of *Alexandrium* spp. in water and the presence of detectable levels of PSP toxins in mussels from Holy Island.

***Dinophysis* spp**. This genus was found in 18 samples from 10 production areas and cell numbers greater than the action level were detected on four occasions during May to August 2007 in the Dee, at Holy Island, and in Morecambe Bay. A maximum concentration of 800 cells/l occurred at Holy Island and coincided with the presence of lipophilic toxins in shellfish flesh.

Prorocentrum lima. This species was detected at concentrations above the action level on two occasions, in August and September, in

Figure 3.117 The presence of Dinophysis *spp. and/or* Prorocentrum *lima (>100 cells/l) in the UK. (a) April 2005–March 2006; (b) April 2006–March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.*





the Blackwater and the Thames estuaries, respectively. No correlation could be established between *P. lima* and positive DSP results.

***Pseudo-nitzschia* spp**. This genus was detected in 570 samples from 49 production areas. Cell concentrations were variable and, in over 98% of cases, did not exceed 50 000 cells/l. Only four samples exceeded the action level of 150 000 cells/l. They were collected from three areas: Colwyn Bay (June), Poole (April) and Teign (April). A correlation between the presence of *Pseudo-nitzschia* spp. and the occurrence of toxins in shellfish flesh could only be established for some of the sites where ASP toxins were detected.

3.7.6.2 Scotland

3.7.6.2.1 April 2005 – March 2006 (shellfish) and September 2005 – March 2006 (harmful algae)

Source: *Biotoxin Monitoring Programme Scotland, PSP Monitoring – 1st April 2005 to 31st March 2006, DSP Monitoring – 1st September to 31st March 2006, Cefas contract report C2511, C2649 and C2576.*

Official Control Monitoring Programme for Presence of Biotoxins in Live Bivalve Molluscs in Scotland 1st April 2005- 31st March 2006, Integrin Advanced Biosystems Ltd.

A total of 831 shellfish samples from 98 inshore sampling locations were collected. Due to changes in the organisation of the water monitoring programme, data are available from September 2005 and, from this time, 256 water samples were collected.

PSP. 466 samples were tested for PSP and toxins were detected in five samples (mussels [3], cockles [1] and queen scallops [1]) collected from five separate areas: Loch Stockinish (July), Culbin Sands (June), Scalpay (July), Clift Sound (January) and Weisdale Voe (September). Concentrations ranged from 33 to 70 µg STX eq./100 g and were therefore all below the action level for PSP (80 µg STX eq./100 g flesh).

DSP. 652 samples were tested for DSP and toxins were detected in 10 samples (all mussels) from four separate sites: Loch Linnhe (May), Loch Striven (September to October), St Abbs (September) and Ronas Voe (September). All sites were subject to temporary harvesting restrictions until additional tests had confirmed the absence of the toxin in subsequent samples.

ASP. 810 samples were tested for ASP and 14, all king scallops, exceeded the action limit. The 14 samples were from three locations: Loch Caolisport (October to December), Ura Firth (June to February) and Loch Sligachan (November).

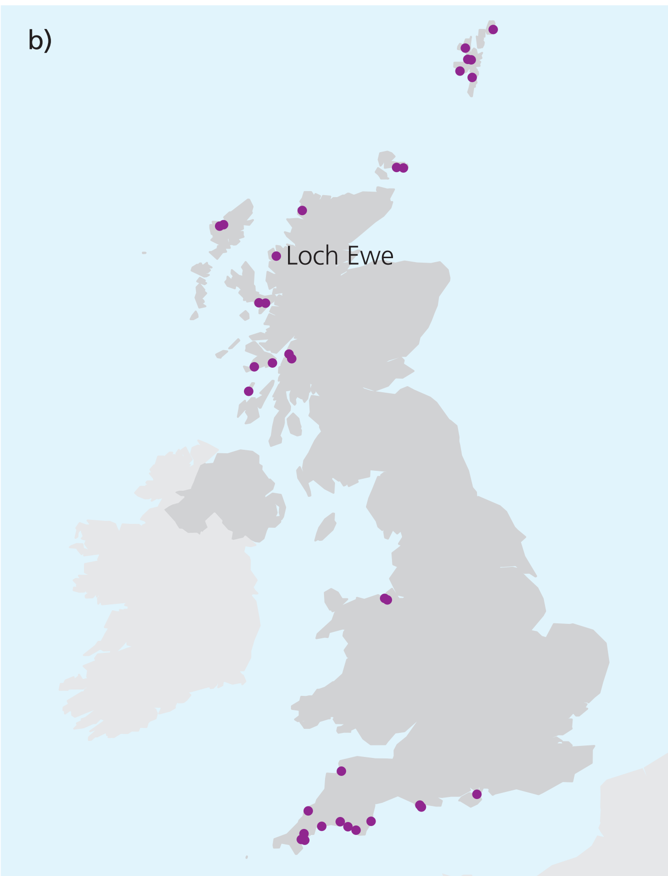
***Alexandrium* spp**. This genus was detected in 22 samples at 7 locations. A maximum concentration of 80 cells/l was detected at Ronas Voe in late October.

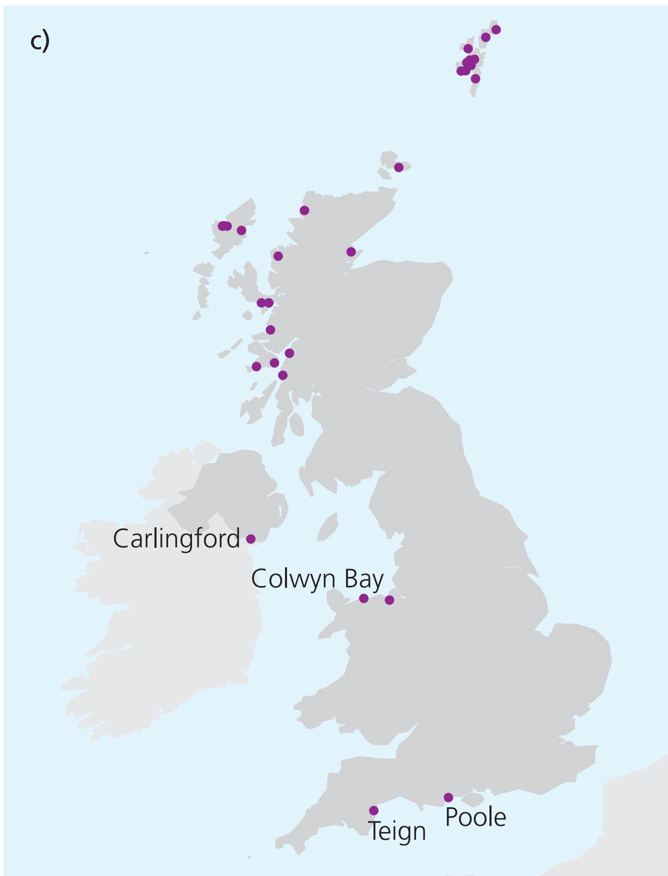
***Dinophysis* spp**. This genus was found in only one sample from Loch Striven, collected in late September, at a concentration of 100 cells/l.

Prorocentrum lima. This species was not detected during the monitoring period.

***Pseudo-nitzschia* spp**. This genus was detected at concentrations above 50 000 cells/l in 8 samples from 6 locations. All but one of these samples was collected in September

Figure 3.118 The presence of Pseudo-nitzschia spp. (>150 000 cells/l England, Wales and Northern Ireland; >50 000 cells/l Scotland) in the UK. (a) April 2005–March 2006; (b) April 2006–March 2007; (c) April 2007–March 2008. © Crown copyright 2010: permission granted by Cefas.





and it was during this month that the highest concentration, over 270 000 cells/l, was detected in a sample from Loch Melfort.

3.7.6.2.2 April 2006 – March 2007

Source: *Biotoxin Monitoring Programme Scotland, PSP Monitoring-1st April 2006 to 31st March 2007, Cefas contract report PAU 179- S02007/PSP & DSP.*

Official Control Monitoring Programme for Presence of Biotoxins in Live Bivalve Molluscs in Scotland 1st April 2006- 31st March 2007, Integrin Advanced Biosystems Ltd.

Monitoring Programme for the Presence of Toxin Producing Plankton in Shellfish Production Areas in Scotland 01 April 2006- 31 March 2007, SAMS contract report SO1007/PAU179A.

A total of 1374 shellfish samples were collected from 111 inshore sampling locations. Of all samples tested, 79% were blue mussels. This constitutes a significant increase over previous years and reflects changes made to the Scottish inshore biotoxin monitoring programme, where mussels are now used as a sentinel species wherever possible. 918 seawater samples collected from 41 coastal locations were analysed for potentially toxic genera or species of phytoplankton.

PSP. 1242 samples were tested for PSP toxins and these were detected in 72 samples (~5% of the total) from 23 separate areas during April to November. While mussels were the most affected species, PSP toxins were also recorded in razorshells and common cockles. PSP episodes were generally of short duration (maximum 3 weeks), although prolonged outbreaks (of more than 8 weeks duration) were also recorded. PSP toxin concentrations recorded during the reporting period ranged from 35 to 413 µg STX eq./100 g, including 15 results above the action level for PSP (80 µg STX eq./100 g flesh).

DSP. 1070 DSP tests were performed and toxins were detected in 70 samples (~5% of the total) from 22 separate sites. Toxin episodes were recorded, in mussels only, between May and October 2006. These episodes were generally long (lasting several weeks) and were usually (but not always) preceded by increased counts of *Dinophysis* spp. Data obtained using LC-MS analysis indicated that, in addition to okadaic acid and dinophysistoxins, other lipophilic toxins such as azaspiracids (1, 2 and 3) and pectenotoxin2 were present in Scottish waters.

ASP. 1392 samples were tested for ASP. Thirteen contained ASP toxins and of these two samples were above the regulatory limit. The two samples exceeding the limit were both from king scallops collected from Loch Caolisport in March 2007.

***Alexandrium* spp**. This genus was detected in about 50% of the samples analysed during August, and the densest blooms occurred in the Shetland Islands between mid-August and early September with a maximum concentration of 6600 cells/l. *Alexandrium* spp. continued to be present at some sites until mid October. Most occurrences of PSP in shellfish tissue were preceded by increased counts of *Alexandrium* spp.

***Dinophysis* spp**. This genus was detected in many samples with a maximum density of over 10 000 cells/l in Shetland (Scalloway) during early August. More than 50% of samples examined in July had this genus present at densities above the action limit of 100 cells/l.

Dinophysis spp. continued to be present throughout August to September and at several sites this was linked to the presence of DSP toxins in the shellfish samples.

Prorocentrum lima. This species occurred infrequently, with a maximum density of 1200 cells/l recorded at Colonsay in late July.

***Pseudo-nitzschia* spp**. Spring and autumn blooms of *Pseudo-nitzschia* occurred during this testing period. At Scapa Bay, concentrations of Pseudo-nitzschia spp. were found to be consistently above 50 000 cells/l from April to mid-June, with a maximum concentration above 1.5 million cells/l during May. The highest concentration observed during this year was 4400 000 cells/l in Loch Ewe during late August. Water samples were not taken from Loch Caolisport during March when ASP was detected in shellfish above the action level so it is not possible to determine if the two occurrences coincided.

3.7.6.2.3 April 2007 – March 2008

Source: *Biotoxin Monitoring Programme Scotland, PSP Monitoring-1st April 2007 to 31st March 2008, Cefas contract report PAU 179- S02007/PSP & DSP.*

Official Control Monitoring Programme for Presence of Biotoxins in Live Bivalve Molluscs in Scotland 1st April 2007- 31st March 2008, Integrin Advanced Biosystems Ltd.

Monitoring Programme for the Presence of Toxin Producing Plankton in Shellfish Production Areas in Scotland 01 April 2007- 31 March 2008, SAMS contract report SO1007/PAU179A.

A total of 2854 shellfish samples from 117 inshore sampling locations were tested, 86% of those being mussels. 1384 water samples were collected from 48 coastal locations.

PSP. 2739 samples were tested for PSP toxins and these were detected in 54 samples from 18 separate areas during April to September. Common mussels were the only affected species. PSP episodes were generally of short duration (maximum of 4 weeks) although prolonged outbreaks (up to 7 weeks) were also recorded. PSP toxin concentrations ranged from 36 to 175 µg STX eq./100 g including 8 results above the action level for PSP (80 µg STX eq./100g flesh).

DSP. 2266 samples were tested for DSP toxins and 84 samples from 29 separate sites recorded positive results. Toxin episodes were recorded in common mussels (83) and Pacific oysters (1) between May and September 2007. The worst affected area was the Shetland Islands, which experienced closures in a large number of production areas. Data obtained from LC-MS analysis indicated that, in addition to okadaic acid and dinophysistoxins, other lipophilic toxins such as pectenotoxin2 and yessotoxins were present in Scottish waters.

ASP. 2230 samples were tested for ASP toxins and 5 were found to contain concentrations above the action level. These consisted of 2 king scallop samples from Loch Caolisport collected during September to October 2007, and mussel samples from Loch Roag (August), Wadbister Voe (July) and Weisdale Voe (August).

***Alexandrium* spp**. This genus was observed around Orkney from mid-April to mid-June (maximum concentration of 6060 cells/l), but its occurrence was less widespread than during the previous monitoring year. The highest concentration observed, 9180 cells/l, was in

the Shetlands in late July. At some monitoring sites along the west coast of Scotland during April to June 2007, and in Shetland during April and July to September 2007, the occurrence of *Alexandrium* spp. was associated with toxicity in shellfish.

***Dinophysis* spp**. This genus exceeded the action limit from late April onwards at a number of sites, appearing most frequently in Argyll & Bute, north-west Scotland, Shetland and Orkney. *Dinophysis* spp. was observed from June to August in Shetland with a maximum concentration of 5300 cells/l. This was associated with the presence of DSP toxins in Shetland shellfish between June and August. The highest concentration of 58 000 cells/l observed was in a sample from Elie, Fife.

Prorocentrum lima. This species was observed at low densities during the monitoring period. The maximum cell density observed was 560 cells/l, in Shetland during mid-June.

***Pseudo-nitzschia* spp**. This genus exceeded 50 000 cells/l in 11% of all samples tested. *Pseudo-nitzschia* spp. were widespread throughout Orkney and the Shetland Islands during June and July. The highest concentration observed, 2.5 million cells/l, was in water samples collected from Shetland (Clift Sound) in early July 2007. In Argyll & Bute (June to August) and Shetland (July to September), occurrences of *Pseudo-nitzschia* spp. at densities above 50 000 cells/l were associated with ASP toxin detection in the shellfish tissue samples. A high concentration of 365 000 cells/l in the Western Isles in late August was also associated with ASP toxicity in shellfish, above the action limit. By October, most *Pseudo-nitzschia* spp. occurrences had ceased, with organisms persisting only around the Orkney Islands and the Scottish east coast.

3.7.6.3 Northern Ireland

3.7.6.3.1 April 2005 – March 2006

Source: *Monitoring for Toxin Producing and Nuisance Microalgae in N. Ireland Coastal Waters – 1 April 2005–31 December 2005 (Report to FSANI)*

Monitoring for Toxin Producing and Nuisance Microalgae in N. Ireland Coastal Waters – 1 January 2006–31 December 2006 (Report to FSANI)

A total of 375 flesh samples were tested for marine biotoxins, consisting of; 215 mussel, 132 oyster, 17 cockle and 11 scallop samples. 759 water samples were also processed for potentially toxic algae.

PSP. 382 samples were tested for PSP toxins, no toxins were detected.

DSP. 404 samples were tested for DSP toxins. One sample was found to be positive which resulted in the closure of a Strangford Lough mussel bed from late April to early May 2005.

ASP. 381 samples were tested for ASP toxins, and monthly samples taken between September 2005 and January 2006 from Strangford Lough had levels in whole flesh above the regulatory limit.

***Alexandrium* spp**. Of the 759 samples processed, thirteen contained *Alexandrium* cells. A maximum abundance of 60 cells/l was recorded from sites in Carlingford Lough in late June and early July.

***Dinophysis* spp**. In 2005 *Dinophysis* spp. were present in 11% of all samples, and a maximum abundance of 640 cells/l was recorded in a water sample from Belfast Lough in mid-July. A positive toxin test in shellfish samples in April was not linked to the presence of DSP producing phytoplankton in the water column and follow up samples gave negative results.

Prorocentrum lima. Only 10 of the 759 samples tested had *P. lima* present. A maximum abundance of 420 cells/l was recorded from Lough Foyle in mid-August, with all other samples below the threshold value of 100 cells/l.

***Pseudo-nitzschia* spp**. This genus was recorded in 50% of samples with a peak abundance of about 70 000 cells/l in mid-September being recorded in a water sample taken from Carlingford Lough.

3.7.6.3.2 April 2006 – March 2007

Source: *Monitoring for Toxin Producing and Nuisance Microalgae in N.Ireland Coastal Waters – 1 January 2006–31 December 2006 (Report to FSANI)*

Monitoring for Toxin Producing and Nuisance Microalgae in N.Ireland

Coastal Waters – 1 January 2007–31 December 2007 (Report to FSANI)

afbini Shellfish toxin report April 06–March 07

A total of 385 samples were tested for marine biotoxins, consisting of 197 mussel, 160 oyster, 13 cockle, 5 clam and 10 scallop samples. 766 water samples from a total of 35 sites were analysed for potentially toxic algae.

PSP. All 371 flesh samples tested for PSP toxins were negative.

DSP. Of the 374 samples tested for DSP toxins, 10 oyster samples were found to yield positive results. DSP toxin positive tests resulted in the subsequent closure of shellfish beds in Dundrum Bay (oyster bed) August 2006 and mid-February to mid-March, Larne Lough in August 2006, Strangford Lough (oyster bed) early February to early March and Carlingford Lough late February to early March.

ASP. 346 samples were tested for ASP, none were found to have levels above the regulatory limit.

***Alexandrium* spp**. Occurrence and abundance of *Alexandrium* spp. was low, with *Alexandrium* recorded in only 15 samples. Maximum cell concentrations of 60 cells/l were recorded in one sample from Belfast Lough in mid-June and one from Carlingford Lough in late July.

***Dinophysis* spp**. This genus was recorded from all water bodies except Lough Foyle. Levels exceeded the action value of 100 cells/l on four occasions, in late June from three sites in Belfast Lough and in early July from Dundrum Bay. A peak abundance of 280 cells/l was recorded in Belfast Lough.

Prorocentrum lima. Cells of *Prorocentrum lima* were detected on 23 occasions during the year, with a maximum abundance of 120 cells/l from a site in Carlingford Lough in mid-September. This was the only occasion when concentrations exceeded the action level of 100 cells/l.

***Pseudo-nitzschia* spp**. A large proportion of samples, 63% of the total, contained *Pseudo-nitzschia* spp., although none exceeded the action level of 150 000 cells/l. The maximum recorded value was about 50 000 cells/l from Dundrum Bay in early October.

3.7.6.3.3 April 2007 – March 2008

Source: *Monitoring for Toxin Producing and Nuisance Microalgae in N.Ireland Coastal Waters – 1 January 2007–31 December 2007 (Report to FSANI)*

afbini Shellfish Toxin Report April 07–08

A total of 387 samples were tested for marine biotoxins, consisting of 234 mussel, 128 oyster, 8 cockle, 10 clam and 7 scallop samples. 822 water samples were analysed for potentially toxic algae.

PSP. 404 samples were tested for PSP toxins, all samples were negative.

DSP. 393 samples were tested for DSP toxins. Two samples were found to be positive resulting in the closure of one oyster bed in Carlingford Lough from late April to early May 2007 and again from late December to mid-January 2008.

ASP. 378 samples were tested for ASP toxins. Scallop samples from Strangford Lough were found to have levels above the regulatory limit in single samples taken in August, September and October.

***Alexandrium* spp**. This genus was detected only in Belfast Lough and Carlingford Lough. Concentrations were low, with a maximum of 120 cells/l recorded from Belfast Lough in mid-June.

***Dinophysis* spp**. This genus was recorded in 82 water samples. Of these, only 5 were above the threshold value of 100 cells/l. All five were from Belfast Lough and were recorded in the period mid-June to early July. Maximum abundance was recorded at 200 cells/l from Belfast Lough in early July.

Prorocentrum lima. This species was recorded on 16 occasions, at low concentrations. Only 2 samples exceeded the action value, with a maximum of 140 cells/l recorded from a Carlingford Lough site in late April and from Strangford Lough in mid-June.

***Pseudo-nitzschia* spp**. This genus was present in 65% of samples. A maximum density of almost 200 000 cells/l was recorded in a water sample taken from Carlingford Lough in late August, although no toxicity was recorded in shellfish samples at the time.

3.7.7 Progress towards the vision of clean and safe seas

The evidence has demonstrated the effectiveness of the current biotoxin monitoring programme within the UK for the protection of human health. There has only been one recorded incident of biotoxin poisoning from the consumption of UK shellfish within the UK during the current reporting period (2005 to 2008). This incident occurred after a biotoxin event was missed by the statutory monitoring programme due to a rapid increase in toxicity in a harvesting area over the space of a single week. Subsequent implementation of weekly monitoring at RMPs during high risk periods should help to mitigate the occurrence of biotoxin outbreaks. However, it is important to highlight that monitoring in itself does not guarantee the absence of biotoxins in marketed shellfish, and it is the Industry which is ultimately responsible for ensuring shellfish safety.

The monitoring programme has provided comprehensive information of the extent of biotoxins in shellfish and potentially toxic algae since 2001. Prior to this, monitoring was less regular and extensive but still provided useful information and was adequate to provide human health protection for the most toxicologically dangerous of the biotoxins, PSP. There has been no indication of any increase in the occurrence of biotoxins in shellfish since 2005 and resultant closures have not increased in frequency. On examination of the detailed data for the monitoring programme from the last three years (Table 3.61), there has been no increase in the percentage of toxic shellfish or potentially toxic algae. However, due to the short time scale examined, it is not possible to determine if this is due to natural variability or indicative of a general trend.

The monitoring programme does highlight ‘hotspots’ of biotoxin activity where toxins or algae are expected to occur most years. Two examples are the Fal Estuary in England (where PSP-causing algae and PSP toxins are annually detected) and the Shetland Islands (where PSP and DSP toxins are detected most years). By continuing to analyse the data from the programme it will be possible to highlight new and emerging ‘hotspots’, which will help in the continued writing and updating of environmental risk assessments and ensure the safety of shellfish consumers.

