UK National Marine Monitoring Programme - Second Report (1999-2001)

Marine Environment Monitoring Group

2004

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UK National Marine Monitoring Programme - Second Report (1999-2001)

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EXECUTIVE SUMMARY

The National Marine Monitoring Programme (NMMP) was initiated by the Marine Environment Monitoring Group (MEMG), formerly known as the Marine Pollution Monitoring Management Group (MPMMG), in the late 1980s to co-ordinate marine monitoring in the UK. The programme was designed to fulfil the UK's mandatory monitoring requirements under the OSPAR Joint Assessment and Monitoring Programme (JAMP) and it also provides data in support of EC Directives.

In recent years the policy underpinning marine monitoring has changed considerably with new and emerging requirements from both OSPAR and EC Directives such as the Water Framework Directive (2000/60/EC). At the same time, the creation of the Department for Environment, Food and Rural Affairs (Defra) in 2001 brought together a number of Government Departments with interests in marine science, providing an opportunity to maximise and co-ordinate Government funded marine science. The Department produced its first Marine Stewardship Report, 'Safeguarding Our Seas' (Defra, 2002), which recognised the importance of an ecosystem-based approach to management of the marine environment in delivering its vision of clean healthy seas. The report sets out a framework to deliver this vision through improved co-ordination of UK marine monitoring, with the aim of supporting more integrated and coherent assessments of the state of the UK marine environment at regular intervals. The first integrated assessment, in the form of a State of the Seas Report, will be produced in 2004.

As part of a review undertaken by the MEMG on behalf of Defra, a strategy for a harmonised and integrated programme of marine environmental monitoring has been set out (MEMG, 2003). This document summarises the key findings and conclusions from the review and presents a number of recommendations which will be implemented by the NMMP Working Group, particularly as it works towards up-dating its operational handbook (the NMMP Green Book). This second report on the National Marine Monitoring Programme is therefore very timely.

Data collected in the first phase of the programme (1992-1995) were reported in 1998 (MPMMG, 1998). Part 1 of this report presents data collected in the second phase (1999-2001). Part 2 of the report presents data collected under a number of other marine monitoring programmes, as case studies. These provide a wider view of trends and distributions of contaminants in the marine environment, presenting a more complete picture of the quality of UK waters.

The second phase of the NMMP initiated long term trend monitoring at selected sites and was modified following the recommendations of the spatial survey (Phase 1):

- The number of replicate samples was optimised to enable trends to be detected without incurring excessive costs;

- Eutrophication monitoring requirements were maintained at levels consistent with the OSPAR Nutrients Monitoring Programme pending the outcome of the application of the OSPAR Comprehensive (eutrophication assessment) Procedure in UK marine waters;

- The programme was expanded to include measures of biological effects at selected sites;

- The analysis of polycyclic aromatic hydrocarbons (PAH) in sediments and mussels was introduced on a routine basis;

- Contaminant monitoring in water was limited to the provision of data from the reference sites monitored for the Dangerous Substances Directive.

Procedures for screening chemical and benthic data were developed to ensure acceptable quality of data submitted to the NMMP. An Ecotoxicological Analytical Quality Control group was established to manage quality control for the expanded range of biological effects techniques included in this second phase of the NMMP.

The 'Driving Force, Pressure, State, Impact and Response' (DPSIR) model has been used as far as possible to structure this report. Trends in inputs of contaminants to UK marine waters as estimated by the OSPAR Riverine Inputs and Direct Discharges (RID) programme are presented to indicate the pressures. Contaminant data are assessed against national and international criteria to describe the state of the marine environment. Biological effects measured on individual organisms and by modification of benthic community structure indicate the impacts of contaminants. Notably, this is the first UK NMMP report that uses tentative Background Reference Concentrations (BRCs) and Ecotoxicological Assessment Criteria (EACs) to evaluate present levels of contamination against pristine conditions. The results must be treated with a great deal of caution because BRC values do not yet reflect regional differences in geochemistry and EAC values have been extrapolated from limited datasets.

The results show that:

- Water borne loads (direct and riverine inputs) of measured contaminants to coastal waters have decreased since 1990;

- Riverine inputs of nitrogen appear to be increasing. However, these nitrogen inputs are strongly correlated with riverine flow and, since 1997, flows have been significantly above the long-term average. When this skewed distribution in flow rates is taken into account, there would appear to be no underlying change in the riverine inputs of nitrogen;

- The highest nitrate concentrations inevitably occur in coastal waters subject to most dilution by nutrient rich river waters, such as the eastern Irish Sea where the size of the river inputs and restricted circulation combine to reduce dilution;

- Contaminant concentrations in waters vary in relation to tidal state and changes in discharges. Concentrations are generally low and not toxic to sensitive test organisms. Higher concentrations are recorded where there are known inputs and impacts are measured occasionally on sensitive test organisms at some sites;

- Historical inputs of contaminants to estuaries still lead to elevated concentrations of mercury, cadmium, Polychlorinated Biphenyls (PCBs) and PAHs in some estuarine sediments. Concentrations are lower at coastal sites. Benthic community structure was determined more by environmental factors than contaminant concentrations. However, localised areas of contaminated sediments in industrialised estuaries were found to be toxic to test organisms;

- Highest concentrations of contaminants in biota continue to be found in industrialised estuaries and those adjacent coastal areas with a known history of contaminant input. Biological effects, such as induction of increased enzyme activity (e.g. EROD), were found in some fish populations from these areas. Imposex in dogwhelks, induced by exposure to tributyl tin (TBT), remains widespread, but reproduction was only impaired significantly in the vicinity of harbours and marinas;

- The data collected to date show that the power of the present programme to detect trends is variable between contaminant and matrix. This information will be used to refine the programme so that, for example, sampling may be increased at 'key' monitoring stations to maximise returns on sampling effort.

In the light of these results, the following amendments to the programme are proposed:

- The programme will be adapted to fulfil future monitoring requirements arising from changes to the JAMP programme and the new EC Water Framework Directive;

- The nutrient monitoring programme will be revised to align more closely with the OSPAR criteria;

- Where contaminant concentrations in sediment exceed assessment criteria levels, sediment bioassays will be used to assess any impact on biota;

- The biological effect of metals and other contaminants on mussels will be assessed through the measurement of metallothionein and scope for growth;

- The programme will be redesigned to target effort at identified ‘key’ monitoring sites.

1. INTRODUCTION

The National Marine Monitoring Programme (NMMP) was initiated in the late 1980s to provide an overview of the quality of the marine environment of the United Kingdom. It was devised by the Marine Environment Monitoring Group (MEMG) (Table 1) and is fulfilled by collaboration between fisheries organisations and environment protection agencies.

The general aims of the NMMP (NMMPWG, 2001) are to:

- Detect long-term trends in physical, biological and chemical variables at selected estuarine and coastal sites
- Report the spatial and temporal distributions of these variables and their inter-relationships
- Support and ensure consistent standards in national and international monitoring programmes for marine environmental quality
- Make recommendations to MEMG as to how new analyses and techniques are best implemented in the United Kingdom
- Co-ordinate, make optimum use of, and gain maximum information from marine monitoring in the United Kingdom
- Provide and maintain a high quality dataset for key chemical and biological variables in the marine environment of the United Kingdom, which is available for national and international fora such as the International Council for the Exploration of the Sea and the European Environment Agency.

In particular, the NMMP is designed to:

- meet international marine monitoring requirements such as the Oslo and Paris Commission (OSPAR) Joint Assessment and Monitoring Programme (JAMP) and Nutrients Monitoring Programme (NEUT)
- collate relevant data gathered for EC Dangerous Substances Directive (76/464/EEC), Shellfish Waters Directive (79/923/EEC), Shellfish Hygiene Directive (91/492/EEC), Urban Waste Water Treatment Directive (91/271/EEC) and Nitrates Directive (91/676/EEC)
- present national data of a consistent quality assured by rigorous Analytical Quality Control (AQC)

The first phase of the NMMP focussed on known impacted estuaries and showed the distribution of contaminants and their effects on sediment dwelling organisms and fish in UK marine waters (MPMMG, 1998).

The second phase began in 1999 and is designed to investigate

- temporal trends in contaminants in estuarine and coastal sites
- the spatial distribution of new contaminants
- the further development of the biological effects programme

Part 1 of this report summarises the results of the first 3 years of temporal monitoring (1999-2001). Although trends are unlikely to be seen after only three years, the data allow an assessment

Table 1. Members of the Marine Environment Monitoring Group

Department/Devolved Administration	• Department for Environment, Food and Rural Affairs (Defra) • National Assembly for Wales (NAW) • Scottish Executive Environment and Rural Affairs Department (SEERAD)
Fisheries/Marine Environment Organisations	• Centre for Environment, Fisheries and Aquaculture Science (CEFAS) • Fisheries Research Services (FRS) • Department of Agriculture and Rural Development for Northern Ireland (DARD)
Environment Protection Agencies	• Environment Agency (EA) • Scottish Environment Protection Agency (SEPA) • Environment and Heritage Service (EHS)
Conservation	• Joint Nature Conservation Committee (JNCC)
Others	• Inter Agency Committee on Marine Science and Technology (IACMST) • Sir Alistair Hardy Foundation for Ocean Sciences (SAHFOS)
MEMG Corresponding members	• Department of Health (DoH) • Department of Trade and Industry (DTI) • Maritime and Coastguard Agency (MCA) • Natural Environment Research Council (NERC) • Water Research Centre (WRC) • Some water companies in England and Wales

of current contaminant levels and indicate the likely performance of the programme in detecting trends in the future. The report is timely since the NMMP is being updated to account for changes in monitoring requirements such as the introduction of a new JAMP and the implementation of the Water Framework Directive (2000/60/EC). The results will be used to ensure that any revisions to the NMMP make best use of the available resources.

Part 2 of this report presents a series of case studies that, where possible, provide integrated assessments relating environmental change to changes in human activities. Some of the studies include data collected before 1999 in order to examine long-term trends. Others describe spatial surveys of new determinands that might be added to the temporal monitoring programme in response to changing legislation and chemical usage.

2. METHODS

The full sample plan and detailed methods are outlined in an operational manual (NMMPWG, 2001). Monitoring requirements are summarised in Table 2.1 and sampling sites are shown in Figure 2.1.

2.1 Components of the NMMP

The NMMP seeks to co-ordinate monitoring of chemical contaminants with biological effects and provide an integrated assessment of the quality of the marine environment, leading to improved environmental management.

2.2 Contaminant monitoring

Contaminants are measured where relevant in waters, sediments and biota to assess their distribution and fate in the environment.

2.2.1 Waters

The main drivers for monitoring seawater are the EC Dangerous Substances Directive and the OSPAR Nutrient Monitoring programme. Samples were collected quarterly to comply with Dangerous Substances Directive requirements and nutrient samples were collected during winter months (December -March) to establish maximum concentrations.

Seawater monitoring provides a 'snap-shot' of environmental quality that depends on tidal movements, inputs and rate of removal. Increasing the number of samples provides a better picture of environmental quality but is not always practicable. *In situ* instrumentation is being used to collect continuous nutrient information at two sites in the Irish and North Sea.

2.2.2 Sediments

When particles carried by rivers meet saline conditions, their surface charge changes and they drop on to the sediment. Estuarine sediments therefore act as sinks for contaminants.

Five replicate samples were collected at each site for analysis of trace metals and organic compounds.

Table 2.1 Monitoring required

Determinand	sediments	shellfish	fish tissue	fish liver	filtered water	unfiltered water
Chemical contaminants						
Nutrients					√	
Metals	√	√	√	√	√	
PCBs	√	√		√		
Organochlorine pesticides		√				√
PAH	√	√				
Biological effects						
benthic macrofauna	√					
Imposex (TBT)		√				
EROD				√		
Oyster Embryo Bioassay						√
Fish disease			√	√		
Metallothionein				√		
DNA adducts				√		

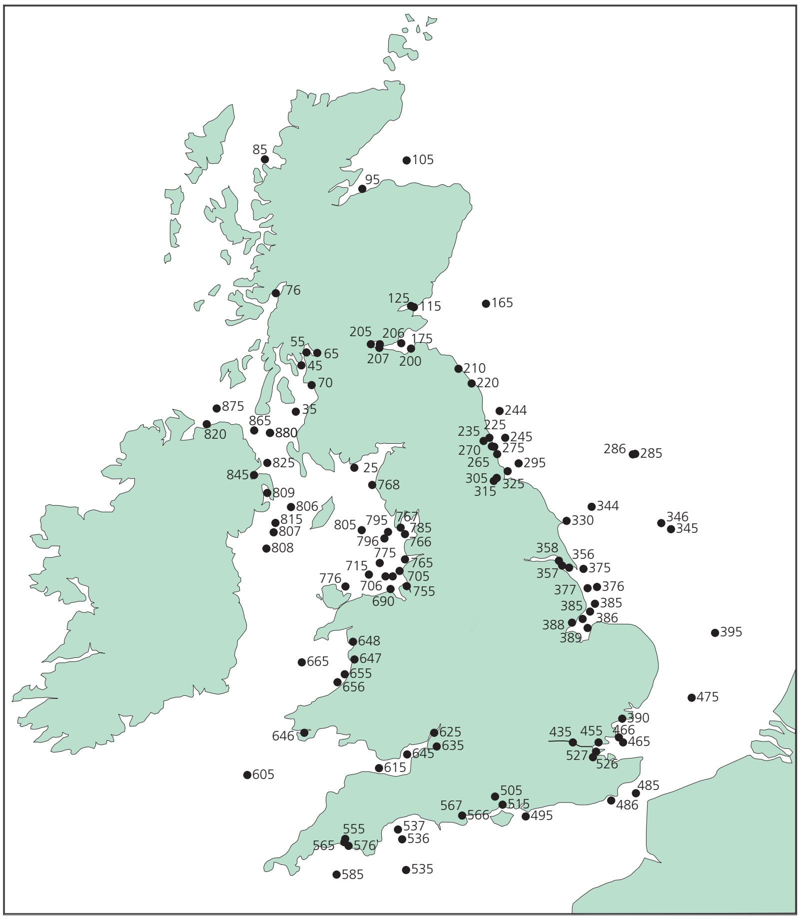


Figure 2.1 Location of sampling sites

Five replicate samples were also collected for benthic community structure analysis at the same time (Section 2.3). Samples were collected annually between February and June. Where possible, the fine fraction (<63 µm) was analysed for trace metals to minimise spatial variability due to particle size. However existing sample preparation methods were used to maintain an existing time series at some sites. Trace metals were extracted using a total digest. Polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) were generally measured on the <2 mm fraction as some volatile compounds may be lost by sieving. Analytical techniques were not proscribed but should achieve the required accuracy and precision stated in the manual (NMMPWG, 2001).

2.2.3 Biota

Biota ingest and absorb contaminants from their surroundings. Contaminants that cannot be excreted remain in the body and are accumulated. Body burdens of contaminants, which are accumulated over time, integrate fluctuations in contaminant concentrations in overlying waters.

- **Shellfish**

The target species is the common mussel (*Mytilus edulis*). Mussels accumulate contaminants from ingested particulate material suspended in overlying waters and across their gill surface. Metal uptake is influenced by mussel age, size, sex and shore position, water temperature, pH, salinity, and ability to absorb or excrete the contaminant (Miller, 1986). The sampling protocol was designed to limit variation in these variables. Three pools of at least 20 individual mussels in the size range 3-6 cm were analysed at each site for a wide range of trace metals and trace organic compounds. Contaminant concentrations vary seasonally, so samples were collected early in the year (February – March) prior to spawning.

- **Fish**

The target species are dab (*Limanda limanda*) and flounder (*Platichthys flesus*). Other species are acceptable if dab and flounder are not present. Fish are mobile and exposed to contaminants mainly through their food but will also accumulate contaminants from the water column. Five pools of five fish were collected from a specific size range to minimise variation due to age. Sampling avoided the spawning season to eliminate seasonal variations. Mercury and arsenic were measured in the flesh. Other contaminants (e.g. PCBs) are lipophilic (i.e. they accumulate in fat) and these were measured in the liver.

2.3 Monitoring of biological effects

Biological effects measure the response of organisms to contaminants, rather than the levels of the contaminants themselves. Biological effects monitoring thus might give a clearer picture of the health and quality of the marine environment. Some techniques are pollutant specific (e.g. imposex in dogwhelks due to tributyl tin (TBT) pollution), and some are more general measures (e.g. fish disease studies).

2.3.1 Benthic macrofauna

Benthic macrofauna are the animals living within (infauna) or on (epifauna) the surface of the sediments. Benthic organisms are effectively sessile so act as integrators of the effects of environmental stresses, whether natural, like salinity changes, or anthropogenic. Benthic communities have long been studied as a measure of environmental quality (Holme and McIntyre, 1971).

2.3.2 Oyster embryo bioassay

Young oysters undergo intense cellular activity during the early stages of development. The presence of contaminants in the surrounding environment adversely affects development of their normal shape. Embryos are exposed to environmental and clean water samples and their development is compared. The difference gives a measure of overall water quality.

2.3.3 Fish disease studies

Sampling and disease reporting followed protocols set up by the International Council for the Exploration of the Seas (ICES) (Bucke *et al.*, 1996). Target fish species are dab (*L. limanda*) and cod (*Gadus morhua*) for offshore locations and the European flounder (*P. flesus*) for inshore or estuarine stations. Fish were examined for both externally visible diseases and internal diseases. Internal lesions indicating exposure to contaminants include foci of cellular alteration (FCA), and benign and malignant tumours.

2.3.4 Imposex studies

The imposex condition has been strongly linked with the presence of organotins in the environment (Gibbs *et al.*, 1987). Imposex is an abnormality in gastropod molluscs in which male sexual characteristics are imposed on the genital systems of females. The female develops a penis and a *vas deferens*, which may block the genital opening so that egg capsules cannot be laid. Affected populations gradually decline and may become eliminated.

2.3.5 The mixed function oxidase test (EROD)

EROD or Ethoxyresorufin-O-deethylase activity provides an indication of the presence of a range of organic contaminants in fish. When absorbed, these contaminants induce synthesis of enzymes known as the cytochrome P450 group. Cytochrome P4501A1 (CYP1A1) is the terminal component and EROD activity is CYP1A1 dependent.

2.3.6 Metallothionein

Marine organisms produce metallothioneins (MT) to convert the toxic forms of some metals into less toxic, bound forms. The bound forms can then be safely stored and may be excreted. The presence of MT indicates exposure to these metals, rather than any impact of exposure on the well-being of the animal. MT is usually measured in the liver of fish (and related organs in invertebrates), as the liver normally contains the highest concentrations of inducing metals.

2.3.7 DNA adducts

DNA adducts are formed when reactive chemicals bind to DNA. The presence of DNA adducts is linked to PAH exposure and the presence of cancerous growths in fish liver.

2.4 Indicators

Indicators summarise complex information or raw data and can be used to assess the effectiveness of alternative policy options and keep policy makers and the public better informed. Under the NMMP, the MEMG has been developing indicators for hazardous substances, eutrophication and litter, as part of a wider initiative to develop a suite of marine indicators to report on the state of the UK marine environment and to measure changes. Some of these indicators have been used for the first time in this report.

2.4.1 Indicators of chemical quality

- Concentrations of hazardous substances compared to Environmental Quality Standards (EQSs)

EQSs are concentrations below which a substance is not believed to be detrimental to aquatic life. These were originally developed for the EC Dangerous Substances Directive (76/464/EEC). The concept is now well established and is incorporated into the Water Framework Directive (2000/60/EC). EQSs are derived using acute toxicity tests on organisms at different trophic levels. To provide a safety factor, the EQS is set substantially below the concentration observed to have a toxic effect on the test organisms. EQSs vary for each substance and can be different for fresh, estuarine or coastal waters. EQSs for the most toxic substances (List 1 EC Dangerous Substances Directive or Annex 1 substances in the EC Water Framework Directive) are set at a European level. The EQSs for less toxic substances are set nationally.

EQSs for water have been used in this report, however it should be noted that these are currently under review, for 33 substances, under the Water Framework Directive. EQSs have not yet been developed for sediments and biota where the 'EQS' is simply a standstill clause (i.e. no upward trends in concentrations). This presents a difficulty when interpreting data in sediments and biota and is recognised as an area that needs further development.

- **Concentrations of hazardous substances compared to Background Reference Concentrations (BRCs)**

BRCs were adopted by OSPAR in 1997 for contaminants in seawater, sediment and biota, as assessment tools for use in Quality Status Reports. BRCs were developed by examining typical concentrations of both naturally occurring and man-made contaminants in remote parts of the OSPAR maritime area. In general, man-made substances are expected to have a background concentration of zero. However, due to their persistence and long-range transport, many substances are detected in remote areas. For naturally occurring substances, the BRC is the range of concentrations that would be anticipated in the absence of any human activity.

Assessments made against the current OSPAR BRCs should be treated with caution. Reservations have been expressed over the current BRCs, due to the difficulties in deriving and applying such assessment criteria over wide geographical areas. In particular, BRC values need to take account of natural variability, for example, in the source rocks for coastal sediments. OSPAR has recognised that the current values require revision and a review is underway at present

- **Concentrations of hazardous substances compared to Ecotoxicological Assessment Criteria (EACs)**

EACs were also adopted by OSPAR in 1997. EACs are the concentrations of substances above which there may be impacts on biota. They are used to identify potential areas of concern and to prioritise substances for attention. The concepts behind EACs and EQSs are similar, however EACs exist for a number of substances in sediments and biota.

As with BRCs, assessments made against current EACs should be treated with extreme caution. Concentrations of a contaminant below the EAC for that contaminant do not guarantee a safe situation. On the other hand, it is not compelling that biological effects occur where an EAC is exceeded. This can only be established through biological investigations in the field. Current EACs are being reviewed along with BRCs. It seems likely that existing EACs will no longer be endorsed, being replaced by new criteria developed using the improved methodologies now available for effects assessment.

2.4.2 Indicators of biological quality

Benthic community data are summarised using the following:

- **Univariate Indicators**

These summarise the fauna composition. Examples are the number of taxa (T), abundance of taxa (A), evenness of taxa distribution (J'), and diversity measured by the Shannon-Wiener index (H').

- **Multivariate Indicators**

Multivariate statistics examine the similarity between different sites in terms of species composition. These similarities can then be examined against environmental variables to link changes in species composition with pollution.

- **Biotic Indices**

These summarise the way the fauna functions. Two biotic indices are used in this report: the UK Infaunal Trophic Index (ITI) and the AZTI Marine Biotic Index (AMBI). The ITI was developed to distinguish the impact of organic deposits from municipal sewage discharges on infauna in fully marine environments.

Biological effects indicators are being developed. A recent workshop examined the JAMP suite of biological effects techniques, the data sets available and their suitability for indicator reporting. It concluded that imposex in dogwhelks, fish disease, EROD, oyster embryo bioassay and sediment bioassays were suitable for immediate development for indicator reporting, but that bile metabolites and metallothionein still require substantial development.

2.5 Framework for indicators

The framework adopted for the development of marine indicators is the DPSIR model, currently used by the European Environment Agency (EEA). The framework assumes cause-effect relationships between interacting components of social, economic, and environmental systems, which are

- Driving forces of environmental change (e.g. industrial production)

Driving force	Pressure	State	Impact	Response
Shipping/transport Industry (including offshore) Agriculture Horticulture	Inputs from point (dredged material/ paint or effluent disposal) and diffuse sources (movement of ships)	Concentrations in sediment or biota	Species effects (imposex) and Population effects	reduction at/of sources

- Pressures on the environment (e.g. discharges of waste water)
- State of the environment (e.g. water quality in estuarine and coastal waters)
- Impacts on population, economy, ecosystems (e.g. reduction in benthic diversity)
- Response of the society (e.g. improved environmental protection)

The DPSIR model has been used to link the driving forces and pressures with observations in the marine environment. Some case studies in Part II of this report have used the model.

The aim of such an approach is:
- to provide information on all of the different elements in the DPSIR chain
- to demonstrate how these elements are interconnected &
- to measure/estimate the effectiveness of management responses.

An example of how the DPSIR model can be used in relation to the problem of TBT pollution from antifouling applications from ships is given above.

The development of marine indicators in the UK is still in its early stages. Before formal adoption, further testing and evaluation is required for some of the proposed indicators and further research and development for others.

It is anticipated that more indicators will be introduced into future NMMP reports and other reports on the state of the UK marine environment.

3. QUALITY CONTROL AND DATA ASSESSMENT

3.1 Chemistry

The remit of the NMCAQC group is to provide quality assurance (QA) for all marine chemistry data submitted as part of the NMMP programme. The group maintains and, where necessary, takes steps to improve the current level of QA. This is achieved through a combination of external and internal AQC programmes. The group objectively evaluates laboratories' performance, and ensures that where necessary, appropriate measures such as workshops are put in place to enable laboratories to improve their performance. The group also ensures that QA is in place for new determinands introduced to the NMMP (e.g. TBT in sediments, chlorophyll).

Laboratories have subscribed to the QUASIMEME inter-laboratory proficiency-testing (PT) scheme since it became available in 1996.

QUASIMEME provides:
- PT samples twice annually for all current NMMP determinands
- development exercises for new determinands
- workshops to help laboratories identify and resolve analytical problems

NMCAQC:
- collates the PT results
- promotes in-house AQC
- Sets accuracy and precision targets
- Ensures all NMMP determinands are covered by QUASIMEME
- Assesses the quality of data submitted to NMMP through application of a data filter.

In the data filter, aspects of external and internal AQC are set as criteria to be met, and these are awarded points (Table 3.1) that are summed to produce a total score. Acceptable scores were set for each determinand matrix combination according to the complexity of the analyses.

Table 3.2 shows the percentages of data for each determinand/matrix combination that met the data filter criteria for the years 1999-2001, for the combined three-year period. The data acceptance rate was greater than 80%, except for organochlorine substances and PAHs in shellfish. NMCAQC is planning workshops hosted by expert laboratories to try to improve performance for these determinands and has asked laboratories to submit improvement plans.

Due to the lack of in-house AQC information, the data filter could not be used for metals and organics in seawater. The acceptance of data for

Table 3.1. Data Filter Criteria

Data Filter Criterion	Score %
Is the laboratory accredited for the specified determinand - is the lab. well organised?	5
Is the Limit of Detection achieved consistent with NMMP requirements? (1)	5
Control charts - are charts plotted for the determinand in question? – a key element of routine QC	15
Comparison of control chart expected value with control chart mean value – an indicator of bias.(1)	15
Number of measurements made on control chart reference material during period of interest – indicator of the power of chart to detect anomalies (2)	15
Control chart reference material standard deviation (s.d.)– a measure analytical precision (1)	10
Number of measurements used to establish s.d.td devn for control chart limits – reliability of s.d. estimate and of control limits	5
Number of breaches of warning limits during period for which limits apply (used to assess validity of control limits) (3)	-5 or 0
Proficiency testing (4)	30
Total	**100**

Table 3.2. Percentage of data accepted into the NMMP database

Determinand Group	Percentage of data accepted		
	1999	2000	2001
Nutrients	93	93	93
Metals in sediments	100	100	100
PCBs in sediments	95	95	94
PAH in sediments	82	83	93
Metals in shellfish	95	93	95
PCBs in shellfish	100	99	98
OCs in shellfish	30	29	31
PAH in shellfish	60	59	62
Metals and PCBs in fish	96	100	97
Metals in waters	93	93	93
Organics in waters	95	95	95

these determinands was based on proficiency test performance. Data was accepted if 50% of the PT samples were within the accepted range (i.e. a Z score <2). Few data were excluded from the database (Table 3.2).

3.2 Biology

The National Marine Biological AQC Scheme (NMBAQC Scheme) has operated since 1994. It aims to ensure good laboratory practices are maintained in order to provide consistent, good quality marine biological data. The scheme is presently focused on benthic invertebrate community analysis and acts as a data quality screen for the UK NMMP programme. Participation in the scheme is mandatory for the 13 laboratories submitting biological data to the NMMP programme but other commercial laboratories also participate.

The scheme was designed to test the taxonomic proficiency of individual laboratories and their ability to process samples.

Components of the scheme include Ring Test and Macrobenthos samples, which are sent to the participants for analysis, and Own Samples and Laboratory Reference samples, which are provided by the participants. All the samples are subject to independent analysis by an external consultant. The scheme also circulates sediment samples for Particle Size Analysis and organises identification workshops.

3.2.1 *Targets and standards*

Benthic Fauna

Data quality is assessed by comparing the percent similarity of the Own Sample as measured by the lab and the auditor using the Bray-Curtis Similarity Index (BCSI). A five-tier system is applied (see Table 3.3). Samples not reaching the required standards are flagged, along with the remaining replicates from the same NMMP site. Labs are encouraged to reanalyse flagged samples to produce data of acceptable quality, which can be submitted to the NMMP database.

Particle Size Analysis

Data is assessed on the basis of z-scores for each lab compared to the means of all labs, after removal of statistical outliers.

Table 3.3. Data quality assessment criteria

BCSI (%)	Description of Results and Actions
100	Excellent
95-<100	Good
90-95	Acceptable
85-90	Poor – remedial action suggested
<85	Fail – remedial action required

3.3 Ecotoxicology

The NMMP Ecotoxicology AQC Group was formed in 1999 to introduce quality assurance for biological effects measurements. The group consists of representatives from the organisations that carry out OSPAR JAMP biological effects techniques. The group advises on the design of biological effects programmes, the quality of data generated and the introduction of new techniques.

Inter-laboratory test materials for biological effects measurements are provided by the Biological Effects Quality Assurance in Monitoring (BEQUALM) programme. BEQUALM is equivalent to the QUASIMEME laboratory proficiency testing scheme used for the measurements of chemical contaminants. (see www.BEQUALM.org.uk)

AQC procedures are now in place for JAMP biological effects measurements:

- Fish disease
- EROD, CYP1A
- DNA adducts
- Metallothionein
- Oyster embryo bioassay
- Sediment bioassays

Test materials are distributed on an annual basis and data is submitted to a lead laboratory for performance evaluation. In addition, participating laboratories are required to monitor in house quality using control charts based on internal laboratory reference material.

Test materials for bile metabolites and the measurement of imposex in whelks are distributed by the QUASIMEME programme and in the foreseeable future will continue to be covered by this scheme.

3.4 Data assessment

Following sample collection and analysis contaminant data were assessed following OSPAR guidelines (OSPAR Commission, 1999). The measurements of each contaminant at each site are summarised by the median log-concentration* for each year. These yearly medians form a time series that can be analysed for trends (Fryer & Nicholson, 1999). The type of trend assessment depends on the length of the time series. With only three years, the yearly median log-concentrations are averaged and then back-transformed to the concentration scale. This value is called the median concentration throughout the report. Although sites with only one year of data were excluded from the assessment, they are sometimes mentioned in the report. Estimates of year-to-year variation in concentration are used to construct upper 90% confidence limits on the median concentrations.

The median concentrations are mapped firstly in comparison to background/reference concentrations (BRCs), and secondly in comparison to ecotoxicological assessment criteria (EACs). The map circles increase in size with the median contaminant concentrations.

Where data are compared to BRCs, the circles are coloured blue if the upper 90% confidence limits on the median concentrations are less than twice the upper BRC, indicating that concentrations are "close-to-background". The circles are coloured pink if the upper 90% confidence limits exceed twice the upper BRC.

Where compared to EACs, the circles are coloured blue if the upper 90% confidence limits on the median concentrations are less than the upper EAC, indicating that concentrations are "unlikely to cause harm to the marine ecosystem". The circles are coloured pink if the upper 90% confidence limits exceed the upper EAC.

[1] *A log-concentration scale is used because log-concentrations tend to be normally distributed with constant variance over time. Also, absolute changes in log-concentration equate to percentage changes in concentration, so trends may be easily compared across different time series.*

4. RESULTS

As outlined above, the NMMP is designed to contribute to the UK's national and international monitoring requirements. These satisfy EC Directives and the OSPAR Joint Assessment and Monitoring Programme (JAMP). JAMP questions relevant to the monitoring are highlighted in the text. Results are discussed according to their matrix (waters, sediments, shellfish and fish). In each matrix, contaminant concentrations are presented followed by biological effects.

4.1 Waters

This section includes results from the OSPAR Riverine Inputs and Direct Discharge (RID) monitoring programme, which quantifies inputs to marine waters from rivers and effluents. Nutrient, trace metal and trace organic concentrations in waters are presented. The biological effects of nutrients have not been measured for the current programme but general water quality has been determined by the oyster embryo bioassay test.

4.1.1 Nutrients

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

This section addresses ***JAMP Issue 2.1: "Are agreed measures effective at reducing nutrient inputs?"***. Winter nutrient concentrations in UK marine waters are also presented because they are one of a suite of criteria to assess eutrophication. There are no agreed BRCs or EACs for nutrients; however suggested draft common assessment criteria for UK waters subject to the OSPAR Common Assessment Procedure for eutrophication are shown in Table 4.1.

Waterborne loads of nutrients entering marine waters via rivers (Riverine inputs) and directly from wastewater discharges on UK coasts and estuaries (Direct inputs) are estimated from spot measurements of concentration and flow. Riverine inputs include wastewater discharges upstream of the sampling point used to calculate river inputs. Monitoring of annual inputs of nutrients to UK tidal waters from riverine and direct sources was initiated in 1990 as part of the OSPAR RID reporting programme. Estimates of nitrogen and phosphorus inputs for 1985 were reported to the North Sea Conference as a baseline for the reduction programme.

Figures 4.1 and 4.2 show that, since 1985, inputs of total phosphorus fell by about 40-50%, but there was no reduction in nitrogen inputs. Closer scrutiny shows that, since 1990, direct inputs of phosphorus and nitrogen have decreased by some 30-40%. This reflects the effectiveness of measures taken to control point source discharges direct to marine waters. Over the same period, riverine inputs of both nutrients have increased but varied significantly in relation to river flow. Flows (Figure 4.1) have varied widely over the period and have been particularly high in recent years. Since 1990, mean flows have been above the long-term average and generally exhibit an upward trend. Nitrogen inputs have closely followed the pattern in river flows.

Elevated nutrient concentrations are part of a suite of indicators used to assess eutrophication. Conventionally, monitoring is restricted to winter months (Dec-Feb) to estimate the maximum nutrient concentration and thus, depending upon retention, to indicate nutrients in the body of water available

Table 4.1. Draft Common Assessment Criteria for the Comprehensive procedure

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

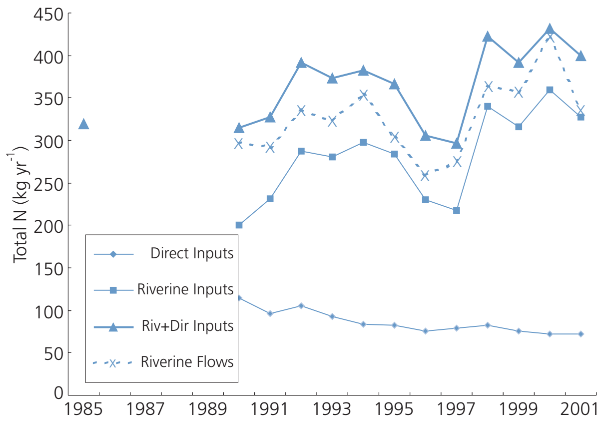


Figure 4.1. Trends in direct and riverine inputs of Total N (ktlyr) to UK coastal waters compared with riverine flows (10^4 llsec)

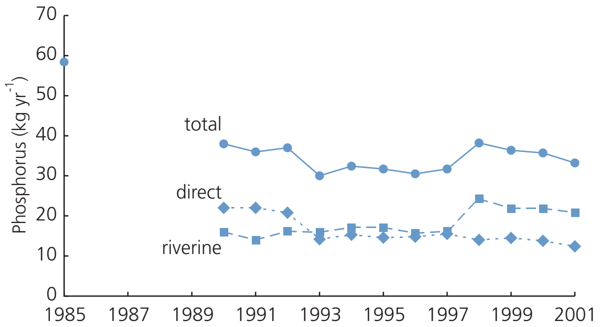


Figure 4.2. Inputs of phosphate phosphorus (ktlyr) to UK coastal waters 1990-2000

to support algal growth. In practice, continuous monitoring has revealed that concentrations continue to rise until conditions are suitable for algal growth (Figure 4.3). For this reason, data collected in early March were accepted from some offshore sites.

Winter nutrient samples were collected for 37 sites around the UK during 1999-2001. This is a small proportion of the total sampling effort in UK coastal waters, with more intensive sampling carried out in many areas. Most NMMP sites were sampled annually, but some coastal sites were only sampled in 1999 and 2001. Measurements were excluded

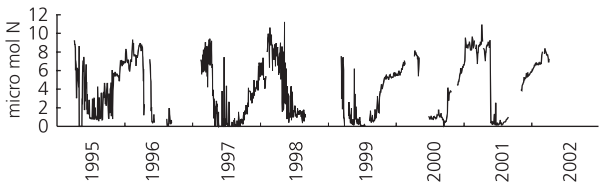


Figure 4.3. Continuous Total Oxidised Nitrogen (nitrite + nitrate) data from the DARD moored sampler in the Irish Sea

from sites where the salinity was less than 30, as concentrations at low salinity sites are high because of riverine inputs and the draft OSPAR assessment criteria are related to waters with a salinity of 34 or higher.

Nutrient concentrations

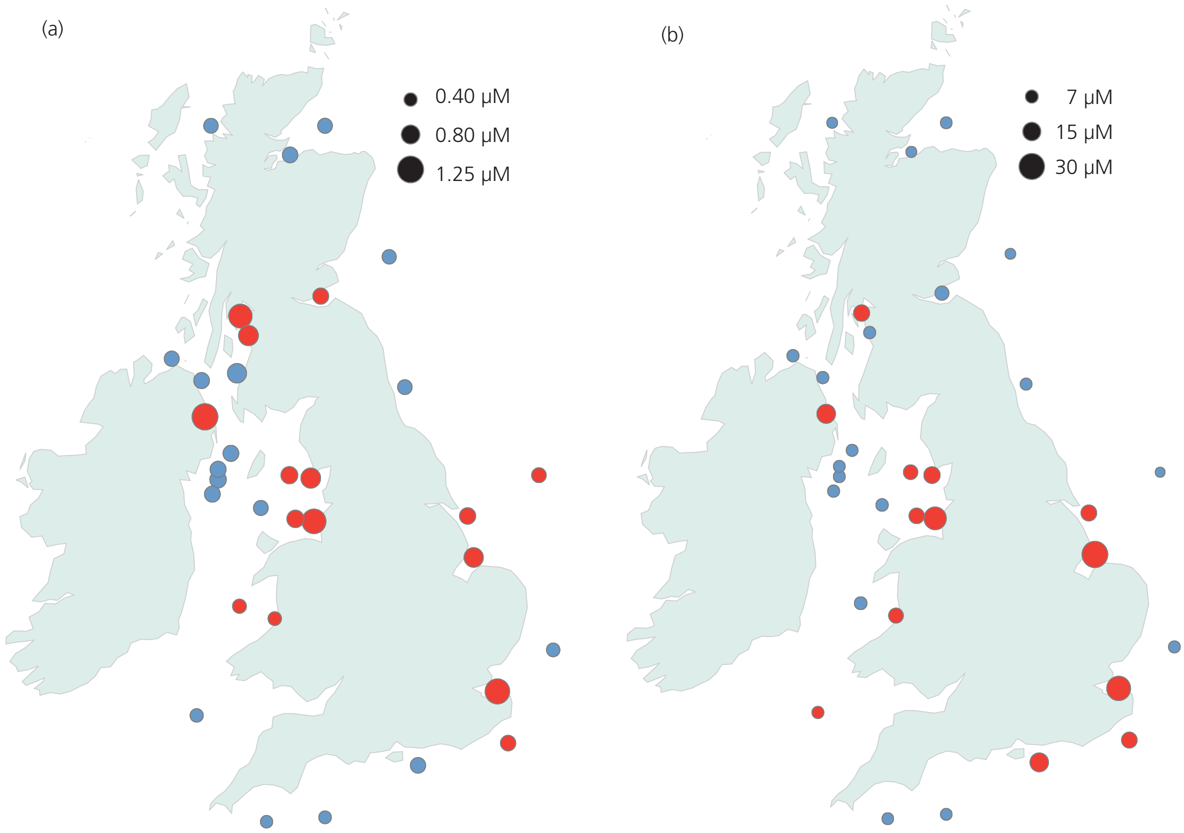
Ammonia

Ammonia is a reduced form of nitrogen, readily oxidised to nitrate. The main sources are wastewater discharges, and benthic inputs in turbid estuaries (Balls, 1992). Direct discharges of sewage contributed over half of the total input to UK coastal waters in 2000. Highest concentrations in the initial spatial survey occurred in low salinity waters in the Mersey, Tees, Tyne and Wear estuaries. Data from these sites were excluded from this survey on the basis of salinity and there are no data for coastal waters in England and Wales, so the only data available is from Scotland and Northern Ireland. Concentrations were highest at Clyde and Belfast Lough stations and low at offshore sites. The median concentration of ammonia ranged from 0.2-10 µM.

Total Oxidised Nitrogen (TOxN)

TOxN is a combination of nitrite and nitrate. Nitrate is the fully oxidised and most stable form of nitrogen in oxygenated waters. Nitrite is a reduced form of nitrogen, intermediate between nitrate and ammonia. Concentrations of nitrite are generally low (<1 µM) but are higher in estuaries with high ammonia concentrations. Nitrate is the main form of nitrogen in coastal waters and median TOxN concentrations were compared to the draft OSPAR assessment criteria for dissolved inorganic nitrogen (DIN).

Median concentrations of TOxN exceeded the relevant assessment criteria in the inshore waters of the Wash, Thames, Clyde, Belfast Lough, Liverpool Bay,Cardigan Bay and Southampton Water (Figure 4.4). The median concentration of TOxN generally decreases as salinity increases and is less than 10 µM in most coastal waters where the salinity is >34, with the exception of the Eastern Irish Sea and Celtic Deep. Nutrient concentrations in nearshore waters of the Eastern Irish Sea are high as a result of large inputs to an area with restricted dilution and dispersion. Long term trend studies off the southwest coast of the Isle of Man show that nitrogen concentrations increased from the 1950s to the 1970s then stablilised (Gowen *et al.*, 2002). Evans *et al* (2001) detected a recent decrease in nitrate concentrations at this location and in the Menai Straits. High variability in the limited data resulted in the 95 percent confidence limit exceeding the criteria in the Celtic Deep.



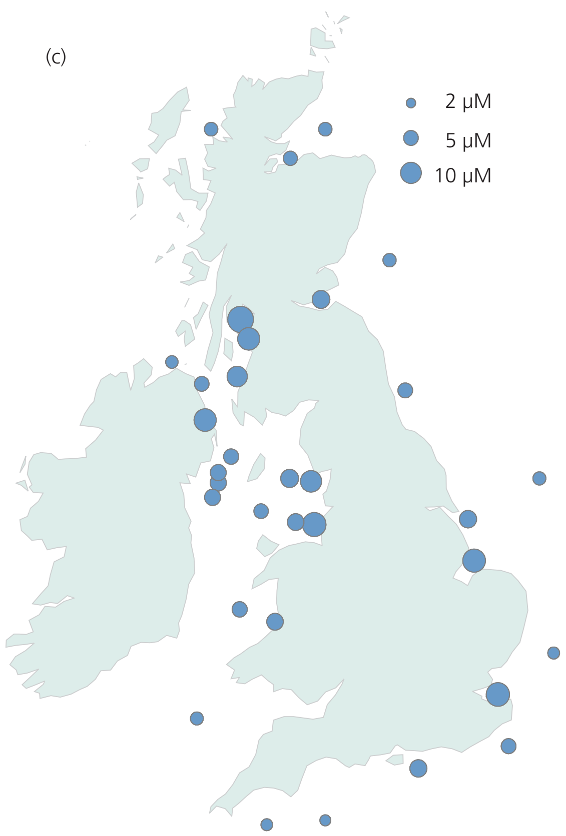


Figure 4.4. *Nutrient concentrations : (a) phosphate, (b) TOxN and (c) silicate*

Phosphate

Direct and riverine inputs of phosphate are a similar magnitude; however, direct inputs have decreased as riverine inputs have increased (Figure 4.2). Riverine inputs are a combination of waste water discharges and run off from the land. They are modified in turbid estuaries by adsorption onto particulate matter at low salinities and release from particulate matter at higher salinities (Balls, 1992). In coastal waters, concentrations decrease with salinity. The median concentration of phosphate ranged from 0.4-1.9 µM and exceeded OSPAR assessment criteria in inshore waters in Belfast Lough, the Thames, Liverpool Bay, Clyde , Humber, Forth and Cardigan Bay (Figure 4.4). One offshore site in the North Sea exceeded the assessment criteria due to variability in the data.

Silicate

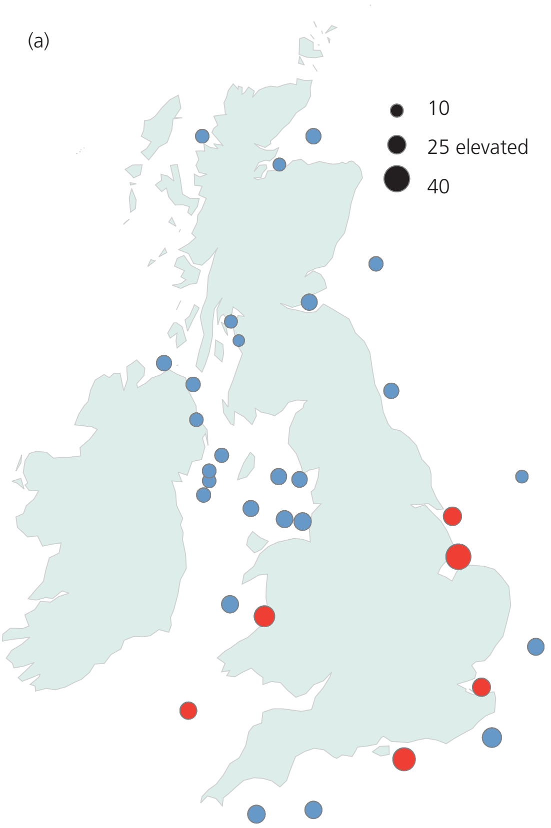
Silica is required for the growth of diatoms, which have a siliceous cell wall and are the dominant algae in the Spring plankton 'bloom'. The concentration of silicate in rivers largely depends on the local geology, with some anthropogenic input. Rivers draining catchments rich in aluminosilicate rocks are high in silicate and anthropogenic inputs arise from its use in detergents. OSPAR have not proposed assessment criteria for silicate. Median silicate concentrations ranged from 2.7-15.4 µM, with high concentrations in the Wash, Thames, Liverpool Bay and Clyde compared to other sites (Figure 4.4).

Nutrient Ratios

Algae are assumed to take up nutrients according to the Redfield ratio of 106C:16N:1P:16Si. Deviation from this ratio may indicate an excess of one nutrient, leading to limitation of algal growth by the other. Although nitrogen is assumed to be the limiting nutrient in coastal waters, it was in excess compared to phosphate at 17 of the 37 sites sampled and at almost all sites compared to silicate (Figure 4.5). The N:P ratio was greater than 25:1 at 6 sites in the Wash, Selsey Bill, Cardigan Bay, Celtic Deep and at South Varne, off the Thames. The N:Si ratio was greater than 2:1 at 13 sites and greater than 3:1 at the Wash and Selsey Bill sites. The Wash and Selsey Bill exhibited the greatest excess of nitrogen.

Summary

Since 1985, UK inputs of phosphorus have reduced, but nitrogen inputs have not diminished. Inputs of nitrogen were influenced by the exceptionally high riverine flows of 1998-2001. The recent significant extension of Nitrate Vulnerable Zone designations under the Nitrate Directive and phosphate stripping at inland sewage treatment works (STWs) is expected to reduce diffuse inputs



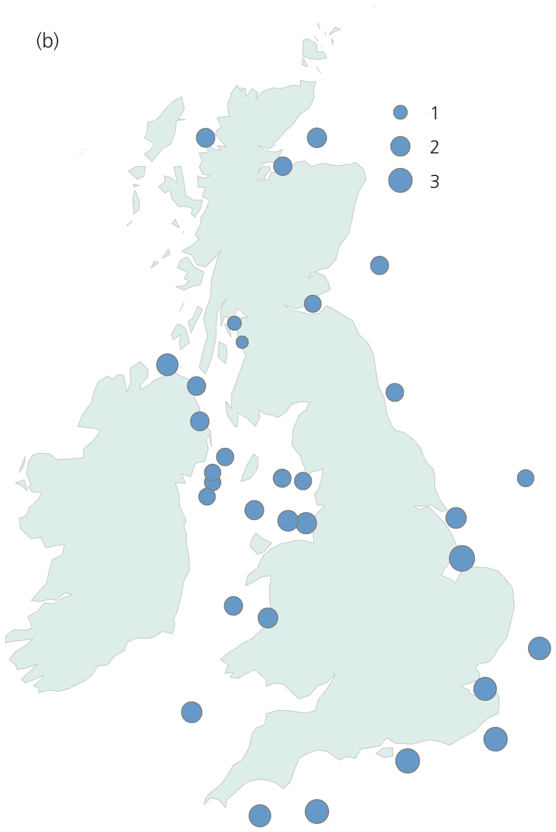


Figure 4.5. Nutrient ratios (a) TOxN:P, (b) TOxN:S

of nitrogen and direct inputs of phosphate to rivers. Currently, the UK is raising standards at many coastal and estuarine STWs, so as to accord with UWWT Directive standards of secondary or tertiary treatment. The STW improvements are, and should continue to be, reflected in reduced direct inputs of nitrogen and phosphorus to coastal waters.

Inputs of ammonia are oxidised to nitrate, which is the main form of inorganic nitrogen present in coastal waters. Riverine inputs of nitrogen are diluted by seawater in estuaries, whereas inputs of phosphate are similarly diluted by seawater, but modified by interaction with particulate matter in turbid estuaries. The concentrations of nutrients in coastal waters depend upon the size of the input and the available dilution. A review of nutrient inputs to estuaries showed relatively high loads of TOxN to the Severn, Mersey, Clyde, Humber, Thames and Solent estuaries, whilst loads to west Wales and northern Scottish estuaries were particularly low (Nedwell *et al.*, 2002). Nitrate concentrations were below OSPAR assessment criteria in most offshore sites except in the Eastern Irish Sea and Celtic Deep. Assessment criteria were also exceeded close to large riverine inputs at the mouths of estuaries.

Nutrient uptake by phytoplankton begins when conditions are suitable for growth, usually in early Spring. Continuous monitoring has shown the short-term variability in nutrient concentrations and highlighted the limitations of spot samples in evaluating the winter nutrient maximum. Continuous measurements of nutrients at the NMMP site in the Irish Sea are described in more detail in the Case Study section of this report.

The effect of nutrients on algal growth is quantified by the measurement of chlorophyll. Chlorophyll measurements for the second phase of the NMMP were limited and are not reported. Temporal trends in phytoplankton in coastal waters are summarised in the Case Study section, using data collected by the Sir Alister Hardy Foundation for Ocean Science continuous plankton recorder. These data show that large scale variations in phytoplankton are associated with interannual variations in climate. Future monitoring for the NMMP should include all OSPAR assessment criteria for eutrophication, including routine measurements of chlorophyll a as an indicator of phytoplankton growth and an assessment of species composition to establish imbalance in the system.

4.1.2 Contaminant concentrations

Contaminants in the marine environment are derived from industrial and sewage effluents, rivers and atmospheric inputs. Inputs have been monitored at the tidal limit of rivers and in direct discharges since 1990. Figure 4.6 shows total inputs of mercury, cadmium, lead, copper, zinc and lindane from rivers, sewage and industrial effluents to UK waters for the period 1990-2001. Inputs are calculated by multiplying the concentration by the flow, to give the 'load'. In some cases the measured concentration is below the limit of detection. The actual concentration is unknown and lies between zero and the limit of detection. These values are used to calculate lower and upper estimates for the load, which is reported as a range.