Table 3.62 presents our expert opinion on the status and trends for algal toxins within the UK seas. We have done this by assigning a single colour for status where: green indicates few or no problems, amber indicates some problems and red indicates many problems. The confidence rating is classified as low (I), medium (II) or high (III) based on the number of indicators available in that region and the agreement between them.

Table 3.61 Percentage of samples that exceeded action levels within the UK from April 2005 to March 2008.

	2005–2006	*2006–2007*	*2007–2008*
England and Wales			
PSP	0.2	0.2	0
DSP	0.4	0.6	0.9
ASP	0	0	0
Alexandrium	12	17	12
Dinophysis and *P. lima*	1	4	1.8
Pseudo-nitzschia	0.4	3.6	0.36
Scotland			
PSP	0	1.2	0.3
DSP	1.8	6.5	3.7
ASP	1.7	0.1	0.22
Alexandrium	Not available	Not available	Not available
Dinophysis and *P. lima*	Not available	Not available	Not available
Pseudo-nitzschia	Not available	Not available	Not available
Northern Ireland			
PSP	0.0	0.0	0.0
DSP	0.3	2.7	0.5
ASP	0.8	0.0	0.8
Alexandrium	1.7	2.0	1.7
Dinophysis and *P. lima*	2.5	0.7	0.9
Pseudo-nitzschia	0.0	0.0	0.1

Table 3.62 Regional assessment summary for algal toxins.

CP2 Region	*Key factors and pressure*	*What the evidence shows*	*Trend*	*Current status*	*Confidence in assessment*	*Forward look*
1	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
2	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
3	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
4	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
5	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
6	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
7	Harmful algal blooms in association with shellfish culture	Fisheries closed when toxin levels breach limits	Not known		**III**	
8	No shellfish culture			**NA**	**0**	

NA Indicates no significant activity in this region.

3.7.8 Forward look and need for further work

The current biotoxin monitoring programme is well established and fulfils the legal requirements under EU Council Regulations. Providing that the scope of the programme does not diminish without a full assessment of risk, maintenance of the programme should be sufficient for the future. However, the programmes employed are not static and development of new risk assessments and analytical and molecular approaches to biotoxin and toxic phytoplankton detection are ongoing. Current work includes fully quantitative HPLC for PSP in all species, LC-MS detection of DSP toxins and molecular determination of *Pseudo-nitzschia* and *Alexandrium* species. These developments in the shellfish testing regimes will increase the information on temporal and spatial risks and the occurrence of toxins and algal species in UK waters. This will aid and enhance the current monitoring programmes.

3.8 Underwater Noise

3.8.1 Key points

i. Introduction

Underwater noise is an issue for most marine mammals, many marine fish, and perhaps some shellfish as underwater sound is important for communication, locating mates, searching for prey, avoiding predators and hazards, and for short- and long-range navigation. Sound can mask biologically relevant signals; it can lead to a variety of behavioural reactions; hearing organs can be affected in the form of hearing loss, and at very high received levels, sound can injure or even kill marine life. Man-made sound sources of primary concern are explosions, shipping, seismic surveys, offshore construction and offshore industrial activities and sonars of various types.

ii. How has the assessment been undertaken?

There is currently not enough evidence to provide a quantitative assessment of underwater noise in UK waters.

iii. Current status of underwater noise and past trends

The current status of underwater noise is unknown due to lack of information from monitoring studies. Similarly, trends cannot be assessed at present due to a lack of knowledge.

iv. What has driven change?

No data are available with which to assess changes in underwater noise levels, but increasing activity in, for example, offshore wind farm developments, is likely to have increased noise levels in their vicinity.

v. What are the uncertainties?

Data are available for a limited number of research studies, but there is no rigorous assessment of uncertainties which would arise during routine monitoring studies.

vi. Forward look

The issue of underwater noise is of concern and urgently requires more research. Future studies should focus on mapping and/or modelling ambient noise, observational and experimental studies, and the further development of frameworks for assessing noise related risks.

3.8.1.1 Overall findings

There is currently not enough evidence to provide a quantitative assessment of underwater noise in UK waters. However, the issue is of concern and needs more research as soon as possible. Noise is an issue as for many marine organisms, including most marine mammals, many marine fish, and perhaps even some invertebrates as sound is important for communication, locating mates, searching for prey, avoiding predators and hazards, and for short- and long-range navigation. There are a variety of sources for underwater sound, some sounds occur naturally, others are man-made. Sound can potentially affect marine organisms in various ways. It can mask biologically relevant signals; it can lead to a variety of behavioural reactions; hearing organs can be affected in the form of hearing loss, and at very high received levels, sound can injure or even kill marine life. From a conservation perspective, estimating the effects of noise disturbances on populations is critical, and there are first attempts to develop population consequences of acoustic disturbance models (PCAD), at least for marine mammals. Man-made sound sources of primary concern with regards to disturbance of marine

life are explosions, shipping, seismic surveys, offshore construction (e.g. for offshore wind farms or hydrocarbon production and transport facilities), and offshore industrial activities (dredging, drilling), sonars of various types, and acoustic deterrent devices. Documented effects on marine life vary greatly from very subtle behavioural changes, to avoidance reactions, hearing loss, injury and even death in extreme cases. With regard to the UK, the distribution and level of sound-generating activities are very difficult to quantify and noise budgets for the different regions are not yet available. There are also big gaps in our understanding of the effects of noise on marine life which could be addressed in future studies, including mapping and/or modelling ambient noise, observational and experimental studies (e.g. controlled exposure experiments), and the further development of risk assessment frameworks.

3.8.1.2 Contribution towards the GES descriptor

The relevant qualitative descriptor for determining Good Environmental Status (GES) under the EU Marine Strategy Framework Directive states that ...*Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*. The available information contributes directly to this descriptor as it informs on the current level of sound-generating activities off the UK and outlines research on the effects of underwater sound on marine life. In cases of more forthcoming information, feasible mitigation measures can be applied to reach GES with regards to underwater sound.

3.8.1.3 Future risks

Studies indicate that ocean acidification resulting from increased uptake of carbon dioxide (CO_2) as a consequence of higher atmospheric concentrations and reduced ventilation results in significant decreases in ocean sound absorption at frequencies lower than about 10 kHz. Ambient noise levels in the ocean within the auditory range critical for environmental, military, and economic interests are set to increase significantly due to the combined effects of decreased sound absorption and increasing inputs of noise from human activities.

3.8.1.4 Cross-cutting issues

Since underwater noise is not an activity *per se*, but rather a pressure resulting from various activities, there are many cross-links to other activities addressed in *Charting Progress 2* such as aggregate extraction and marine construction.

3.8.2 Introduction

3.8.2.1 Description of the topic

Water is an ideal medium for sound: acoustic waves travel four times faster in water than in air and attenuation is much less underwater than above. Not surprisingly, many marine organisms, including most marine mammals (whales, dolphins, porpoises, pinnipeds), many marine fish, and even some invertebrates, use sound to communicate, to locate mates, to search for prey, to avoid predators and hazards, and for short- and long-range navigation (for reviews see Tyack and Clark, 2000; Popper et al., 2001, 2003; Dudzinski et al., 2002; Würsig and Richardson, 2002). Sound can potentially affect marine organisms in various ways. It can mask biologically relevant signals such as echolocation clicks or communication calls; it can lead to a variety of behavioural reactions; hearing organs can be affected in the form of hearing loss, and at very high received levels, sound can injure or even kill marine life (Richardson et al., 1995).

Concerns on the potential adverse effects of anthropogenic sounds on marine life have been raised from within the scientific community since the 1970s, and research on the topic expanded in the 1980s (e.g. Payne and Webb, 1971; Richardson et al., 1985). During the past decade the topic has been investigated extensively by a number of scientific institutions, Government agencies and intergovernmental bodies (reviews by Richardson et al., 1995; Council, 2003; NRC, 2005; Würsig and Richardson, 2002; Popper et al., 2004; Hastings and Popper, 2005; Hildebrand, 2005; ICES-AGISC, 2005; Wahlberg and Westerberg, 2005; Thomsen et al., 2006b; Madsen et al., 2006; Southall et al., 2007; Nowacek et al., 2007; Weilgart, 2007). These studies have documented both the presence and absence of physiological effects and behavioural responses of marine life to various sound signals. This has set the scene for discussions among scientists, stakeholders and policy makers on how to address potential impacts of underwater sound and how to develop meaningful mitigation measures within regulatory frameworks.

3.8.2.2 Environmental impacts

A variety of sound sources in the marine environment occur naturally, such as vocalisations of marine mammals, fish and certain crustaceans, sounds that are induced by precipitation, wind, currents and waves. In addition there are acoustic events such as sub-sea volcanic eruptions, earthquakes and lightning strikes (Richardson et al., 1995; Würsig and Richardson, 2002). Some can reach quite high source levels of more than 200 dB re 1μPa peak to peak (Møhl, 2003) (see Box 3.5). Snapping shrimp influence ambient noise levels in tropical and subtropical waters to a high

Box 3.5:
Sound

Sound waves are pressure fluctuations, compressions and rarefaction of the molecules in the medium through which the sound wave propagates. The unit for pressure is the Pascal (i.e. Newton per square metre). Each sound wave has both a pressure component (in Pascals) and a particle motion component, indicating both the velocity (m/s) and the acceleration (m/s^2) of the molecules in the sound wave. Depending on their receptor mechanisms, marine life is sensitive to either pressure or particle motion or both. The pressure can be measured with a pressure sensitive device such as a hydrophone (underwater microphone). Due to the wide range of pressures and intensities and also taking the physiology of marine life into account, it is customary to describe sound through the use of a logarithmic scale. The most generally used logarithmic scale for describing sound is the decibel scale (dB). The sound pressure level (SPL) of a sound of pressure P is given in decibels (dB) by: SPL (dB) = 20 log10 (P/P0). P is the measured pressure level and P0 is the reference pressure. The reference pressure in underwater acoustics is defined as 1 μPa. As the dB value is given on a logarithmic scale, doubling the pressure of a sound leads to a 6 dB increase in sound pressure level. As the reference pressure for in air measurements is 20 μPa, underwater dB and in air dB cannot be compared without applying conversion factors. First, 26 dB must be subtracted from the underwater dB due to the difference in reference units; second, another 36 dB must be subtracted to account for the difference in acoustic impedance between air and water. An underwater sound pressure level of 200 dB re 1 μPa would therefore correspond to 138 dB re 20 μPa in air (see Urick, 1983 and OGP, 2008 for more details).

degree and might also contribute to ambient noise levels in some areas at higher latitudes (Wenz, 1962; Council, 2003; Hildebrand, 2005).

Man-made sound sources of primary concern in impact assessments are underwater explosions, shipping, seismic exploration, offshore construction (e.g. for offshore wind farms or hydrocarbon production and transport facilities) and offshore industrial activities (dredging, drilling), sonars of various types and acoustic devices designed to deter marine mammals from approaching (so called acoustic harassment or deterrent devices, AHDs, ADDs). Emitted frequencies range from low frequency engine sound < 100 Hz to very high frequency echo sounders of several hundred kHz (Lepper et al., 2004). Source levels also vary widely and can reach more than 250 dB re 1µPa in the case of some offshore construction activities, seismic exploration and explosives (reviews by Richardson et al., 1995; Nowacek et al., 2007; Thomsen et al., 2006b).

Impact assessments are generally concerned with those man-made activities that overlap in frequency with the hearing range of marine organisms in question (reviews by Richardson et al., 1995; Nowacek et al., 2007). A sound is audible when the receiver is able to perceive it over background noise. The threshold of hearing that varies with frequency also determines audibility. The frequency dependent hearing sensitivity is expressed in the form of a hearing curve (audiogram), which in fish and marine mammals usually exhibits a U-shaped form. Determining if and how far an animal can hear a sound is the first important step in any impact assessment (Richardson et al., 1995; Popper et al., 2003). Marine mammals are – in general – more sensitive and have a wider range of hearing than marine fish. However, some fish, like cod and herring have acute hearing at lower frequencies (< 1 KHz), that rivals or exceeds that of cetaceans (see Figure 3.119, Figure 3.120).

When evaluating the effects of underwater sound, many parameters are important. For example, peak pressure, received acoustic energy, signal duration, spectral type, frequency (range), duty cycle, directionality, and signal rise times. For species that are only sensitive to particle motion (e.g. a variety of fish), this must also be considered. Possible effects can vary depending on a range of internal and external factors, and can be broadly divided into masking (obscuring of sounds of interest by interfering sounds, generally at similar frequencies), response, and discomfort, hearing loss and injury. Hearing loss can either be temporary threshold shift (TTS) or permanent threshold shift (PTS). In extreme cases, and at very high received sound pressure levels, that are usually close to the source, very intense sounds might also lead to non-auditory tissue injury or even death of the exposed organism (Richardson et al., 1995; Popper et al., 2003, 2004; Madsen, 2005; Nowacek et al., 2007; Southall et al., 2007; see Box 3.6).

3.8.2.3 Socio-economic impacts

Effects on the environment due to underwater sound exposure can potentially lead to further socio-economic impacts. For example, many marine construction activities, such as pile driving for offshore wind farm foundations, are planned within or near spawning areas of commercially important fish species such as sole (*Solea solea*) and herring (*Clupea harengus*). Behavioural responses of these species to pile-driving sound might prevent fish from reaching breeding or spawning sites, finding food, and acoustically locating mates. This could result in long-term effects on reproduction and

Figure 3.119 Representative audiograms of some toothed whales (Odontocetes) that occur in UK waters. The audiograms show a typical U-shaped form with lower sensitivity (in dB re: 1μPa) at lower frequencies (in kHz) compared to frequencies above 10 kHz. Harbour porpoises, for example, hear best above 100 kHz. Audiograms can be obtained either in behavioural experiments or through external measurements of the auditory brainstem response. Sources: Johnson (1967); Nachtigall et al. (1995); Szymanski et al. (1999); Kastelein et al. (2002, 2003). © Crown copyright 2010: permission granted by Cefas.

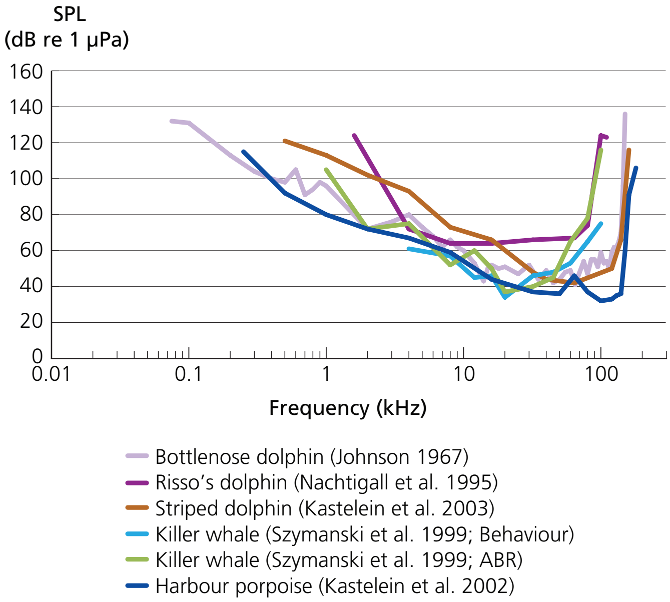
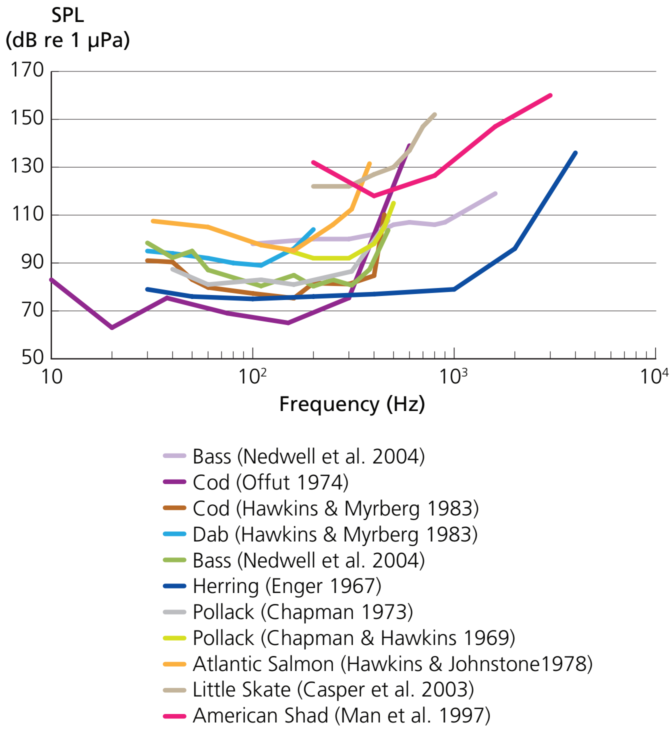


Figure 3.120 Audiograms of fish regularly occurring in UK waters (note the different x-axis scale relative to Figure 3.119. Sources: Enger (1967); Chapman and Hawkins (1969, 1973); Offutt (1974); Hawkins and Johnstone (1978); Hawkins and Myrberg Jr (1983); Mann et al. (1997); Casper et al. (2003); Nedwell et al. (2004). © Crown copyright 2010: permission granted by Cefas.



population parameters and might in turn affect the fishery. Furthermore, avoidance reactions might result in displacement away from potential fishing grounds and result in reduced catches, as has been shown to be the case in herring due to seismic survey activity (Engås et al., 1996). The UK waters are also home to a rich diversity of cetacean species with some, such as the bottlenose dolphin, forming resident populations in distinct areas. Whale watching has become an important socio-economic factor in some regions of the UK, and negative effects due to sound exposure from other sources might have a detrimental effect on this relatively new industry.

3.8.2.4 European legislation and regulation

The EU Marine Strategy Framework Directive sets out a framework for the development of national strategies, aimed at achieving Good Environmental Status (GES) in the marine environment by 2021 at the latest. The Directive sets out eleven high level descriptors of Good Environmental Status. Descriptor number 11 states that *...Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*. Amendments to the Habitats Regulations for

Box 3.6: Theoretical zones of noise influence

In terms of the theoretical zones of noise influence (see Figure 3.121), the zone of audibility is the largest and the one leading to the death of the receiver, the smallest. This model has been used very often as a first guidance in impact assessments where the zones of noise influences are determined based on sound propagation modelling or sound pressure level measurements on the one hand, and information on the hearing capabilities of the species in question on the other. As sound spreads in principle omnidirectionally from the source, the zones of noise influences, given as distance from the source, indicate a radius rather than a straight line. It should be noted here that this model gives only a very rough estimate of the zones of influence as sound in the seas is always three-dimensional. The interference, reflection and refraction patterns within sound propagation will inevitably lead to much more complex sound fields than those based on the model by Richardson et al. (1995). This complexity may lead to effects such as increases of received sound energy with distance, especially when multiple sound sources are used simultaneously (i.e. seismic surveys).

Figure 3.121 Theoretical zones of noise influence (after Richardson et al. 1995). © Crown copyright 2010: permission granted by Cefas.

England and Wales and the new Offshore Marine Regulations came into force on 21 August 2007. Both Regulations revised the definition of disturbance of European Protected Species (EPS). These are species listed on Annex IV to the EU Habitats Directive and include all species of cetacean and turtle occurring in European waters, as well as the Atlantic sturgeon. It is now an offence to deliberately disturb wild animals of EPS status in such a way as to be likely significantly to affect i) the ability of any significant group of animals of that species to survive, breed, or rear or nurture their young; or ii) the local distribution or abundance of that species (see also Section 3.8.3.2 and www.jncc.gov.uk).

The EU Environmental Impact Assessment Directive 85/337/EEC (as amended by 97/11/EC) requires Member States to adopt all measures necessary to ensure that, before consent is given, projects likely to have significant effects on the environment by virtue of their nature, size or location are made subject to a requirement for development consent and an assessment with regard to their effects. In line with the requirements of the Directive an environmental impact assessment (EIA) is to be carried out in support of applications to develop certain types of project as listed in the Directive at Annexes I and II, such as for example marine dredging and offshore wind farms. It is then left to the discretion of regulators in each country to amend EIAs to include underwater sound assessments. In some countries, including the UK, regulators ask for a specific assessment of sound-related disturbance for offshore wind farms.

3.8.3 Developments since *Charting Progress*

3.8.3.1 Developments in key pressures, understanding and management measures

Underwater noise was not specifically addressed in *Charting Progress*, although it was mentioned as an issue that needed to be considered in the future. Since the publication of *Charting Progress* in 2005, scientific studies and reviews on the potential impact of underwater sound on marine life have been further advanced, particularly fuelled by research on the impacts of sonar on cetaceans (see Cox et al., 2006) and investigation on the effects of construction activities on marine mammals and fish (Hastings and Popper, 2005; Madsen et al., 2006; Thomsen et al., 2006b). In the UK, the Inter-Agency Committee on Marine Science and Technology (IACMST) published a report on underwater sound and marine life that synthesised the current information and also outlined priorities for future research on sound related impacts (IACMST, 2006).

For marine mammals, some progress has been made in developing noise exposure criteria, that is, sound levels that should not be exceeded to avoid temporary threshold shift (TTS) and injury. Based on a comprehensive review of studies undertaken until 2006, a group of US scientists proposed a number of noise exposure criteria for marine mammals that will ultimately guide regulators, perhaps also those in other countries in the coming years (Southall et al., 2007). The proposed injury criteria are composed both of unweighted peak pressures and M-weighted sound exposure levels which are an expression for the total energy of a sound wave. The M-weighted function also takes the known or derived species-specific audiogram into account. For three functional hearing categories of cetaceans, proposed injury criteria are an unweighted 230 dB re 1μPa peak to peak for all types of sounds and an M-weighted sound exposure level of 198 or 215 dB re 1 μPa^2 •s for pulsed and non-pulsed sounds. For pinnipeds the respective criteria are 218 dB 1μPa peak to peak and 186 or 203 re 1 μPa^2 •s (M-weighted). It should be noted that these values are considered controversial within the scientific community as they are based on very limited data sets with respect to noise-induced injury in marine mammals.

There are similar attempts to develop criteria for fish (Popper et al., 2006) but these assessments are not yet finalised. (Popper et al. 2006 define a double exposure criteria of 208 db re 1 μPa peak and 198 dB re 1μPa SEL. However, they note that these values are very preliminary.)

In the UK, some measures are currently employed to mitigate exposure of marine life during noisy activities. These are employed singly or in combination and are often required as part of operating guidelines or as a condition of a licence/permit. Management measures are in place for the construction of offshore wind farms (EIAs, restriction of pile driving activities to outside spawning periods, soft-start procedures). Seismic surveys are regulated by the Department for Energy and Climate Change, and effects of underwater sound are taken into account in regulatory advice, leading in some cases to seasonal and/or geographical restrictions. Since 2005, the Joint Nature Conservation Committee (JNCC) has further updated its guidelines for minimising acoustic disturbance to marine mammals from seismic surveys (www.jncc.gov.uk/marine; JNCC, 2004). The guidelines provide advice on the planning stage, on the application of a soft start procedure during surveys (during a soft-start procedure, the energy of the emitted

sound is slowly increased), the use of marine mammal observers and the application of passive acoustic monitoring techniques (PAM) as a mitigation measure. The UK Royal Navy updated and applies a variety of mitigation measures to reduce potential effects from mid-frequency active sonar on cetaceans. These involve a risk assessment during planning stages, monitoring during operations (visual and passive acoustic monitoring), and specific actions to mitigate against adverse effects (cessation or modification of transmission, reducing power, modification of source levels and application of mitigation zones).

The JNCC has recently produced a guidance document on the deliberate disturbance of marine European Protected Species (EPS) for English and Welsh territorial waters and the UK offshore marine area. This applies the amendments to the Habitats Regulations for England and Wales and the new Offshore Marine Regulations (see Section 3.8.2.4). The first few sections introduce the offence, provide some of the background to JNCC's advice on the interpretation of the new legislation, and present guidance on how to assess the likelihood of disturbing marine EPS. The next sections focus on species- and activity-specific guidance. Guidance is provided per species on what would constitute a 'significant group' and what species would be most sensitive to the effects of disturbance on their local distribution and abundance in UK waters. The activities section provides information on known impacts, highlights active areas of research and makes links to existing good practice guidelines or those under development, in UK waters. Good practice guidelines on seismic surveys, pile driving and explosive use are included as annexes. The document is currently being finalised after a period of public consultation which ended in June 2008 (see http://www.jncc.gov.uk/page-4227; JNCC, 2004).

3.8.3.2 Relevant research/activities

On an international level, including work undertaken in the UK, relevant research activities addressing the effects of underwater have been numerous and results will be addressed in Section 3.8.4.4. This section will concern important conceptual developments. Most notably, the Marine Board of the European Science Foundation produced a paper on the effects of anthropogenic sound on marine mammals that set out to i) define a strategic framework for future assessments, ii) provide guidance about prioritisation of research, and iii) suggest a process of implementation for future research on the effects of sound on marine mammals (Boyd et al., 2008). The paper lists sources of sound and hazards to marine mammals (see Table 3.63). It also sets out a risk assessment framework that can be applied to various activities and which is composed of several steps that build on one another (hazard identification, dose-response assessment, exposure assessment, risk characterization). This particular framework might become very important if applied in a regulatory context, for example in setting up an EIA framework for the assessment of the effects of pile driving sound on harbour porpoises and harbour seals.

Investigating whether and how underwater sound affects populations is crucial from a conservation perspective (for a review see Thomsen et al. 2008). Consequently, research has focussed on whether anthropogenic sound has a significant effect on populations. This is crucial in assessing the impacts of sound in relation or addition to other stressors, either to assess cumulative impacts and/or to focus

Table 3.63 Types of anthropogenic sound sources that could affect marine mammals (Boyd et al. 2008).

Source	*Effects of greatest concern*
Vessels	Masking Habitat displacement
Air guns	Masking Physical trauma Hearing loss Behavioural change Habitat displacement Behaviourally-mediated effects
Intense low- or mid-frequency sonar	Physical trauma Hearing loss Behavioural change Behaviourally-mediated effects
Pile driving	Physical trauma Hearing loss Behavioural change Behaviourally-mediated effects
Other sonars (depth finders, fish finders)	Masking Hearing loss Behavioural change Behaviourally-mediated effects
Dredges	Behavioural change Behaviourally-mediated effects Habitat displacement
Drills	Hearing loss Behavioural change Behaviourally-mediated effects
Bottom-towed fishing gear	Behavioural change Behaviourally-mediated effects Habitat displacement
Explosions	Physical trauma Hearing loss Behavioural change Behaviourally-mediated effects
Recreational vessels	Masking Behavioural change Behaviourally-mediated effects
Acoustic deterrents	Behaviourally-mediated effects
Over flying aircraft (including sonic booms)	Behaviourally-mediated effects

protection efforts. Competition within and across species, fisheries, natural predation, contamination, diseases, reduction in prey availability or geographical shifts of vital prey species, and chemical pollution can all impact marine life (reviews for marine mammals in Perrin et al. 2002). For example, Read et al. (2006) roughly estimated that worldwide fisheries kill several hundreds of thousands of cetaceans as by-catch every year. It is therefore evident, that potential impacts of sound must be considered on a wider perspective, addressing the consequences of acoustic disturbance on populations (for a recent review see Thomsen et al. 2008).

For marine mammals, NRC (2005) developed a population consequence of acoustic disturbance model (PCAD model, Box 3.7). The model involves different steps from sound source characteristics through behavioural change, life functions impacted, and effects on vital rates, to population consequences. Most of the functions of the model are currently unknown, but attempts to address noise related disturbances in a wider context will become more important in the future.

3.8.4 Presentation of the evidence

3.8.4.1 Monitoring period

There is no specific monitoring of underwater sound in UK waters to date. Any relevant literature data to date were included.

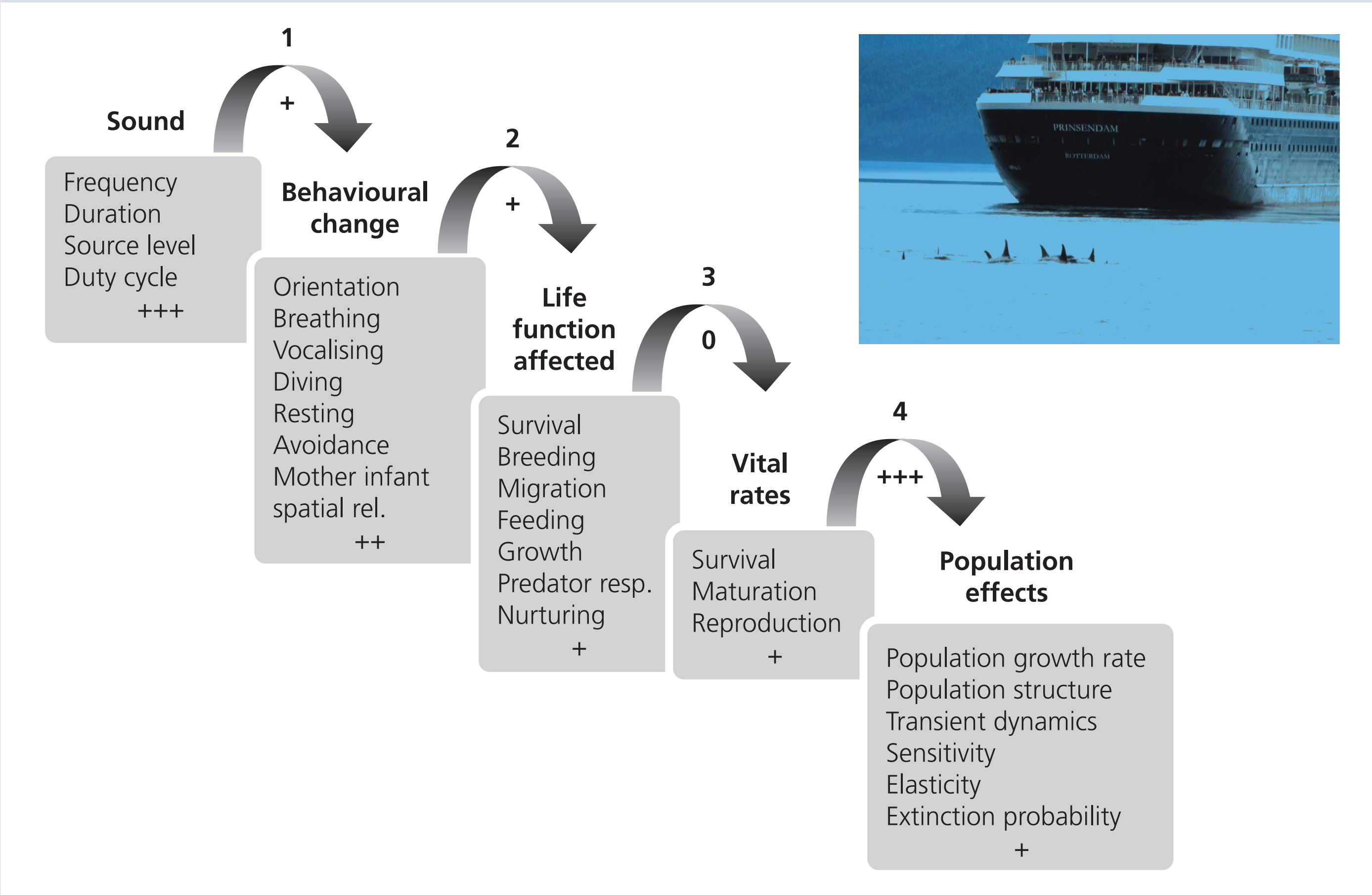
3.8.4.2 Data sources

Data for this overview were derived from peer-reviewed sources and accessible reports. In particular, papers were used that were incorporated in the Overview of the Impacts of Anthropogenic Underwater Sound in the Marine Environment for OSPAR (OSPAR, 2009b) Another

Box 3.7: Population Consequences of Acoustic Disturbance (PCAD) model

The conceptual Population Consequences of Acoustic Disturbance model (Figure 3.122) describes several stages required to relate acoustic disturbance to effects on a marine mammal population. Five groups of variables are of interest, and transfer functions specify the relationships between the variables listed. For example, how sounds of a given frequency affect the vocalization rate of a given species of marine mammal under specified conditions. Each box lists variables with observable features, such as sound, behavioural change, life function affected, vital rates, and population effect. In most cases, the causal mechanisms of responses are not known. For example, survival is included as one of the life functions that could be affected to account for such situations as the beaked whale stranding, in which it is generally agreed that exposure to sound leads to death. The causal steps between reception of sound and death are by no means known or agreed on, but the result is clear. The '+' signs at the bottoms of the boxes indicate how well the variables can be measured. The indicators between boxes show how well the 'black box' nature of the transfer functions is understood; these indicators scale from '+++' (well known and easily observed) to '0' (unknown).

Figure 3.122 Population Consequences of Acoustic Disturbance (PCAD) model. © Crown copyright 2010: permission granted by Cefas.



focus was placed on the results of the various Strategic Environmental Impact Assessments that have been undertaken off the UK, starting in 2001 (see www.offshore-sea.uk.org and Harland et al., 2005, Harland and Richards, 2006; Richards et al., 2007).

3.8.4.3 Adequacy and confidence of the data

Many studies on sound pressure levels and investigations into effects have been published in non-refereed sources that are in some cases difficult to evaluate. There is also the issue that researchers often do not state what measurement units were applied in the respective investigation which makes comparisons across studies sometimes difficult (see discussion by Richardson et al., 1995). While these studies have been included in this review their conclusions should be treated with caution (for reviews, see Richardson et al., 1995; Council, 2003; Hastings and Popper, 2005; Nowacek et al., 2007).

3.8.4.4 Trends

3.8.4.4.1 Human activities generating sound: sound profiles

Relevant activities generating underwater sound might be divided into *offshore construction activities* (e.g. pile driving for offshore wind farms and oil and gas platforms), *offshore industrial activities* (e.g. marine aggregate extraction, offshore wind farm operation), *seismic surveys, sonar* (military, research, private), *shipping* (large commercial vessels and small to medium-sized crafts and ships), and *other activities* (*acoustic deterrent* devices in mariculture, wave and tidal generators). Table 3.64 provides an overview of the acoustic properties of these sources.

Explosions are used in construction and occasionally in the removal of unwanted subsea structures. As indicated in Table 3.64, they are one of the strongest point sources of anthropogenic sound in the seas. Pile driving, such as for the construction of harbours (see Hawkins, 2006) and offshore wind turbines, involves short and comparably very loud signals, mostly at lower frequencies below 1 kHz (see Table 3.64). Oil and gas rigs generate sound by conduction of the sound from machinery on the platform into the water column. Data on sound fields around oil and gas rigs are extremely sparse but some measurements indicate that mostly lower frequencies (< 1 kHz) are emitted. Aggregate dredging activities can be quite loud, sound being generated by the operation of the dredge and sediment disturbance. Measurements by Defra (2003) indicate that mostly lower frequencies are emitted (Figure 3.123).

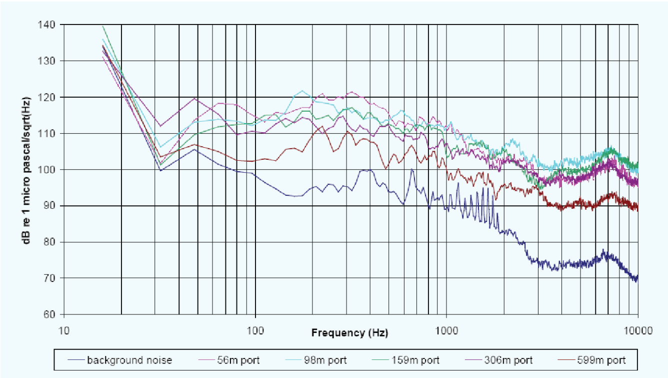
Sound from operational offshore wind farms of the current capacities (2 MW) is relatively low and again most prominent in the lower frequency range (Thomsen et al., 2006b; Nedwell et al., 2008). Large commercial vessels (container/cargo ships, super tankers, cruise liners) produce relatively loud and predominately low frequency sounds. Although the exact characteristics depend on vessel type, size, operational mode and implemented sound reduction measures, the strongest energy tends to be below several hundred Hz with broadband source levels generally in the 180 to 190 dB range (OSPAR, 2009b). Small craft and boats and medium-sized vessels (e.g., recreational craft, jet skis, speed boats, operational work boats, hovercraft, support and supply ships, many research vessels, fishing vessels) produce source levels of approximately 160 to 180 dB re: 1μPa, depending on speed and other operational characteristics (OSPAR, 2009b;

Table 3.64 Overview of the acoustic properties of some anthropogenic sounds.

Sound	*Source level (dB re 1μPa-m)*[a]	*Bandwidth (Hz)*	*Major amplitude (Hz)*	*Duration (ms)*	*Directionality*	*Source*
Offshore construction activities						
TNT (1–100 lbs)	272 – 287 Peak	2 – 1000	6 – 21	~ 1 – 10	Omnidirectional	Urick, 1983; Richardson et al., 1995
Pile driving	228 Peak / 243 – 257 P-to-P	20 – > 20 000	100 – 500	50	Omnidirectional	OSPAR, 2009b
Offshore industrial activities						
Dredging	168 – 186 rms	30 – > 20 000	100 – 500	Continuous	Omnidirectional	Thomsen et al. 2009
Drilling	145 – 190 rms[b]	10 – 10 000	< 100	Continuous	Omnidirectional	OSPAR, 2009b
Wind turbine	142 rms	16 – 20 000	30 – 200	Continuous	Omnidirectional	Thomsen et al., 2006b
Seismic surveys						
Airgun array	260 – 262 P-to-P	10 – 100 000	10 – 120	30 – 60	Vertically focussed[a]	OSPAR, 2009b; OGP/IAGC 2007; Zimmer, 2004
Sonar						
Military sonar low-frequency	215 Peak	100 – 500		600 – 1000	Horizontally focussed	OSPAR, 2009b; ICES-AGISC 2005; Zimmer, 2004
Military sonar mid-frequency	223 – 235 Peak	2800 – 8200	3500	500 – 2000	Horizontally focussed	OSPAR, 2009b; ICES-AGISC 2005; Zimmer, 2004
Echosounders	235 Peak	Variable	Variable 1500 – 36 000	5 – 10 ms	Vertically focussed	OGP/IAGC 2007
Shipping						
Large vessels	180 – 90 rms	6 – > 30 000	> 200	Continuous	Omnidirectional	OSPAR, 2009b
Small- and medium-sized boats and ships	160 – 180 rms	20 – >10 000	> 1000	Continuous	Omnidirectional	OSPAR, 2009b
Other activities						
Acoustic deterrent / harassment devices	132 – 200 Peak	5000 – 30 000	5000 – 30 000	Variable 15 – 500 ms	Omnidirectional	OSPAR, 2009b
Tidal and wave energy devices[c]	165 – 175 rmsc	10 – 50 000	-	Continuous	Omnidirectional	OSPAR, 2009b

[a] Nominal source; [b] higher source levels from drill ships use of bow thrusters; [c] projection based on literature data levels back-calculated at 1m.

Figure 3.123 Spectrum of dredging sound recorded at various distances from the source in relation to ambient (background) noise levels. Source: Defra (2003). © Crown copyright 2010.



see Table 3.64). Seismic surveys generate very loud airgun pulses with predominately low frequencies that are directed downward into the seabed. Yet, a considerable amount of acoustic energy, especially at the higher frequencies, is also emitted horizontally (reviewed by OSPAR, 2009b). Sonar comprises echsounders, fish-finding sonars and military sonar. Echosounders are in place in most vessels, from small leisure craft to the largest commercial ships. They work in a wide range of frequencies (frequency band of highest amplitude = 1.5 to 36 kHz) with high source levels of more than 200 dB re: 1 μPa (see Table 3.64). The energy is directed downward into the seabed, but a proportion is also emitted horizontally. Military sonars generate sounds either in the low frequency range for long-range surveillance or in the mid-frequency range to find and track underwater targets (ICES-AGISC, 2005). Source levels are considerable (Table 3.64). These include acoustic deterrent devices used to scare seals away from fish farms (source levels see Table 3.64), and wave and tidal energy generators. Available information on the acoustic signatures of the latter activities is limited. However, tidal turbines appear to emit broadband sound covering the frequency range 10 Hz to 50 kHz with significant narrow band peaks in the spectrum. Depending on size, it is likely that tidal current turbines will produce broadband source levels of between 165 and 175 dB re: 1μ Pa (OSPAR, 2009b).

3.8.4.4.2 Documented effects of sound generating activities

Owing to the lack of studies, documented effects of underwater sound on marine life will be addressed on a wider level, including those studies undertaken in the UK.

Offshore construction and industrial activities

Modelling studies indicate that pile driving sound might be audible to marine mammals over very large distances of more than 80 km, under certain circumstances (David, 2006; Madsen et al., 2006; Thomsen et al., 2006b; Nedwell et al., 2008). Studies during the construction of Danish offshore wind farms at Horns Reef (North Sea) and Nystedt (Baltic) indicated long-range (~15 to 20 km) changes in behaviour and avoidance in harbour porpoises, which was of short duration at one site (Horns Reef) and perhaps persistent at the other (Nystedt; Tougaard et al., 2003a,b, 2005; Carstensen et al., 2006). Yet, there were methodological pitfalls in both studies, and the results should be interpreted with caution[3]. Little is known about the effects of dredging on marine mammals and the evidence for

3 For example, there was no documentation of received sound pressure levels. Further, absolute abundance of porpoises near Nysted was low from the start, and hence findings of lower numbers at some point in time might be incidental (Tougaard et al., 2005). Another note of caution applies to the use of pingers and seal-scarers before ramming to deter porpoises and seals from the vicinity of the construction sites (Tougaard et al., 2003a, 2005). In particular, the seal scarers might have caused avoidance response in porpoises at some distances, since the source levels used were reportedly rather high with carrier frequencies well within the hearing range of porpoises (SL = approx. 189 dB peak to peak re: 1 μPa broadband; carrier frequencies of 13 to 15 kHz; Lofitech, Norway, pers. comm.). Since harbour porpoises have very acute hearing in that frequency range, it cannot be ruled out that effects were caused by a combination of the mitigation measures employed, along with the pile driving.

behavioural reactions in baleen is circumstantial (see Richardson et al., 1995; Nowacek et al., 2007). Playbacks of operational wind farm sound through an underwater speaker indicated short-range responses in harbour porpoises (Koschinski et al., 2003), yet some of the methods used were criticised (Madsen et al., 2006). Injuries and death due to pile driving and explosions are documented in a variety of fish species. However, behavioural effects in fish due to construction activity have not been investigated fully (reviews in Hastings and Popper, 2005; Thomsen et al., 2006a; OSPAR, 2009b).

Sonar

The full effects of sonar on cetaceans are not well known, mostly due to the difficulty of studying the interaction, and to a lesser extent because the details of sonar equipment and usage are not easy to obtain. The use of high-intensity mid-frequency sonar has lead to mortality in cetaceans in some places (e.g. Bahamas, Canary Islands). It appears that beaked whales are the most affected species. However strandings are caused probably not by injuring individuals with high sound pressure levels, but by an extreme behavioural response that leads to a decompression like syndrome (ICES-AGISC, 2005; Cox et al., 2006; Nowacek et al., 2007; OSPAR, 2009b). The scientific assessment of the biological impacts of sonar indicates that sonar is not a major current threat to marine mammal populations generally, nor will it ever be likely to form a major part of ocean sound (ICES-AGISC, 2005). ICES-AGISC (2005) also noted that sonar can place individual whales at risk, and has affected the local abundance of beaked whales. This latter point has raised concerns especially from non-governmental organisations such as the National Resource Defence Council (USA) and the Whale and Dolphin Conservation Society (UK; see for example Weilgart, 2007; Parson et al., 2008). There have been few studies on the effects of sonar on fish. Results indicate a fair amount of resistance to sound exposure with little or no tissue damage, hearing loss in the form of temporary threshold shift (TTS) that recovers quickly and some rather anecdotal observations of behavioural reactions (Exposure levels = up to 193 dB re: 1µPa; species: e.g. northern pike (*Esox lucius*), broad whitefish (*Coregonus nasus*), lake chub (*Couesius plumbeus*), herring (*Clupea harengus*), Atlantic cod (*Gadus morhua*); rainbow trout (*Oncorhyncus mykiss*); Jørgensen et al., 2005; ICES-AGISC, 2005; Popper et al., 2005a, 2007; see review in OSPAR, 2009b).