There was a reduction of about 70% in the mercury load during 1990-2001, due to controls on discharges from chlor-alkali plants and other industries. Cadmium loads fell by about 75% over the same period. The UK input of lead shows no trend over the past decade following the 60% reduction in inputs between 1985 and 1990 as reported by the UK to the Fifth North Sea Conference. This decrease correlated closely with the steep fall in emissions to air, due to the phasing out of lead in petrol. The loads of copper and zinc to estuarine and coastal waters fluctuated during the 1990s with an overall downward trend for both metals.

Lindane or gamma-hexachlorocyclohexane (gamma-HCH) was used widely as an insecticide and preservative until 2002 when it was banned as an agricultural pesticide in the EU. Its sale and supply were also banned in the UK. Inputs of lindane declined by 60-80% between 1990 and 2001.

Metals and organics in water

Concentrations of contaminants in marine waters are compared with the Environmental Quality Standards (EQSs) set in response to the EC Dangerous Substances Directive (76/464/EEC). Under the Directive, discharges containing dangerous substances and their receiving waters are monitored. Results are presented here for the national background reference or 'control' sites for each estuary, which form part of the NMMP, and not for sites close to discharges.

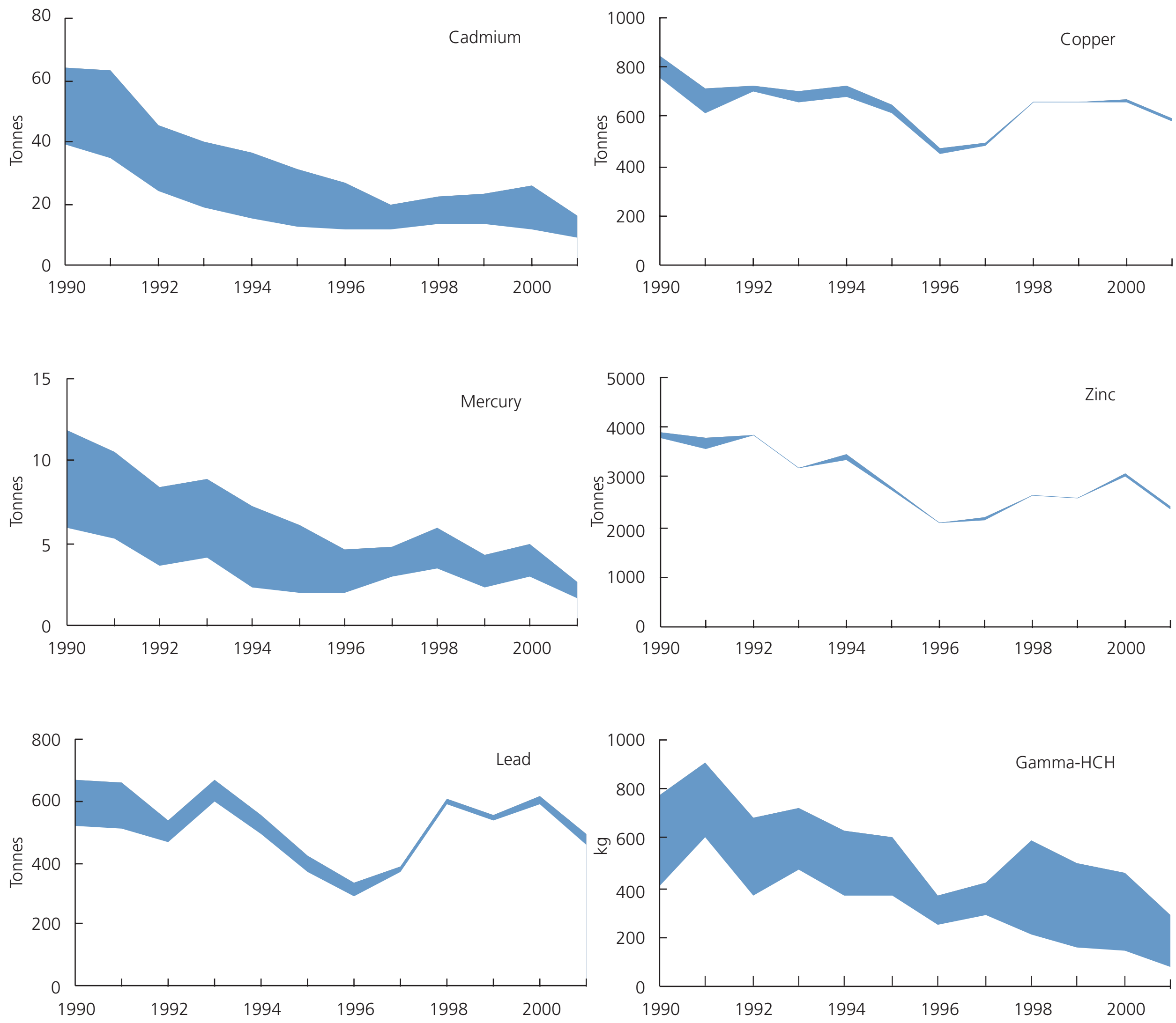


Figure 4.6. UK inputs to the sea, 1990 to 2001

Although compliance with EQSs is assessed by comparison with annual average concentrations, the results reported here are presented as median concentrations for consistency with the other contaminant groups. Where there were fewer than six results for a substance at any site over the three-year period these were excluded from the final assessment. Some subjective comparisons are also made with survey results for 1993 to 1995, presented in the first NMMP report (MPMMG, 1998).

Estuarine concentrations of contaminants are dominated by riverine and direct (industrial and sewage effluent) inputs. Atmospheric deposition of contaminants is less significant and is not discussed here, although it is of major importance to the North East Atlantic as a whole (OSPAR Commission, 2000a). Salinity is an important variable as concentrations of metals, such as cadmium and copper, tend to decrease with increasing salinity as they are diluted in outer estuaries and offshore waters (OSPAR Commission, 2000c). However, since data are only reported here for the national background reference ('control') site in most estuaries, these gradients are not shown by the present data set.

Metals

Median metals concentrations were below the EQSs at all sites (Table 4.2). However the upper 90% confidence limits exceeded the EQS for copper at one site in the Thames estuary and for zinc in the Tyne estuary and in the Wash.

Mercury concentrations were below detection (10 ng l^{-1} or lower) in most samples. The sources of mercury include industrial effluents and use in products such as dental fillings, batteries, lighting

Table 4.2 Summary of metals concentrations in filtered waters, 1999 to 2001

Metal	BRC, µg l^{-1}	EQS, µg l^{-1}	Range of median concentrations across sites, µg l^{-1}	Sites assessed[1]
arsenic	-	25	1.1 - 3.0	9
boron	-	7,000	700 - 4,086	8
cadmium	0.004 - 0.025	2.5	0.012 - 0.25	17
chromium	0.09 - 0.12	15	0.157 - 1.5	8
copper	0.05 - 0.36	5	0.738 - 4.73	18
lead	0.005 - 0.02	25	0.086 - 5.98	17
mercury	0.0001 - 0.0005	0.3	0.003 -0.011	12
nickel	0.16 - 0.26	30	0.345 - 3.05	18
zinc	0.03 - 0.45	40	1.26 - 26.2	20

[1] Sites with adequate monitoring data

and medical instruments, although the release of mercury from historically contaminated dredged material may now be the largest source in the UK (OSPAR Commission, 2000e).

Cadmium concentrations in the Severn Estuary declined from nearly 0.4 µg l^{-1} in 1995 to about 0.09 µg l^{-1} in 2001, well below the EQS but still about four times higher than the upper BRC. Historical contamination of the Severn was caused by metal smelting in Avonmouth and south Wales. Elsewhere, concentrations were similar to BRCs. Cadmium in discharges from the non-ferrous metals and fertiliser industries has been reduced and further action on cadmium in waste, phosphate fertilisers and releases from metal industries has been recommended (OSPAR Commission, 2002).

Although lead is widely used in the manufacture of plastics and batteries, its use has declined in recent years. Estuarine concentrations of dissolved lead should be low, due to the tendency of this metal to bind to particulate matter. Concentrations were below EQS at all sites, but high compared to the upper BRC in the Tyne and Tees estuaries, with median concentrations up to 6 µg l^{-1}.

Dissolved copper concentrations in 2001 were similar to those in 1995, reflecting the lack of reduction in inputs from rivers, industrial and sewage effluents to estuaries. Median concentrations were below the EQS but greater than twice the upper BRC.

Zinc inputs also showed little change since 1995 and concentrations in estuaries were typically an order of magnitude above BRCs, except in the Tyne where concentrations were close to the EQS.

Organic compounds

Where results for organic compounds in waters were reported, concentrations were extremely low, as organic compounds are not very soluble in water. Many results were below detection limits and only total hexachlorocyclohexane (HCH) generally exceeded the limit of detection and approached the EQS (Table 4.2). No BRCs have been derived for organics in water.

The highest median concentrations of total HCH were measured in the Wash and the Thames Estuary. Concentrations were lower in other southern English estuaries (Figure 4.7). A strong

Table 4.3 Summary of concentrations of organic substances in unfiltered waters

Substance	EQS, ng l^{-1}	Range of medians, ng l^{-1}	Sites assessed[1]
total HCH	20	4.8 - 16.7	8
ppDDT	10	0.5 - 5[2]	10
dieldrin	10	0.2[2] – 5.0[2]	12
hexachlorobenzene	30	0.2[2] – 2.9[2]	12
hexachlorobutadiene	100	1.7[2] – 4.2[2]	7
pentachlorophenol	2,000	15[2] – 334[2]	10
trifluralin	100	10[2] – 11.4[2]	7

[1] Sites with adequate monitoring data
[2] Median is based on results recorded as the limit of detection, which varied both between and within sites

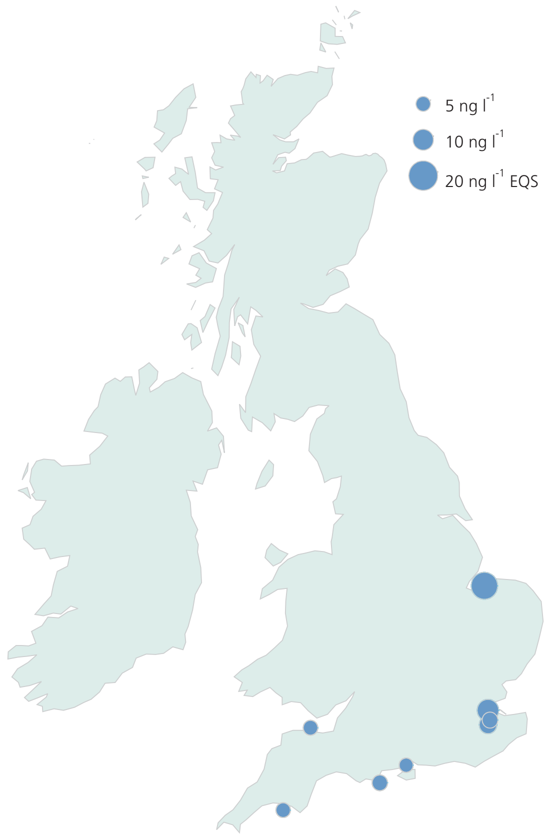


Figure 4.7 Median concentrations of total HCH relative to the EQS. The median concentrations were all below the EQS with 90% confidence

gradient in concentrations in offshore waters has been reported ranging from about 0.3 ng l^{-1} off North West Scotland to between 1 and 2 ng l^{-1} in the southern North Sea and English Channel (OSPAR Commission, 2000a).

4.1.3 Biological effects in waters - oyster embryo bioassay

Seventeen estuarine NMMP sites around the English and Welsh coasts were monitored routinely in 1999-2001 for oyster embryo toxicity, covering all major estuaries in England and Wales. Temporal detail is varied but all sites were sampled on more than one occasion. A small study produced data for the Clyde estuary.

Oyster embryos were exposed for 24 hours to estuarine water from each site at various intervals throughout the year. The exposure period, though short, embraces intense cellular activity in which a number of critical physiological and biochemical processes may be impaired, causing abnormal growth or development of embryos. Water of low salinity was modified to bring it within the optimal range for oyster development (salinity 22-35). Results are expressed as Percentage Net Response (PNR), the net abnormality in the sample relative to the 'background' response in a control exposure to artificially prepared seawater.

Results

England and Wales (Figures 4.8 and 4.9)

At all 17 sites there was at, one time or another, toxicity (positive PNR). In the majority of estuaries the toxicity was occasional, showed no trend, and does not indicate a sustained deterioration in biological water quality.

Time series from 1999 to 2001 from the Tyne and Wear estuaries each displayed significant toxicity (PNR >20%) on one occasion but little or no effect on other occasions. The Tees estuary (Phillips Buoy) showed significant effects in 1999, but none thereafter.

No significant effects were observed in samples taken from the Humber estuary at Spurn Head.

Effects were recorded at Mucking in the Thames estuary throughout 1999 and 2000, but only one survey showed a significant PNR value (May 2000), whereas all samples in 2001 showed little effect. Low toxicity was found in the Medway and Southampton Water, with the exception of one sample from Sun Pier on the Medway in Feb 2001 (PNR 39%). Other sites on the Medway (Burham) and in Southampton Water (East site 515, Brambles) were only surveyed during 2000.

There were low PNR values in South West and Welsh estuaries, especially Poole Harbour South Deep, few of these results exceeded 10%, and none indicated serious deterioration in water quality (PNR > 20%).

At the five sites in the North West of England, results from 2001 revealed no significant toxicity in the Mersey or Ribble estuaries, or from coastal sites.

Scotland

The Clyde was surveyed three times from May to August in 1999 along a 30 km transect from the tidal weir in Glasgow. Water quality was investigated at 1, 3, 4, 6 and 9 metres, to reflect the stratified nature of the estuary. This was spatially a more detailed study than the English ones, but covers a considerably shorter time period.

There was no toxicity in the first surveys in May and early August, but the third survey in mid-August showed significant toxicity (PNR >20%) at the Kelvin confluence (2 mile station, 6 m depth), and at Rothesay Dock (6 mile station, 2 m and 6 m depth).

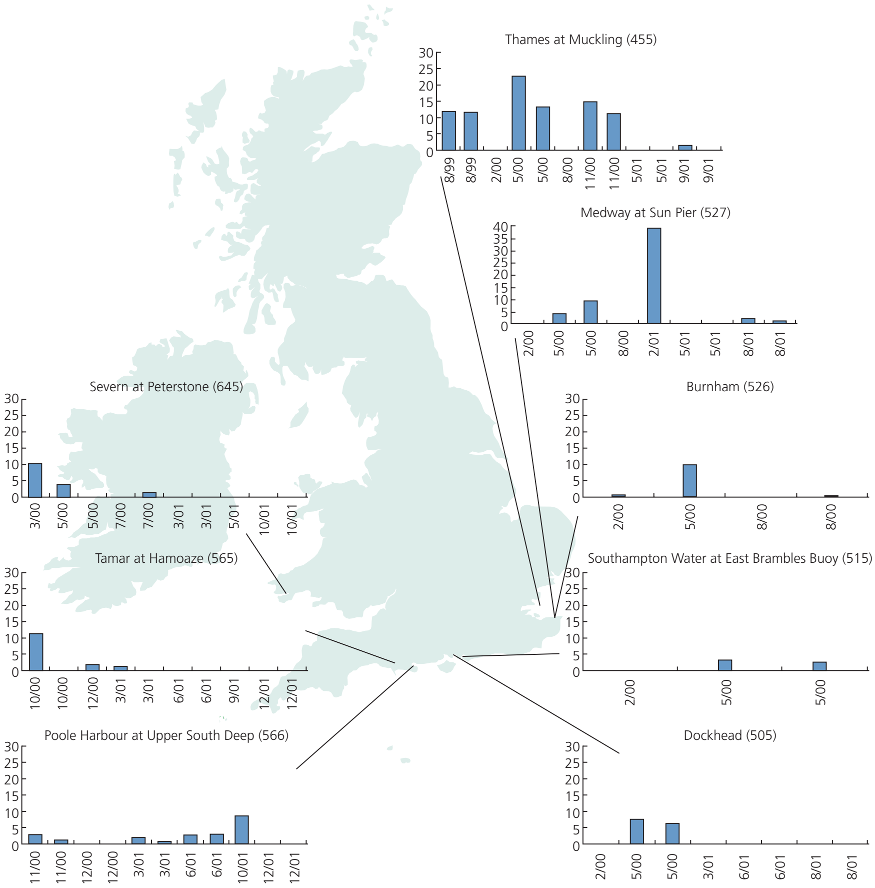


Figure 4.8. Oyster embryo larval development results: Southern England and Wales, 1999-2001

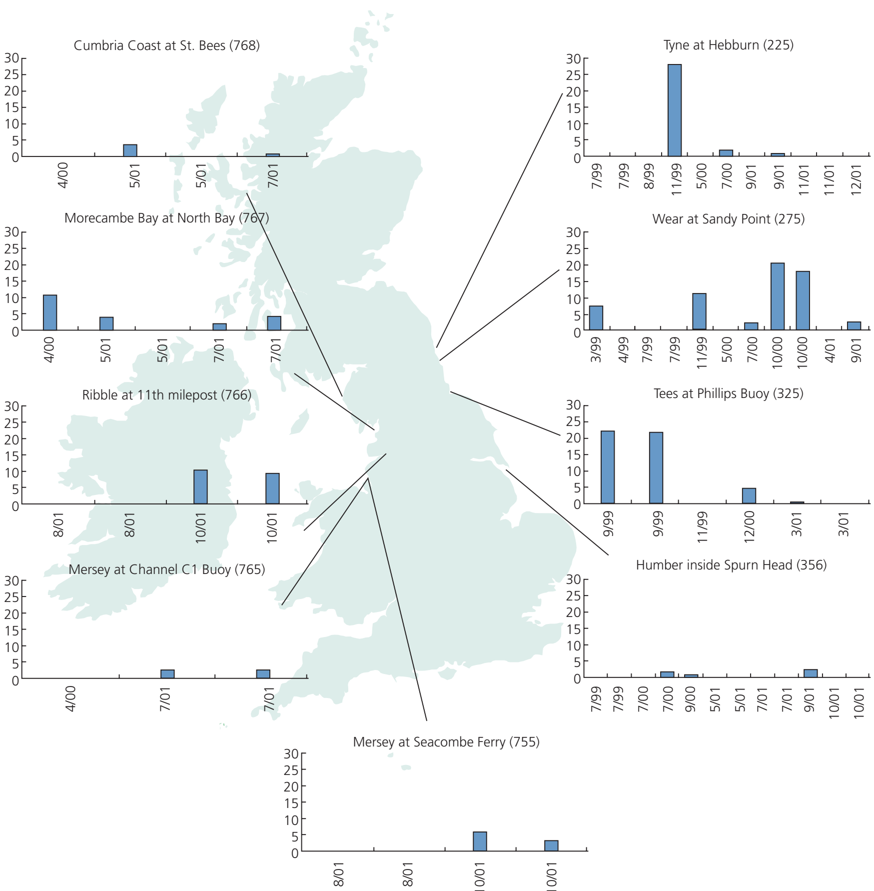


Figure 4.9. Oyster embryo larval development results: Northern England, 1999-2001 Summary

Summary
In UK estuaries the concentrations of metals and organic substances rarely exceeded the standards set under the Dangerous Substances Directive. No median values exceeded any EQS for the period 1999 to 2001.

Despite reductions in direct and riverine inputs of cadmium, mercury, zinc and lindane, concentrations of these substances remain higher in industrialised UK estuaries than in more remote areas. Metals concentrations are typically several times BRCs, with the highest concentrations in English estuaries. The lack of clear correspondence between trends in loads and estuarine concentrations over time may partly be explained by the low precision with which both are estimated (OSPAR Commission, 2000b). Some contaminants in the water column, such as cadmium, remain in the dissolved phase, while others such as mercury and lead are adsorbed onto particulate material and deposited as sediments or transported out of the estuary.

UK estuarine biological water quality measured by toxicity to oyster embryo-larval development was generally very good. The oyster embryo larval test is a robust measure of the biological water quality of discrete water samples. However, because samples were collected quarterly at a limited number of sites in each estuary, it is difficult to draw conclusions on biological water quality of an entire estuary, or on temporal trends. The surveys in English and Welsh estuaries carried out according to NMMP Green Book guidance meet only the minimum requirement for estuarine monitoring, especially with regard to spatial coverage within estuaries.

The surveys on the Clyde highlight the variability of biological water quality with location, time and depth, and the need for detailed spatial surveys of each estuary prior to developing a longer-term temporal trend, monitoring programme.

4.2 Sediments

The extent to which trace metals and trace organic compounds discharged to the marine environment are deposited into the sediments depends on their solubility. Trace organic compounds in particular are not generally soluble in water and are more likely to be detected in sediments. The impact of these contaminants on sediment dwelling organisms is assessed by examination of the benthic community structure and sediment bioassay tests.

4.2.1 Contaminants in sediments

This section addresses two JAMP questions:

JAMP Issue 1.2: What are the concentrations and fluxes of mercury, cadmium and lead in sediment and biota?
and
JAMP Issue 1.7. Do high concentrations of PCBs pose a risk to the marine ecosystem?

In order to show a more complete picture, the distribution of all monitored metals is discussed.

Metals
Trace metals in marine sediments are associated with the fine-grained aluminosilicates, a major constituent of which is aluminium. The abundance of aluminium in marine sediments is related to the geology of the sediment and is not regarded as a contaminant. The relative abundance of other metals to aluminium are expressed as metal/aluminium ratios. These are used to compare sediments from different geographical areas having different geologies and to indicate enrichment due to inputs of metals as a result of man's activities.

Sediment metals concentrations were assessed in relation to BRCs, and EACs. The sediment BRCs are the ranges of metal/aluminium ratios in fine sediments or the fine fraction of sediments from uncontaminated areas. EACs are expressed as ranges of absolute metals concentrations, and were developed to identify potential areas of concern and to indicate which substances may cause harm to biota. OSPAR consider EACs for metals in sediments as having 'provisional' status, and recommend that they should not be regarded as firm standards or as triggers for remedial action. OSPAR also states that caution should be exercised in their use, since they do not take into account specific long-term biological effects such as carcinogenicity, genotoxicity or reproductive disruption due to hormone imbalances, or combined toxic effects.

The OSPAR BRCs and EACs for metals in fine-grained marine sediments (OSPAR Commission, 2000d) are shown in Table 4.4. The sediment concentrations of these metals are compared to the BRCs as metal/aluminium ratios in Figure 4.10.

Median mercury/aluminium ratios were elevated to more than twice the upper BRC at most sites and were much higher at some sites. For example, the median ratios were 51, 52 and 76 times higher than the upper BRC in the Tees, Mersey and Thames estuaries.

Median cadmium/aluminium ratios were below twice the upper BRC at a number of sites. However, several estuaries showed ratios elevated above twice the upper BRC, and some to more than five times the upper BRC. The largest median ratios were 15, 18 and 30 times the upper BRC in the Tyne, Thames and Ribble estuaries.

Table 4.4. BRCs and EACs for metals in fine-grained marine sediments

Metal	Sediment BRCs as metal/aluminium ratios (multiply by 10^{-4})	EACs (mg kg^{-1} dry weight)
Mercury	0.0034–0.0066	0.05–0.5
Cadmium	0.007–0.03	0.1–1
Lead	1.8–4.0	5–50
Arsenic	2.0 – 4.5	1 –10
Copper	2.2 – 5.7	5 – 50
Chromium	9.0 - 20	10 – 100
Nickel	4.4 – 9.1	5 – 50
Zinc	8.8 - 18	50 – 500

Median lead/aluminium ratios were also elevated to more than twice the upper BRC at most sites (Figure 4.10). Median values were 11, 20 and 27 times higher than the upper BRC in the Tyne, Tees and Wear estuaries.

For arsenic, chromium and nickel the median sediment metal/aluminium ratios exceeded twice the upper BRC thresholds at only a few sites. However, there was evidence of more widespread contamination by copper and zinc. The highest median copper/aluminium ratios were all about 5 times higher than the upper BRC and these were found at two sites in the Tamar and one in the Thames. The highest median zinc/aluminium ratios were found in the Tyne, the Wear and the Mersey, with enrichment factors of 5 to 6 times the upper BRC threshold.

The median metal concentrations for sediments for 1999-2001 were compared to the OSPAR EACs as absolute concentrations (Figure 4.11).

The median mercury concentrations exceeded the upper EAC at several sites and were more than five times higher than this value in the Thames and Tees estuaries.

Median cadmium concentrations were above the upper EAC at several sites, and above three times the upper EAC in the Thames.

Median lead concentrations were above the upper EAC at about half of the NMMP sites and higher than five times the upper EAC in the Tyne, Wear, and Tees estuaries.

Median arsenic concentrations were above the upper EAC at most sites. At two sites in the Tamar, the concentrations were more than 15 and 8 times higher than the upper EAC. This is a result of the local mineralization; a century ago, half of the world's arsenic was produced in the area. Arsenic inputs remain high, due to drainage from disused mines.

Sediment chromium, nickel and copper concentrations were above the upper EAC at a number of sites. Sediment zinc concentrations were below the upper EAC at most sites.