Seismic surveys

Documented effects in marine mammals include TTS at different exposure levels in bottlenose dolphins (*Tursiops truncatus*), white whales (*Delphinapteras leucas*), and harbour porpoises (Finneran et al., 2002; Lucke et al., 2008) and behavioural reactions in a wide variety of species at various distances from the sources (for a UK study see Stone and Tasker, 2006; for reviews, see Richardson et al., 1995; Nowacek et al., 2007; OSPAR, 2009b). Stone and Tasker (2006) analysed 1625 sightings of marine mammals occurring during 201 seismic surveys in UK waters between 1998 and 2000. They found that sighting rates of white-sided dolphins, white-beaked dolphins, a grouping of 'all small odontocetes' and a grouping of 'all cetaceans' were significantly lower during periods of shooting compared with non-shooting periods on surveys with large airgun arrays. However, throughout the course of the surveys, sighting rates were not found to differ significantly, indicating that any behavioural responses were short-term in nature.

Studies on fish show both the presence and absence of injuries due to exposure to seismic sounds (see OSPAR, 2009b; McCauley et al., 2003; Popper et al., 2005b). There was also TTS in three freshwater species which was quickly recoverable (Popper et al., 2005b). Behavioural changes have been found in a variety of investigations, both in experiments (e.g. Hassel et al., 2004) and catch rate studies (e.g. Engås et al., 1996). The extent of seismic-induced mortality for commercial species is estimated to be so low that it is considered not to have significant negative impacts on recruitment to the populations. Yet, based on the few existing studies showing a reduction in catch rates during sound exposure behavioural response is indicated within a radius of several kilometres from the sound source. If fish that are on their way to the spawning grounds are exposed to this type of sound, or if they are exposed to the sound during the actual spawning, the effects can have an impact on the fish's spawning success and thereby the recruitment (reviewed in OSPAR, 2009b).

Shipping

Studies on the effects of shipping sound on marine animals are observational and therefore in many cases difficult to interpret. Studies so far indicate behavioural response (changes in motor behaviour, communicative sounds, use of echolocation signals) in some dolphin and whale species exposed to approaching vessels as well as to sound from different vessels (Janik and Thompson, 1996; Lesage et al., 1999; Au and Green, 2000; Nowacek et al., 2001; van Parijs and Corkeron, 2001; Williams et al., 2002; Hastie et al., 2003; Buckstaff, 2004; Southall, 2005; Foote et al., 2004; Aguilar Soto et al., 2006). However, experiments using acoustic tags and controlled exposure on North Atlantic right whales indicated no response to playbacks of vessel sound, perhaps due to habituation to the noisy environment (Nowacek et al., 2004). One main problem with regard to the effects of shipping sound on marine mammals might be masking, i.e. the interference with echolocation and communication sounds (for recent reviews, see Janik, 2005; OSPAR, 2009b). Prolonged exposure to shipping sound has been shown to induce TTS in fish with recovery being a function of frequency and duration of exposure. Behavioural reactions have been documented in a few cases and the potential for masking is high, as many fish species use sound for communication (OSPAR, 2009b). Another issue is recreational traffic. Given that most recreational activity takes place in shallow coastal waters, where sound attenuates rapidly, cetaceans, for example, may not perceive vessel sounds as potentially threatening. Subsequently, they risk physical damage and there is evidence of boat collision in many stranded UK cetaceans (Evans et al., 2003).

Other activities

There are indications for rather large-scale habitat exclusions of odontocetes (a non-target species) in response to commercially available acoustic harassment devices (AHDs; Morton and Symonds, 2002; Olesiuk et al., 2002), and therefore these devices could be considered a potential concern, provided that no mitigation measures are implemented. Although there is the potential that some pingers could exclude porpoise from their habitat, it should be noted that investigations indicate a rather short range of response of few hundred metres from the device. It has to be considered that the benefits of reducing lethal by-catch may outweigh the costs imposed by behavioural changes in some populations and habitats. Since relatively high frequencies are used, both AHDs and pingers are less likely to impact on fish behaviour, except

perhaps for individuals within the immediate vicinity of the device. Hearing damage may only be caused by some of the high intensity sound sources in species with good hearing sensitivity. Calculations of impact zones vary markedly based on exposure criteria used. Repeated exposure for extended amount of times (e.g. as a result of overlapping sound fields from different devices) may pose a risk for some odontocetes (reviewed in OSPAR, 2009b).

3.8.4.4.3 Scale and level of sound generating activities off the UK

In most cases the scale and level of sound generating activities off the UK are difficult to quantify, not to mention the difficulties in assessing individual sound budgets (see Hildebrand 2005 for a worldwide attempt). The sections that follow thus provide a rather broad description of trends in each activity over the past decade or so.

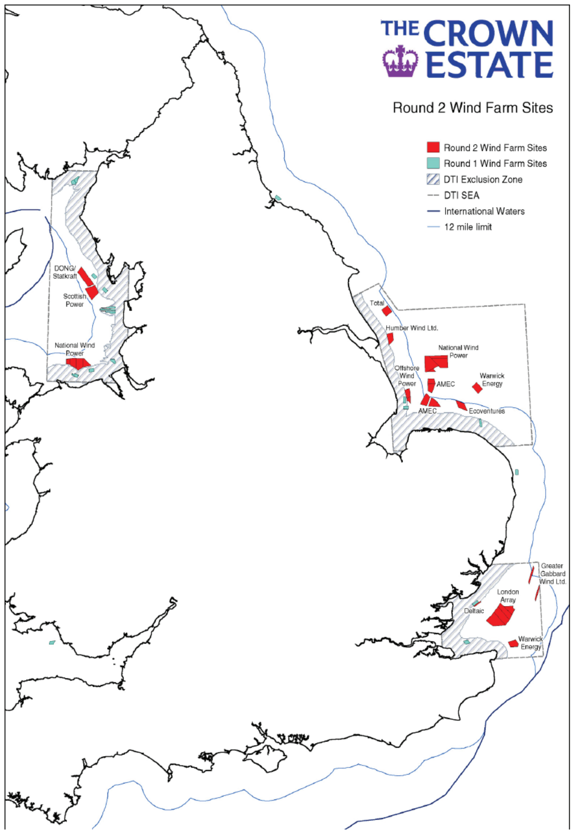
Offshore construction activities

Construction for offshore wind farms is a new activity off the UK which is steadily increasing. By December 2009, there were ten offshore wind farms operating 228 turbines with a total installed capacity of 0.7 GW of electricity (two are small demonstrator projects). A further 1.7 GW of capacity was under construction; 4.9 GW had been consented, and about 9.9 GW was in planning and pre-planning processes across the UK. Most of the offshore wind farms planned or built are in relatively near shore areas (Figure 3.124).

Offshore industrial activities

Offshore industrial activities are carried out to a very large extent along the UK shores with centres off the east coast (oil and gas production, aggregate dredging) and the south coast (aggregate dredging; see Thomsen et al., 2009 for an overview). Production of offshore oil and gas began on the UK continental shelf in 1964 with the first licences granted and the first well drilled in the central North Sea soon after. The first North Sea gas field began production in 1967 (BERR, 2008). Drilling activity has remained high in the North Sea with the number of exploration wells drilled peaking in 1990 at 159. Development drilling activity has remained high in recent years with 201 wells drilled in 2006 after a development drilling peak of 289 wells in 1998 (BERR, 2008). Figure 3.125 shows the principal infrastructure off the UK east coast, including all oil and gas terminals, pipelines and fields, licensed areas, and wind farm sites (BERR, 2008). There are currently 284 oil and gas installations in production. The first platform

Figure 3.124 Overview of offshore wind farm development in UK waters. © The Crown Estate.



installations were predominantly in the Southern North Sea, followed later by increased activity in the Northern North Sea, the Moray Firth and the Irish Sea. The largest increase in platform numbers occurred during the late 1980s, with over 80% of current platforms in production by 1997. There has been an increase in the number of platforms to the west of Shetland from one to three over the past ten years.

Aggregate dredging in Britain is found within distinct regions around the coasts of England and Wales, presumably producing localised areas of dredging sound. There are currently eight regions off the coast of England and Wales licensed for marine aggregate dredging. The amount of material dredged varies considerably across regions. Figure 3.126 shows the cumulative total of material dredged from each region between 1999 and 2007. Relatively large amounts of material were removed from the East, the South and the Humber. The smallest amount extracted was from the Eastern English Channel region.

For the distribution of offshore wind farms along the UK coast, see Figure 3.124.

Shipping

Shipping activity off the UK is difficult to quantify. Some attempts have been made by the Strategic Environmental Impact Assessments commissioned by the Department of Trade and Industry between 2001 and 2004, with some data for SEA 2 to SEA 7 (see www.offshore-sea.org.uk). An overview of all shipping routes from the COAST database – developed in 1995/96 to detail information on shipping traffic – can be seen in Figure 3.127. A detailed analysis of the shipping activity in each of the regional areas for *Charting Progress 2* is beyond the scope of this report. Yet, Figure 3.127 indicates an overall high level of activity within the Southern North

Figure 3.125 Principal infrastructure on the UK continental coast, as of 2008 Source: BERR (2008). Crown Copyright; cropped to show UK east coast structures. © Crown copyright 2010: DECC.

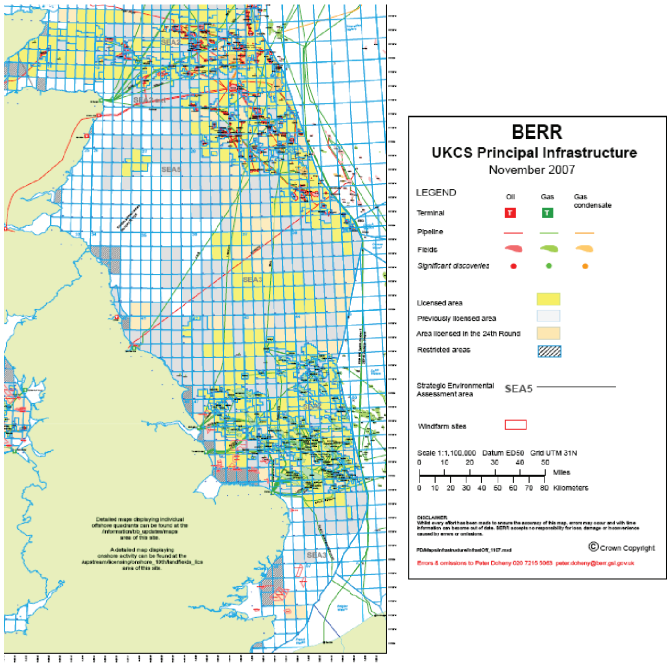
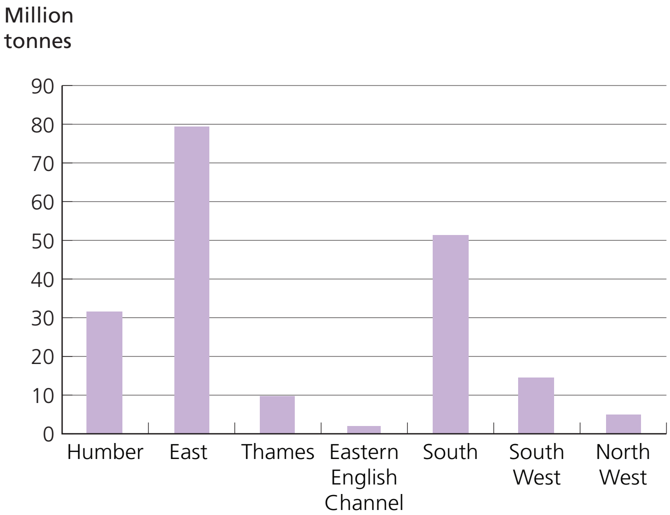


Figure 3.126 Total amounts of aggregate extracted from each region, 1999 – 2007. Source: BMAPA (2000, 2001, 2002, 2003, 2004, 2005, 2006a, b, 2007, 2008); The Crown Estate (2006b,a); taken from Thomsen et al. (2009). © Crown copyright 2010.



Sea and coastal areas (including SEA 2 to SEA 5 experiencing very busy traffic). Regions 1 to 5 have the highest levels of shipping traffic.

Seismic surveys

Seismic surveys have been carried out in the North Sea since 1963, with the majority being 2D line transects. With the developing 3D technology, surveying began in 1978 with high numbers of 3D surveys concentrated in the Southern North Sea and Northern central North Sea. Over the past 10 years, activity has begun moving into areas west of the Shetland Islands and into the Irish Sea. The industry splits areas of the UK coast up into quadrants for reference. The amount of seismic activity within quadrants 11 to 57 (UK East-North and partly South Coast) has varied between 1997 and 2003 (Figure 3.128). The greatest activity was seen in 1997 when 10 705 km and 6441 km^2 of 2D and 3D surveys were made, respectively. Overall, 2000 was the ‘quietest’ year in terms of surveys, with only 210 km and 463 km^2 of 2D and 3D seismic activity, respectively (Thomsen et al., 2008; ASCOBANS, 2005).

Sonar

The usage of military sonar off the UK is scarcely known, not least since the data are in most cases classified. Harland et al. (2005) and Harland and Richards (2006) provided technical information on underwater ambient noise in SEA 6 and SEA 7 and noted that no published information could be identified on the statistics of sonar use (see also Richards et al., 2007). However, ICES-AGISC (2005) noted that most usage of sonar has been confined to comparatively well-defined exercise areas.

Figure 3.127 Major shipping lanes in the North Sea. Source: CORDAH (2001). © CORDAH.

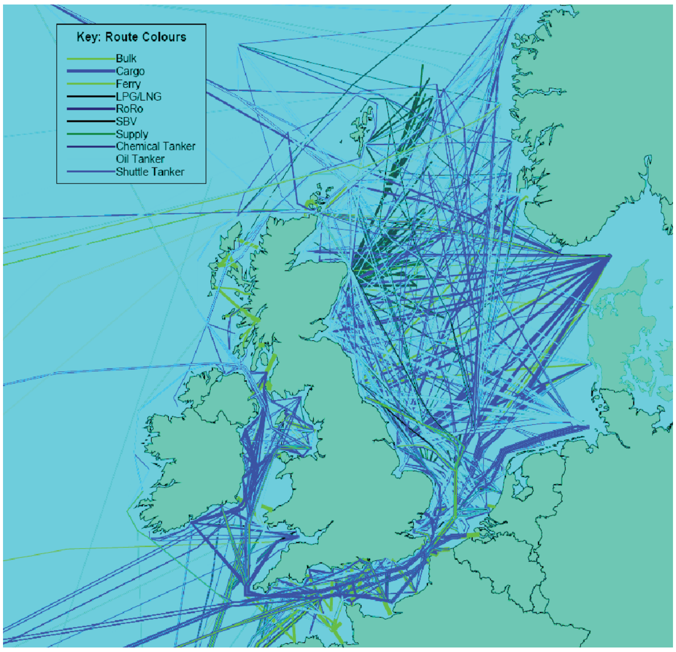
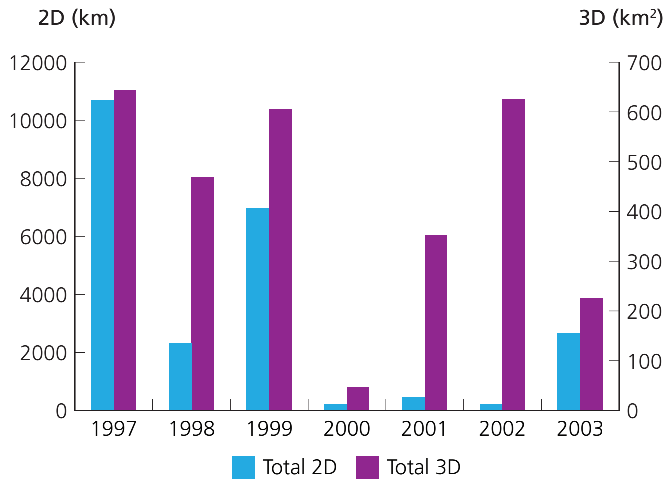


Figure 3.128 Total length of 2D surveys and total area of 3D surveys carried out within Quadrants 11 to 57 of the UK continental shelf between 1997 and 2003. Adapted data from ASCOBANS (2005). © Crown copyright 2010.



3.8.4.5 Gaps in knowledge

Despite some progress, there are considerable gaps in our understanding of the effects of underwater noise on marine life. These address both the sender as well as the receivers:

- There are in many cases not enough data on the level of certain sound generating activities in the UK; it is completely lacking for military sonar, and very sparse for shipping (see Harland et al., 2005; Harland and Richards, 2006)
- These and many other sources are as yet only very insufficiently documented. This applies in particular to activities relevant to UK waters such as marine aggregate dredging, drilling and seismic exploration (in particular the contribution of higher frequencies to the overall sound spectrum)
- There is very little information on ambient noise levels along the UK coast, and it is therefore impossible in many cases to establish a baseline against which potential effects can be assessed (see also IACMST, 2006)

> Ambient noise is usually defined as the background noise from all sources except those close enough to be individually detectable. Richardson et al. (1995), Harland et al. (2005) and Harland and Richards (2006) defined ambient noise as the sound received by an omnidirectional sensor which is not from the sensor itself or the manner in which it is mounted.

- In many cases, the sound pressure levels that an organism receives during an activity are unknown, due to a lack of measurements (Nowacek et al., 2007)
- Behavioural responses due to sound exposure are poorly understood, as there are only a very limited number of controlled exposure studies available (see Southall et al., 2007 for a review)
- Our knowledge of hearing in marine life is very limited as there are only a few audiograms from a limited number of species both for marine mammals and fish. Studies suggest a relatively high level of variation both across and within species (reviews by Hastings and Popper, 2005; Southall et al., 2007)
- Information on how underwater sound affects populations of marine life is completely lacking (see Thomsen et al., 2008 for a recent review).

3.8.4.6 Contribution towards the GES descriptor

The importance of underwater sound has been highlighted in the EU Marine Strategy Framework Directive (MSFD), which includes underwater sound as one of eleven high level descriptors of good environmental status (GES). The relevant GES descriptor states that *...Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment*.

The available information contributes directly to this descriptor as it informs on the current level of sound-generating activities off the UK and outlines research on the effects of underwater sound on marine life. In cases of more forthcoming information, feasible mitigation measures can be applied to reach GES with regards to underwater sound.

3.8.5 What the evidence tells us

3.8.5.1 Assessment criteria

It is currently not possible to derive assessment criteria with regard to underwater noise as there are too many uncertainties concerning how marine life perceives sound and what units are relevant for assessments. There is currently a working group for GES descriptor 11 within ICES working on this issue. The report from this group was published in 2010.

3.8.5.2 Status

It is currently not possible to assess the status of underwater noise in the UK seas based on the information provided.

3.8.5.3 Climate change

Studies by Hester et al. (2008) indicated that ocean acidification resulting from increased uptake of CO_2 as a consequence of higher atmospheric concentrations and reduced ventilation results in significant decreases in ocean sound absorption for frequencies lower than about 10 kHz with a frequency dependent decrease in sound absorption (α = dB/km) exceeding 12%. Under likely projections of future fossil fuel CO_2 emissions and other sources, a pH change of 0.3 units or more can be anticipated by the middle of the 21st century, resulting in a decrease in α by almost 40%. Ambient noise levels in the ocean within the auditory range critical for environmental, military, and economic interests are set to increase significantly due to the combined effects of decreased absorption and increasing sources from human activities.

Studies indicate that ocean acidification resulting from increased uptake of CO_2 as a consequence of higher atmospheric concentrations and reduced ventilation results in significant decreases in ocean sound absorption at frequencies lower than about 10 kHz. Ambient noise levels in the ocean within the auditory range critical for environmental, military, and economic interests are set to increase significantly due to the combined effects of decreased sound absorption and increasing inputs of noise from human activities.

3.8.6 Forward look and need for further work

Looking at the issue of effects of underwater sound in more general terms, Hastings and Popper (2005), IACMST (2006) and Southall et al. (2007) gave a detailed list of recommendations for future research. There are several areas where research on the effects of underwater sound on marine life should be prioritised, including:

- Establishment of standardised protocols for testing the extent to which sources radiate sound in the marine environment. This needs to include a system for depositing data in appropriate formats, to be used in future models predicting ambient noise levels in the seas
- Acoustic measurements of relevant sound sources. Detailed measurements of source levels, frequency content, and radiated sound field around intense and/or chronic sound sources
- Ambient noise measurements. Systematic measurements of underwater ambient noise to quantify how human activities are affecting the acoustic environment
- Hearing measurements. Much more autographic data (sensitivity of individuals vs. frequency of sound) is needed both for marine fish and marine mammals to assess impacts

- Marine life behavioural responses to sound exposure. Measurements of behavioural reactions to various underwater sound types are urgently needed to establish more robust cause-effect relationships. In this regard controlled exposure experimental playbacks (CEEP) should be undertaken
- Effects of sound exposure on marine life hearing (i.e. masking, PTS and TTS). It is necessary to further investigate the physiological effects of underwater sound. This can be done using controlled exposure experiments
- The applicability of existing regulations and treaties for the protection of the marine environment specifically to cover underwater sound should be investigated. Where necessary, amendments should be proposed
- Building of a modern regulatory, risk-based framework relating to sound in the marine environment. This should be based on existing legislation and the application of the precautionary principle. Its purpose should be to provide agreed impact/harm criteria, eliminate confusion over terminology, and enable more consistent mitigation measures.