Organics

Commercial formulations of polychlorinated biphenyls (PCBs), such as Aroclors, have been widely used in transformers, capacitors, hydraulic fluids and as plasticisers in paints, plastics and sealants. The inputs of PCBs to the marine environment include energy production, combustion industry, production processes and waste (landfill, incineration, waste treatment and 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 transportation 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 (larger surface:volume ratio) have a

Figure 4.10 Median metal/aluminium ratios in sediment for (a) mercury, (b) cadmium, (c) lead and (d) arsenic in relation to their BRCs

Figure 4.10 continued: Median metal/aluminium ratios in sediment for (e) chromium, (f) nickel, (g) copper and (h) zinc in relation to their BRCs

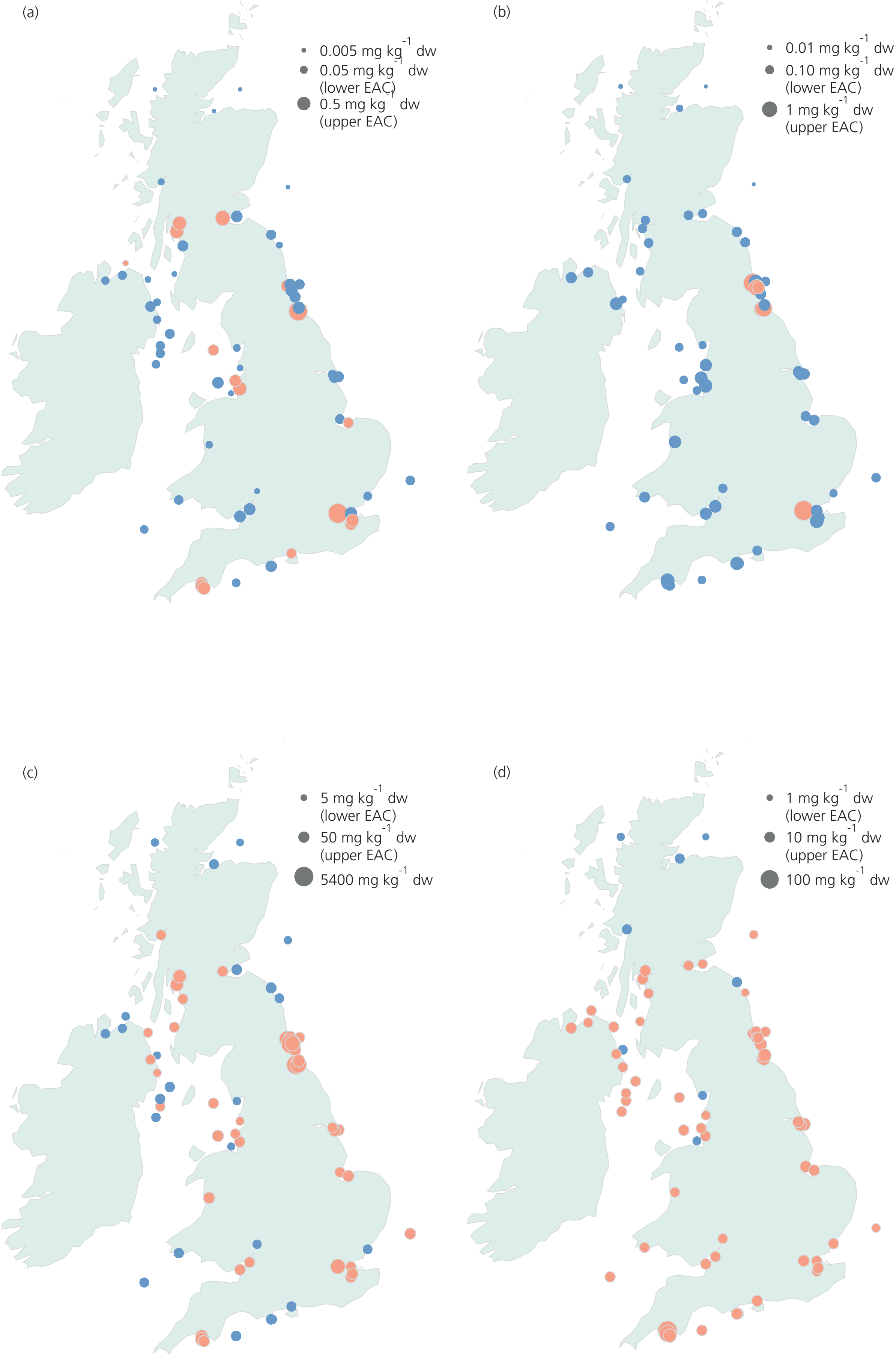


Figure 4.11. Median concentrations of (a) mercury, (b) cadmium, (c) lead and (d) arsenic in sediments, relative to their provisional EACs

Figure 4.11. continued: Median concentrations of (e) chromium, (f) nickel, (g) copper and (h) zinc in sediments, relative to their EACs

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 PCB congeners of which the ICES 7 (CBs 28, 52, 101, 118, 153, 138, 180) were selected for measurement due to their relatively high concentrations in technical mixtures, wide chlorination range and persistence. Where all ICES 7 CBs are not measured, CB 153 can be used as representative of all other CBs. CB 153 occurs in all technical formulations and lies in the middle of the range of physical parameters such as solubility* and molecular size. It is not easily metabolised and, therefore, is often the dominant CB in biota samples.

Sediment CB concentrations were reported at 56 sites. However, CB 180 concentrations were not available for some sites. The sum of the ICES 7 CBs and CB 153 concentrations are summarised in Table 4.5. In the initial 1992-1995 spatial survey, concentrations of the Σ7CBs in the sediments from offshore sites in England and Wales were all less than 5 µg kg^{-1} (CEFAS 1998) and the work was not repeated in 1999-2001.

The lowest concentrations were found at the Tay/Forth offshore site and highest at Woolwich in the Thames. In general, the lowest concentrations for all of the ICES 7 CBs were found at the Scottish offshore sites, and the highest in southeast England.

The Scottish offshore sites showed a predominately atmospheric input of CBs with the lower chlorinated CBs (CB 52 and 101) dominating the profiles. Most estuarine sites contained a higher proportion of the more highly chlorinated CBs, with the hexa-CBs dominating (CB 138 and 153).

Currently there are no BRCs for CBs in sediment in UK waters. The EAC for the sum of the ICES 7 CBs is provisional and ranges from 1 to 10 µg kg^{-1} dry weight. Only six sites had median concentrations below 1 µg kg^{-1} dry weight (Figure 4.12). Three were Scottish offshore sites (Minches, Moray Firth, Forth/Tay) two were in Northern Ireland (North Channel, North Antrim coast) and one in Wales (Severn). The median concentration was above

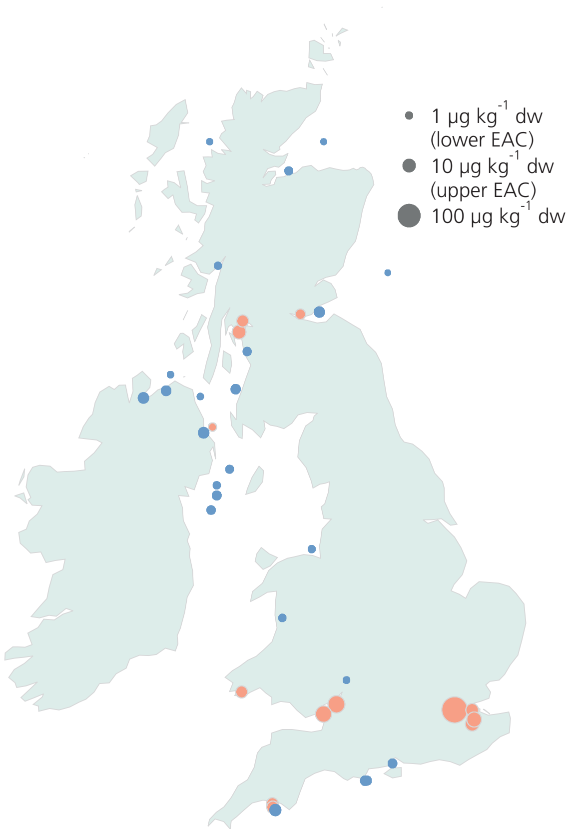


Figure 4.12. Median concentrations of the sum of the ICES 7 CBs in sediment (ug/kg dry weight) and their relationship to provisional EACs

the upper EAC at seven sites, one in Scotland (Clyde), two in Wales (Severn) and four in the south of England (Thames and Medway). At one site (Thames), the median concentration was above 50 µg kg^{-1} dry weight. The upper 90% confidence limit was above the upper EAC at 6 additional sites.

In the initial 1992-1995 spatial survey CB 153 concentrations were reported in sediment from 67 sites. However, few results were above detection limits. Highest concentrations were found in the Severn with concentrations up to 25 µg kg^{-1} (Table 4.5). In the 1999-2001 survey, the highest concentrations were found in the Thames (24.6 µg kg^{-1} dry weight) and the Mersey (15.4 µg kg^{-1} dry). Lower concentrations were found in the Severn (4.7 and 4.2 µg kg^{-1} dry weight).

* *Described by the octanol/water partition coefficient (log K_{ow})*

Table 4.5. Summary of PCB data for sediments

CB	CB 153	Sum of ICES 7 PCBs
EAC $\mu g\ kg^{-1}$ dry weight (provisional)		1 -10
Median concentration range 1999-2001, $\mu g\ kg^{-1}$ dry weight	0.02 – 24.6	0.42 – 184.2
No. of sites assessed	56	37
Sites with median concentration > upper EAC		7
Sites with upper 90% confidence limit > upper EAC		13
Concentration range 1992-1995 $\mu g\ kg^{-1}$ dry weight	<2.5 - 25	

PAH
Determinations of a suite of 10 PAH were undertaken in surface sediments to address the JAMP issue

JAMP issue 1.10 What are the concentrations of PAHs in the maritime area?

The PAH determined were: naphthalene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*a*]pyrene, benzo[*ghi*]perylene and indeno[1,2,3-*cd*]pyrene.

Data were available from most NMMP sites. Total PAH concentrations ranged from below the limit of detection at the offshore Moray Firth site to 207,000 $\mu g\ kg^{-1}$ dry weight in the Medway at Sun Pier in 1999. EACs have been proposed for 8 of the 10 PAH determined (Table 4.6). The sum of the 10 PAH determined was >10,000 $\mu g\ kg^{-1}$ (approximately the sum of the upper EAC values) at a number of sites: the Tyne at Hebburn, Wear at Alexandra Bridge, Tees at Bamlett's Bight, Tees at no. 23 buoy, Thames at Woolwich, Medway at Burnham, and the Bann estuary in Northern Ireland. These high concentrations indicate potential for harm to sediment dwelling animals at these sites.

Table 4.6 indicates that there are a number of sites at which the upper EAC for individual PAH were exceeded. On the Medway at Sun Pier, concentrations greater than 5 times the upper, provisional EAC were recorded in 1999 for benz[*a*]anthracene and benzo[*a*]pyrene but the concentrations had decreased to below the upper EAC in 2000 and 2001. Exceedance of the EAC generally tended to be higher for larger PAH compounds, which suggested that the major contribution derived from combustion sources.

The distribution of pyrene in sediments is shown in Figure 4.13. Concentrations above the upper, provisional EAC were generally found at locations close to combustion sources.

Table 4.6. EAC values (µg/kg dw) for PAH (OSPAR)

PAH	lower EAC	upper EAC	median concentration > upper EAC	Upper 90% confidence limit > upper EAC
Naphthalene[1]	50	500	1	3
Phenanthrene[1]	100	1,000	4	10
Anthracene[1]	50	500	1	6
Fluoranthene[2]	500	5,000	0	2
Pyrene[2]	50	500	11	18
benz[a]anthracene[2]	100	1,000	1	7
Chrysene[2]	100	1,000	2	9
benzo[a]pyrene[2]	100	1,000	3	8

Note : [1] indicates that the EAC for this compound is firm
[2] indicates that the EAC for this compound is provisional

4.2.2 *Biological effects of contaminants in sediments*

Benthic Community Structure

The benthos comprises those animals that live in marine sediments. Community structure depends heavily on salinity and on sediment structure and may also be affected by contamination. In this report, 28 estuarine and 31 coastal sites have been analysed, focusing on spatial data from all sites for 2000 and temporal trends at only two sites from 1997 to 2001. Relationships have been investigated between the benthic communities and environmental parameters of water depth, sediment type and the sedimentary concentration of contaminants.

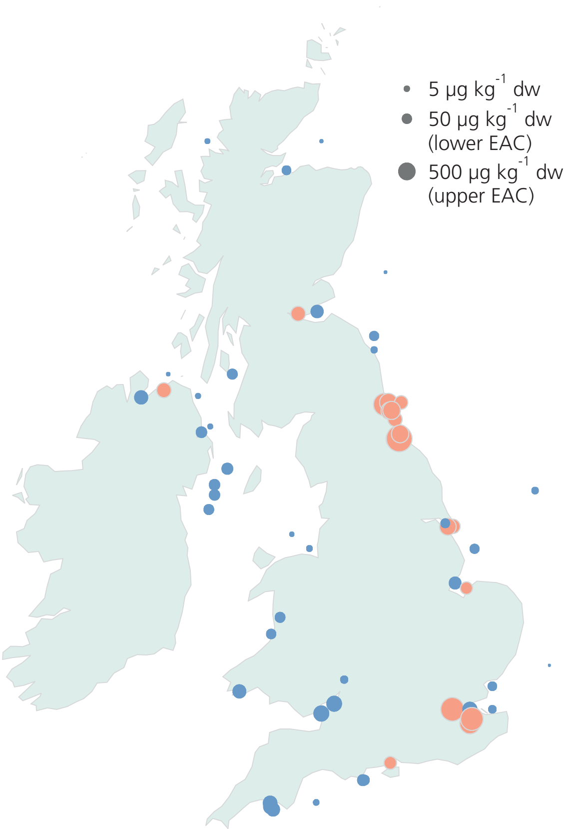


Figure 4.13. Median pyrene concentrations in sediment

Macrofaunal community data were standardised by the exclusion of: colonial or epifaunal species such as hydroids or barnacles; nematode worms; juvenile fauna unassigned to a species; and species represented by only a single specimen in a sample.

Spatial Analysis Results - 2000

Tables 4.7 and 4.8 present univariate statistics (as mean values per site surely sample) for coastal and estuarine sites for 2000 as follows: number of species (S); abundance of all specimens (A); evenness (J'- allocation of abundance values across the species); and Shannon Wiener index (H' - a measure of diversity). Low evenness or Shannon Wiener diversity values were associated with samples dominated numerically by few species and may indicate a naturally stressed or anthropogenically impacted community.

The status of the benthic fauna was also assessed using functional indices such as the UK infaunal trophic index (ITI, Codling and Ashley, 1992) and the AZTI marine biotic index (AMBI, Borja *et. al.*, 2000). The ITI classifies the fauna into different groups according to their mode of feeding, while the AMBI classifies the fauna according to their sensitivity to pollution. The ITI values range from 0 to 100. Values of 60 or above are regarded as 'normal', between 60 and 30 as 'changed', and below 30 as 'degraded'. The AMBI scores range from 0 to 7, with low scores indicating an unstressed healthy benthic community, and high scores indicating stressed or polluted sites.

Coastal Sites

The maximum number of species was 94 at Strangford Lough, while the minimum was only 3 at the Ribble. These sites also represented the maximum and minimum abundance of individuals, 2876 and 11 respectively. Other sites with low species numbers and abundance included the Irish Sea Buoy, Dundrum Bay and the North Antrim Coast. The fauna at Strangford Lough was dominated by the porcelain crab *Pisidia longicornis*, associated with extensive beds of horse mussel, *Modiolus modiolus*. The dominant species at the Ribble site was a crustacean, *Haustorius arenarius*, which prefers mobile sand.

The Shannon Wiener diversity index ranged from 0.93 at the Ribble to 3.11 at Cloch in the Clyde, a deepwater site with mixed sediment. The highest evenness, 0.95, was at Dundrum Bay. The lowest evenness of 0.45 off Humber/Wash related to the dominance of the brittle star, *Amphiura filiformis*, accounting for 75% of the total abundance, but this site also had the highest ITI of 88.69, suggesting that it was not degraded. The lowest ITI, 7.5, was recorded at an intertidal site, Budle Bay, where sub-surface deposit feeders ('opportunist species') dominated, including the annelid worms *Capitella* species and *Tubificoides* species - indicating a degraded community impacted by organic enrichment. This site had the highest AZTI biotic index (5.68) because of the presence of these pollution tolerant species.

Multivariate methods of data analysis may provide a more sensitive measure of community change than univariate methods (Warwick and Clarke, 1991).

Group average cluster analysis was first used to produce a Bray Curtis similarity dendrogram so as to identify groups of similar stations. Multidimensional scaling (MDS) was then used to create an ordination plot. The MDS ordination plot arranges sites according to the similarity of their faunal composition, the most similar sites appearing closest together.

Table 4.7. NMMP coastal data collected in 2000.
Number of species (S), abundance (A), Peilou evenness (J'), Shannon Wiener diversity (H'), infaunal trophic index (ITI), and AZTI marine biotic index (AMBI).
Mean value per 0.1m^2 (based on 5 replicates unless otherwise stated)

Site	Site Name	S	A	J'	H'	ITI	AMBI
45	Clyde – Off Cumbrae	10	18	0.9	2.02	57.20	2.38
55	Clyde – Cloch Point	48	263	0.8	3.11	60.30	1.67
70	Clyde – Irvine Bay	20	97	0.76	2.25	74.11	0.90
76	Loch Linnhe	12	68	0.65	1.6	61.51	2.41
175	Forth – Kingston Hudds	40	303	0.66	2.44	78.96	0.67
220*	Northumberland - Budle Bay	7	534	0.77	1.46	7.50	5.68
245	Off the Tyne	20	76	0.84	2.49	58.56	2.80
270	Durham - Off Seaham	24	105	0.81	2.56	72.64	0.64
345	Off Humber/Wash	15	102	0.45	1.2	88.69	0.29
388	Wash – off Boston	14	165	0.7	1.82	41.38	2.55
389	Wash – Cork Hole	12	74	0.71	1.74	65.59	3.11
475**	Thames – Gabbard **	21	114	0.61	1.88	63.46	2.45
536	Lyme Bay	44	558	0.58	2.21	52.36	0.95
576	Off Tamar	53	1,165	0.63	2.49	51.92	2.42
646	Milford Haven	34	586	0.57	2.05	39.63	2.65
655	Cardigan Bay	19	61	0.84	2.46	61.10	2.34
715	Liverpool Bay	37	196	0.67	2.43	60.41	1.53
765	Mersey C1 Buoy	11	89	0.74	1.76	48.84	0.77
766	Ribble	3	11	0.78	0.93	77.58	0.25
767	Morecambe Bay	7	17	0.9	1.74	71.77	0.65
768	Cumbria Coast – St Bees	33	575	0.53	1.84	65.49	0.73
805	SE Isle of Man	32	202	0.67	2.33	74.48	0.85
807	Irish Sea NMP5	7	16	0.87	1.69	26.32	1.79
808	Irish Sea Buoy	8	13	0.94	1.92	49.19	2.01
809	Strangford Lough	94	2,876	0.52	2.34	77.17	1.24
815	Dundrum Bay	8	12	0.95	1.96	55.56	2.19
815	Dundrum Bay	8	13	0.91	1.84	45.62	2.52
825	Belfast Lough – Outer	21	97	0.77	2.34	67.91	0.47
845	Belfast Lough – Inner	48	996	0.68	2.61	65.47	1.39
865	North Channel NC2	30	105	0.8	2.66	82.67	1.15
875	North Antrim Coast	8	13	0.91	1.69	58.86	2.78
	Mean	24	307	0.74	2.06	60.07	1.75
	Median	20	102	0.76	2.02	61.10	1.67
	Maximum	94	2,876	0.95	3.11	88.69	5.68
	Minimum	3	11	0.45	0.93	7.50	0.25
	Standard deviation	19	558	0.13	0.47	16.76	1.13

**extrapolated to 0.1 m^2 from 0.01 m^2 cores, ** mean of three replicates only*

Table 4.8. NMMP estuarine data collected in 2000.
Number of species (S), abundance (A), Peilou evenness (J'), Shannon Wiener diversity (H'),
infaunal trophic index (ITI), and AZTI marine biotic index (AMBI).
Mean value per $0.1m^2$

Site	Site Name	S	A	J'	H'	ITI	AMBI
208	Forth estuary	43	1,418	0.55	2.07	78.51	0.67
210	Tweed - Yarrow Slake	11	65,175	0.35	0.83	2.55	5.86
225	Tyne – Hebburn	24	39,874	0.44	1.41	1.94	5.48
235	Tyne - Ferry Crossing	21	50,086	0.55	1.68	4.20	5.12
275	Wear- Sandy Point	27	9,340	0.57	1.87	17.93	4.44
305	Tees – Bamlett's Bight	12	622	0.60	1.50	12.22	5.41
315	Tees – No 23 Bouy	30	5,792	0.52	1.79	5.54	4.15
325	Tees Estuary – Phillips buoy	28	2,828	0.60	1.99	50.06	1.32
356	Humber – Spurn Head	29	6,048	0.43	1.44	38.49	4.72
357	Humber – Grimsby Roads	36	33,242	0.22	0.80	60.25	4.30
358	Humber – Sunk Island	8	68	0.56	1.16	37.51	0.15
435	Thames - Woolwich	13	1,442	0.40	1.03	1.31	1.33
455	Thames – Mucking	14	7,322	0.56	1.48	30.59	3.37
526	Medway – Burham	9	30,920	0.72	1.59	0.46	2.35
527	Medway – Sun Pier	23	5,092	0.46	1.45	57.00	2.69
555	Tamar – Warren Point	47	9,086	0.58	2.22	40.17	3.85
565	Tamar – Hamoaze	68	74,322	0.25	1.04	64.86	2.26
566	Poole Harbour – South Deep	93	81,204	0.45	2.06	41.45	3.74
567	Poole Harbour – Wytch	12	185,100	0.27	0.66	14.56	5.22
625	Severn – Purton	2	710	0.05	0.03	100.00	0.03
635	Severn – Bedwin	13	5,550	0.53	1.36	55.71	1.97
645	Severn – Peterstone	12	2,488	0.15	0.39	3.60	5.59
647	Dovey – Ynys-Hir	10	3,510	0.50	1.15	68.98	2.98
648	Mawddach – Bontddu	13	3,400	0.60	1.54	62.51	3.04
690	Dee – Mostyn Bank	19	912	0.51	1.50	85.62	0.10
755	Mersey – Seacombe Ferry	9	68	1.90	0.78	62.75	1.79
820	Bann	6	8,322	0.62	1.11	56.73	3.93
880	Lough Foyle	23	2,710	0.65	2.03	48.47	3.21
	Mean	23	22,738	0.52	1.36	39.43	3.18
	Median	17	5,671	0.53	1.44	40.81	3.29
	Maximum	93	185,100	1.90	2.22	100.00	5.86
	Minimum	2	68	0.05	0.03	0.46	0.03
	Standard deviation	20	39,806	0.31	0.54	29.03	1.78

The cluster analysis identified four main coastal groups, clearly discernible in the MDS plot (Figure 4.14). Group A relates to sites with a low number of species and abundance, in coarse sediment with a low proportion of fine material. Group B contains high salinity sites dominated by opportunist species. Group C sites have very high silt and clay content, above 97%. Group D contains the remaining sites, with high diversities and heterogeneous substrates offering a wide range of habitats for colonisation.

Which environmental factors best explain the macrofaunal community patterns? Spearman rank correlation analysis showed a statistically significant correlation between community patterns and water depth or sediment characteristics, but no significant correlation with contaminant levels. The influence of sediment type on the macrofaunal community is

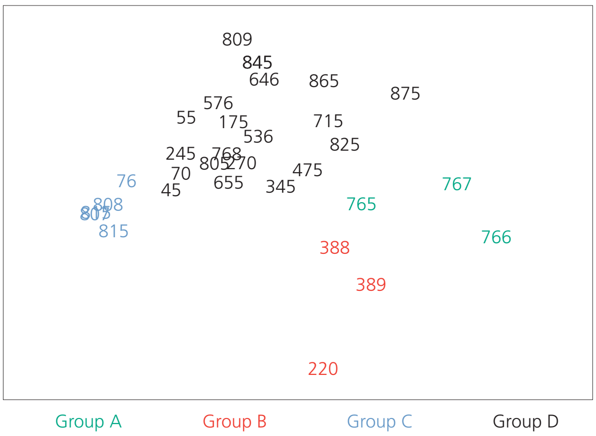


Figure 4.14 MDS plot of the NMMP 2000 coastal macrobenthic dataset showing groups identified by cluster analysis

clearly shown by superimposition of the percentage silt/clay content on the MDS plot in Figure 4.15 Muddy sites with high silt/clay content separated clearly from those with predominantly coarser sediments.

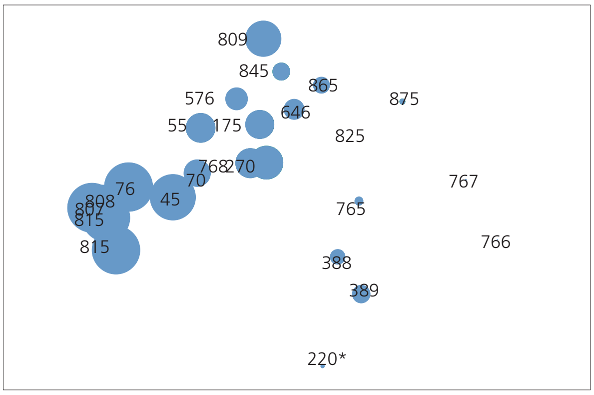


Figure 4.15 MDS plot of the NMMP 2000 coastal macrobenthic dataset and relative percentage silt/clay content of sediment. (Circle size indicates silt/clay content)

Estuarine Sites
The number of species varied from 2 at Purton to over 60 at Tamar and Poole Harbour. The highest abundance at Poole Harbour was over 100,000 invertebrates per 0.1 m^2. Abundance exceeded 10,000 individuals per 0.1 m^2 at sites in the Tweed, Tyne, Humber, Medway and Tamar estuaries. The macrofaunal communities at these sites were dominated by annelid worms including oligochaetes and the polychaetes *Capitella* species, *Manayunkia aestuarina* and *Aphelochaeta marioni*. These taxa indicate highly stressed estuarine environments with low salinities and some organic enrichment.

Diversity and evenness indices ranged from 0.03 at Purton in the Severn to 2.22 at Warren Point in the Tamar and from 0.05 at Purton to 1.90 in the Mersey respectively. Low diversity and evenness at Purton related to the dominance of crustacean *Bathyporeia* species, 99.4% of the population. Comparatively high ITI and low AMBI at this site suggest that, despite the dominance of *Bathyporeia*, this community is natural rather than impacted.

Although both estuarine and coastal sites exhibited a wide range of AMBI, estuarine sites generally had higher AMBI, suggesting more stress either natural from salinity change or from contaminants or both.

Cluster analysis identified six groups, shown on the MDS plot (Figure 4.16). An excluded Group A contains a single site, (Severn, Purton) with very low diversity and only two species, one of which is extremely dominant. Group B represents the Mersey site, a well balanced but impoverished community. Group C contains sites with low diversity dominated by one or two species. Group D represents sites with moderate diversity and a heterogeneous sediment environment. Group E represents the Thames at Woolwich, with low diversity and dominated by two oligochaete worm species. Group F contains sites with low diversity but very high abundances (>8000 individuals per 0.1 m^2).

Spearman Rank correlation analysis found significant correlations between the estuarine macrofauna data and sediment parameters but none with contaminant levels.

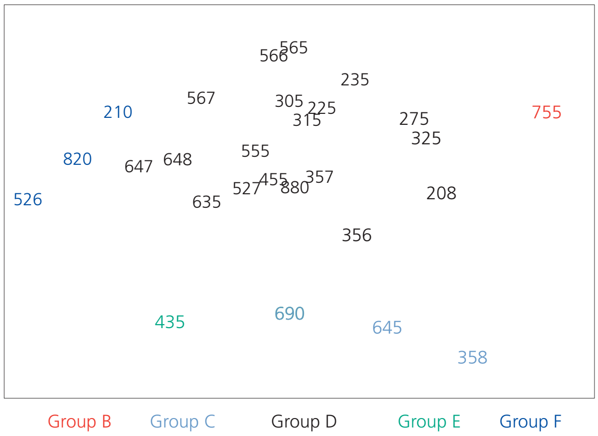


Figure 4.16 Results from multidimensional scaling (MDS) analysis of the NMMP2000 estuarine macrobenthic dataset. (An outlying site 625 is omitted)

Temporal Analysis (1997–2001)
Coastal sites in the inner and outer Belfast Lough were analysed for temporal trends because five years data (1997 – 2001) were available (Table 4.9).

In the inner lough there were declining trends in the number of species, abundance, Shannon Wiener diversity, and AMBI scores. However, the evenness and ITI values showed little change. No trend was apparent in the outer lough. The high faunal abundances in 1997/8 were attributable to dominance by the annelid worms, *Cirratulus cirratus*, *Cirriformia tentaculata*, and *Mediomastus fragilis*, as well as the amphipod crustaceans, *Ampelisca diadema* and *Photis longicaudata*. The presence of these annelids suggested quite high

Table 4.9. NMMP data 1997-2001 Number of Species (S), abundance (A), Pielou evenness (J'), Shannon Wiener diversity (H'), trophic index (ITI) and biotic index (AMBI) for Site 845, Belfast Lough. Mean value per $0.1m^2$

Year	S	A	J'	H'	ITI	AMBI
Inner Lough						
1997	59.00	3,302.00	0.67	2.71	63.67	2.56
1998	66.80	3,921.80	0.65	2.72	64.82	1.92
1999	61.40	1,804.80	0.62	2.53	58.93	1.75
2000	48.40	994.80	0.68	2.64	65.51	1.40
2001	30.00	420.00	0.68	2.30	56.09	1.21
Outer Lough						
1997	20.20	95.80	0.78	2.29	56.06	0.68
1998	20.00	96.40	0.84	2.46	65.36	0.82
1999	30.60	183.20	0.76	2.58	71.56	0.71
2000	19.60	93.60	0.77	2.26	68.03	0.46
2001	12.40	34.00	0.87	2.18	70.64	0.61

nutrient input, although the ITI value did not reflect this. Populations of these species had diminished considerably by 2001.

Cluster analysis of inner lough data identified five distinct groups, coinciding with the five sampling years (Figure 4.17). Within any one sampling year the 5 replicates were similar, but over the five year period there was a consistent trend as depicted by the arrow.

Spearman rank correlation analysis revealed significant correlations between changes in the macrofauna and the sedimentary concentrations of organic nitrogen and organic carbon from 1999 to 2001.

Major human changes in 1997 to 2002 include the cessation of sewage sludge disposal in 1998, with consequent changes at Belfast sewage treatment works, reduction of major discharges from a fertiliser plant, increased dredging activities, and expansion of sea-bed mussel culture. Although trends were not evident in the outer lough, the benthic trends seen in the inner lough might therefore be related to a general decrease in its nutrient loading. The Belfast Lough case study puts these changes into a wider context.

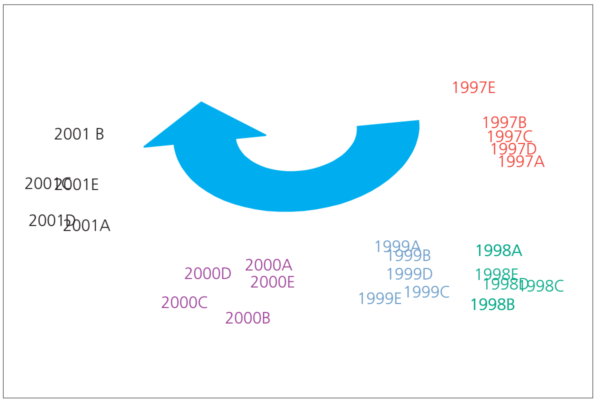


Figure 4.17 Results of multidimensional scaling (MDS) analysis of the temporal data for 1997-2001 at the inner Belfast Lough

Corophium/Arenicola sediment bioassay

Sediment bioassays measure the acute toxicity of pore water or contaminants bound to sediment, to sediment dwelling organisms. Two have been developed for this purpose using the ploychaete *Arenicola marina* and the crustacean *Corophium volutator* (Thain and Bifield, 2002 and Roddie and Thain, 2002). Both animals live in the sediment: *A. marina* ingests sediment and *C. volutator* grazes on sediment particles. In the bioassays the animals are exposed under controlled conditions to collected sediments. In both bioassays, mortality is measured after 10 days, and feeding behaviour is also measured for *A. marina*.

Eighty-four sediment samples were collected from four estuaries Mersey (19), Southampton Water (21), Tyne (22) and Tees(22). Twenty-one samples exhibited *C. volutator* mortality in excess of 20%, twelve of which occurred in samples from the Tees estuary (Figure 4.18). *C. volutator* mortalities of 100% were recorded at three locations, one on Southampton Water and two on the Tees. Four sediments exhibited mortality to *A. marina* in excess of 20%, all from the Tees estuary, one of which was 100%. At several sites sediment toxicity was confined to one bioassay, as at Seal Sands using *A. marina*. This emphasises the value of carrying out joint bioassays.

This work establishes a baseline for further monitoring in these estuaries and will help design surveys elsewhere. Where persistent toxicity is found, complementary sediment chemical contaminant analysis will be necessary.

Summary

Sediment metal/aluminium ratios were highest in industrialised estuaries. Mercury and lead were above background levels at most sites, whereas arsenic, chromium and nickel were close to background levels at most sites. A similar pattern was observed when absolute concentrations were compared to provisional EACs, except in the case of arsenic. Although As/Al ratios were mostly less than twice the BRC, absolute concentrations of arsenic were above the EAC range. This is because arsenic occurs naturally in rocks in the UK and is particularly high in ironstone deposits such as occur in the Tamar estuary.

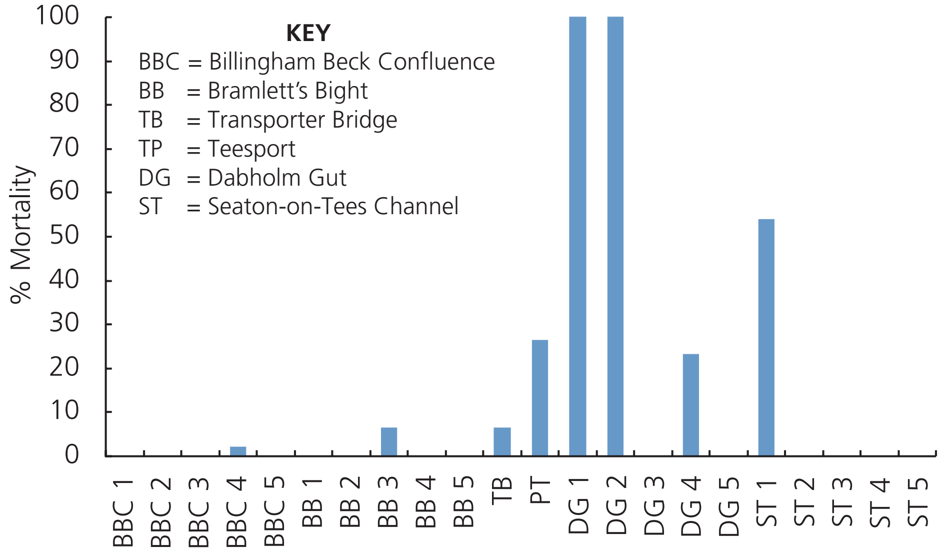


Figure 4.18 C. Volutator mortality in sediment samples from the Tees

Although point source inputs of metals have shown significant reductions with time, sediment metals concentrations will take longer to change, due to retention by strong metal-sulphide and metal-organic binding mechanisms. These historically contaminated sediments will be covered by recent deposits of less contaminated particulate material.

CB concentrations were lowest in sediments at the Scottish sites. A higher proportion of lower chlorinated CBs was observed at these sites and is consistent with atmospheric transportation being the main input route. Inshore sites contained a high proportion of the more chlorinated CBs consistent with a particulate bound, waterborne source. Sediments from most sites, including five offshore sites, had median CB concentrations above the provisional lower EAC indicating a potential impact on sediment dwelling animals.

Wide spatial differences in sediment concentrations of PAHs were observed, with the highest concentrations in industrialised estuaries and close to large urban conurbations, consistent with the fact that these compounds are derived from combustion sources.

The 2000 benthic community data showed clearly the differences between estuarine and coastal sites. Estuarine sites had larger populations but lower diversity. The sieving of coastal samples with a 1 mm mesh and of estuarine samples with 0.5 mm mesh may account, in part, for the higher abundances seen in estuarine samples. Estuarine sites had lower ITI results than coastal ones, suggesting a community bias towards surface deposit feeders in estuaries and a larger proportion of surface-detrital or water column feeders at coastal sites. However, although the ITI was developed to distinguish the impact of organic sewage deposits on fully marine fauna, high riverine organic input to estuaries depresses ITI values naturally. Lower estuarine ITI values therefore do not necessarily indicate degraded communities.

Detailed examination of the coastal benthic community data showed that univariate measures varied widely. Sites with low diversity tended to occur in areas of coarse sediment such as Morecambe Bay and off the Humber/Wash, or in very fine homogeneous sediment such as the north-western Irish Sea and Loch Linnhe. The lack of correlation between macrofaunal patterns and sediment contaminants reflects the relatively uncontaminated character of the coastal sites.

The estuarine benthic community data suggested that sites with greater diversity, evenness and ITI were in areas with small salinity gradients, low sediment mobility and low organic enrichment. In contrast, high biotic index scores at sites with high abundance of opportunist species indicated modified or heavily modified benthic communities. Spearman rank correlation showed that sediment particle size characteristics exerted more influence on the macrofaunal communities than contaminants.

A spatial survey of sediment toxicity in 4 estuaries showed localised toxicity at all sites.

4.3 Shellfish

Contaminants were measured in mussels to address the following JAMP issues.

JAMP issue 1.2: What are the concentrations and fluxes of mercury, cadmium and lead in sediment and biota?

JAMP issue 1.7: Do high concentrations of PCBs pose a risk to the marine ecosystem?

and

JAMP issue 1.11: Do PAHs affect fish and shellfish ?

The effect of TBT on dogwhelks was the only biological effect measured in shellfish .

Metals

Nine metals (mercury, cadmium, lead, copper, zinc, nickel, chromium, arsenic and silver) were measured in mussels. Concentrations were compared to BRCs (see Table 4.10). There are no EACs or EQS for metals in mussels. A conclusion from the spatial NMMP survey in 1993-95 was that efforts should be made to close significant gaps in the mussel data available for some estuaries (MPMMG, 1998). This has been achieved in the temporal trend-monitoring phase, with 22 sites sampled.

Table 4.10. BRCs for metals in mussels

Metal	BRC range (mg kg^{-1} wet weight)
Mercury	0.005-0.010
Cadmium	0.070-0.110
Lead	0.010-0.190
Copper	0.76-1.10
Zinc	11.6-30

Figure 4.19 shows the median mussel concentrations for mercury, cadmium and lead and copper and zinc for 1999-2001. However, no data were submitted for mussels from the Severn estuary, where the highest cadmium levels were reported in 1993-95 (MPMMG, 1998). Bryan et al. (1985) reported that mussels are 'unreliable' indicator species for copper, zinc, arsenic and silver, and have 'moderate' usefulness for nickel. Mussel body burdens of these metals should therefore be interpreted carefully.

Median mercury, cadmium and lead concentrations exceeded the BRCs at most sites. However, median copper and zinc concentrations were generally below the BRCs (see Figure 4.19).

The lowest mercury concentration reported was slightly above the upper BRC at 14.7 µg kg^{-1} wet weight, in Belfast Lough. All other sites were above background concentrations. Median concentrations above 50 µg kg^{-1} wet weight were observed on the Northumberland Coast, in the Blackwater estuary, in the Thames estuary, in Morecambe Bay and at St. Bees Head in Cumbria, with the highest median concentration of 126 µg kg^{-1} wet weight in the Medway estuary.

The lowest cadmium median concentration was observed in the Wash, at 102 µg kg-1 wet weight, and low median concentrations were also recorded in the Forth and Tay estuaries, in the Firth of Clyde and in Poole Harbour. Median concentrations were close to background at 9 sites. Median concentrations above 500 µg kg^{-1} wet weight were recorded in the Humber and Thames estuaries, with the highest median concentration of 1,178 µg kg^{-1} wet weight in the Medway estuary.

The lowest median lead concentration was within the BRC range, at 135 µg kg^{-1} wet weight in Lough Foyle. Median concentrations were close to background at only two sites. Median concentrations were above 1,000 µg kg^{-1} wet weight in the Clyde and Forth estuaries, on the Northumberland Coast, in the Tamar estuary and in Morecambe Bay, with the highest median concentration of 2,217 µg kg^{-1} wet weight in the Medway estuary.

Median concentrations for copper and zinc were low, except in the Blackwater, Tamar and Medway for copper, and in the Medway for zinc.

The minimum and maximum median concentrations reported for arsenic, chromium, nickel and silver in mussels for 1999-2001 are listed in Table 4.11. There are currently no BRCs, EACs or EQSs available for these metals in mussels.

The high chromium concentrations in the Clyde estuary are consistent with previous results (Burt *et al.*, 1992; Miller, 1986). The elevated silver levels in Belfast Lough may be due to photographic processing waste.

Chlorinated Biphenyls (CBs)

CB concentrations in mussels collected between 1999 and 2001 were reported for twenty-one inshore and estuarine sites, on a wet weight basis. Data were not reported for all ICES 7 CBs at all sites. The results are summarised in Table 4.12 and shown in Figure 4.20 for CB 153 and in Figure 4.21 for the sum of the ICES 7 CBs. The lowest concentration for the sum of the ICES 7 CBs was found at St. Bees on the Cumbrian coast and the highest at Sun Pier, Medway, in the south east of

Figure 4.19. Median (a) mercury, (b) cadmium, (c) lead and (d) copper concentrations in mussels, relative to their BRCs.

(e)

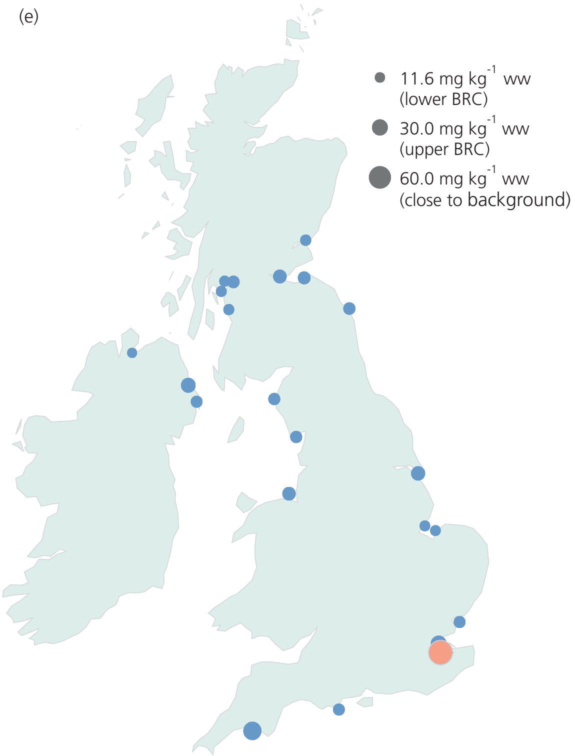


Figure 4.19. continued: Median (e) zinc concentrations in mussels, relative to their BRCs

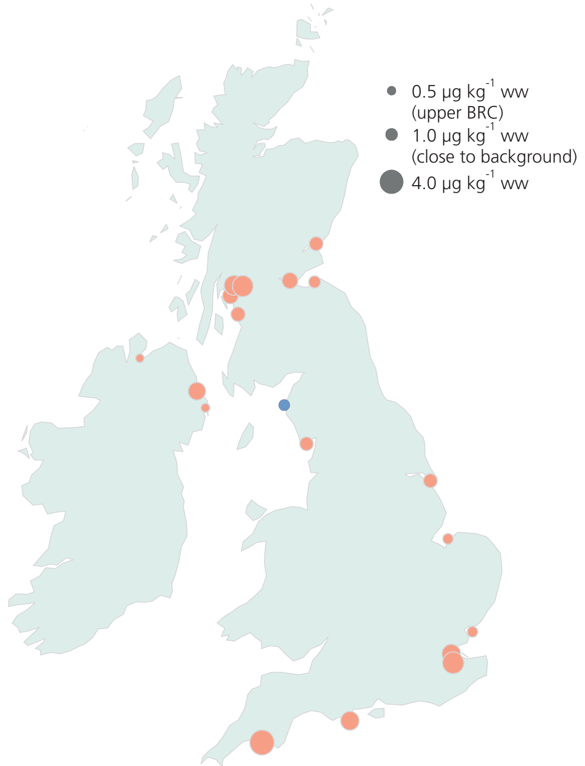


Figure 4.20. Median concentrations of CB 153 in mussels (µg kg^{-1} wet weight) and their relationship to BRCs

Table 4.11. Mussel concentrations for arsenic, chromium, nickel, and silver

Metal	Minimum median Concentration (µg kg^{-1} wet weight)	Site	Maximum median Concentration (µg kg^{-1} wet weight)	Site
Arsenic	945	Poole Harbour	7,470	Medway
Chromium	111	The Wash	2,835	Clyde Estuary
Nickel	174	Morecambe Bay	2,478	Medway
Silver	<50	Several	318	Belfast Lough

Table 4.12. Summary of PCB concentrations in mussels

	CB 153	Sum of ICES 7 CBs
BRC, µg kg^{-1} wet weight	0.1 – 0.5	0.35 – 1.7
EAC µg kg^{-1} wet weight		0.75- 7.5
Range of median concentrations, µg kg^{-1} wet weight	0.55 – 4.63	2.55 –15.86
No. of sites assessed	19	16
Sites with median concentration > upper EAC	NR[1]	7
Sites with upper 90% confidence limit on median concentration > upper EAC	NR[1]	11
Concentration range 1993-1998 µg kg^{-1} wet weight	0.006 – 12.8	

Note [1] NR = not relevant as there is no EAC for CB 153 in mussels

England. CB 153 was reported at nineteen sites with the lowest concentration again found on the Cumbrian coast and the highest at Warren Point on the Tamar (Table 4.12).

The CB profiles were similar at most sites with the penta- and hexa-CBs (CB101, CB118, 138 and 153) dominating. In general the 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.

OSPAR BRCs have been established for CB 153 and the sum of the ICES 7 CBs in mussels (Table 4.12). At most sites the median concentrations of CB 153 (Figure 4.20) and the sum of the ICES 7 CBs were above twice the upper BRCs. Median CB 153 concentrations were above 3 µg kg^{-1} wet weight at 3 sites (Warren Point in the Tamar, Sun Pier in the Medway, and Port Glasgow in the Clyde). These sites also had the highest median concentrations for the sum of the ICES 7 CBs, with values above 10 µg kg^{-1} wet weight.

EACs have been established for the sum of the ICES 7 CBs for mussels, with a range of 5–50 µg kg^{-1} dry weight. For comparison, this range (Table 4.12) was converted to a wet weight basis using a conversion factor of 0.15 (based on the median water content of mussels of 85%). The median concentrations for the sum of the ICES 7 CBs were estimated to be below the upper EAC with 90% confidence at only five sites (Figure 4.21).

Polycyclic Aromatic Hydrocarbons (PAH)
The PAH determined were: naphthalene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*a*]pyrene, benzo[*ghi*]perylene and indeno[1,2,3-*cd*]pyrene.

Mussel data were available from 19 sites. Flouranthene was the only PAH found above the limit of detection at all sites. The determination of PAH in biota has recently been introduced to the programme so relatively few data are available and this limits comparison between sites. Maximum concentrations for individual PAH are given in Table 4.13.

For all PAH determined, the concentrations were below the upper provisional OSPAR EAC (Table 4.14). A few anthracene and one pyrene value exceeded the lower provisional EAC. These data suggested that toxic effects were unlikely in the areas sampled. The concentrations of pyrene found in mussels is shown in Figure 4.22 (See note to Table 4.13).

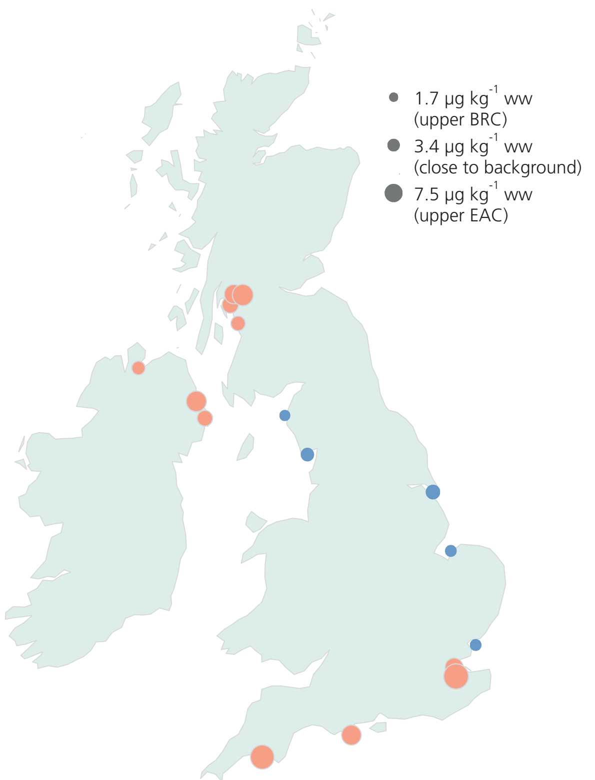


Figure 4.21. ICES 7 CBs relative to BRC and EACs

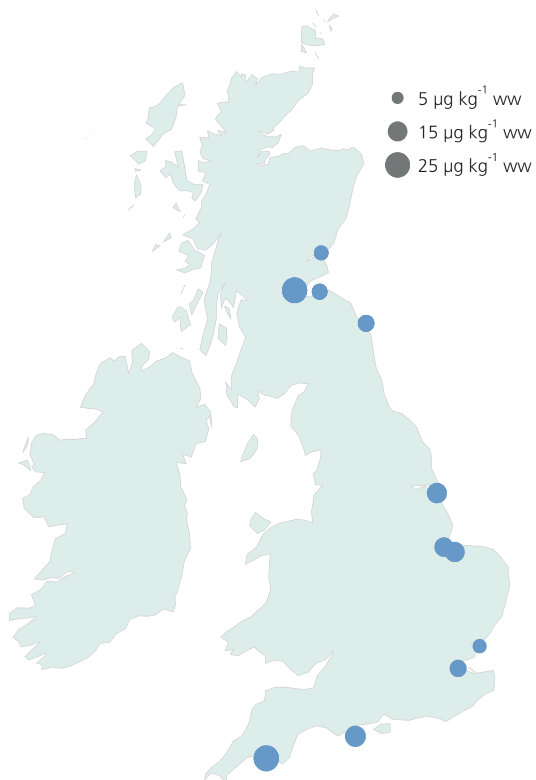


Figure 4.22 Pyrene in mussels

Table 4.13. Maximum PAH concentrations in mussels (µg/kg wet weight)

PAH compound	Maximum concentration	Site
Naphthalene	69	55 (Clyde)
Phenanthrene	94	325 (Tees)
Anthracene	3	220 (NE coast)
Fluoranthene	133	65 (Clyde)
Pyrene	197	65 (Clyde)[1]
benz[a]anthracene	5	357 (Humber)
chrysene/triphenylene	309	65 (Clyde)
Benzo[*a*]pyrene	5	357 (Humber)
Benzo[*ghi*]perylene	59	65 (Clyde)
indeno[1,2,3-*cd*]pyrene	10	220 (NE coast)

Note [1] only one year of data was available for this site so the data were excluded from Figure 4.22.