Abbreviations

afbini	Agri-Food and Biosciences Institute, Northern Ireland
Ag	Silver
AGR	Advanced gas-cooled nuclear reactor
AHAs	Associated harvesting areas (for shellfish under the biotoxins programme)
Al	Aluminium
AOAC	Association of Official Analytical Chemists
As	Arsenic
ASP	Amnesic Shellfish Poisoning
BAC	Background Assessment Concentration
BDE	Brominated diphenyl ether congener
BFR	Brominated flame retardant
Bq	Becquerel
BRC	Background Reference Concentration (OSPAR)
CB	Chlorobiphenyl congener
Cd	Cadmium
Cefas	Centre for Environment, Fisheries and Aquaculture Science
CEMP	Co-ordinated Environmental Monitoring Programme (OSPAR)
CSSEG	Clean and Safe Seas Evidence Group
CSEMP	Clean Seas Environmental Monitoring Programme (UK)
Cu	Copper
DECC	Department of Energy and Climate Change
Defra	Department for Environment, Food and Rural Affairs
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DRZ	Diagonal radioactive zones
DSD	Dangerous Substances Directive (EU)
DSP	Diarrhetic Shellfish Poisoning
DTA	Direct Toxicity Assessment Programme (Environment Agency)
dw	Dry weight
EA	Environment Agency
EAC	Environmental Assessment Concentration
EARP	Enhanced Actinide Removal Plant (at Sellafield)
EcoQO	Ecological Quality Objective
EDC	Endocrine-disrupting compound
EQS	Environmental Quality Standard
ERL	Effects Range – Low
ERM	Effects Range – Median
EROD	Ethoxyresorufin-*o*-deethylase (an enzyme)
GES	Good Environmental Status (under the EU MSFD)
HBCD	Hexabromocyclododecane
HBDSEG	Healthy and Biologically Diverse Seas Evidence Group
HD	Habitats Directive (EU)
Hg	Mercury

ICES	International Council for the Exploration of the Sea
IMO	International Maritime Organization
JAMP	Joint Assessment and Monitoring Programme (OSPAR)
JNCC	Joint Nature Conservation Committee
K_d	Vertical extinction coefficient for irradiance
LOD	Limit of detection
LoQ	Limit of quantification
lw	Lipid weight
LWR	Light water nuclear reactor
MBA	Mouse bioassay
MCS	Marine Conservation Society
MSFD	Marine Strategy Framework Directive (EU)
ND	Nitrates Directive (EU)
Ni	Nickel
NIEA	Northern Ireland Environment Agency
NMMP	UK National Marine Monitoring Programme (forerunner to CSEMP)
NVZ	Nitrate Vulnerable Zone (under the EU Nitrates Directive)
OSPAR	Convention for the Protection of the Marine Environment of the North-east Atlantic
PAHs	Polycyclic aromatic hydrocarbons
Pb	Lead
PBDE	Polybrominated diphenyl ether
PCBs	Polychlorinated biphenyls
PSEG	Productive Seas Evidence Group
PSP	Paralytic Shellfish Poisoning
PTS	Permanent threshold shift (in hearing loss)
RID	Riverine Inputs and Direct Discharges programme (OSPAR)
RMPs	Representative monitoring points (for biotoxins analysis)
SEPA	Scottish Environment Protection Agency
SRD	Sewage related debris
STX	Saxitoxin
SWD	Shellfish Waters Directive (EU)
TBT	Tributyltin
THORP	Thermal Oxide Reprocessing Plant (at Sellafield)
TNORM	Technologically-enhanced naturally occurring radioactive material
TTS	Temporary threshold shift (in hearing loss)
UKMMAS	UK Marine Monitoring and Assessment Strategy
UWWTD	Urban Wastewater Treatment Directive (EU)
VDSI	Vaf Deferenf Sequence Index
VTG	Vitellogenin
WFD	Water Framework Directive (EU)
ww	Wet weight
Zn	Zinc

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References

Aarkrog, A., Boelskifte, S., Dahlgaard, H., Duniec, S., Hallstadius, L., Holm, E. and Smith, J.N. 1987. Technetium-99 and caesium-134 as long distance tracers in Arctic waters. Estuar. Coast. Shelf Sci. 24, 637-647.

Aas, E., Baussant, T., Balk, L., Liewenborg, B. and Andersen, O.K., 2000a. PAH metabolites in bile, cytochrome P4501A and DNA adducts as environmental risk parameters for chronic oil exposure: a laboratory experiment with Atlantic cod. Aquat. Toxicol. 51, 241-258.

Aas, E., Beyer, J. and Goksøyr, A., 2000b. Fixed wavelength fluorescence (FF) of bile as a monitoring tool for polyaromatic hydrocarbon exposure in fish: An evaluation of compound specificity, inner filter effect and signal interpretation. Biomarkers 5, 9-23.

Aas, E., Liewenborg, B., Grøsvik, B.E., Campus, L., Jonsson, G., Børseth, J.F. and Balk, L., 2003. DNA adduct levels in fish from pristine areas are not detectable or low when analysed using the nuclease P1 version of the ^{32}P-postlabelling technique. Biomarkers 8, 445-460.

ACOPS, 2008. Annual survey of reported discharges attributed to vessels and offshore oil & gas installations operating in the United Kingdom pollution control zone 2007. Advisory Committee on the Protection of the Sea, London, March 2008.

Aguilar Soto, N., Johnson, M., Madsen, P., Bocconcelli, A., Tyack, P. and Borsani, F. 2006. Does shipping noise affect the foraging behaviour of Cuvier's beaked whale (*Ziphius cavisrostris*)? Mar. Mammal Sci. 22, 690-699.

Akcha, F., Leday, G. and Pfohl-Leszkowicz, A., 2004. Measurement of DNA adducts and strand breaks in dab (*Limanda limanda*) collected in the field: effects of biotic (age, sex) and abiotic (sampling site and period) factors on the extent of DNA damage. Mutat. Res. 552, 197-207.

Allchin, C.R., Law, R.J. and Morris, S., 1999. Polybrominated diphenylethers in sediments and biota downstream of potential sources in the UK. Environ. Pollut. 105, 197-207.

Allott, R., 2005. Assessment of compliance with the public dose limit – principles for the assessment of total retrospective doses. National Dose Assessments Working Group NDAWG/2/2005. Environment Agency, Food Standards Agency, HPA, NII, Chilton.

Allott, R. and Copplestone, D., 2008a. Update on habitats assessments for England and Wales. NDAWG Paper 13-04. Environment Agency, Food Standards Agency, HPA, NII, Chilton.

Allott, R. and Copplestone, D., 2008b. Impact of radioactive substances on Ribble and Alt estuarine habitats. Technical Report NMA/TR/2008/02. Environment Agency, Bristol and London.

AOAC, 1990. AOAC Official Methods of Analysis. pp. 881-882. AOAC International.

AOAC, 2005. Official method 2005.06 paralytic shellfish poisoning toxins in shellfish. Prechromatographic oxidation and liquid chromatography with fluorescence detection, First action 2005. AOAC International.

Ariese, F., Beyer, J., Jonsson, G., Visa, C.P. and Krahn, M.M., 2005. Review of analytical methods for determining metabolites of polycyclic aromatic compounds (PACs) in fish bile. ICES Tech. Mar. Environ. Sci. 39, 1-41.

ASCOBANS, 2005. Report on information on seismic survey activities by the United Kingdom 1997-2003 ASCOBANS 12th Advisory Committee Meeting, Brest, France, 12 - 14 April 2005.

Au WWL and Green, M., 2000. Acoustic interaction of humpback whales and whale-watching boats. Mar. Environ. Res. 49, 469-481.

Ayres, P.A., 1975. Mussel poisoning in Britain with special reference to paralytic shellfish poisoning. Environ. Health 83, 261-265.

Ayres, P.A. and Cullum, M., 1978. Paralytic shellfish poisoning: An account of investigations into mussel toxicity in England 1968-77. Fisheries Research Technical Report No. 40, MAFF Directorate of Fisheries Research, Lowestoft.

Balk, L., Liewenborg, B., Larsen, B.K., Aas, E., Hylland, K. and Sanni, S., 2006. Large hydrophobic adducts and strand breaks analysed in hepatic DNA from Atlantic cod (*Gadus morhua*) caged at the Statfjord oilfield. In: Biological Effects of Contaminants in Pelagic Ecosystems. Edited by Hylland, K., Lang, T., Thain, J., Vethaak, A.D. and Wosniok, W. Society of Environmental Toxicology and Chemistry (SETAC), pp. 277-295.

Balls, P.W., Hull, S., Miller, B.S., Pirie, J.M. and Proctor, W., 1997. Trace metals in Scottish estuarine and coastal sediments. Mar. Pollut. Bull. 34, 42-50.

Bann, R.A., Steenwinkel, M.S.T., Van den Berg, P.T.M., Roggeband, P.T.M. and Van Delf, J.H.M., 1994. Molecular dosimetry of DNA damage induced by polycyclic aromatic hydrocarbons; relevance for exposure monitoring and risk assessment. Human Exposure Toxicol. 13, 880-887.

Barnes, D., 2002. Biodiversity: Invasion by marine life on plastic debris. Nature 416, 808.

Barnes, D. and Milner, P., 2005. Drifting plastic and its consequences for sessile organism dispersal in the Atlantic Ocean. Mar. Biol. 146, 815-825.

Barucca, M., Canapa, A., Olmo, E. and Regoli, F., 2006. Analysis of vitellogenin gene induction as a valuable biomarker of estrogenic exposure in various Mediterranean fish species. Environ. Res. 101, 68-73.

Baumann, P.C., 1998. Epizootics of cancer in fish associated with genotoxins in sediment and water. Mutat. Res. 411, 227-233.

Beresford, N., Brown, J., Copplestone, D., Garnier-Laplace, J., Howard, B., Larsson, C.-M., Oughton, D., Pröhl, G. and Zinger, I., 2007. D ERICA: an INTEGRATED APPROACH to the assessment and management of environmental risks from ionising radiation. Description of purpose, methodology and application. A Deliverable Report for the Project "ERICA" (Contract No. FI6R-CT-2004-508847) within the EC's VIth Framework Programme. Swedish Radiation Protection Authority, Stockholm, pp. 82.

BERR, 2008. Department for Business, Enterprise and Regulatory Reform. http://www.og.dti.gov.uk/

Best, M.A., Wither, A.W. and Coates, S., 2007. Dissolved oxygen as a physico-chemical supporting element in the Water Framework Directive. Mar. Pollut. Bull. 55, 5364.

Bhavsar, S.P., Hayton, A., Reiner, E.J. and Jackson, D.A., 2007. Estimating Dioxin-like polychlorinated biphenyl toxic equivalents from total polychlorinated biphenyl measurements in fish. Environ. Toxicol. Chem. 26, 1622-1628.

Bignell, J.P., Dodge, M.J., Feist, S.W., Lyons, B., Martin, P.D., Taylor, N.G.H.A., Stone, D., Trevalent, L. and Stentiford, G.D., 2008. Mussel histopathology: effects of season, disease and species. Aquatic Biology, 2: 1-15.

BMAPA, 2000.The area involved - 2nd annual report. British Marine Aggregate Producers Association.

BMAPA, 2001. The area involved - 3rd annual report. British Marine Aggregate Producers Association.

BMAPA, 2002. The area involved - 4th annual report. British Marine Aggregate Producers Association.

BMAPA, 2003. The area involved - 5th annual report. British Marine Aggregate Producers Association.

BMAPA, 2004. The area involved - 6th annual report. British Marine Aggregate Producers Association.

BMAPA, 2005. The area involved - 7th annual report. British Marine Aggregate Producers Association.

BMAPA, 2006a. Aggregates from the sea: drawing strength from the depths. British Marine Aggregate Producers Association.

BMAPA, 2006b. The area involved - 8th annual report. British Marine Aggregate Producers Association.

BMAPA, 2007. The area involved - 9th annual report. British Marine Aggregate Producers Association.

BMAPA, 2008. The area involved - 10th annual report. British Marine Aggregate Producers Association.

Borja, A., Muxika, I. and Franco, J., 2003. The application of a marine biotic index to different impact sources affecting soft-bottom benthic communities along European coasts. Mar. Pollut. Bull. 46, 835-845.

Borja, A., Josefson, A.B., Miles, A., Muxika, I., Olsgard, F., Philips, G., Rodriguez, J.G. and Rygg, B., 2007. An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Frame Work Directive. Mar. Pollut. Bull. 55, 42-52.

Boyd, I., Brownell, B., Cato, D., Clarke, C., Costa, D., Eveans, P.G.H., Gedamke, J., Genrty, R., Gisiner, B., Gordon, J., Jepson, P., Miller, P., Rendell, L., Tasker, M., Tyack, P., Vos, E., Whitehead, H., Wartzok, D. and Zimmer, W., 2008. The effects of anthropogenic sound on marine mammals – A draft research strategy. European Science Foundation and Marine Board.

Brown, J.E., Iosjpe, M., Kolstad, A.K., Lind, B., Rudjord, A.L. and Strand, P. 2002. Temporal trends for technetium-99 in Norwegian coastal environments and spatial distribution in the Barents Sea. J. Environ. Radioactiv. 6, 49-60.

Bucke, D., Vethaak, A.D., Lang, T. and Mellergaard, S., 1996. Common diseases and parasites of fish in the North Atlantic: Training guide for identification. ICES TIMES 19. 27pp.

Buckstaff, C., 2004. Effects of boats on dolphin vocal behaviour. Mar. Mammal Sci. 20, 709-725.

Bull, J.C., Jepson, P.D., Ssuna, R.K., Deaville, R., Allchin, C.R., Law, R.J. and Fenton, A., 2006. The relationship between polychlorinated biphenyls in blubber and levels of nematode infestations in harbour porpoises, *Phocoena phocoena*. Parasitology 132, 565-573.

Carstensen, J., Henriksen, O.D. and Teilmann, J., 2006. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). Mar. Ecol. Prog. Ser. 321, 295-308.

Casper, B.M., Lobel, P.S. and Yan, H.Y., 2003. The hearing sensitivity of the little skate, *Raja erinacea*: a comparison of two methods. Environ. Biol. Fishes 68, 371-379.

CEC, 1991a. Council Directive of 31 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC). Official Journal of the European Communities L 375 1-8.

CEC, 1991b. Council Directive of 21 May 1991 concerning urban waste water treatment (91/271/EEC). Official Journal of the European Communities L 135 40-52.

CEC, 1992. Council Directive of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora (92/43/EEC). Official Journal of the European Communities L 206 7-50.

CEC, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal of the European Communities L 327 1-73.

Cefas, 2005. Monitoring of the quality of the marine environment, 2002-2003. Science Series Aquatic Environment Monitoring Reports. Centre for Environment, Fisheries and Aquaculture Sciences, Lowestoft. Report no. 57. 64pp.

Cefas, 2006 Monitoring the quality of the marine environment, 2003-2004. Science Series Aquatic Environment and Monitoring Report. Centre for Environment, Fisheries and Aquaculture Sciences, Lowestoft. Report no. 58. 168pp.

Cefas, 2007 Monitoring the quality of the marine environment, 2004-2005. Science Series Aquatic Environment and Monitoring Report. Centre for Environment, Fisheries and Aquaculture Sciences, Lowestoft. Report no. 59. 75pp.

Chapman, C.J. and Hawkins, A.D. 1969. The importance of sound in fish behaviour in relation to capture by trawls. FAO Fish. Rep. 62, 717-729.

Chapman, C.J., and Hawkins, A.D., 1973. A field study of hearing in cod, *Gadus morhua*. J. Comp. Physiol. A 85, 147-167.

Charlesworth, M. and Service, M. 1999. Nutrient Inputs, Trophic Status and Mathematical Modelling of Nitrogen and Phosphorus in the Tidal Lagan and Belfast Lough. Report by Queen's University, Belfast & Department of Agriculture Northern Ireland for EHS.

Charlesworth, M. and Service, M., 2000. An assessment of metal contamination in northern Irish coastal sediments. Biology and the Environment: Proc. Roy. Irish Acad. 100B, 1-12.

Cheshire, A. and Adler, E. et al., 2009. UNEP/IOC Guidelines on Survey and Monitoring of Marine Litter. UNEP Regional Seas Reports and Studies, No. 186; IOC Technical Series No. 83: xii + 120 pp.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser. 210, 223-253.

Council, N.N.R., 2003. Ocean Noise and Marine Mammals. National Academies Press.

Cox, T.M., Ragen, T.J., Read, A.J., Vos, E., Baird, R.W., Balcomb, K., Barlow, J., Caldwell, J., Cranford, T., Crum, L., D'Amico, A., D'Spain, G., Fernández, A., Finneran, J., Gentry, R., Gerth, W., Gulland, F., Hildebrand, J., Houser, D., Hullar,, T., Jepson, P.D., Ketten, D., MacLeod, C.D., Miller, P., Moore, S., Mountain, D., Palka, D., Ponganis, P., Rommel, S., Rowles, T., Taylor, B., Tyack, P., Wartzok, D., Gisiner, R., Mead, J., Lowry, L. and Benner, L., 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetac. Res. Manag. 7, 77-187.

CSTT, 1994. Comprehensive studies for the purposes of Article 6 of DIR 91/271 EEC, the Urban Waste Water Treatment Directive. Published for the Comprehensive Studies Task Team of Group Coordinating Sea Disposal Monitoring by the Forth River Purification Board, Edinburgh.

CSTT, 1997. Comprehensive studies for the purposes of Article 6 & 8.5 of DIR 91/271 EEC, the Urban Waste Water Treatment Directive, second edition. Published for the Comprehensive Studies Task Team of Group Coordinating Sea Disposal Monitoring by the Department of the Environment for Northern Ireland, the Environment Agency, the OAERRE page 40 version of: July 4, 2002 Scottish Environment Protection Agency and the Water Services Association, Edinburgh.

Darnerud, P., Eriksen, G.S., Johannesson, T., Larsen, P.B. and Viluksela, M., 2001. Polybrominated diphenyl ethers: occurrence, dietary exposure and toxicology. Environ. Health Perspect. 109, 49-68.

David, J.A., 2006. Likely sensitivity of bottlenose dolphins to pile-driving noise. Water Environ. J. 48-54.

DECC, 2009. UK Strategy for Radioactive Discharges 2006 – 2030. Department of Energy and Climate Change.

Defra, 2002. UK Strategy for Radioactive Discharges 2001 – 2020. Department for Environment, Food and Rural Affairs.

Defra, 2003. Preliminary investigation of the sensitivity of fish to sound generated by aggregate dredging and marine construction. Project AE0914 Final Report. Department for Environment, Food and Rural Affairs.

Defra, 2006. Code of Practice on Litter and Refuse. Department for Environment, Food and Rural Affairs.

Defra, 2008. UK Strategy for Radioactive Discharges 2006 - 2030 – consultation document. Department for Environment, Food and Rural Affairs.

Desantis, S., Corriero, A., Cirillo, F., Deflorio, M., Brill, R., Griffiths, M., Lopata, A.L., de la Serna, J.M., Bridges, C.R., Kime, D.E. and De Metrio, G., 2005. Immunohistochemical localization of CYP1A, vitellogenin and zona radiata proteins in the liver of swordfish (*Xiphias gladius* L.) taken from the Mediterannean Sea, South Atlantic, South Western Indian and Central North Pacific Oceans. Aquat. Toxicol. 71, 1-12.

Devlin, M.J., Best, M., Coates, D., Bresnan, E., O'Boyle, S., Park, R., Silke, J., Cusack, C. and Skeats, J., 2007. Establishing boundary classes for the classification of UK marine waters using phytoplankton communities. Mar. Pollut. Bull. 55, 91-103.

Devlin, M.J., Barry, J., Mills, D.K., Gowen, R.J., Foden, J., Sivyer, D. and Tett, P., 2008. Relationships between suspended particulate material, light attenuation and Secchi depth in UK marine waters. Estuar. Coast. Shelf Sci. 79, 429-439.

DPAG, 2008. Dounreay Particle Advisory Group – Fourth Report. Scottish Environment Protection Agency.

Drury, D.F. and Wheeler, D.C., 2008. Applications of a Serratia marcescens bacteriophage as a new microbial tracer of aqueous environments. J. Appl. Microbiol. 53, 137-142.

Dudzinski, K.M., Thomas, J.A. and Douaze, E., 2002. Communication. In: Encyclopedia of Marine Mammals. Edited by Perrin, W.F., Wuersig, B., Thewissen, J.G.M. Academic Press. pp. 248-268.

EC, 2005. Commission Regulation (EC) No 208/2005 of 4 February 2005 amending Regulation (EC) No 466/2001 as regards polycyclic aromatic hydrocarbons. Official Journal of the European Union, Brussels.

EC, 2006. Commission Regulation No. 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. OJ L 364/5.

EC, 2008. Directive 2008/56/EC of European Parliament and of the Council. Establishing a framework for community action in the field of marine environmental policy.

Encams, 2003. The Management of Britain's Resort Beaches. August 2003.

Engås, A., Løkkeborg, S., Ona, E. and Soldal, A.V., 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). Can. J. Fish. Aquat. Sci. 53, 2238-2249.

Enger, P.S., 1967. Hearing in herring. Comparative Biochemistry and Physiology 22, 527-538.

Environment Agency, 2001. Investigation of litter problems in the Severn Estuary/Bristol Channel area. R & D Technical Report E1-082/TR.

Environment Agency, 2008. Briefing note – enhanced beach monitoring for radioactive particles near Sellafield site – June 2008 update. Environment Agency.

Environment Agency, Food Standards Agency, Northern Ireland Environment Agency and Scottish Environment Protection Agency, 2008. Radioactivity in Food and the Environment, 2007.

Ericson, G., Lindesjoo, E. and Balk, L., 1998. DNA adducts and histopathological lesions in perch (*Perca fluviatilis*) and northern pike (*Esox lucius*) along a polycyclic aromatic hydrocarbon gradient on the Swedish coastline of the Baltic Sea. Can. J. Fish. Aquat. Sci. 55, 815-824.

Ericson, G., Noaksson, E. and Balk, L., 1999. DNA adduct formation and persistence in liver and extrahepatic tissues of northern pike (*Esox lucius*) following oral exposure to benzo[*a*]pyrene, benzo[*k*]fluoranthene and 7H-dibenzo[*c,g*]carbazole. Mutat. Res. 427, 136-145.

Ericson, G., Skarphedinsdottir, H., Zuanna, L.D. and Svavarsson, J., 2002. DNA adducts as indicators of genotoxic exposure in indigenous and transplanted mussels, *Mytilus edulis* L. from Icelandic coastal sites. Mutat. Res. 516, 91-99.

European Communities, 2005. Commission Regulation (EC) No 2073/2005 of 15 November 2005 on microbiological criteria for foodstuffs. Official Journal of the European Communities, L 338, 22.12.2005, 1-26.

Evans, P.G.H., Anderwald, P. and Baines, M., 2003. Cetacean Status Review. English Nature, Countryside Council for Wales.

Feist, S.W., Lang, T., Stentiford, G.D. and Koehler, A., 2004. Use of liver pathology of the European flatfish dab (*Limanda limanda* L.) and flounder (*Platichthys flesus* L.) for monitoring. ICES Tech. Mar. Environ. Sci. 38: 42pp.

Finneran, J.J., Schlundt, C.E., Dear, R., Carder, D.A. and Ridgway, S.H., 2002. Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic watergun. J. Acoust. Soc. Am. 111, 2929-2940.

Fleisher, J.M., Kay, D., Salmon, R.L., Jones, F., Wyer, M.D. and Godfree, A.F. 1996. Marine waters contaminated with domestic sewage: nonenteric illnesses associated with bather exposure in the United Kingdom. Am. J. Public Health 86, 1228-1234.

Foden, J., Sivyer, D.B., Mills, D.K. and Devlin, M.J. 2008. Spatial and temporal distribution of chromophoric dissolved organic matter (CDOM) fluorescence and its contribution to light attenuation in UK waterbodies. Estuar. Coast. Shelf Sci. 79, 707-717.

Folmar, L.C., Gardner, G.R., Schreibman, M.P., Magliulo-Cepriano, L., Mills, L.J., Zaroogian, G., Gutjahr-Gobell, R., Haebler, R., Horowitz, D.B,, and Denslow, N.D., 2001. Vitellogenin-induced pathology in male summer flounder (*Paralichthys dentatus*). Aquat. Toxicol. 51, 431-441.

Foote, A.D., Griffin, R.M., Howitt, D., Larsson, L., Miller, P.J.O. and Hoelzel, A.R., 2004. Whale-call response to masking boat noise. Nature 428, 910.

Förlin, L. and Haux, C., 1990. Sex differences in hepatic cytochrome P-450 monooxygenase activities in rainbow trout during an annual reproductive cycle. J. Endocrinol. 124, 207-213.

Fossi, M.C., Casini, S., Ancora, S., Moscatelli, A., Ausili, A. and Notarbartolo-di-Sciara, G., 2001. Do endocrine disrupting chemicals threaten Mediterranean swordfish? Preliminary results of vitellogenin and zona radiata proteins in *Xiphias gladius*. Mar. Environ. Res. 52, 477-483.

Fossi, M.C., Casini, S., Marsili, L., Neri, G., Mori, G., Ancora, S., Moscatelli, A., Ausili, A. and Notarbartolo-di-Sciara, G., 2002. Biomarkers for endocrine disruption in three species of Mediterannean large pelagic fish. Mar. Environ. Res. 54, 667-671.

Fossi, M.C., Casini, S., Marsili, L., Ancora, S., Mori, G., Neri, G., Romeo, T. and Ausili, A., 2004. Evaluation of ecotoxicological effects of endocrine disrupters during a four-year survey of the Mediterranean population of swordfish (*Xiphias gladius*). Mar. Environ. Res. 58, 425-429.

Fowler, M.E., 1987. Zoo animals and wildlife. In: Veterinary Cancer Medicine. Edited by Theilen, G.H. and Madewell, B.R. Lea & Febiger. pp. 649-662.

French, B.L., Reichart, W.L., Hom, T., Nishimoto, M., Sanborn, H.R. and Stein, J.E., 1996. Accumulation and dose-response of hepatic DNA adducts in English sole (*Pleuronectes vetulus*) exposed to a gradient of contaminated sediments. Aquat. Toxicol. 36, 1-16.

Geoghegan, F., Katsiadaki., I., Williams, T.D. and Chipman, J.K., 2008. A cDNA microarray for the three-spined stickleback, *Gasterosteus aculeatus* L., and analysis of the interactive effects of oestradiol and dibenzanthracene exposures. J. Fish Biol. 72, 2133-2153.

George, S.G., Wright, J. and Conroy, J., 1995. Temporal studies of the impact of the Braer oil spill on inshore feral fish from Shetland, Scotland. Arch. Environ. Con. Tox. 29, 530-534.

Gibbs, P.E., Bryan, G.W., Pascoe, P.L. and Burt, G.R., 1987. The use of the dog-whelk, Nucella lapillas, as an indicator of tributyltin (TBT) contamination. J. Mar. Biol. Assoc. 67, 507-523.

Gilbert, C., 1996. The cost to local authorities of coastal and marine pollution – a preliminary appraisal. In: Recent Policy Developments and the Management of Coastal Pollution. pp. 12-14. Edited by Earll, R.C. Marine Environmental Management and Training.

Goessler, W. and Pavkov, M., 2003. Accurate quantification and transformation of arsenic compounds during wet ashing with nitric acid and microwave assisted heating. Analyst 128, 796-802.

Goksøyr, A., Andersson, T., Buhler, D.R., Stegeman, J.J., Williams, D.E. and Forlin, L., 1991. Immunochemical cross-reactivity of beta-Naphthoflavone-inducible Cytochrome P450 (P450ia) in liver-microsomes from different fish species and rat. Fish Physiol. Biochem. 9, 1-13.

Gomez-Bautista, M., Ortega-Mora, L.M, Tabares, E., Lopez-Rodas, V. and Costas E., 2000. Detection of infectious Cryptosporidium parvum oocysts in mussels (*Mytilus galloprovincialis*) and cockles (*Cerastoderma edule*). Appl. Environ. Microb. 66, 1866-1870.

Gowen, R.J., Hydes, D.J., Mills, D.K., Stewart, B.M., Brown, J., Gibson, C.E., Shammon, T.M., Allen, M. and Malcolm, S.J., 2002. Assessing trends in nutrient concentrations in coastal shelf seas: a case study in the Irish Sea. Estuar. Coast. Shelf Sci. 54, 927–939.

Gubbins, M.J., Greathead, C., Amundrud, T., Gillibrand, P., Tett, P., Inall, M., Hawkins, A.J.S. and Davies, I.M., 2008.Towards the determination of the carrying capacity of Scottish sea lochs for shellfish aquaculture. ICESCM2008/H13, 7pp.

Hagger, J.A., Jones, M.B., Lowe, D., Leonard, D.R.P., Owen, R. and Galloway, T.S., 2008. Application of biomarkers for improving risk assessments of chemicals under the Water Framework Directive: A case study. Mar. Pollut. Bull., 56, 1111-1118.

Hall, A.J. and Thomas, G.O., 2007. Polychlorinated biphenyls, DDT, polybrominated diphenyl ethers, and organic pesticides in United Kingdom harbour seals (*Phoca vitulina*) – mixed exposures and thyroid homeostasis. Environ. Toxicol. Chem. 26, 851-861.

Hall, A.J., Law, R.J., Wells, D.E., Harwood, J., Ross, H.M., Kennedy, S., Allchin, C.R., Campbell, L.A. and Pomeroy, P.P., 1992. Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 phocine distemper epizootic. Sci. Total Environ. 115, 145-162.

Hall, A.J., Kalantzi, O.I. and Thomas, G.O., 2003. Polybrominated diphenyl ethers (PBDEs) in grey seals during their first year of life – are they thyroid endocrine disrupters? Environ. Pollut. 126, 29-37.

Hall, A.J., Hugunin, K., Deaville, R., Law, R.J., Allchin, C.R. and Jepson, P.D., 2006. The risk of infection from polychlorinated biphenyl exposure in the harbor porpoise (*Phocoena phocoena*): a case-control approach. Environ. Health Perspect. 114, 704-711.

Harshbarger, J.C., 2004. Chronology of oncolygy in fish, amphibians and invertebrates. Bulletin of the European Association of Fish Pathology, 24: 62-81.

Harland, E.J. and Richards, S.D., 2006. SEA 7 Technical Report: Underwater ambient noise, QinetiQ, [available at www.offshore-sea.org.uk].

Harland, E.J., Jones, S.A.S. and Clarke, T., 2005. SEA 6 Technical Report: Underwater ambient noise QinetiQ. [available at www.offshore-sea.org.uk].

Harvey, J.S. and Parry, J.M., 1998. The analysis of DNA adduct formation, removal and persistence in the common mussel *Mytilus edulis* exposed to 4-nitroquinoline 1-oxide. Mutat. Res. 399, 31-42.

Hassel, A., Knutsen, T., Dalen, J., Skaar, K., Løkkeborg, S., Misund, O.A., Østensen, Ø., Fonn, M., Haugland, E.K., 2004. Influence of seismic shooting on the lesser sand eel (*Ammodytes marinus*). ICES J. Mar. Sci. 61, 1165-1173.

Hastie, G.D., Wilson, B., Tufft, L.H. and Thompson, P.M., 2003. Bottlenose dolphins increase breathing synchrony in response to boat traffic. Mar. Mammal Sci. 19, 74-84.

Hastings, M.C. and Popper, A.N., 2005. Effects of sound on fish. Contract 43A0139 Task Order 1, California Department of Transportation.

Hawkins, A.D., 2006. Assessing the impact of pile driving upon fish 2005 International Conference on Ecology and Transportation North Carolina State University, Raleigh. 22pp.

Hawkins, A.D. and Johnstone, A.D.F., 1978. The hearing of the Atlantic salmon (*Salmo salar*). J. Fish Biol. 13, 655-673.

Hawkins, A.D. and Myrberg, Jr A.A., 1983. Hearing and sound communication under water. In: Bioacoustics: A Comparative Approach. Edited by Lewis, B. Academic Press, pp. 347-405.

Heffernan, J., Barry, J., Devlin, M. and Fryer, R., 2010. *A simulation tool for designing nutrient monitoring programmes for eutrophication assessments* Environmetrics 21, 3-20.

Hester, K.C., Peltzer, E.T., Kirkwood, W.J. and Brewer, P.G., 2008. Unanticipated consequences of ocean acidification: A noisier ocean at lower pH. Geophys. Res. Lett. 35:L19601, doi: 19610.11029/12008GL034913.

Hildebrand, J.A., 2005. Impacts of anthropogenic sound. In: Marine Mammal Research: Conservation Beyond Crisis. Edited by Reynolds, J.E., Perrin, W.F., Reeves, R.R., Montgomery, S. and Ragen, T. The Johns Hopkins University Press, pp. 101-124.

Hines, A., Oladirana, G.S., Bignell, J.P., Stentiford, G.D. and Viant, M.R., 2007. Direct sampling of organisms from the field and knowledge of their phenotype: key recommendations for environmental metabolomics. Environ. Sci. Technol. 41, 3375-3381.

Hiramatsu, N., Matsubara, T., Fujita, T., Sullivan, C.V. and Hara, A., 2006. Multiple piscine vitellogenins: biomarkers of fish exposure to estrogenic endocrine disruptors in aquatic environments. Mar. Biol. 149, 35-47.

HPA, 2005. Ionising Radiation Exposure of the UK Population: 2005 Review. Health Protection Agency, HPA-RPD-001.

Hunt, G.J., Smith, B.D. and Camplin, W.C., 1997. Recent changes in liquid radioactive waste discharges to the Irish Sea from Sellafield, part 1: Inputs and uptake by coastal biota. Radioprotection – Colloques 32, 17-22.

IACMST, 2006. Underwater sound and marine life. IACMST Working Group Report No.6. Inter-Agency Committee on Marine Science and Technology - National Oceanographic Centre, Southampton.

IAEA, 1999. Application of radiological exclusion and exemption principles to sea disposal. International Atomic Energy Agency. IAEA-TECDOC-1068.

ICES, 2007. Report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO). ICES Document CM 2007/MCC: 04. International Council for the Exploration of the Sea.

ICES, 2008. Report of the Marine Chemistry Working Group. International Council for the Exploration of the Sea.

ICES-AGISC, 2005. Report of the Ad-hoc Group on the Impact of Sonar on Cetaceans and Fish, International Council for the Exploration of the Sea.

ICRP, 2007. The 2007 Recommendations of the International Commission on Radiological Protection. ICRP Publication 103. Annals of the ICRP 37 (2-4).

IPCC, 2007. Climate Change 2007: Synthesis Report. Contribution of working groups I, II, and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland, 104 pp

Irion, G. and Muller, G., 1990. Lateral distribution and sources of sediment associated metals in the North Sea. In: Facets of Modern Biogeochemistry. Edited by Ittekkot, V., Kempe, S., Michaelis, W. and Spritzy, A. Springer-Verlag, pp. 175-201.

Janik, V.M., 2005. Underwater acoustic communication networks in marine mammals. In: Animal Communication Networks. Edited by McGregor, P.K. Cambridge University Press, pp. 390-415.

Janik, V.M. and Thompson, P.M., 1996. Changes in surfacing patterns of bottlenose dolphins in response to boat traffic. Mar. Mammal Sci. 12, 597-602.

Jennings, S., and Kaiser, M.J., 1998. The effects of fishing on marine ecosystems. Adv. Mar. Biol. 14, 201-352.

Jepson, P.D., Bennett, P.M., Deaville, R., Allchin, C.R., Baker, J.R. and Law, R.J., 2005. Relationships between PCBs and health status in harbour porpoises (*Phocoena phocoena*) stranded in the United Kingdom. Environ. Toxicol. Chem. 24, 238-248.

JNCC, 2004. Guidelines for Minimising Acoustic Disturbance to Marine Mammals from Seismic Surveys. Joint Nature Conservation Committee.

Johnson, C.S., 1967. Sound detection thresholds in marine mammals. In: Marine Bioacoustics II. Edited by Tavolga, W.N. Pergamon, pp. 247-260.

Johnson, L., Lomax, D., Myers, M., Swanson, P., Felli, L., West, J. and O'Neill, S., 2005. Xenoestrogen exposure and altered reproductive timing in Puget Sound English sole. Presentation at the 26th Annual SETAC Conference, Baltimore, Maryland, USA, November 2005. Abstract only. On-line citation (March 2007): http://abstracts.co.allenpress.com/pweb/setac2005/document/?ID=57009

Jonsson, G., Beyer, J., Wells, D. and Ariese, F., 2003. The application of HPLC-F and GC-MS to the analysis of selected hydroxy polycyclic hydrocarbons in two certified fish bile reference materials. J. Environ. Monit. 5, 513-520.

Jørgensen, R., Olsen, K.K., Falk-Petersen, I.-B. and Kanapthippilai, P., 2005. Investigations of potential effects of low frequency sonar signals on survival, development and behaviour of fish larvae and juveniles. Norwegian College of Fishery Science University of Tromsø.

Kalantzi, O.I., Hall, A.J., Thomas, G.O. and Jones, K.C., 2005. Polybrominated diphenyl ethers and selected organochlorine chemicals in grey seals (*Halichoerus grypus*) in the North Sea. Chemosphere 58, 345-354.

Kastelein, R.A., Bunskoek, P., Hagedoorn, M. and Au, W.W.L., 2002. Audiogram of a harbor porpoise (*Phocoena phocoena*) measured with narrow-band frequency modulated signals. J. Acoust. Soc. Am. 112, 334-344.

Kastelein, R.A., Hagedoorn, M., Au, W.W.L. and de Haan, D., 2003. Audiogram of a striped dolphin (*Stenella coeruleoalba*). J. Acoust. Soc. Am. 113, 1130-1137.

Kay, D., Fleisher, J.M., Salmon, R.L., Jones, F., Wyer, M.D., Godfree, A.F., Zelenauch-Jacquotte, Z. and Shore, R., 1994. Predicting likelihood of gastroenteritis from sea bathing: results from randomised exposure. Lancet. 344, 905-909.

Kay, D., Lee, R., Wyer, M.D. and Stapleton, C., 2009a. Integrated catchment studies: source identification and modelling. In: Management of Shellfish Harvesting Waters for Public Health Protection. Edited by Rees, G., Pond, K. Kay, D. and Domingo, S. Published jointly by International Water Association and World Health Organization. Chapter 10 (in press).

Kay, D., Lee, R., Wyer, M.D. and Stapleton, C.S., 2009b. Profiling shellfish harvesting waters for public health protection. In: Management of Shellfish Harvesting Waters for Public Health Protection. Edited by Rees, G., Pond, K. Kay, D. and Domingo, S. Published jointly by International Water Association and World Health Organization. Chapter 12 (in press).

Kay, D., Kershaw, S., Lee, R., Wyer, M.D., Watkins, J. and Francis, C., 2009c. Results of field investigations into the impact of intermittent sewage discharges on the microbiological quality of wild mussels (*Mytilus edulis*) in a tidal estuary. Wat. Res. 42, 3033-3046.

Kelly-Gerreyn, B.A., Hydes, D.J., Hartman, M.C., Siddorn, J., Hyder, P. and Holt, M.W., 2007. The phosphoric acid leak from the wreck of MV Ece in the English Channel in 2006: assessment with a ship of opportunity, an operational ecosystem model and historical data. Mar. Pollut. Bull. 54, 850-862.

Kennington, K., Shammon, T.M., Kraberg, A. Wither, A., Jones, P. and Hartnoll, R.G., 2003. The distribution of nutrient and phytoplankton in the eastern Irish Sea during 2001. Environment Agency Technical Report E-1049/TR5.

Kennington, K., Wither, A., Shammon, T.M., Jones, P., Harrison, A. Kraberg, A and Hartnoll, R.G., 2004. The distribution of nutrient and phytoplankton in the eastern Irish Sea during 2002. Environment Agency Technical Report E-1049/TR6.

Kennington, K., Wither, A., Shammon, T.M., Jones, P., Harrison, A. and Hartnoll, R.G., 2005. The distribution of nutrient and phytoplankton in the north-eastern Irish Sea during 2003. Environment Agency Technical Report E-1049/TR7

Kerckhof, F., 2004. Belgian report for the seventh meeting of the OSPAR Pilot Project Steering Group on Monitoring Marine Litter.

Kershaw, P.J., Heldal, H.E., Mork, K.A. and Rudjord, A.L., 2004. Variability in the supply, distribution and transport of the transient tracer technetium-99 in the NE Atlantic. J. Mar. Sys. 44, 55-81.

KIMO, 2000. Impacts of Marine Debris and Oil: Economic and Social Costs to Coastal Communities. KIMO, Shetland.

KIMO, 2008. Fishing for Litter. Scotland Final Report 2005 – 2008.

Kirby, M.F., Allen, Y.T., Dyer, R.A., Feist, S.W., Katsiadaki, I., Matthiessen, P., Scott, A.P., Smith, A., Stentiford, G.D., Thain, J.E., Thomas, K.V., Tolhurst, L. and Waldock, M.J., 2004. Surveys of plasma vitellogenin and intersex in male flounder (*Platichthys flesus*) as measures of endocrine disruption by estrogenic contamination in United Kingdom estuaries: temporal trends, 1996 to 2001. Environ. Toxicol. Chem. 23, 748-758.

Kirby, M.F., Smith, A.J., Barry, J., Katsiadaki, I. Lyons, B. and Scott, A.P., 2006. Differential sensitivity of flounder (*Platichthys flesus*) in response to oestrogenic chemical exposure: An issue for design and interpretation of monitoring and research programmes. Mar. Environ. Res. 62, 315-325.

Knowles, J.F., Smith, D.L. and Winpenny, K., 1998. A comparative study of the uptake, clearance and metabolism of technetium in lobster (*Homarus gammarus*) and edible crab (*Cancer pagurus*). Rad. Protect. Dosim. 75, 125-129.

Koehler, A., 2004. The gender-specific risk to liver toxicity and cancer of flounder (*Platichthys flesus* (L.)) at the German Wadden Sea coast. Aquat. Toxicol. 70, 257-276.

Koschinski, S., Culik, B.M., Henriksen, O.D., Tregenza, N., Ellis, G.M., Jansen, C. and Kathe, G., 2003. Behavioural reactions of free-ranging harbour porpoises and seals to the noise of a simulated 2 MW windpower generator. Mar. Ecol. Prog. Ser. 265, 263-273.

Lafferty, K.D., Porter, J.W. and Ford, S.E., 2004. Are diseases increasing in the ocean? Ann. Rev. Ecol. Ev. System. 35, 31-54.

Laist, D.W., 1997. Impacts of marine debris: entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. In: Marine Debris: Sources, Impacts and Solutions. Edited by Coe, J. and Rogers, D.B. Springer Series on Environmental Management.

Lang, T. and Dethlefsen, V., 1996. Fish disease monitoring – a valuable tool for pollution assessment? ICES CM 1996/E 17,18 pp.

Larsson, C-M., Jones, C., Gomez-Ros, J.M. and Zinger, I., 2004. Framework for assessment of environmental impact of ionising radiation in major European ecosystems, FASSET Project, EC Contract No FIGE-CT-2000-00102.

Law, R. (Compiler), 2008. Environmental monitoring conducted in Lyme Bay following the grounding of MSC Napoli in January 2007, with an assessment of impact. Science Series, Aquatic Environment Monitoring Report, Cefas, Lowestoft, 61: 36pp.

Law, R.J., Kelly, C.A., Baker, K.L., Langford, K.H. and Bartlett, T., 2002. Polycyclic aromatic hydrocarbons in sediments, mussels and crustacea around a former gasworks site in Shoreham-by-Sea, UK. Mar. Pollut. Bull. 44, 903-911.

Law, R.J., Allchin, C.R. and Mead, L.K., 2005. Brominated diphenyl ethers in twelve species of marine mammals stranded in the UK. Mar. Pollut. Bull. 50, 356-359.

Law, R.J., Jepson, P.D., Deaville, R., Reid, R.J., Patterson, I.A.P., Allchin, C.R. and Jones, B.R., 2006a. Collaborative UK Marine Mammals Strandings Project: summary of contaminant data for the period 1993-2001. Science Series Technical Report, Cefas Lowestoft, 131: 72pp.

Law, R.J., Bersuder, P., Allchin, C.R. and Barry, J., 2006b. Levels of the flame retardants hexabromocyclododecane and tetrabromobisphenol A in the blubber of harbour porpoises (*Phocoena phocoena*) stranded or bycaught in the U.K., with evidence for an increase in HBCD concentrations in recent years. Environ. Sci. Technol. 40, 2177-2183.

Law, R.J., Bersuder, P., Mead, L.K. and Jepson, P.D., 2008a. PFOS and PFOA in the livers of harbour porpoises (*Phocoena phocoena*) stranded or bycaught around the UK. Mar. Pollut. Bull. 56, 792-797.

Law, R.J., Bersuder, P., Barry, J., Wilford, B.H., Allchin, C.R. and Jepson, P.D., 2008b. A significant downturn in levels of HBCD in the blubber of harbor porpoises (*Phocoena phocoena*) stranded or bycaught in the UK: an update to 2006. Environ. Sci. Technol. 42, 9104-9109.

Law, R.J., Bersuder, P., Barry, J., Deaville, R., Reid, R.J. and Jepson, P.D., 2010a. Chlorobiphenyls in the blubber of harbour porpoises (*Phocoena phocoena*) from the UK: levels and trends 1991-2005. Mar. Pollut. Bull. 60, 470-473.