Table 4.14. Provisional EACs for mussels set by OSPAR (µg/kg dry weight and as a wet weight estimate)

PAH compound	Lower EAC		Upper EAC	
	µg kg^{-1} dry weight	µg kg^{-1} wet weight	µg kg^{-1} dry weight	µg kg^{-1} wet weight
naphthalene	500	75	5,000	750
phenanthrene	5,000	750	50,000	7,500
anthracene	5	0.75	50	7.5
fluoranthene	1,000	150	10,000	1,500
pyrene	1,000	150	10,000	1,500
benzo[*a*]pyrene	5,000	750	50,000	7,500

The mussel data were assessed as 'total PAH' using summed PAH concentrations as determined, and as benzo[*a*]pyrene equivalents (BaPEs). Total PAH concentrations ranged from 10 to 682 µg kg^{-1} wet weight, with the highest concentration at Port Glasgow in the Clyde estuary. The equivalency approach is a technique for assessing the risk posed by a number of different compounds present in a single exposure source (Bolger *et al*., 1996). Instead of calculating individual risks for each component, one compound with a known level of toxicity (benzo[*a*]pyrene in this case) is used as a standard. The concentrations of each of the other compounds are adjusted on the basis of their estimated, comparative levels of toxicity to calculate equivalent concentrations as benzo[a]pyrene. Summation of the equivalent concentrations produces a single number from which the risk can be estimated. This approach allows the assessment of PAH concentrations in relation to their carcinogenic potential, which is a longer-term risk than that indicated by the EACs. Only six of the compounds determined under the NMMP currently have toxic equivalency factors (TEFs) assigned (Law *et al*., 2002). Both the highest summed PAH concentrations (682 µg kg^{-1} wet weight) and the highest TEFs (0.02 to 32 µg kg^{-1} wet weight benzo[a]pyrene equivalents) occurred in the Clyde estuary at Port Glasgow in 2001, indicating this to be a contaminated site.

It would be easier to determine if PAH contamination at each site is due to oil or combustion sources if a large suite of PAH were determined within the NMMP in the future, including alkylated PAH. Alkylated PAH predominate in oils, whereas the parent PAH compounds do so for combustion sources.

Summary

Concentrations for several metals were elevated in mussels, mainly in industrialised estuaries including the Tyne, Tees, Thames, Mersey, Clyde and Belfast Lough. The Medway estuary is a significant 'hotspot' for mercury, cadmium, lead, arsenic, nickel and zinc due to inputs from local industry at Rochester and Gillingham, and naval activity at Chatham docks.

The BRC range was exceeded at all sites indicating that concentrations are above background levels. In addition, median CB concentrations in mussels were above the lower EAC at all sites indicating potential cause for concern.

Although PAH concentrations in sediments were high and exceeded the upper EACs at several sites, concentrations of PAH were below the relevant EACs in mussels, indicating that mussels may not be affected by the high concentrations in sediments.

4.3.1 TBT-specific biological effects

This section deals with ***JAMP issue 1.3: to what extent do biological effects occur in the vicinity of major shipping routes, offshore installations, marinas and shipyards?***

The main marine inputs of tributyltin (TBT) compounds come from biocides in antifouling paints on ships. TBT leaches slowly from the paint into the water. In recent years, it has only been used on vessels more than 25 m long, but historically it was widely used on smaller vessels, yachts and fish farm equipment.

Females of the common dog whelk, *Nucella lapillus*, respond to TBT by developing male sexual characteristics, principally a penis and *vas deferens* - a condition known as imposex. In severe cases, it leads to sterility and death and subsequent decline in dog whelk populations. The development of imposex is a very sensitive biomarker of exposure to TBT.

The degree of imposex in adult females is normally expressed as the relative penis size index (RPSI, %), in comparison to adult males in the same population, and as the vas deferens sequence index (VDSI), the average extent of the development of a vas deferens by a group of females. *Vas deferens* development has been divided into a sequence of stages from 0 to 6. A VDSI greater than 4 indicates that some of the mature females cannot reproduce.

In 1997, imposex was surveyed using OSPAR Guidelines (OSPAR, 1998) along the west coast of Great Britain and the east coast of Ireland. Away from likely TBT inputs, RPSI and VDSI in open coastal sites were low and gave no concern for the health of the populations or individuals. More detailed surveys in and around ports and marinas showed the degree of imposex to differ considerably. In general, the VDSI declined away from ports and exceeded 4 only within a few hundred metres of the input.

In 1998–99, imposex was surveyed around North Sea coasts and 23 sites were sampled on the east and south coasts. All populations showed some imposex and all females were usually affected, with small numbers of sterile females at 8 sites. At most sites, intensity of imposex had declined between 1992 and 1998.

Summary

Recent decisions within the International Maritime Organisation (IMO) prohibit application of new TBT-based coatings to ships after 2003 and require removal of existing TBT coatings by 2008. It is likely that imposex conditions around harbours and shipping lanes will improve further, with deterioration at open coastal sites now being unlikely.

4.4 Fish

Contaminants were measured in fish to address the following JAMP issues.

JAMP Issue 1.2: What are the concentrations and fluxes of mercury, cadmium and lead in sediment and biota?

JAMP Issue 1.7. Do high concentrations of PCBs pose a risk to the marine ecosystem?

and

JAMP 1.11 Do PAH affect fish and shellfish ?

Mercury and arsenic were measured in fish flesh and cadmium, lead and PCBs were measured in fish liver. Metallothionein was determined in fish livers as an indicator of exposure to metals. Finfish do not accumulate PAH as they are rapidly transformed and excreted via the bile. Exposure of finfish to PAH was assessed by determination of EROD activity in liver. DNA adducts and fish disease were measured as indicators of exposure to both PCBs and PAH.

Metals

Mercury was measured in fish flesh, and cadmium and lead in fish liver. Arsenic was reported for fish muscle only where data were required for the EC Fisheries Product Directive.

Most reported data were for dab at offshore sites and flounder in estuaries. Data were also reported for plaice at offshore sites around Scotland and in Morecambe Bay, and in whiting at St. Bees on the Cumbrian coast. At most sites similar size fish were analysed each year.

Mercury in fish flesh

The BRCs for mercury in the flesh of flat fish, including dab, flounder and plaice, are 30-70 µg kg^{-1} wet weight. Figure 4.23 shows the median mercury concentrations for 1999-2001.

The pattern of results was similar to that reported in the spatial NMMP survey of 1993-95 (MPMMG, 1998). The highest median concentrations were

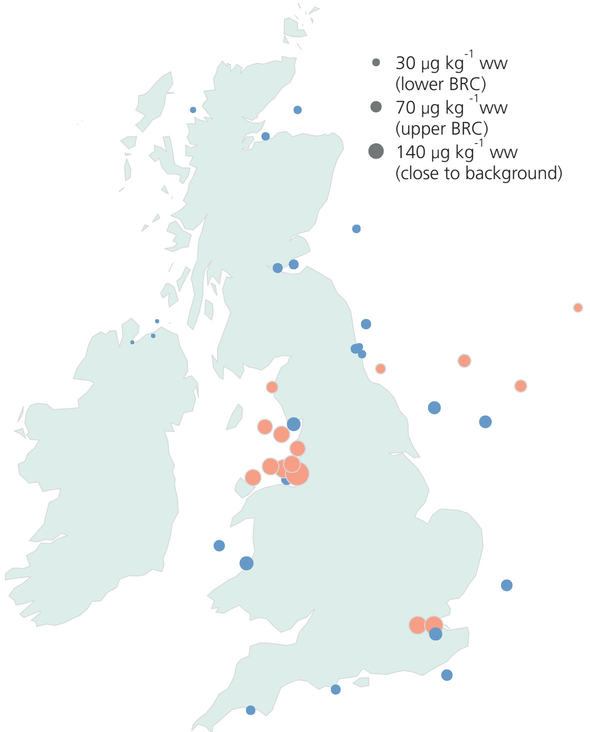


Figure 4.23. Median mercury concentrations in fish flesh

found in the Mersey estuary (374 and 188 µg kg^{-1} wet weight), Liverpool Bay (221 and 187 µg kg^{-1} wet weight) and Morecambe Bay (178 and 108 µg kg^{-1} wet weight). Median concentrations were also high at sites in the Thames estuary (227 and 212 µg kg^{-1} wet weight). Median concentrations were close to background at 24 of the 39 sites.

Cadmium and lead in fish liver

There are no BRCs or EACs for cadmium or lead in fish liver. Median concentrations of cadmium in fish liver (see Figure 4.24) were generally below 150 µg/kg wet weight, consistent with results reported by CEFAS (1998) for a spatial survey around England and Wales in 1995-96. Higher median cadmium concentrations were measured in the Thames, Moray Firth, Dogger Central, Inner Cardigan Bay and West Dogger (166, 178, 196, 334, 439 µg kg^{-1} wet weight respectively).

Median lead concentrations were below 200 µg kg^{-1} wet weight at most sites. Higher median concentrations, in the range 200-700 µg kg^{-1} wet weight, were observed in flounder from sites in the Tyne, the Wear, and in dab and flounder

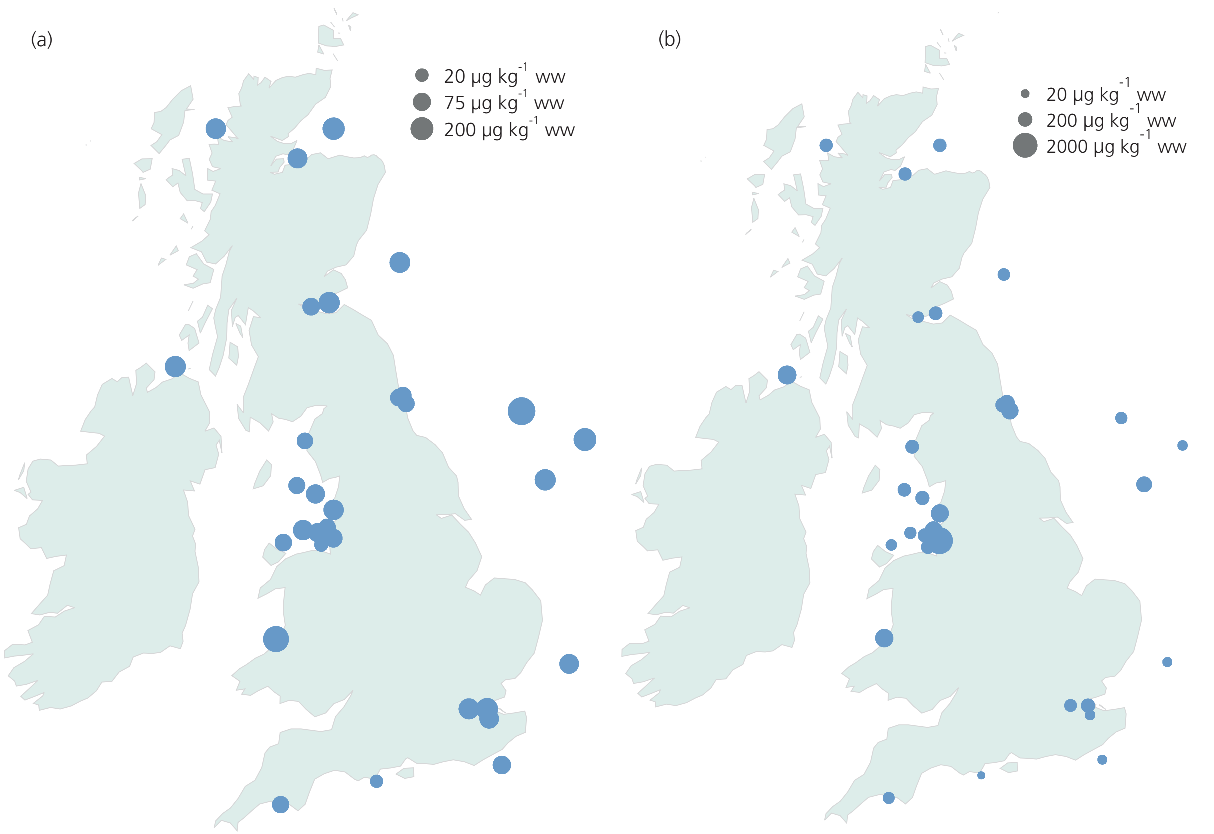


Figure 4.24 Median concentrations of (a) cadmium and (b) lead in fish liver

from sites in the Mersey and Ribble. Median lead concentrations were also in this range at sites off Humber, Cardigan Bay and North Antrim coast. The highest median concentration was 2.79 mg kg^{-1} wet weight in flounder from the inner Mersey estuary.

Arsenic in fish flesh

There are no BRCs, EACs or EQSs for arsenic in fish flesh. The results showed no clear pattern (Figure 4.25), as in the spatial survey of 1993-95 (MPMMG, 1998). The highest concentrations were in plaice from the offshore site in the Moray Firth, suggesting that the levels reflect inputs from both anthropogenic and non-anthropogenic sources.

Metal-specific biological effects in fish

Marine organisms have developed a range of strategies to cope with different concentrations of essential or toxic metals. In general, detoxification depends on the conversion of toxic - often inorganic and ionic - forms into less toxic bound forms. The bound forms such as intracellular phosphatic granules may be stored safely and subsequently excreted.

One detoxification mechanism is the induction of metallothioneins (MT). Metallothionein synthesis may be particularly stimulated by exposure via food or water to mercury, copper, cadmium and zinc. Some animals may also hold essential elements such as copper and zinc in other forms such as metalloenzymes, but excess levels are normally sequestered by MT.

Formation of MT is a natural response to exposure to certain metals and provides protection against their possible toxicity. The presence of MT is an indication of exposure to these metals, rather than a measure of subsequent health. MT is usually measured in the liver of fish, as this normally contains the highest concentrations of inducing metals.

Measurement of MT, within the NMMP started in 2001. Initial surveys measured MT in five male and five female dab (*Limanda limanda*) livers at 16 offshore sites. The livers were assayed by a standard radio-Cd saturation assay. Results are expressed as ng MT/mg protein in liver homogenate.

There was high variability at each site with factors of 10 or more between the maxima and minima, making statistical comparisons difficult (Figure 4.26). Nevertheless, concentrations tended to be higher in females (median 511 ng MT/mg protein, interquartile range 369–712 ng MT/mg protein) than in males (median 306 ng MT/mg protein, interquartile range 221–421 ng MT/mg protein).

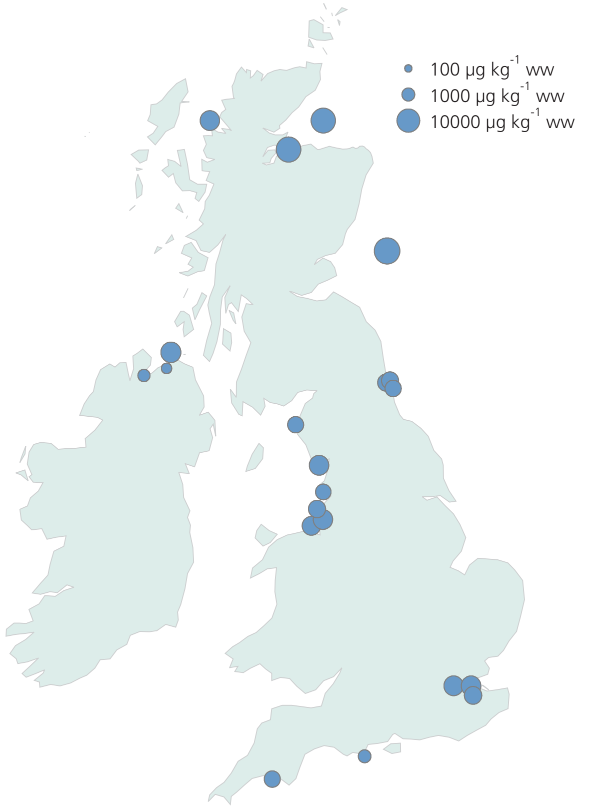


Figure 4.25. Median arsenic concentrations in fish flesh

There was no correlation between the median concentrations of MT in males and females at each site.

The livers were also analysed for copper, cadmium, lead, zinc and mercury content. Overall the relationship between metal concentration and metallothionein was not strong; however significant positive correlations were found between metallothionein and the concentration of zinc and log transformed copper, mercury and cadmium in females. The concentration of metallothionein in males was significantly positively correlated with log transformed copper and lead.

Summary

The fish data showed results consistent with previous reports, with greater than background mercury concentrations in the Mersey estuary, Liverpool Bay and Morecambe Bay, as a result of historical discharges of mercury from the chlor-alkali industry. Cadmium and lead concentrations in fish liver were generally low, with higher concentrations, typically in industrialised estuaries. Relatively high concentrations of cadmium were also measured at Dogger Bank in the North Sea.

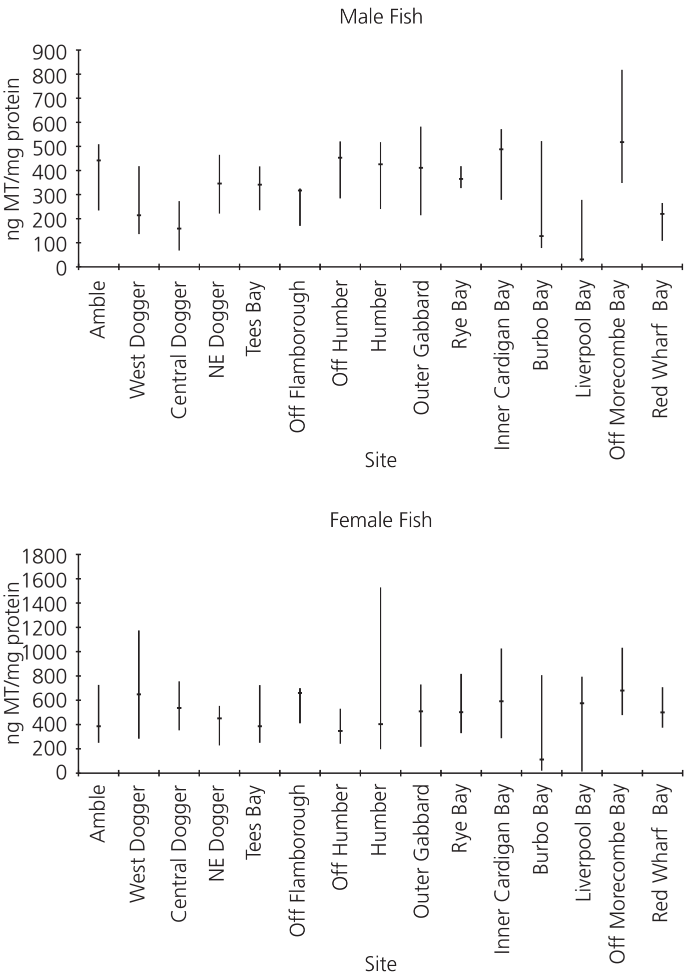


Figure 4.26. Concentrations (median and range) metallothionien in dab livers

Metallothionein concentrations in fish were highly variable, but positively correlated with copper in both males and females.

CBs

CB data were reported in fish liver at 27 sites. At most sites, dab, flounder or plaice were collected. CB data in fish are normally reported on a lipid weight basis. Data from three sites were excluded, as supporting lipid data were unavailable. Figure 4.27 shows the median concentrations for the sum of the ICES 7 CBs, normalised to lipid, Median concentrations ranged from 219 μg kg^{-1} lipid weight (Minches) to 4100 μg kg^{-1} lipid weight (Thames Mucking). The highest concentrations were found in the Thames, Mersey, Medway, Tamar (Warren Point) and Liverpool Bay.

Fish liver from Scottish sites contained the highest proportion of lower chlorinated CBs, with CB 52 and 101 dominating the profiles. This suggests a predominately atmospheric input. Most estuarine sites contained a higher proportion of the more highly chlorinated CBs, with the hexa-CBs dominating (CB 138 and 153). In coastal areas particulate bound waterborne sources dominate

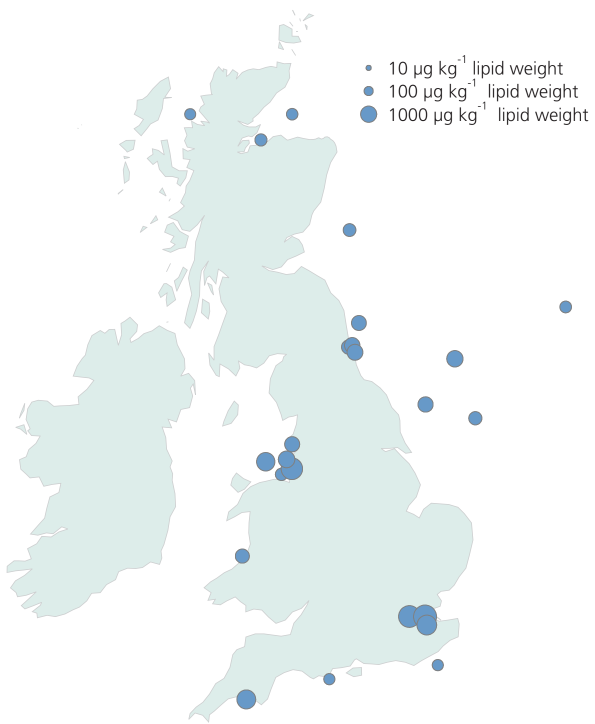


Figure 4.27 Median concentrations of the sum of the ICES 7 CBs in fish liver (μg kg^{-1} lipid weight)

and therefore there is a higher proportion of the more hydrophobic, highly chlorinated CBs. In the open sea, atmospheric deposition is more important and therefore the less chlorinated CBs tend to dominate the profiles.

Currently there are no BRCs for CBs in fish liver. The EAC for ICES 7 CBs for fish (whole) is 1 – 10 µg kg^{-1} wet weight, however there is no firm EAC available for fish liver.

Results for the 1993 to 1995 spatial survey were reported on a wet weight basis, for CB 153 only, at 78 sites. In this survey concentrations of CB 153 were greater than 20 µg kg^{-1} wet weight at more than half of the sites. The two highest concentrations were recorded in fish liver from Liverpool Bay (160 and 170 µg kg^{-1} wet weight). Similarly high concentrations were found in samples collected during the 1999-2001 survey, with median CB 153 concentrations in fish liver from 11 of the 27 sites being greater than 20 µg kg^{-1} wet weight. The median concentration at Liverpool Bay was 74 µg kg^{-1} wet weight.

EROD analysis

This section deals with ***JAMP issue 1.11: do PAHs affect fish and shellfish?***

Fish detoxify a number of organic contaminants, specifically polycyclic aromatic hydrocarbons (PAH) and some polychlorinated biphenyls (PCB), by a metabolic pathway - the mixed function oxygenase (MFO) enzyme system – that is induced by exposure to such compounds. Induction of the MFO system may be deleterious via its transformation of parent carcinogenic and genotoxic compounds into active forms. The activity of ethoxyresorufin-O-deethylase (EROD) in fish liver is a measure (expressed as pM/min/mg protein) of such exposure.

Dab at coastal and offshore sites off England and Wales were obtained by 3-m beam or Granton trawl. The ICES standard assay was used (Stagg and McIntosh, 1998). Scottish estuarine data are reported in the Forth and Clyde EROD case study (Part 2 of this report)

Mean EROD activities were calculated for each site for each year (male and female values amalgamated) and are presented in Figures 4.28 to 4.30.

1999
Fish were collected from 17 sites in May. EROD levels were higher in males than females at all sites, 6-7 times higher at some sites (Off Tees, Flamborough and NW Dogger). The highest male: female EROD ratios were recorded in the North Sea.

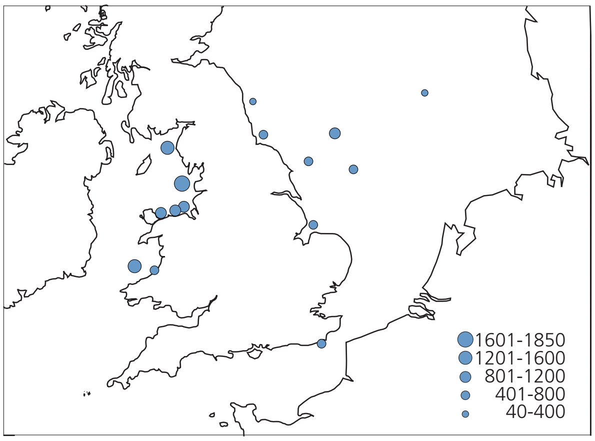


Figure 4.28. Mean EROD activity (pM/min/mg protein) in dab in 1999

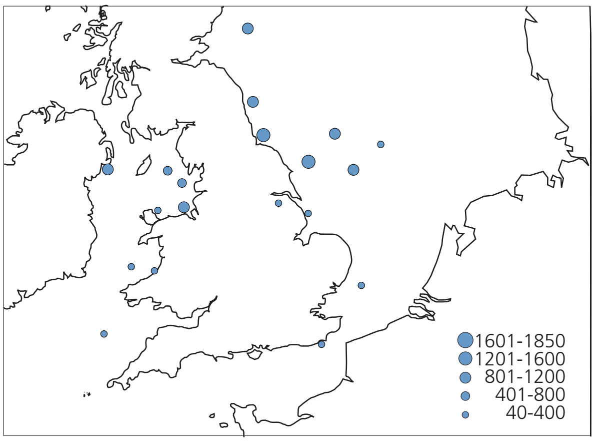


Figure 4.29. Mean EROD activity (pM/min/mg protein) in dab in 2000

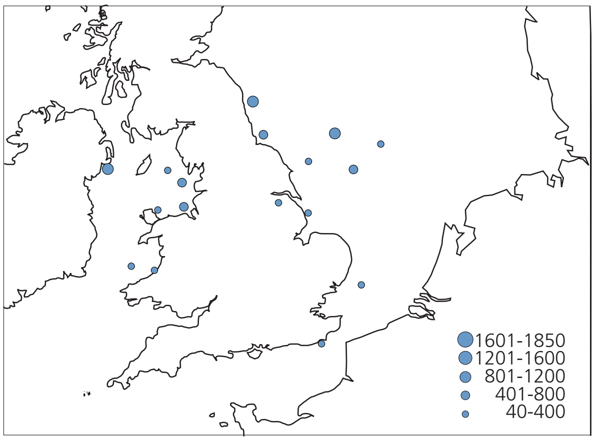


Figure 4.30. Mean EROD activity (pM/min/mg protein) in dab in 2001

These inter-sex differences in EROD relate to sampling in early May, close to the end of the spawning period. The reproductive cycle of North Sea dab is: pre-spawning (Sept-Dec), spawning (Jan-April), post-spawning (May) and resting (June-August) (Kirby *et al.*, 1999) with some year to year variability. Male levels are highest during the spawning period and lowest during the post-spawning/resting phase (May to August); females do not show as strong a trend (Lange *et al.*, 1999).

In male fish, high levels were found off Morecambe, St Bees Head and Outer Cardigan Bay (1854, 1617 and 1574 pM/min/mg protein respectively) (Figure 4.28). High values were also found in females at the Liverpool Bay and Irish Sea stations, but were low in the North Sea.

2000

Dab were sampled at 19 sites in June. The data were similar for both sexes, reflecting the reproductive state of the fish at the time, so were combined.

The highest mean values on the East Coast were off Tees, off Flamborough, West Dogger, off Humber and in the Firth of Forth, with 1570, 1253, 1147, 1047 and 1014 pM/min/mg protein respectively (Figure 4.29). Highest values on the West Coast were at near-shore sites in Liverpool Bay (Off Morecambe and Burbo Bight) and Dundrum Bay, east coast of Ireland. Low mean values were found at the South of Humber (46 pM/min/mg protein) and Rye Bay (134 pM/min/mg protein) sites.

The 2000 data generally relate to known areas of organic contamination although correlations of size and reproductive parameters with EROD suggest that the recorded activities may also have been influenced by non-contaminant sources.

2001

Dab were sampled from 19 sites in June/July. The data showed, overall, slightly reduced EROD values (Figure 4.30). No site showed a mean value above 1000 pM/min/mg protein and the highest values (≥800 pM/min/mg protein) were found in the east and northeast. A mean value of 858 pM/min/mg protein at Dundrum Bay, although similar to that recorded in 2000, should be treated with caution as it derived from only 5 fish. Low values were again recorded in samples from Rye Bay and Cardigan Bay.

DNA Adducts

In fish, the detoxification of some PAHs (e.g. benzo[*a*]pyrene and benzo[*a*]anthracene) produces highly reactive metabolites, termed DNA adducts, capable of binding to DNA. Such chemical modification of the DNA is considered to be one of the initial steps of chemical carcinogenesis. A causal link between sediment PAH contamination, DNA adduct formation and liver pathology in benthic fish species has previously been demonstrated. An association between PAH exposure, DNA adduct formation and cancerous lesion development in fish has also been revealed by a number of field and laboratory studies in both Europe and North America. Analysis of DNA adducts in liver therefore provides a sensitive biomarker of PAH exposure.

Flatfish (dab, *Limanda limanda*, and flounder, *Platichthys flesus*) greater than 15 cm long were collected and their livers were immediately removed and stored in liquid nitrogen. Samples were analysed by 32P-postlabelling assay for DNA adducts (Randerath *et al.*, 1981). The DNA adduct levels are shown in Table 4.15.

Dab collected from coastal and offshore NMMP sites

Adducts were detected at all sites apart from outer Cardigan Bay (1996 and 1998) and off the Tyne (1998). The DNA adduct profiles indicated exposure to complex mixtures of genotoxins and the levels varied both spatially and temporally. For example, dab caught in outer Cardigan Bay during 1999 and 2000 contained profiles characteristic of exposure to complex mixtures of carcinogenic metabolites.

It is clear that coastal dab are exposed at some times and in some places to sufficient genotoxins to induce detectable levels of DNA adducts. Dab migrate through unpolluted and polluted waters, so it is not possible to link specific sites to the induction of DNA adducts.

Flounder collected from the Alde, Tyne, Tees and Mersey

Significantly higher levels of adducts were detected in the contaminated Tyne estuary compared with fish from the reference Alde site during both spring and autumn. PAH bile metabolite analysis of flounder caught in May and October confirmed that fish were exposed to higher levels of PAH compared with those in the Alde. Differences between adduct levels in the Mersey and Tees during the autumn were not statistically significant but were elevated over those in the Alde.

DNA adduct data are likely to reflect local contamination because flounder are usually found in their home estuaries for up to 8 months of the year. Fish from the Tyne in spring and autumn revealed aromatic or hydrophobic DNA adducts in the liver, typical of exposure to a complex mixture of PAH. This supports previous work (Lyons *et al.*, 1999) showing that Tyne fish stocks are exposed to high levels of sediment-associated PAH and that a proportion of the bioavailable PAH was being metabolised to carcinogenic metabolites. Significantly, Tyne flounder show more pre-cancerous lesions, specifically hepatocellular foci of cellular alteration, compared with fish from unpolluted sites (DETR, 2001).

Table 4.15. Levels of hepatic DNA adducts (DNA adducts per 108 undamaged nucleotides) in dab and flounder from UK coastal waters

Date	Station	NMMP	DNA adducts per 10^8 undamaged nucleotides
Dab (offshore/coastal)			
6/2000 [c]	Offshore Forth/Tay	165	3.2 ± 0.8 [a] (7) [b]
6/2000	Amble	244	7.6 ± 2.0 (10)
6/1999	Amble	244	2.3 ± 1.0 (10)
6/2000	West Dogger	285	5.4 ± 1.0 (10)
6/1996	West Dogger	285	5.6 ± 2.3 (4)*
6/2000	Flamborough	344	2.0 ± 0.5 (10)
6/2000	Off Humber	345	4.3 ± 1.1 (10)
6/1996	Off Humber	345	7.6 ± 1.0 (4)*
6/1999	Rye Bay	486	2.1 ± 0.4 (10)
6/2000	Celtic deep	605	10.6 ± 2.1 (10)
6/2000	Outer Cardigan Bay	665	6.0 ± 1.3 (10)
6/1999	Outer Cardigan Bay	665	10.0 ± 1.9 (9)
6/1998	Outer Cardigan Bay	665	0.0 ± 0.0 (5)*
6/1996	Outer Cardigan Bay	665	0.0 ± 0.0 (5)*
6/2000	Burbo Bight	705	2.3 ± 1.3 (10)
6/1999	Burbo Bight	705	6.1 ± 1.4 (10)
6/1998	Burbo Bight	705	5.6 ± 2.2 (10)
6/1996	Burbo Bight	705	16.2 ± 9.4 (4) *
6/2000	Liverpool Bay	715	7.5 ± 1.2 (10)
6/2000	Red Wharf Bay	776	8.2 ± 1.9 (10)
6/1999	Red Wharf Bay	776	12.7 ± 2.2 (10)
6/1998	Off Tyne		0.0 ± 0.0 (5)*
6/1996	Off Tyne		4.0 ± 2.6 (5)*
Flounder (Estuaries)			
5/2000	Tyne	/	7.4 ± 3.1 (6)
10/2000	Tyne	/	14.1 ± 2.7 (6)
10/2000	Tees	/	5.0 ± 1.5 (6)
10/2000	Mersey	/	7.4 ± 3.1 (5)
5/2000	Alde	/	1.7 ± 0.4 (6)
10/2000	Alde	/	3.0 ± 0.9 (6)

[a] *Mean adduct levels ± SE*
[b] *Numbers in parentheses represent number of individual liver samples analysed*
[c] *Sample date*
* *Number in parentheses represents number of pooled samples analysed (3 fish per pool)*

Fish pathology and disease biomarkers
Fish diseases and pathological changes in the liver have long been used as indicators of environmental stress on fish populations. The International Council for the Exploration of the Sea (ICES) has contributed to standardised methodologies. Externally, conditions used for monitoring include acute and healing ulcerations, lymphocystis, epidermal hyperplasia/papilloma and hyperpigmentation. Internally, liver lesions comprising nodules and larger tumours have become routine to monitor the health of offshore fish species, such as the dab (*Limanda limanda*). Although the aetiology of certain diseases is known - for example an iridovirus is known to cause lymphocystis - that of others remains uncertain. However, measurement of these diverse conditions in individual fish helps monitor the overall health of a population.

Sampling and disease reporting protocols followed those recommended by ICES (Bucke *et al.*, 1996). Dab (*L. limanda*) were examined for externally visible diseases. Fish greater than 20 cm long were dissected and, where macroscopic liver nodules were present, they were removed and preserved in 10% neutral buffered formalin (NBF) before examination to see if they were malignant. Standard sections of liver tissue from up to 50 dab were also collected and examined for histopathology. Although the range of possible lesions in flatfish is large, the categories here have been restricted to those that are indicative of contaminant exposure. Grade 1 indicates 'no abnormalities detected', Grade 2 'foci of cellular alteration (FCA)', Grade 3 'benign tumours' (adenoma, cholangioma, hemangioma) and Grade 4 'malignant tumours' (carcinoma).

Table 4.16 summarises the prevalence of externally visible diseases (lymphocystis, ulceration, epidermal hyperplasia/papilloma, hyperpigmentation) and of liver nodules greater than 2 mm in diameter.

Liver nodules were most prevalent at the sites in Liverpool Bay, Flamborough and West Dogger and relatively high at other sites on the Dogger (including North and East Dogger and the Hospital Ground). The high prevalence at inner Cardigan Bay was based on a limited number of fish and greater confidence would require larger sample sizes. In 2000, liver nodules were found in seven fish from the Firth of Forth during the only visit to that location and were not detected at any other Scottish sites visited between 1999 and 2001. Data on the prevalence of the four grades of liver pathology from Rye Bay, West Dogger, Burbo Bight (Liverpool Bay) and Cardigan Bay are given in Figures 4.31 to 4.34 respectively. A greater proportion of livers appeared normal (Grade 1) at Rye Bay (the reference site) compared to other sites, particularly in 1999, but gross liver pathology (liver nodules) and a very low proportion of Grade 4 pathology were detected in 2001. Histological examination of liver from fish captured at this site revealed Grade 2 and 3 pathology in all three years. In contrast, the proportion of dab sampled from the West Dogger and Burbo Bight sites exhibiting Grade 4 pathology was considerably higher. At West Dogger, the prevalence of gross liver nodules was reasonably consistent (Figure 4.32) while at Burbo Bight and Cardigan Bay, the proportion of fish exhibiting gross liver nodules was higher in 2000 and 2001 than in 1999 (Figures 4.33 and 4.34). At Cardigan Bay, the proportion of fish displaying Grade 2 liver pathology remained fairly consistent and there was an increase in the prevalence of Grade 3 liver pathologies.

Summary

CB concentrations were lowest at the Scottish sites. A higher proportion of lower chlorinated CBs was observed at these sites and is consistent with atmospheric transportation being the main input route. Inshore sites contained a high proportion of the more chlorinated CBs consistent with a particulate bound, waterborne source.

No comparisons to EACs could be made for fish liver. However, median concentrations in all estuarine sites were greater than 500 µg kg^{-1} lipid weight. These data highlight that at inshore and estuarine sites, particularly in the south of England and Liverpool Bay, CB concentrations continue to be of concern.

Sites in Liverpool Bay and off the North East coast of England have shown consistent elevated EROD activity. These results suggest that fish populations in these areas have undergone long-term exposure to MFO-inducing chemicals. It has been shown that flounder (*Platichthys flesus*) populations in the estuaries of the Mersey, Tyne and Tees exhibit significantly elevated levels of EROD activity (Kirby *et al.*, 1999). The data presented here suggest that coastal populations of other fish species, such as dab, may also be affected. The potential influence of season and specimen size on EROD activity reinforces the requirement to adhere to the NMMP monitoring protocol.

Dab and flounder both exhibited patterns of DNA adducts indicative of exposure to complex mixtures of PAH. However, it may be that other environmental contaminants were contributing to the overall genotoxic response. The migration of dab is a complicating factor. DNA adducts are persistent and the detection of DNA adducts in dab at a particular location may be a consequence of previous exposure to contaminants elsewhere. This highlights the need for integrated monitoring where biomarkers of recent exposure over days or weeks, such as EROD and bile metabolites, are used alongside DNA adducts reflecting cumulative contaminant exposure over months. DNA adducts are associated with increases in pre-cancerous and cancerous lesions in marine flatfish. Around the UK, some dab and flounder populations are being exposed to carcinogenic contaminants partly responsible for their observed pre-cancerous lesion

Fish disease monitoring showed externally visible diseases in dab to remain at generally low levels. In particular, the results from Scottish waters indicate lower levels of disease than are found more generally in the North and Irish Seas. Overall, the prevalence of external diseases in dab was similar to that previously described from the same sampling locations in recent years (Anon, 2000, 2001). The dab captured off Flamborough, in some areas of the Irish Sea and on the Dogger Bank continue to exhibit higher levels of disease than fish from the Rye Bay reference site. Hyperpigmentation was a prominent condition in dab from several areas in the North Sea, but had low prevalence in the Irish Sea and in the English Channel at Rye Bay. The prevalence of hyperpigmentation has increased at the Cardigan Bay and West Dogger sites during the 1990s (CEFAS, 1998 and 2000). Ulceration was a common pathology in dab from several sites,

Table 4.16. External disease and liver nodule prevalence per year and site, as percent.

Sampling Site	Year	Nearest NMMP Site	No. Fish Examined	Lympho-cystis	Ulceration	Epidermal-Papilloma	Hyper-pigmentation	Liver Nodules
Rye Bay	1999	486	346	1.2	9.8	1.7	4.3	0.0
	2000		202	0.0	4.5	0.5	5.4	0.0
	2001		272	0.0	4.4	0.0	0.4	1.2
Inner Cardigan Bay	2000*	656	231	2.6	9.1	1.3	8.2	25.0
	2001*		112	0.0	6.3	1.8	2.7	25.0
Smiths Knoll	2001	395	153	0.0	0.7	0.7	0.0	0.0
Red Wharf Bay	1999	776	265	1.9	9.8	1.1	0.4	0.6
	2000		223	0.0	9.9	0.9	0.9	4.0
	2001		294	0.0	13.9	0.3	0.0	0.5
Inner Liverpool Bay (Burbo Bight)	2000	706	206	3.9	14.0	0.5	1.5	3.5
Liverpool Bay 2	1999	715	290	1.0	16.9	1.0	0.3	5.3
	2000		291	1.4	28.2	2.1	1.4	4.2
	2001		259	0.0	18.1	0.8	0.4	5.0
Morecambe Bay	2000	795	263	1.1	14.1	0.8	0.4	0.0
	2001		271	0.0	11.8	1.1	0.0	0.6
Off Flamborough	1999	344	242	7.9	2.5	2.1	16.5	8.0
	2000		257	8.9	1.6	2.7	10.5	3.5
	2001		166	3.6	0.0	0.6	3.0	5.4
Outer Humber	2000	345	261	5.7	1.9	1.1	11.9	1.9
Silver Pit	2001	~	271	4.8	2.2	1.5	4.4	1.2
West Dogger	1999	286	376	8.5	5.3	2.4	22.1	8.7
	2000		267	6.0	4.5	2.6	35.2	11.4
	2001		262	1.5	4.2	0.8	7.6	5.0
Hospital Ground	2000	287	275	3.6	22.2	1.1	25.1	5.7
	2001		259	3	10	2	29	8
North Dogger	2001	284	319	1.9	8.2	0.9	10.3	5.9
Amble	2000	244	234	7.7	1.3	1.7	7.7	0.7
	2001		157	8.9	1.3	1.9	1.9	2.0
East Dogger	2001	288	237	9.7	9.3	0.8	1.7	4.4
St Bees	1999	768	253	2.0	7.1	2.8	0.0	4.0
	2000		147	8.2	4.8	1.4	4.8	0.0
Burbo Bight	1999	705	355	1.7	9.9	2.8	0.3	4.6
	2000		234	2.1	17.9	1.7	0.4	1.5
	2001		225	0.0	20.0	0.9	0.0	4.9
Off Tees	1999	295	188	5.9	1.1	3.7	5.9	0.0
	2000		147	8.2	4.8	1.4	4.8	0.0
	2001		150	3.3	2.0	2.0	0.7	2.2
Firth Of Forth	2000	~	224	6.3	1.8	5.4	17.0	5.6
Outer Gabbard	2000	475	174	0.6	4.6	2.3	7.5	0.9
	2001		218	0.0	1.4	1.4	0.5	0.8
Indefatigable Bank	2001	~	265	0.4	4.2	0.4	8.3	3.1
SE Isle Of Man	2001	805	158	2.5	14.6	0.6	0.0	0.0
Bell Rock	1999	155	~	5.3	0.7	1.5	~	~
	2000		2367	2.9	0.6	0.9	~	~
	2001		1085	5.0	0.5	2.7	3.8	~
St Abbs	1999	155	~	1.6	2.1	1.5	~	~
	2000		552	2.9	1.6	1.3	~	~
	2001		513	6.2	1.4	2.1	1.4	~
Moray Firth	2000	95	3426	4.2	1.0	2.0	~	~
	2001		1687	4.2	1.6	2.9	2.0	~
East Orkney	1999	~	~	2.9	0.6	0.5	~	~
	2000		858	3.6	0.5	1.2	~	~
	2001		515	1.9	0.0	2.3	1.8	~

** Prevalence of liver nodules calculated from only 32 and 12 fish in 2000 and 2001 respectively*

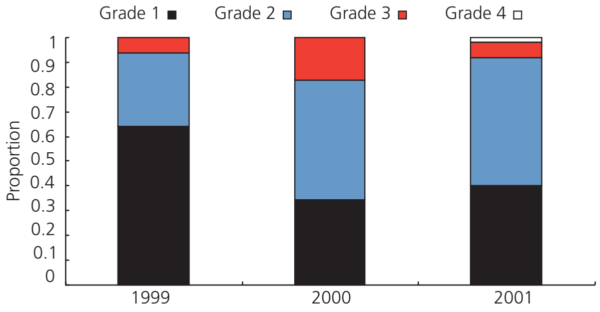


Figure 4.31. Chart showing the proportion of grades of liver pathology in dab sampled from the Rye Bay reference site

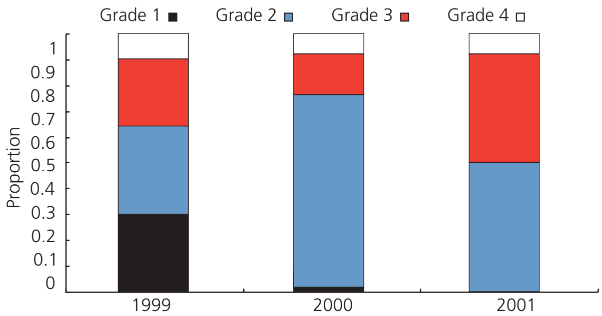


Figure 4.32. Chart showing the proportion of grades of liver pathology in dab sampled from West Dogger

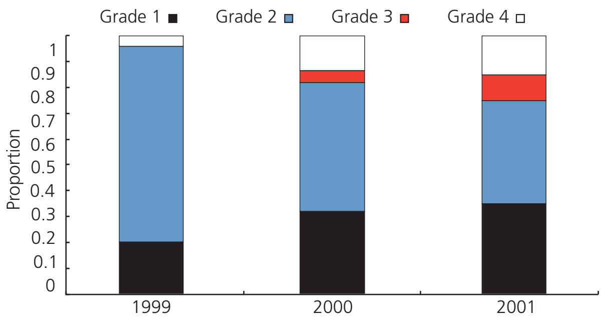


Figure 4.33. Chart showing the proportion of grades of liver pathology in dab sampled from Burbo Bight (Inner Liverpool Bay)

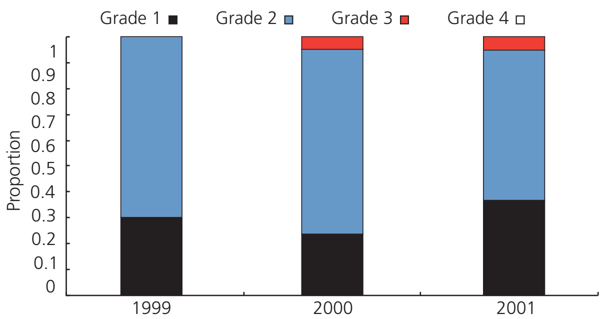


Figure 4.34. Chart showing the proportion of grades of liver pathology in dab sampled from Cardigan Bay

with the highest prevalence on the Dogger Bank and from areas in the Irish Sea (including Liverpool Bay and Burbo Bight). Lymphocystis was seen at most sites, with the highest prevalence in dab from the North Sea (most notably off Flamborough, off the Tees estuary and at sites on the Dogger Bank). Epidermal papilloma was recorded at low prevalence in fish from all sites. This pathology was most common in fish from the Firth of Forth, West Dogger, off Flamborough, off Tees, and inner Cardigan Bay sites. The presence of macroscopic nodules or tumours was noted in fish captured from numerous sites, with the highest prevalences from the Dogger Bank and Flamborough sites. This is consistent with previous findings (CEFAS, 1998 and 2000).

Glossary

Adenoma - a benign tumour of glandular tissue (including liver).

Aetiology - assignment of a cause or reason.

Carcinoma - a malignant tumour derived from epithelial cells.

Cholangioma - a benign tumour of bile ducts.

Epidermal hyperplasia/papilloma - an externally visible disease appearing as pale raised areas of skin. Possibly caused by a virus.

Hemangioma - a benign tumour of endothelial (lining) cells.

Hyperpigmentation - condition of increased pigmentation where the upper surface exhibits dark green/black areas (increased numbers of pigmented cells containing melanin). On the underside, affected areas mainly appear white with some melanisation.

Liver nodule - a discrete pathology of the liver greater than 2mm in diameter.

Lymphocystis - a disease caused by an iridovirus giving rise to multiple small nodules, usually on the surface of the fish.

Neoplasia - an abnormal mass of tissue.

4.5 Trend detection

The ability of the NMMP to detect temporal trends has been estimated using the first three years of data. The 10-year detectable trend was calculated for each contaminant time series (Nicholson *et al*,, 1997) and the results were summarised by the median 10-year detectable trend for each determinand/matrix combination (Table 4.17). After ten years' monitoring, the NMMP is likely to detect a 6% annual decrease or increase in metal sediment concentrations (equivalent to a 43% decrease or 69% increase over ten years). However, the NMMP is only likely to detect a 20% annual decrease or increase in CB sediment concentrations (an 87% decrease or 416% increase over ten years).

The performance of the NMMP was also summarised by the median number of years required to detect a 50% reduction in concentration for each determinand / matrix combination (Table 4.18). After 10 years of monitoring, 50% reductions are only likely to be detected in metal concentrations in sediment, shellfish, and water. After 20 years of monitoring, 50% reductions in nutrient concentrations and in PAH concentrations in sediment are also likely to be detected. However, even after 20 years of monitoring, a 50% reduction in CB concentrations would probably not be detected by the present sampling regime.

The 1999-2001 monitoring data will now be used to redesign the NMMP with high power to detect trends in all groups of contaminants. This implies an increase in sampling effort at 'key' monitoring stations, and a reduction in monitoring frequency at other 'sentinel' stations where contaminant levels are close to background.

Table 4.17. Percent annual change in contaminant concentration required to detect trend after 10 years

	Sediment	Shellfish	Seaweed	Fish	Water
Metals	6%	6%	16%	13%	5%
CBs	20%	15%		17%	
PAHs	11%	16%			
pesticides		18%			12%*
nutrients					9%

* *This value is unreliable because most pesticide measurements in water were below the limit of detection.*

Table 4.18. Numbers of years required to detect a 50% reduction in contaminant concentrations

	Sediment	Shellfish	Seaweed	Fish	Water
Metals	7	7	>20	>20	6
CBs	>20	>20		>20	
PAH	20	>20			
pesticides		>20			>20
nutrients					14

5. CONCLUSIONS AND RECOMMENDATIONS

Riverine inputs and direct discharges to tidal waters of many contaminants have decreased since monitoring was initiated by OSPAR in 1990. Discharges of cadmium and mercury in particular have decreased substantially as a result of controls on industrial discharges. The environmental monitoring data presented in this report cover the 3 years from 1999-2001 so it is not possible to comment on long term trends associated with trends in inputs. The results have shown the spatial distribution of contaminants and their associated biological effects and have been used to address the relevant JAMP issues. Examples of longer term trends in environmental contaminants associated with trends in inputs at selected sites are presented in Part 2 of the report.

JAMP issue 2.1 Are agreed measures effective at reducing nutrient inputs?

Inputs of phosphate have almost halved since 1990 but there has not been a corresponding decrease in inputs of nitrogen. Agricultural inputs of nitrogen are expected to decrease as a result of the recent designations of Nitrate Vulnerable Zones, which impose controls on agricultural practices. The area of coastal waters impacted by these inputs depends on the relative size of the input and available dilution. Highest winter nitrate concentrations occurred in the Wash, Thames, Liverpool Bay and Eastern Irish Sea. The biological effects of these inputs were not included in the NMMP, but spring maximum chlorophyll concentrations have been found to be correlated to riverine nutrient loads to coastal waters.

Do contaminant inputs lead to the exceedence of Environmental Quality Standards in UK waters?

Riverine and direct discharges of most contaminants have decreased in recent years as a result of controls on point source discharges. Contaminant concentrations in estuaries were below EQSs but above background concentrations. Oyster embryo bioassay tests confirmed that water quality in estuaries was generally good but variable at some locations, with sporadic effects occurring at the same site sampled at different times.

JAMP issue 1.2 What are the concentrations and fluxes of mercury, cadmium and lead in sediment and biota?

Historical inputs have left a legacy of contaminated sediments in some areas. Concentrations of mercury, cadmium and lead in sediments were more than twice the background reference concentration at most sites and particularly high in industrialised estuaries where the concentrations were greater than the ecotoxicological assessment criteria. However, there was no correlation between concentrations of metals and benthic community structure, which was largely determined by the physical nature of the sediments, salinity and water depth. The lack of correlation of benthic communities with contaminants suggests that the stress caused by contaminants is relatively small compared to variability in physical parameters between sites and that the current provisional EACs for metals in sediments require revision. However, it was possible to correlate improvements in the benthic community structure with a substantial reduction in organic inputs in Belfast Lough.