Law, R.J., Barry, J., Bersuder, P., Barber, J.L., Deaville, R., Reid, R.J. and Jepson, P.D., 2010b. Levels and trends of BDEs in blubber of harbor porpoises (*Phocoena phocoena*) from the UK, 1992-2008. Environ. Sci. Technol. 44, 4447-4451.

Lee, R.J. and Younger, A.D., 2003. Determination of the relationship between faecal indicator organisms and the presence of human pathogenic micro-organisms in shellfish. In: Molluscan Shellfish Safety. Edited by Villalba, A., Reguera, B., Romalde, J.L., and Beiras, R. Consellería de Pesca e Asuntos Marítimos da Xunta de Galicia and Intergovernmental Oceanographic Commission of UNESCO; Santiago de Compostela, Spain. pp. 247-252.

Lees, D., 2000. Viruses and bivalve shellfish. Int. J. Food Microbiol. 59, 81-116.

Leonard, K.S., McCubbin, D., Brown, J., Bonfield, R, and Brooks, T., 1997. Distribution of Tc-99 in UK coastal waters. Mar. Pollut. Bull. 34, 628-636.

Leonard, K.S., McCubbin, D., McMahon, C.A., Mitchell, P.I. and Bonfield, R., 1998. $^{137}Cs/^{90}Sr$ ratios in the Irish Sea and adjacent waters: a source term for the Arctic. Rad. Protect. Dosim. 75, 207-212.

Leonard, K.S., McCubbin, D., McDonald, P., Service, M., Bonfield, R. and Conney, S., 2004. Accumulation of technetium-99 in the Irish Sea. Sci. Total Environ. 322, 255-270.

Lepper, P.A., Turner, V.L.G., Goodson, A.D. and Black, D., 2004. Source levels and spectra emitted by three commercial aquaculture anti-predation devices. Seventh European conference on Underwater Acoustics.

Lesage, V., Barrette, C., Kingsley, M.C.S. and Sjare, B., 1999. The effect of vessel noise on the vocal behavior of belugas in the St. Lawrence River Estuary, Canada. Mar. Mammal Sci. 15, 65-84.

Lin, E.L.C., Cormier, S.M. and Torsella, J.A., 1996. Fish biliary polycyclic aromatic hydrocarbon metabolites estimated by fixed-wavelength fluorescence: Comparison with HPLC-fluorescent detection. Ecotox. Environ. Safety 35, 16-23.

Lindahl, P., Ellmark, C., Gafvert, T., Mattsson, S., Roos, P., Holm, E. and Erlandsson, B., 2003. Long term study of technetium-99 in the marine environment on the Swedish west coast. J. Environ. Radioactiv. 67, 145-156.

Long, E.R., MacDonald, D.D., Smith, S.L. and Calder, F.D., 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environ. Manag. 19, 81-97.

Loring, D.H., 1991. Normalisation of heavy metal data from estuarine and coastal sediments. ICES J. Mar. Sci. 48, 101-115.

Lucke, K., Lepper, P., Blanchet, M.A. and Siebert, U., 2008, How tolerant are harbour porpoises to underwater sound? In: Marine Mammals and Seabirds in Front of Offshore Wind Energy / MINOS - Marine Warm-Blooded Animals in North and Baltic Seas. Edited by Wollny-Goerke, K. and Eskildsen, K. Teubner, Wiesbaden, Germany, pp. 59-77.

Luoma, S.N., 2008. Silver nanotechnologies and the environment: Old problems or new challenges? PEN 15, September 2008. The Woodrow Wilson International Center for Scholars – Project on Emerging Nanotechnologies.

Luoma, S.N., Ho, Y.B. and Bryan, G.W., 1995. Fate, bioavailability and toxicity of silver in estuarine environments. Mar. Pollut. Bull. 31, 44-54.

Lyons, B.P., Stewart, C. and Kirby, M.K., 1999. The detection of biomarkers of genotoxin exposure in the European flounder (*Platichthys flesus*) collected from the River Tyne Estuary. Mutat. Res. 446, 111-119.

Lyons, B.P., Stewart, C. and Kirby, M.K., 2000. ^{32}P-postlabelling analysis of DNA adducts and EROD induction as biomarkers of genotoxin exposure in dab (*Limanda limanda*) from British coastal waters. Mar. Environ. Res. 50, 575-579.

Lyons, B.P., Bignell, J., Stentiford, G.D. and Feist, S.W., 2004a. The viviparous blenny (*Zoarces viviparus*) as a bioindicator of contaminant exposure: application of biomarkers of apoptosis and DNA damage. Mar. Environ. Res. 58, 757-762.

Lyons, B.P., Stentiford, G.D., Green, M., Bignell,J., Bateman, K., Feist, S.W., Goodsir, F. Reynolds, W.J. and Thain, J.E., 2004b. DNA adduct analysis and histopathological biomarkers in European flounder (*Platichthys flesus*) sampled from UK estuaries. Mutat. Res. 552, 177-186.

Lyons, B.P., Stentiford, G.D., Bignell, J., Goodsir, F., Sivyer, D.B., Devlin, M.J., Lowe, D., Beesley, A., Molina, S.B., Pascoe, C.K., Moore, M.N. and Garnacho, E., 2006. A biological effects monitoring survey of Cardigan Bay using flatfish histopathology, cellular biomarkers and sediment bioassays: Findings of the Prince Madog prize 2003. Mar. Environ. Res. 62, S342-S346.

Madsen, P.T., 2005. Marine mammals and noise: problems with root mean square sound pressure levels for transients. J. Acoust. Soc. Am. 117, 3952-3957.

Madsen, P.T., Wahlberg, M., Tougaard, J., Lucke, K. and Tyack, P., 2006. Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. Mar. Ecol. Prog. Ser. 309, 279-295.

Malins, D.C., Krahn, M.N., Myers, M.S., Rhodes, L.D., Brown, D.W., Krone, C.A., McCain, B.B. and Cham, S, 1985. Toxic chemicals in sediments and biota from a creosote-polluted harbour. Relationships with hepatic neoplasms and other hepatic lesions in English sole (*Parophyrs vetulus*). Carcinogenesis 6, 1463-1469.

Mann, D.A., Lu, Z. and Popper, A.N., 1997. A Clupeid Fish can detect ultrasound. Nature 48:341.

Marine Environment Monitoring Group, 2004. UK National Marine Monitoring Programme – second report (1999-2001). 137pp.

Mato, Y., Isobe, T., Takada, H., Kanehiro, H., Ohtake, C. and Kaminuma, T., 2001. Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. Environ. Sci. Technol. 35, 318-324.

McCauley, R.D., Fewtrell, J. and Popper, A.N., 2003. High intensity anthropogenic sound damages fish ears. J. Acoust. Soc. Am. 113, 631-642.

McCubbin, D., Leonard, K.S., McDonald, P., Bonfield, R. and Boust, D., 2006. Distribution of technetium-99 in sub-tidal sediments of the Irish Sea. Cont. Shelf Res. 26, 458-473.

McCubbin, D., Jenkinson, S.B., Leonard, K.S., Bonfield, R.A. and McMeekan, I.T., 2008. An assessment of the availability of Tc-99 to marine foodstuffs from contaminated sediments. Project R01062/C2170 . RL09/08. Cefas, Lowestoft.

McHugh, B., Law, R.J., Allchin, C.R., Rogan, E., Murphy, S., Foley, M.B., Glunn, D. and McGovern, E., 2007. Bioaccumulation and enantiomeric profiling of organochlorine pesticides and persistent organic pollutants in the killer whale (*Orcinus orca*) from British and Irish waters. Mar. Pollut. Bull. 54, 1724-1731.

MCS, 2007. Beachwatch 2007 – Nationwide Beach Clean and Survey Report. MCS, Ross-on-Wye, UK. Marine Conservation Society.

Miller, B.S., 1986. Trace metals in the common mussel *Mytilus edulis* (L.) in the Clyde Estuary. Proceedings of the Royal Society of Edinburgh, 90B: 377-391.

Mills, L.J. and Chichester, C., 2005. Review of evidence: are endocrine-disrupting chemicals in the aquatic environment impacting fish populations? Sci. Total Environ. 343, 1-34.

Møhl, B., 2003. Sperm whale sonar rivals tactical sonar with source levels at 235 dB. In: Proceedings of the ECS workshop on active sonar and cetaceans. Edited by Evans, P.G.M. and Miller, L.A. ECS Newsletter No 42. European Cetacean Society, pp. 41-42.

MON, 2003. Meeting on the Working Group on Monitoring (MON). Copenhagen 16-18 December, 2003, OSPAR Commission, London.

MON, 2007. Meeting on the Working Group on Monitoring (MON). Copenhagen 4-7 December, 2007, OSPAR Commission, London.

Morton, A.B. and Symonds, H.K. 2002. Displacement of *Orcinus orca* (L.) by high amplitude sound in British Columbia, Canada. ICES J. Mar. Sci. 59, 71-80.

MPMMG, 1997. Comprehensive studies for the purposes of Articles 6 & 8.5 of Dir 91 / 271 EEC. The Urban Waste Water Treatment Directive. Marine Pollution Monitoring Management Group.

Mrosovsky, N., Ryan, G.D. and James, M.C., 2009. Leatherback turtles: The menace of plastic. Mar. Pollut. Bull. 58, 287-289.

Murray, L.H. and Lee, R.J., 2009. Risk management of shellfisheries. In: Shellfish Safety and Quality. Edited by Shumway, S. and Rodrick, G. Woodhead Publishing Limited.

Myers, M.S., Landahl, J.T., Krahn, M.M., Johnson, L.L. and McCain, B.B., 1990. Overview of studies on liver carcinogenesis in English sole from Puget Sound: evidence for a xenobiotic chemical etiology. I. Pathology and epizootiology. Sci. Total Environ. 94, 33-50.

Myers, M.S., Landahl, J.T., Krahn, M.M. and McCain, B.B., 1991. Relationship between hepatic neoplasms and related lesions and exposure to toxic chemicals in marine fish from the U.S. West Coast. Environ. Health Perspect. 90, 7-15.

Myers, M.S., Olson, O.P., Johnson, L.L., Stehr, C.S., Hom, T. and Varanasi, U., 1992. Hepatic lesions other than neoplasms in subadult flatfish from Puget Sound, WA; relationship with indices of contaminant exposure. Mar. Environ. Res. 34, 45-51.

Myers, M.S., Stehr, C.M., Olson, O.P., Johnson, L.L., McCain, B.B., Chan, S.-L. and Varanasi, U., 1994. Relationships between toxicopathic lesions and exposure to chemical contaminants in English sole (*Parophrys vetulus*), starry flounder (*Pleuronectes stellatus*), and white croaker (*Genyonemus linatus*) from selected marine sites on the Pacific Coast, USA. Environ. Health Perspect. 102, 200-215.

Nachtigall, P.E., Au, W.W.L., Pawlowski, J.L. and Moore, P.W.B., 1995. Risso's dolphin (*Grampus griseus*) hearing thresholds in Kaneohe Bay, Hawaii. In: Sensory Systems of Aquatic Mammals. Edited by Kastelein, R.A., Thomas, J.A. and Nachtigall, P.E. De Spil Publ, pp. 49-53.

Nash, J.P., Kime, D.E., Van der Ven, L.T.M., Wester, P.W., Brion, F., Maack, G., Stahlschmidt-Allner, P. and Tyler, C.R., 2004. Long-term exposure to environmental concentrations of the pharmaceutical ethynylestradiol causes reproductive failure in fish. Environ. Health Perspect. 112, 1725-1733.

NDAWG, 2008. Guidance note 1 – overview of guidance on the assessment of radiation doses from controlled releases. National Dose Assessments Working Group. Environment Agency, Food Standards Agency.

Nedwell, J.R., Edwards, B., Turnpenny, A.W.H. and Gordon, J., 2004. Fish and marine mammal audiograms: a summary of available information. Subacoustech report ref. 534R0214. Submitted to Chevron Texaco Ltd.

Nedwell, J.R., Parvin, S.J., Edwards, B., Workman, R., Brooker, A.G. and Kynoch, J.E., 2008. Measurement and interpretation of underwater noise during construction and operation of offshore windfarms in UK waters. Report for COWRIE, Newbury, UK.

Nixon, S.W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia 41, 199-219.

Nowacek, S., Wells, R. and Solow, A., 2001. Short-term effects of boat traffic on bottlenose dolphins, *Tursiops truncatus* in Sarasota Bay, Florida. Mar. Mammal Sci. 17, 673-688.

Nowacek, D., Johnson, M.P. and Tyack, P.L., 2004. North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. Proc. Roy. Soc. Lond., B 271, 227-231.

Nowacek, D.P., Thorne, L.H., Johnston, D.W. and Tyack, P.L., 2007. Responses of cetaceans to anthropogenic noise. Mammal Rev. 37, 81-115.

NRC, 2005. Marine mammal populations and ocean noise – determining when noise causes biologically significant effects. National Academies Press.

Occhipinti-Ambogi, A., 2007. Global change and marine communities: Alien species and climate change. Mar. Pollut. Bull. 55, 342-352.

Offutt, G.C., 1974. Structures for the detection of acoustic stimuli in the Atlantic codfish, *Gadus morhua*. J. Acoust. Soc. Am. 56, 665-671.

OGP, 2008. Fundamentals of Underwater Sound. Report No. 406, May 2008. Oil & Gas Producers.

OGP/IAGC, 2007. Seismic surveys and marine mammals. International Association of Oil and Gas Producers / International Association of Geophysical Contractors, London.

Oliver, J.D., 2005. Wound infections caused by *Vibrio vulnificus* and other marine bacteria. Epidemiol. Infect. 133, 383-391.

Olesiuk, P.F., Nicol, L.M., Sowden, M.J. and Ford, J.K.B., 2002. Effects of the sound generated by an acoustic harrassment device on the relative abundance and distribution of harbour porpoises (*Phocoena phocoena*) in Retreat Passage, British Columbia. Mar. Mammal Sci. 18, 843-862.

OSPAR, 1995. Assessment and monitoring. The Joint Assessment and Monitoring Programme. OSPAR Commission.

OSPAR, 1998a. OSPAR Strategy with regard to Hazardous Substances. OSPAR 1998-16, OSPAR Commission.

OSPAR, 1998b. Ministerial meeting of the OSPAR Commission, Sintra, July 1998 – Main results. OSPAR, London.

OSPAR, 2001a (2004 update). Certain brominated flame retardants-Polybrominated diphenyl ethers, polybrominated biphenyls, hexabromo cyclododecane. OSPAR Priority Substances. OSPAR Commission.

OSPAR, 2001b. Draft common assessment criteria and their application within the Comprehensive Procedure of the Common Procedure. Meeting of the Eutrophication Task Group, London, 9-11 October 2001. OSPAR Commission.

OSPAR, 2003a. Joint Assessment Monitoring Programme Guidelines for Contaminant-specific Biological Effects Monitoring. Reference number 2003-10. OSPAR Commission.

OSPAR, 2003b. The OSPAR Integrated Report 2003 on the Eutrophication Status of the OSPAR Maritime Area based upon the first application of the Comprehensive Procedure Includes 'baseline'/assessment levels used by Contracting Parties and monitoring data. MMC 2003/2/4. OSPAR Commission.

OSPAR, 2004. Proposal for Assessment Criteria for TBT – Specific Biological Effects. ASMO 04/3/3.

OSPAR, 2005. Draft Report on the North Sea Pilot Project on Ecological Quality Objectives. Meeting of the Biodiversity Committee, Bonn, 21-25 February 2005. OSPAR Commission.

OSPAR, 2007a. PBDE congeners to be determined in the marine environment as part of the OSPAR Co-ordinated Environmental Monitoring Program (CEMP).

OSPAR, 2007b. Second periodic evaluation of progress towards the objectives of the OSPAR radioactive substances strategy. OSPAR Commssion.

OSPAR, 2009a. Deposition of air pollutants around the North Sea and the North-East Atlantic in 2007. Comprehensive Atmospheric Monitoring Programme. Publication No. 449. OSPAR Commission.

OSPAR, 2009b. Overview of the Impacts of Anthropogenic Underwater Sound in the Marine Environment, OSPAR Commission. Publication no. 441.

OSPAR, 2008a. OSPAR CEMP (Coordinated Environmental Monitoring Program), 2008-8.

OSPAR, 2008b. Assessment of the Environmental Impact of Offshore Wind-farms, OSPAR Commission. Publication no. 385.

OSPAR, 2010. Quality Status Report, 2010. OSPAR Commission.

OSPAR JAMP, 1998a JAMP guidelines for general biological effects monitoring. Joint Assessment and Monitoring Programme. Oslo and Paris Commissions. 38 pp.

OSPAR JAMP, 1998b JAMP guidelines for contaminant-specific biological effects monitoring. Joint Assessment and Monitoring Programme. Oslo and Paris Commissions. 38 pp.

OSPAR WKIMON, 2008. ICES/OSPAR Workshop on Integrated Monitoring of Contaminants and their Effects in Coastal and Open-sea Areas (WKIMON IV) 5-7 February 2008. ICES Report 200608.

Painting, S.J., Devlin, M.J., Malcolm, S.J., Parker, E.R., Mills, D.K., Mills, C., Tett, P., Wither, A., Burt, J., Jones, R. and Winpenny, K., 2007. Assessing the impact of nutrient enrichment in estuaries: susceptibility to eutrophication. Mar. Pollut. Bull. 55, 74-90.

Parker, J.G., Rosell, R.S. and MacOscar, K.C., 1988. The phytoplankton production cycle in Belfast Lough. J. Mar. Biol. Assoc. UK 68, 555-564.

Parson, E.C.M., Dolman, S.J., Wright, A.J., Rose, N.A. and Burns, W.C.G., 2008. Navy sonar and cetaceans: just how much does the gun need to smoke before we act? Mar. Pollut. Bull. 56, 1248-1257.

Payne, J.F., 1976. Field evaluation of benzopyrene hydroxylase induction as a monitor for marine petroleum pollution. Science 191, 945-946.

Payne, R. and Webb, D., 1971. Orientation by means of long range acoustic signalling in baleen whales. Ann. NY. Acad. Sci. 188, 110-141.

Pearce, D.J., Sivyer, D.B., Read, J., Greenwood, N., Platt, K, Rawlinson, M.B. and Mills, D.K., 2002. SmartBuoy: A marine environmental monitoring buoy with a difference, 158 KB, http://www.cefas.co.uk/data/marine-monitoring.aspx

Pemberton, D., Brothers, N.P. and Kirkwood, R., 1992. Entanglement of Australian fur seals in man-made debris in Tasmanian waters. Wildlife Res. 19, 151-159.

Perl, T.M., Bedard, L., Kosatsky, T., Hockin, J.C., Todd, E.C. and Remis, R.S., 1990. An outbreak of toxic encephalopathy caused by eating mussels contaminated with domoic acid. New England Journal of Medicine 322, 1775-1780.

Perrin, W.F., Wuersig, B. and Thewissen, J.G.M., 2002. Encyclopedia of Marine Mammals, Vol 1. Academic Press.

Popper, A.N., Salmon, M. and Horch, K.W., 2001. Acoustic detection and communication by decapod crustaceans. J. Comp. Physiol. A 187, 83-89.

Popper, A.N., Fay, R.R., Platt, C. and Sand, O., 2003. Sound detection mechanisms and capabilities of teleost fishes. In: Sensory Processing in Aquatic Environments. Edited by Collin, S.P. and Marshall, N.J. Springer Verlag, pp. 3-38.

Popper, A.N., Fewtrell, J., Smith, M.E. and McCauley, R.D., 2004. Anthropogenic sound: Effects on the behavior and physiology of fishes. Mar. Technol. Soc. J. 37, 35-40.

Popper, A.N., Halvorsen, M.B., Miller, D., Smith, M.E., Song, J., Wysocki, L.E., Hastings, M.C., Kane, A.S. and Stein, P., 2005a. Effects of SURTASS Low Frequency Active sonar on fish. J. Acoust. Soc. Am. 117, 2440.

Popper, A.N., Smith, M.E., Cott, P.A., Hanna, B.W., MacGillivary ,A.O., Austin, M. and Mann, D.A., 2005b. Effects of exposure to seismic airgun use on hearing of three fish species. J. Acoust. Soc. Am. 117, 3958-3971.

Popper, A.N., Carlson, T.J., Hawkins, A.D. and Southall, B.L., 2006. Interim criteria for injury of fish exposed to pile driving operations: a white paper. [available at: www.wsdot.wa.gov/NR/rdonlyres/84A6313A-9297-42C9-BFA6-750A691E1DB3/0/BA_PileDrivingInterimCriteria.pdf].

Popper, A.N., Halvorsen, M.B., Kane, A., Miller, D.L., Smith, M.E., Song, J., Stein, P. and Wysocki, L.E., 2007. The effects of high-intensity, low-frequency active sonar on rainbow trout. J. Acoust. Soc. Am. 122, 623-635.

Quilliam, M.A., Xie, M. and Hardstaff, W.R.. 1995. Rapid extraction and cleanup for liquid chromatographic determination of domoic acid in unsalted seafood. Journal of AOAC International 78, 543-554.

Read, A.J., Drinker, P. and Northridge, S.P., 2006. By-catches of marine mammals in U.S. and global fisheries. Conserv. Biol. 20, 163-169.

Rees, H.L., Waldock, R., Matthiessen, P. and Pendle, M.A., 2001. Improvements in the epifauna of the Crouch estuary (United Kingdom) following a decline in TBT concentrations. Mar. Pollut. Bull., 42, 137-144.

Rees, H.L., Boyd, S.E., Schratzberger, M. and Murray, A., 2006. Role of benthic indicators in regulating human activities. Environmental Science and Policy 9, 496-508.

Rees, H.L., Eggleton, J.D., Rachor, E. and Vanden Berghe, E. (Eds), 2007. Structure and dynamics of North Sea benthos. ICES Cooperative Report No. 288. International Council for the Exploration of the Sea, 258 pp.

Reichert, W.L., Myers, M.S., Peck-Miller, K., French, B., Anulacion, B.F., Collier, T.K., Stein, J.E. and Varanasi, U., 1998. Molecular epizootiology of genotoxic events in marine fish: linking contaminant exposure, DNA damage, and tissue-level alterations. Mutat. Res. 411, 215-225.

Reichert, W.L., French, B.L. and Stein, J.E., 1999. Biological effects of contaminants: Measurements of DNA adducts in fish by ^{32}P-postlabelling. ICES Tech. Mar. Environ. Sci. No. 25.

Reid, J.B., Evans, P.G.H. and Northridge, S.P., 2003. Atlas of cetacean distribution in north-west European waters. Joint Nature Conservation Committee. 76 pp.

Reiss, H., and Kroncke, I., 2005. Seasonal variability of infaunal community structures in three areas of the North Sea under different environmental conditions. Estuar. Coast. Shelf Sci. 65, 253-274.

Richards, S.D., Harland, E.J. and Jones, S.A.S., 2007. Underwater noise study supporting Scottish Executive Strategic Environmental Assessment for marine renewables, QinetiQ. [available at www.seaenergyscotland.net]

Richardson, W.J., Fraker, M.A., Würsig, B. and Wells, R.S., 1985. Behaviour of bowhead whales, *Balaena mysticetus* summering in the Beaufort Sea: reactions to industrial activities. Biol. Conserv. 32, 195-230.

Richardson, W.J., Malme, C.I., Green, Jr C.R. and Thomson, D.H., 1995. Marine Mammals and Noise, Vol 1. Academic Press

Roberts, S., 2002. Clean Coast Scotland Community Council Questionnaire Report, April 2002.

Robinson, C.D. and Scott, A.P., 2006. Draft OSPAR Background Document for Biological Effects Monitoring Techniques: Fish vitellogenin as a biomarker of exposure to xenoestrogens. Fisheries Research Services Collaborative Report No. 09/06., p. 29. Fisheries Research Services.

Rodhe, W., 1969. Crystallization of Eutrophication Concepts in North Europe. National Academy of Sciences.

Rowlatt, S.M. and Lovell, D.R., 1994. Lead, zinc and chromium in sediments around England and Wales. Mar. Pollut. Bull. 28, 324-329.

Rozan, T.F. and Hunter, K.S., 2001. Effects of discharge on silver loadings and transport in the Quinnipiac River, Connecticut. Sci. Total Environ. 279, 195-205.

Ryan, P.G., 1990. The effects of ingested plastic and other marine debris on seabirds. pp. 623-634. In: Proceedings of the Second International Conference on Marine Debris, 2-7 April 1989, Honolulu, Hawaii. NOAA Tech. Mem. NMFS, NOAA-TM-NMFS-SWFSC-154.

Sazykina, T.G. and Kryshev, I.L., 2003. MARINA II Annex A, Appendix F. Assessment of the impact of radioactive substances on marine biota of north European waters. European Commission, Radiation Protection 132.

Scanlan, C.M., Foden, J., Wells, E. and Best, M.A., 2007. The monitoring of opportunistic macroalgal blooms for the water framework directive. Mar. Pollut. Bull. 55, 162-171.

Schets, F.M., Harold, H.J.M., Berg, V.D., Engels, G.B., Lodder, W.J. and Husman, A.M.R., 2007. Cryptosporidium and Giardia in commercial and non-commercial oysters (*Crassostrea gigas*) and water from the Oosterschelde, the Netherlands. Int. J. Food Microbiol. 113, 189-194.

Schiewe, M.H., Weber, D.D., Myers, M.S., Jacques, F.J., Reichert, W.L., Krone, C.A., Malins, D.C., McCain, B.B., Chan, S.-L. and Varanasi, U., 1991. Induction of foci of cellular alteration and other hepatic lesions in English sole (*Parophrys vetulus*) exposed to an extract of an urban marine sediment. Can. J. Fish. Aquat. Sci. 48, 1750-1760.

Schropp, S.J., Lewis, F.G., Windom, H.L., Ryan, J.D., Calder, F.D. and Burney, L.C., 1990. Interpretation of metal concentrations in estuarine sediments of Florida using Aluminium as a reference element. Estuaries 13, 227-235.

Scott, T.M., Rose, J.B., Jenkins, T.M., Farrah, R. and Lukasik,, J., 2002. Microbial source tracking: current methodology and future directions. Mini-review. Appl. Environ. Microbiol. 68, 5796-5803.

Scott, A.P., Katsiadaki, I., Witthames, P.R., Hylland, K., Davies, I.M., McIntosh, A.D. and Thain, J., 2006a. Vitellogenin in the blood plasma of male cod (*Gadus morhua*): a sign of oestrogenic endocrine disruption in the open sea? Mar. Environ. Res. 61, 149-170.

Scott, A.P., Kristiansen, S.I., Katsiadaki, I., Thain, J., Tollefsen, K.E., Goksøyr, A. and Barry, J., 2006b. Assessment of oestrogen exposure in cod (*Gadus morhua*) and saithe (*Pollachius virens*) in relation to their proximity to an oilfield. In: Biological Effects of Contaminants in Marine Pelagic Ecosystems. Edited by Hylland, K., Lang, T. and Vethaak, D. pp. 329-339. SETAC Press.

Scott, A.P, Sanders, M. Stentiford, GD, Reese, RA, and Katsiadaki I., 2007. Evidence for estrogenic endocrine disruption in an offshore flatfish, the dab (*Limanda limanda* L.). Mar. Environ. Res. 64, 128-148.

Service, M. and Durrant, A.E., 1996. Belfast Lough Nutrient Budget. Report by Queen's University, Belfast & Department of Agriculture Northern Ireland for EHS.

Sheader, D.L., Williams, T.D., Lyons, B.P. and Chipman, J.K., 2006. Oxidative stress response of European flounder (*Platichthys flesus*) to cadmium determined by a custom cDNA microarray. Mar. Environ. Res. 62, 33-44.

Sheavely, S.B. and Register, K.M., 2007. Marine debris & plastics: environmental concerns, sources, impacts and solutions. J. Poly. Environ. 15, 301-305.

Skjoldal, H.R., Gool, S.V., Offringa, H., Dam, C.V., Water, J., Degré, E., Bastinck, J., Pawlak, J., Lassen, H., Svelle, M., Nilsen, H.-G. and Lorentzen, H., 1999. Workshop on Ecological Quality Objectives (EcoQOs) for the North Sea. Scheveningen - Netherlands. 1-3 September 1999, TemaNord, 1999:591.

Sleiderink, H.M., Beyer, J., Scholtens, E., Goksyr, A., Nieuwenhuize, J., Van Liere, J.M., Everaarts, J.M. and Boon, J.P., 1995. Influence of temperature and polyaromatic contaminants on CYP1A levels in North Sea dab (*Limanda limanda*). Aquat. Toxicol. 32, 189-209.

Smedes, F., 2007. Methods using passive sampling techniques in sediment for the estimation of porewater concentrations and available concentrations for hydrophobic contaminants, ICES Annual Science Conference, Helsinki, Finland, September 17-21, 2007, ICES CM 2007/ J: 07.

Somerville, S., Miller,K. and Mair, J., 2003. Assessment of the aesthetic quality of a selection of beaches in the Firth of Forth, Scotland. Mar. Pollut. Bull. 46, 1184-1190.

Southall, B.L., 2005. Shipping Noise and Marine Mammals: a Forum for Science, Management, and Technology. Final Report of the National Oceanic and Atmospheric Administration (NOAA) International Symposium. [www/nmfs.noaa.gov/pr/acoustics/shipnoise.htm]

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene, C.R.J., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A. and Tyack, P., 2007. Marine mammal noise exposure criteria: initial scientific recommendations. Aquat. Mamm. 33, 411-521.

Stein, J.E., Reichert, W.L., Nishimoto, M. and Varanasi, U., 1990. Overview of studies on liver carcinogenesis in English sole from Puget Sound; evidence for a xenobiotic chemical etiology II: Biochemical studies. Sci. Total Environ. 94, 51-69.

Stentiford, G.D. and Feist, S.W., 2005. First reported cases of intersex (ovotestis) in the flatfish species, dab (*Limanda limanda*): Dogger Bank, North Sea. Mar. Ecol. Prog. Ser. 301, 307-310.

Stentiford, G.D., Longshaw, M., Lyons, B.P., Jones, G., Green, M. and Feist, S.W., 2003. Histopathological biomarkers in estuarine fish species for the assessment of biological effects of contaminants. Mar. Environ. Res. 55, 137-159.

Stentiford, G.D., Johnson, P.J., Martin, A., Wenbin, W., Ward, D.G., Viant, M., Lyons, B.P. and Feist, S.W., 2005. Liver tumours in wild flatfish: a histopathological, proteomic and metabolomic study. OMICS: J. Integr. Biol. 9, 281-299.

Stentiford, G.D., Bignell, J.P., Lyons, B.P. and Feist, S.W., 2009. Site-specific disease profiles in fish and their use in environmental monitoring. Mar. Ecol. Prog. Series. 381, 1-15.

Stone, C.J. and Tasker, M.L., 2006. The effects of seismic airguns on cetaceans in UK waters. J. Cetac. Res. Manag. 8, 255-263.

Sunnotel, O., Snelling, W.J., McDonough, N., Browne, L., Moore, J.E., Dooley, J.S.G. and Lowery, C.J., 2007. Effectiveness of standard UV depuration at inactivating Cryptosporium parvum recovered from spiked Pacific oysters (*Crassostrea gigas*). Appl. Environ. Microbiol. 73, 5083-5087.

Swift, D.J. and Nicholson, M.D., 2001. Variability in the edible fraction content of cobalt-60, technetium-99, silver-110m, caesium-137 and americium-241 between individual crabs and lobsters from Sellafield (north eastern Irish Sea). J. Environ. Radioactiv. 54, 311-326.

Szymanski, M.D., Bain, D.E., Kiehl, K., Pennington, S., Wong, S. and Henry, K.R., 1999. Killer whale (*Orcinus orca*) hearing: Auditory brainstem response and behavioural audiograms. J. Acoust. Soc. Am. 106, 1134-1141.

Takada, H., Mato, Y., Endo, S., Yamashita, R. and Zakaria, M., 2006. Pellet Watch: Global monitoring of persistent organic pollutants using beached plastic resin pellets.

Tett, P., Portilla, E., Inall, M., Gillibrand, P., Gubbins, M. and Amundrud, T., 2007a. Modelling the assimilative capacity of sea lochs (final report on SARF 12). Scottish Aquaculture Research Forum Report 12, 29pp.

Tett, P., Gowen, R., Mills, D., Fernandes, T., Gilpin, L., Huxham, M., Kennington, K., Read, P., Service, M., Wilkinson, M. and Malcolm, S., 2007b. Defining and detecting undesirable disturbance in the context of marine eutrophication. Mar. Pollut. Bull. 55, 282-297.

Tett P., Carreira, C., Mills, D.K., van Leeuwen, S., Foden, J., Bresnan, E., and Gowen, R.J. (2008). Use of a Phytoplankton Community Index to assess the health of coastal waters. ICES J. Mar. Sci. 65, 1475-1482.

Teuten, E.L., Rowland, S.J., Galloway, T.S. and Thompson, R.C., 2007. Potential for plastics to transfer hydrophobic contaminants. Environ. Sci. Technol. 41, 7759-7764.

Thain, J. and Bifield, S., 2001. Biological effects of contaminants: Sediment bioassay using the polychaete *Arenicola marina*. ICES TIMES no. 29. 16pp.

Thain, J. and Roddie, B., 2001. Biological effects of contaminants: *Corophium* sp. sediment bioassay and toxicity test. ICES TIMES no. 28. 21pp.

The Crown Estate, 2006a. Marine Aggregates: Crown Estate Licenses - Summary of Statistics 2005.

The Crown Estate, 2006b. Review of UK marine aggregate extraction activities: Historic patterns of marine aggregate extraction (tonnes) 2000-2005.

Thomas, K.V., Balaam, J., Barnard, N., Dyer, R., Jones, C., Lavender, J. and McHugh, M., 2002. Characterisation of potentially genotoxic compounds in sediments collected from United Kingdom estuaries. Chemosphere 49, 247-258.

Thomas, K.V., Balaam, J., Hurst, M.R., Nedyalkova, Z. and Mekenyan, O., 2004. Potency and characterization of estrogen-receptor agonists in United Kingdom estuarine sediments. Environ. Toxicol. Chem. 23, 471-479.

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D. and Russell, A.E., 2004. Lost at sea: Where does all the plastic go? Science 304, 838.

Thomsen, F., Laczny, M. and Piper, W., 2006a. A recovery of harbour porpoises (*Phocoena phocoena*) in the southern North Sea? A case study off Eastern Frisia, Germany. Helgoland Marine Research 60, 189-195.

Thomsen, F., Lüdemann, K., Kafemann, R. and Piper, W., 2006b. Effects of offshore wind farm noise on marine mammals and fish, biola, Hamburg, Germany on behalf of COWRIE Ltd, Newbury, UK.

Thomsen, F., McCully, S.R., Weiss, L., Wood, D., Warr, K., Kirby, M., Kell, L. and Law, R., 2008. Cetacean stock assessment in relation to exploration and production industry sound: current knowledge and data needs. Report for E&P Sound and Marine Life Programme International Association of Oil and Gas Producers.

Thomsen, F., McCully, S.R., Wood, D., White, P. and Page, F., 2009. A generic investigation into noise profiles of marine dredging in relation to the acoustic sensitivity of the marine fauna in UK waters: PHASE 1 Scoping and review of key issues, Aggregates Levy Sustainability Fund / Marine Environmental Protection Fund (ALSF/MEPF), Lowestoft, UK.

Tougaard, J., Carstensen, J., Henriksen, O.H., Skov, H. and Teilmann, J., 2003a. Short-term effects of the construction of wind turbines on harbour porpoises at Horns Reef. Technical report to Techwise A/S. Hedeselskabet.

Tougaard, J., Ebbesen, I., Tougaard, S., Jensen, T. and Teilmann, J., 2003b. Satellite tracking of harbour seals on Horns Reef. Use of the Horns Reef wind farm area and the North Sea. Report request. Commissioned by Tech-Wise A/S. Fisheries and Maritime Museum, Esbjerg, Denmark.

Tougaard, J., Carstensen, J., Teilmann, J., Bech, N.I., Skov, H. and Henriksen, O.D., 2005. Effects of the Nysted Offshore wind farm on harbour porpoises. Technical Report to Energi E2 A/S. NERI, Roskilde, Denmark.

Traynor, I.M., Plumpton, L., Fodey, T.L., Higgins, C. and Elliott, C.T., 2006. Immunobiosensor detection of domoic acid as a screening test in bivalve molluscs: Comparison with liquid chromatography-based analysis. J. AOAC Int. 89, 868-872.

Tyack, P.L. and Clark, C.W., 2000. Communication and acoustic behavior of dolphins and whales. In: Hearing by Whales and Dolphins. Edited by Au, W., Popper, A.N. and Fay, R. Springer Handbook of Auditory Research Series. pp. 156-224.

Tysklind, N., Taylor, M.I., Lyons, B.P., McCarthy, I.D. and Carvalho, G.R., 2009. Development of 30 microsatellite markers for dab (*Limanda limanda* L.): a key UK marine biomonitoring species. Mol. Ecol. Resour. 9, 951-955.

Urick, R., 1983. Principles of underwater sound, Vol 1. McGraw Hill.

Van der Oost, R., Van Schooten, F.J., Ariese, F. and Heida, H., 1994. Bioaccumulation, biotransformation and DNA binding of PAHs in feral eel (*Anguilla anguilla*) exposed to polluted sediments. Environ. Toxicol. Chem., 13, 859-870.

Van Egmond, H.P., Aune, T., Lassus, P., Speijers, G.J.A. and Waldock, M., 1993. Paralytic and diarrhoeic shellfish poisons: occurrence in Europe, toxicity, analysis and regulation. J Natural Toxins 2, 41-83.

Van Franeker, J.A. and S.N.S. Fulmar Study Group, 2008. Fulmar Litter EcoQO Monitoring in the North Sea – Results to 2006. Wageningen IMARES Report No. C033/08, IMARES Texel, 53 pp.

van Parijs, S.M. and Corkeron, P.J., 2001. Boat traffic affects the acoustic behavior of Pacific humpback dolphins, *Sousa chinensis*. J. Mar. Biol. Assoc. UK 81, 533-538.

van Weers, A.W., 2003. MARINA II Annex A, Appendix C. NORM inputs from offshore oil and gas production. European Commission, Radiation Protection 132.

Vethaak, A.D., Jol, J.G., Meijboom, A., Eggens, M.L., Reinhalt, T., Wester, P.W., van de Zande, T., Bergman, A., Dankers, N., Ariese, F., Bann, R.A., Everts, J.M., Opperhuizen, A. and Marquenie, J.M., 1996. Skin and liver diseases induced in flounder (*Platichthys flesus*) after long-term exposure to contaminated sediments in large scale mesocosms. Environ. Health Perspect. 104, 1228-1229.

Wahlberg, M. and Westerberg, H., 2005. Hearing in fish and their reactions to sound from offshore wind farms. Mar. Ecol. Prog. Ser. 288, 295-309.

Wang, X.L., Zwiers, F.W. and Swail, V.R., 2004. North Atlantic Ocean wave climate change scenarios for the 21st century. J. Clim. 17, 2368-2383.

Ward, D.G., Wei, W., Cheng, Y., Billingham, L.J., Martin, A., Johnson, P.J., Lyons, B.P., Feist, S.W. and Stentiford, G.D., 2006. Plasma proteome analysis reveals the geographical origin and liver tumor status of dab (*Limanda limanda*) from U.K. marine waters. Environ. Sci. Technol. 40, 4031-4036.

Webster, L., McIntosh, A.D., Dalgarno, E.J., Megginson, C., Shepherd, N.J. and Moffat, C.F., 2003. The polycyclic aromatic hydrocarbon composition of mussels (*Mytilus edulis*) from Scottish coastal waters. J. Environ. Monit., 5, 150-159.

Webster, L., Russell, M., Walsham, P. and Moffat, C.F., 2006. A review of brominated flame retardants (BFRs) in the aquatic environment and development of an analytical technique for their analysis in environmental samples, FRS Internal Report No 06/06.

Weilgart, L., 2007. The impacts of anthropogenic ocean noise on cetaceans and implications for management. Can. J. Zool. 85, 1091-1116.

Wenz, G.M., 1962. Acoustic ambient noise in the ocean: Spectra and sources. J. Acoust. Soc. Am. 34, 1936-1956.

Weston, K., Jickells, T.D., Fernand, L. and Parker, R., 2004. Nitrogen cycling in the southern North Sea: consequences for total nitrogen transport. Estuar. Coast. Shelf Sci. 59, 559-573.

Weston, K., Greenwood, N., Fernand, L., Pearce, D.J. and Sivyer, D.B., 2008a. Environmental controls on phytoplankton community composition in the Thames plume, U.K. J. Sea Res. 60, 262-270.

Weston, K., Fernand, L., Nicholls, J., Marca-Bell, A., Mills, D., Sivyer, D. and Trimmer, M., 2008b. Sedimentary and water column processes in the Oyster Grounds: a potentially hypoxic region of the North Sea. Mar. Environ. Res. 65, 235-249.

WHO, 2001. Bathing Water Quality and Human Health: Faecal Pollution. World Health Organisation.

WHO. 2003. Guidelines for safe recreational water environments. Volume 1, Coastal and fresh waters.

Whomersley, P., Schratzberger, M., Huxham, M., Bates, H. and Rees, H.L., 2007. The use of time series data in the assessment of macrobenthic community change after the cessation of sewage sludge disposal in Liverpool Bay (UK). Mar. Pollut. Bull. 54, 32-41.

Whyte, J.J., Jung, R.E., Schmitt, C.J., and Tillitt, D.E., 2000. Ethoxyresorufin-*O*-deethylase (EROD) activity in fish as a biomarker of chemical exposure. Crit. Rev. Toxicol. 30, 347-570.

Williams, R., Bain, D.E., Ford, J.K.B. and Trites, A.W., 2002. Behavioural responses of male killer whales to a 'leapfrogging' vessel. . J. Cetac. Res. Manag. 4, 305-310.

Windom, H.L., Schropp, S.J., Calder, F.D., Ryan, J.D., Smith, R.G., Burney, L.C., Lewis, F.G. and Rawlinson, C.H., 1989. Natural trace metal concentrations in estuarine and coastal marine sediments of the Southeastern United States. Environ. Sci. Technol. 23, 314-320.

Wittman, R.J. and Flick, G.J., 1995. Microbial Contamination of Shellfish: Prevalence, Risk to Human Health, and Control Strategies. Ann. Rev. Public Health 16, 123-140.

Wood, M.D., Marshall, W.A., Beresford, N.A., Jones, S.R., Howard, B.J., Copplestone, D. and Leah, R.T., 2008. Application of the ERICA integrated approach to the Drigg coastal sand dunes. J. Environ. Radioactiv. 99, 1484 -1495.

Woodhead, R.J., Law, R.J. and Matthiessen, P., 1999. Polycyclic aromatic hydrocarbons in surface sediments around England and Wales, and their possible biological significance. Mar. Pollut. Bull. 38, 773-790.

Wosniok, W., Lang, T., Dethlefsen, V., Feist, S.W., McVicar, A.H., Mellergaard, S. and Vethaak, A.D., 2000. Analysis of ICES long-term data on diseases of North Sea dab (*Limanda limanda*) in relation to contaminants and other environmental factors. ICES CM 2000/S:12. 15 pp.

Würsig, B. and Richardson, W.J., 2002. Effects of noise. In: Encyclopedia of Marine Mammals. Edited by Perrin, W.F., Würsig, B. and Thewissen, J.G.M. Academic Press, pp. 794-802.

Yasumoto, T., Murata, M., Oshima, Y., Matsumoto, G. and Clardy, J., 1984. Diarrhetic shellfish poisoning. In: Seafood Toxins. American Chemical Society, Symposium Series, 262.

Young, A.K., McCubbin, D., Thomas, K., Camplin, W.C., Leonard, K.S. and Wood, N., 2003. Po-210 concentrations in UK seafood. RSC, ERA II, pp. 143-149.

Zimmer, W.M.X., 2004. Sonar systems and stranding of beaked whales. In: Proceedings of the workshop on active sonar and cetaceans. ECS Newsletter Special Issue No 42. European Cetacean Society pp. 8-13.

Zou, T., Taylor, M., DeVito, M.J. and Crofton, K.M., 2002. Developmental exposure to brominated diphenyl ethers results in thyroid hormone disruption. Toxicol. Sci. 66, 105-116.

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