Baseline sediment bioassays have identified sites in the Mersey, Tyne and Tees estuaries where pore waters were toxic to test organisms. The extent and degree of toxicity were particularly pronounced in the Tees.

Concentrations of mercury, cadmium and lead were above background in mussels from industrialised estuaries including the Tyne, Tees, Thames, Mersey, Clyde and Belfast Lough, reflecting the pattern observed in sediments . The results indicated a significant ‘hotspot’ for mercury, cadmium, lead, arsenic, nickel and zinc in the Medway estuary, possibly due to a combination of inputs from local industry at Rochester and Gillingham, and naval activity at Chatham docks. The biological effect of metal contamination in mussels was not measured but the bioavailability of metals will be assessed in future through the measurement of metallothionein and possible biological effects determined by the scope for growth technique.

The fish data showed greater than background mercury concentrations in flesh from samples from the Mersey estuary, Liverpool Bay and Morecambe Bay, as a result of historical discharges of mercury from the chlor-alkali industry. Cadmium and lead concentrations in fish liver were generally low, with some higher concentrations, typically in industrialised estuaries. Higher concentrations of cadmium were also measured at Dogger Bank in the North Sea. There was no correlation between mercury, cadmium and lead concentrations in fish liver and metallothionein, but there was a positive correlation with copper.

JAMP issue 1.7. Do high concentrations of PCBs pose a risk to the marine ecosytem?

Production of PCBs stopped in the 1970s and they were removed from sale in 1986. However PCBs still enter the marine environment from the disposal of old equipment. PCBs were present in all sediment samples. Concentrations were lowest at Scottish offshore sites and highest in the Thames and Medway. The higher proportion of lower chlorinated CBs observed at the Scottish offshore sites is consistent with atmospheric transportation being the main input route. Inshore sites contained a high proportion of the more chlorinated CBs consistent with a particulate-bound, waterborne source. Concentrations in sediments were above ecotoxicological assessment criteria in the Clyde, Severn, Thames, Medway, but no contaminant effect was observed on the benthos.

Median CB concentrations in mussels were above the lower EAC at all sites but their effects were not measured. Median concentrations of CBs in fish liver were relatively high in the Thames, Medway, Tamar, Mersey and Liverpool Bay.

JAMP issue 1.10 What are the concentrations of PAHs in the maritime area?

JAMP issue 1.11 Do PAHs affect fish and shellfish

Wide spatial differences in sediment concentrations of PAH were observed, with the highest concentrations in industrialised estuaries and close to large urban conurbations, consistent with derivation of these compounds from combustion sources. Although concentrations in sediments were high and exceeded the upper EACs at several sites, concentrations were below the relevant EACs in mussels. This discrepancy between sediments and mussels suggests revision of the provisional EACs for PAH is required.

A suite of methods was used to determine the possible biological effects of PAH in fish including EROD, DNA adducts and fish diseases. Fish caught in coastal waters showed low levels of exposure to PAH, whereas fish caught in industrialised estuaries showed higher levels of exposure. These results are consistent with higher concentrations of contaminants in estuarine sediments, in particular the Mersey and Tyne.

JAMP issue 1.3: TBT- to what extent do biological effects occur in the vicinity of major shipping routes, offshore installations, marinas and shipyards?

Imposex in dogwhelks is directly related to TBT contamination. The effects were widespread but reproduction appeared only to be significantly impaired in the immediate vicinity of marinas and harbours where TBT was used as an anti-foulant. Environmental TBT concentrations in dogwhelk flesh were not measured but such monitoring will commence in 2003.

Although concentrations of some contaminants in waters, sediment and biota were found to be above current OSPAR BRCs and EACs in some areas it was often difficult to relate contaminant levels with measured biological effects. OSPAR has recognised that the current values for BRCs and EACs require revision and is in the process of addressing this issue. These values may not therefore be appropriate for the UK and the assessments made against the current criteria should be treated with caution.

The ability of the NMMP to detect temporal trends varied widely across groups of contaminants. For example, the NMMP would probably detect a 50% reduction over ten years in metal concentrations in sediment, shellfish or water. However, the same reduction in metal concentrations in fish, or in CB or PAH concentrations in any matrix, would probably not be detected by the current sampling regime. The NMMP will now be redesigned to have higher power to detect trends in all groups of contaminants. This implies an increase in sampling effort at 'key' monitoring stations and a reduction in monitoring frequency at other 'sentinel' stations where contaminant levels are close to background.

In the light of known future developments and the results obtained from this second phase of the NMMP, it is recommended that:

- the programme should be adapted to fulfil future monitoring requirements arising from changes to the OSPAR JAMP and the EC Water Framework Directive;
- the nutrients monitoring programme should be revised to align more closely with any OSPAR assessment criteria for eutrophication;
- the development of biological techniques should continue and be further integrated with chemical measurements undertaken for the programme;
- where contaminant concentrations in sediment exceed assessment criteria levels, sediment bioassays should be used to assess whether there may be any impact on biota;
- The biological effect of metals and other contaminants on mussels should be assessed through the measurement of metallothionein and scope for growth.
- The programme should be redesigned to target effort at identified 'key' monitoring sites.

6. REFERENCES

ANON, 2000. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating disposal of wastes at sea, 1997. Science Series, Aquatic Environment Monitoring Report. CEFAS Lowestoft, 52: 39-44.

ANON, 2001. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating disposal of wastes at sea, 1998. Science Series, Aquatic Environment Monitoring Report. CEFAS Lowestoft 53: 30-33.

Balls, P.W., 1992. Nutrient behaviour in two contrasting Scottish estuaries, the Forth and Tay. Oceanologica Acta, 15: 261-277.

Bolger, M., Henry, S.H. and Carrington, C.D., 1996. Hazard and risk assessment of crude oil contaminants in subsistence seafood samples from Prince William Sound. American Fisheries Science Symposium, 18: 837-843.

Borja, A., Franco, J. and Perez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environment. Marine Pollution, 40(12): 1110-1114.

Bryan, G.W., Langston, W.J., Hummerstone, L.G., and Burt, G.R., 1985. A guide to the assessment of heavy-metal contamination in estuaries using biological indicators. Marine Biological Association of the UK. Occasional Publication No. 4.

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 Techniques in Marine Environmental Science, 19: 27pp.

Burt, G.R., Bryan, G.W., Langston, W.J., and Hummerstone, L.G., 1992. Mapping the Distribution of Metal Contamination in UK Estuaries. Plymouth Marine Laboratory Report. Plymouth.

CEFAS, 1998. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1995 and 1996. Science Series Aquatic Environment Monitoring Report. CEFAS Lowestoft, 51: 116pp.

CEFAS, 2000. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1997. Aquatic Environment Monitoring Report, MAFF Directorate of Fisheries Research, Lowestoft, 52: 92pp.

Codling, I.D. and Ashley, S.J., 1992. Development of a biotic index for the assessment of pollution status of marine benthic communities. Final Report to SNIFFER and NRA. NR 3102/1.

DETR, 2001. Pathology biomarkers in estuarine fish species for the assessment of biological effects of contaminants. CDEP 84/5/287.

Evans G.L., Williams, P. J., Le B. and Mitchelson-Jacob, G., 2001. Interdecadal Variability of Nutrients in the Irish Sea School of Ocean Sciences, University of Wales, Bangor.

Fryer, R.J. and Nicholson, M.D., 1999. Using smoothers for comprehensive assessments of contaminant time series in marine biota. ICES Journal of Marine Science, 56: 779 -790.

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. Estuarine, Coastal and Shelf Science, 54: 927-939

Holme, N., AND McIntyre, A.D. (eds)., 1971. Methods for the Study of Marine Benthos, IBP Handbook 16. Oxford, Blackwell Scientific Publications.

Kirby, M.F., Matthiessen, P., Neall, P., Tylor, T., Allchin, C.R., Kelly, C.A., Maxwell, D.L., Thain, J.E., 1999. Hepatic EROD activity in flounder (*Platichthys flesus*) as an indicator of contaminant exposure in English estuaries. Marine Pollution Bulletin, 38: 676 -686.

Lange, U., Goksoyr, A., Siebers, D. and Karbe, L., 1999. Cytochrome P450 1A-dependent enzyme activities in the liver of dab (*Limanda limanda*): kinetics, seasonal changes and detection limits. Compendium of Biochemistry and Physiology B., 123: 361-371.

Law, R.J., Kelly, C., Baker, K., Jones, J., McIntosh, A.D. and Moffat, C.F., 2002. Toxic equivalency factors for PAH and their applicability in shellfish pollution monitoring studies. Journal of Environmental Monitoring, 4: 383-388.

Lyons, B.P., Stewart, C. and Kirby, M.F., 1999. The detection of biomarkers of genotoxin exposure in the European flounder (*Platichthys flesus*) collected from the River Tyne Estuary.

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.

MPMMG, 1998. National Monitoring Programme. Survey of the Quality of UK Coastal Waters. Marine Pollution Monitoring Management Group. Aberdeen. ISBN 09532838 3 6.

Nicholson, M.D., Fryer, R.L. and Ross, C.A., 1997. Designing monitoring programmes for detecting temporal trends in contaminants in fish and shellfish. Marine Pollution Bulletin, 34: 821-826.

NMMPWG, 2001., National Marine Monitoring Programme, Green Book. www.marlab.ac.uk

Nedwell, D.B., Dong, L.F., Sage, A. and Underwood, G.J.C. 2002. Variations of the nutrients loads to the mainland U.K. estuaries: correlation with catchment areas, urbanization and coastal eutrophication. Estuarine, Coastal and Shelf Science. 54: 951-970.

OSPAR Commission, 1999. Report of an assessment of trends in the concentrations of certain metals, PAHs and other organic compounds in the tissues of various fish species and blue mussels: OSPAR Ad Hoc Working Group on Monitoring 1998.

OSPAR Commission, 2000a. Quality Status Report 2000. OSPAR Commission, London, 108pp.

OSPAR Commission, 2000b. Quality Status Report 2000, Region II — Greater North Sea. OSPAR Commission, London, 136pp.

OSPAR Commission, 2000c. Quality Staus Report 2000. Region III - Celtic seas. OSPAR Commission, London, 116pp.

OSPAR Commission, 2000d. Background Reference Concentrations (BRCs) and Ecotoxicological Assessment Criteria (EACs). London. December 2000.

OSPAR Commission, 2000e. OSPAR background document on mercury and organic mercury compounds. OSPAR Commission, London, 28pp.

OSPAR Commission, 2002. OSPAR background document on cadmium. OSPAR Commission, London, 56pp.

Randerath, K., Reddy, M.V. and Gupta., R.C., 1981. ^{32}P-postlabelling test for DNA damage. Proceedings of the National Academy of Science USA, 78: 6126-6129.

Warwick, R.M. and Clarke, K.R., 1991. A comparison of some methods for analysing changes in benthic community structure. Journal of the Marine Biological Association of the United Kingdom, 71(1): 225-244.

UK CASE STUDIES

This section reports marine environmental monitoring data that are not currently included in the NMMP. There are 3 different themes:

1. Spatial surveys of determinands such as plankton or brominated flame retardants that may be included in the future NMMP.

2. Long term trends at specific sites such as Belfast Lough and the estuaries of the Forth, Tyne, Tees, Thames, Humber and Mersey. These case studies discuss data gathered before the start of the NMMP and discuss contaminants not included in the NMMP.

3. Monitoring of general interest, such as litter, the Milford Haven oil spill and disposal ground monitoring.

Wherever possible the structure of case studies has been based on the DPSIR model, progressing from driving forces and pressures to the state of the environment and the impact/biological effect of contaminants.

1. MARINE LITTER

Litter may be defined as visual and tangible pollution from the inappropriate disposal of waste. It is a serious and persistent environmental problem, posing a hazard to beach users, recreational water users and wildlife. It looks unsightly, spoils our enjoyment of Nature and may have serious economic impact on coastal communities. It is of great public concern and is largely preventable, as its source is solely anthropogenic. The general functioning of society, producing waste as an inevitable by-product, is the main driving force in the problem of litter pollution. Litter only arrives on our beaches because waste is allowed to escape into the environment.

The main sources of litter

The Marine Conservation Society (MCS) in its 2001 'Beachwatch' report estimated that about 37% of collected litter items were attributable to tourism or beach visitors. Fly tipping, the illegal deposit of any waste onto land, is also a common problem.

Sewage related debris is a problem on some beaches. About 1.5-2.0 billion sanitary protection products are flushed down UK. toilets every year, with about 60-100 million condoms. Many of these end up in rivers, the sea and our beaches. The range of coastal litter reflects the wide range of contributory sources. Coastal litter was quantified by the MCS: a beach cleaning exercise at 194 sites in September 2001 found 11,894 crisp and sweet packets, 12,485 caps and lids, 9,507 plastic drink bottles, 6,647 cigarette stubs, 5,278 drink cans, 4,849 plastic bags, 6,389 bits of glass, 1,584 fast food containers, 1,126 balloons, 358 4 or 6 pack yolks. The total weight of litter debris collected from 141.3 km of beaches was 8.3 tonnes.

Shipping litter comes in many shapes and sizes. The International Convention for the Prevention of Pollution from Ships 1973, or MARPOL (2), was modified in 1978 and ratified in June 1994 by 69 countries, including the UK. It regulates the types and quantities of operational and cargo wastes that may be discharged from ship to sea, taking into account the ecological sensitivity of different sea areas. One requirement of the MARPOL ruling is that under no circumstances are plastics to be disposed of at sea. However, everyday domestic ship waste is often disposed of at sea rather than bringing it back into port for proper disposal –MCS estimate that 3% of litter collected during its Beachwatch survey could be attributed to this source. Cargo, bulk and containerised, may wash overboard in storms and contributes substantial individual items of coastal litter, and poses a potential shipping hazard. Fishing nets and lines are lost by accident or are dumped deliberately. These items may either wash ashore or sink to the sea bottom.

State of the environment

The MCS Beachwatch survey is the best long-term indication of trends on UK beaches. Despite a decline in litter in the last four years, it remains around 50% higher than in 1994 (Figure 1).

The National Aquatic Litter Group (NALG) is a consortium working to reduce litter pollution. The EA, SEPA and EHSNI are all partners. NALG has developed a standardised monitoring and assessment protocol for assessing the aesthetic state of beaches. The protocol has a standard survey unit and a four grade classification scheme from A (very good) to D (poor), based on the

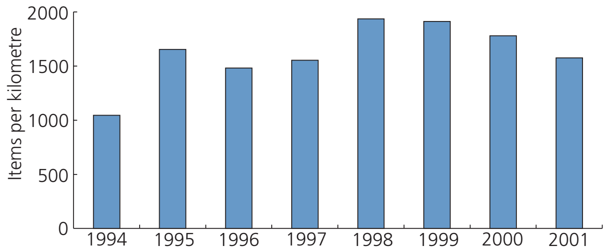


Figure 1. Beachwatch Surveys 1994-2001. Number of items per kilometre (Source: Marine Conservation Society)

assessment of a range of parameters including sewage related debris, gross and general litter, potentially harmful debris, oil, dog faeces and large accumulations of litter.

During 2000-2002 the EA conducted annual snapshot surveys of each of its designated coastal bathing waters, using the protocol. For consistency, and to measure trends, the surveys were in July. In 2002, the survey covered 472 beaches.

Most beaches were graded A or B (very good or good) with 77% in 2000 rising to 82% in 2002 (Figure 2). At the same time, the number of grade D beaches fell from 10% in 2000 to 5% in 2002. Although these were 'snapshot' surveys, this is encouraging.

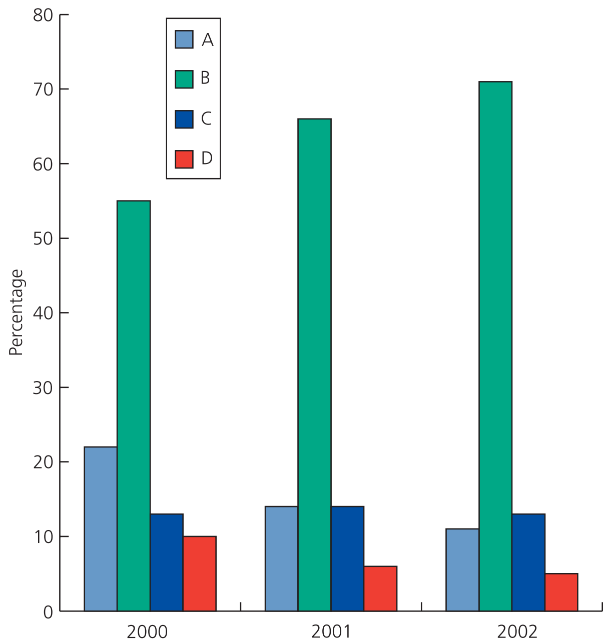


Figure 2. Beach Aesthetics Surveys (England & Wales) (Source: Environment Agency)

Impacts of marine litter

Litter on our coasts may have a serious aesthetic, environmental, health and economic impact. Marine litter may kill and injure marine mammals, seabirds and other forms of marine and coastal life. Plastic litter kills an estimated 100,000 marine mammals and turtles world-wide every year - including 30,000 seals, and up to one million seabirds, either through entanglement or via ingestion.

An autopsy by the Cotentin Cetaceans Study Group and the University of Caen analysed the stomach contents of a whale stranded on a beach in Normandy, and found nearly a kilogram of plastic bags and packaging. They found one plastic and foil crisp bag and two supermarket plastic bags - all from the UK, seven coloured dustbin bag fragments, seven transparent bags and one food container.

Some litter is long-lived and active for decades. It consists to a very great extent of plastics, metal and glass, materials that do not break down easily or quickly.

Responses to litter pollution

The key to controlling marine litter is to tackle it at source. The main regulatory control of shipping litter is MARPOL. Land sourced litter is controlled by a number of regulations, the most important of which are the Environment Act, 1990 (England, Scotland and Wales) and the 1994 Northern Ireland Litter Order. Another key theme is educational initiatives such as the 'Bag it and Bin It' campaign that aims to educate people into not flushing litter down the toilet towards our beaches and riverbanks. Encouraging participation in initiatives such as the MCS Beachwatch survey also helps to educate and inform the public about the problems of aquatic litter. The EA also seeks public participation in providing data on beach litter by helping to monitor beaches using the NALG protocol and entering such data directly onto its website through an interactive link.

The answer to reducing the threat, general unsightliness and costs of marine litter and debris is through a combination of regulation, education and co-operation between the public and interested groups and organisations.

2. CONTINUOUS NUTRIENT MONITORING AT THE NMMP BUOY IN THE NORTH WESTERN IRISH SEA

Automatic water samplers have been used to collect continuous data at the NMMP buoy (site 808) in the Irish Sea, where concerns about eutrophication have focused attention on biological processes affecting the recycling of nutrients. The samplers allowed the determination of nutrients to traceable standards but the analytical quality relied on preserving samples in their original state and getting good agreement between sampler data and coincident spot samples. Short-term trials taking independent samples at the same time and place as the sampler showed no difference between the two sets of samples. Long-term data revealed more variability between results.

Nitrate data have proved robust and reliable and of most interest since phosphorus is not a limiting nutrient. Nevertheless, it seems desirable to make future measurements of phosphorus and to develop the autosampler so that a full range of nutrients may be measured reliably.

Measurements restricted to one depth and one station (Figure 3) showed seasonal cycles of spring uptake and winter replenishment and possible interannual variability. Maximum values occured in the winter when the water was well mixed and were probably better described than the minima.

Summary

Problems of deployment, filtering and mooring security were resolved but the prevention of biofouling and the validation of samples in situ has been difficult. Biofouling has now largely been overcome and the sampler may be accepted as part of a long-term programme giving a context for short-term cruises. The data acquired by the samplers and associated thermistor strings have shown the difficulties of accounting for water movements in a heterogeneous water mass. Large changes in nutrients from day to day and apparent changes in water temperature reflected advection - the passage of different water masses. These results have modified our concept of the gyre in this area as an isolated water mass: major events in both surface and deep waters during the summer implied some ingress of water to the region.

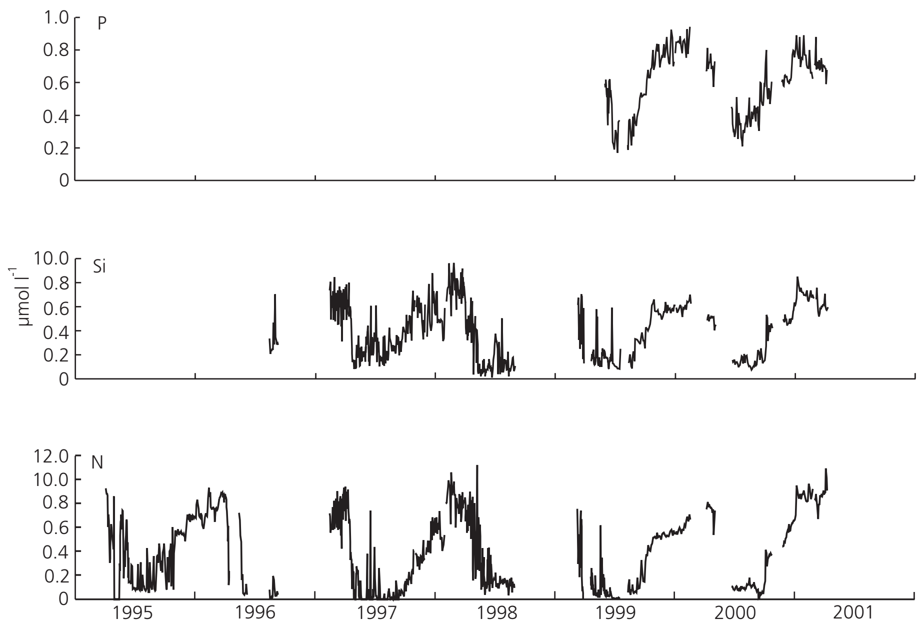


Figure 3. Autosampler data for P, Si and N: μmol l^{-1}

3. TRENDS IN PHYTOPLANKTON IN UK COASTAL WATERS FROM ANALYSIS OF CONTINUOUS PLANKTON RECORDER (CPR) DATA

Distinguishing between natural and anthropogenic variability

It is sometimes assumed that local anthropogenic inputs are the major factor determining long-term variations in coastal nutrient concentrations, and hence trophic status (Allen *et al.*, 1998, Gowen *et al.*, 2002). However, a recent Irish Sea study concluded that climate is a significant factor. This contradiction, based on the same nutrient data, highlights the importance of natural background variations in accounting for nutrient concentrations by climate change, as well as by local anthropogenic inputs.

Many phytoplankton responses thought to be associated with changes in nutrients or nutrient ratios may have equally plausible natural explanations. Distinguishing between natural and anthropogenic changes is problematic for biological populations and nutrient concentrations, requiring background data with wide spatial and long temporal coverage.

The importance of spatial coverage is exemplified by the North Sea, where about 90% of the inputs of the major nutrients, nitrate and phosphate, derived from the North Atlantic (NSTF, 1993). In coastal areas such as the Wadden Sea, German Bight, Kattegat and eastern Skagerrak, nutrient enrichment and oxygen de-oxygenation (OSPAR, 2000) have been attributed to localised inputs whereas natural large-scale effects are in fact responsible.

These spatial aspects are highlighted by long-term biological patterns, where the dominant external force depends on the scale of study. For example, small-scale biological patterns may be driven by biological interactions, whereas large-scale patterns may be caused predominantly by physical or climatic factors. Therefore, one of the disadvantages of small-scale ecological studies is that larger-scale responses are obscured and spuriously attributed to local effects.

The continuous plankton recorder (CPR) survey, started in 1931 in the North Sea, is one of few long-term biological monitoring programmes and the only one at a wide spatial scale giving systematic coverage of the north-east Atlantic and UK shelf seas. Because the survey extends sufficiently to detect different regional variability it may help determine whether localised anthropogenic effects have a wider-scale impact and it may help separate anthropogenic from climatic signals. This natural variability, influenced by global warming, should be considered in assessing the ecological state of UK coastal waters.

CPR data have recently provided information on harmful algal blooms, monitoring and documenting the spread of non-indigenous plankton species, and describing changes in marine biodiversity. Some case studies using CPR data are summarised below.

Recent case studies by the CPR survey

Eutrophication versus natural variability

Sudden increases in phytoplankton biomass seen after the mid-1980s in the North Sea and west of Ireland between 52°N and 58°N (Reid *et al.*, 1998) were accompanied by a decreasing trend in the Northwest. Phytoplankton abundance increased in the late 1980s/ early 1990s, with many species occurring one or two months earlier than normal (Figure 4).

This stepwise increase in chlorophyll levels in the North Sea might naively be seen as evidence of eutrophication. However, an identical pattern occurred west of the British Isles. This suggests a strong overriding climatic signal common to regional areas of the north-east Atlantic and coastal areas of the North Sea, rather than eutrophication. Indeed, long term changes in phytoplankton biomass and phytoplankton community correlated well with the changing climate of the North Atlantic and trends in the North Atlantic Oscillation (NAO) index (Edwards *et al.*, 2001).

Climate change and biogeographical shifts

While the CPR survey has recorded a rapid rise in the incidence of sub-tropical plankton species around the British Isles over the last decade, it has also shown a strong biogeographical shift in biodiversity, with northward extension of more than 10° in latitude of warm-water species, associated with a decrease in colder-water species (Beaugrand, *et al.*, 2002). These shifts correlate with the NAO index and Northern Hemisphere temperature, suggesting a link to climate change and global warming.

Harmful algal blooms

The CPR surveys identify about 170 phytoplankton taxa. Within the CPR database there are numerous

Table 1. Known harmful and detrimental phytoplankton taxa recorded by the CPR survey in the North Atlantic and UK coastal waters

Species/genus	Associated harmful/detrimental effects	Time-series
Ceratium furca	Hypoxia/anoxia	1948 –
Coscinodiscus wailesii	Production of mucilage.	First recorded in 1977 (invasive)
Dinophysis spp	Diarrhectic shellfish poisoning (DSP).	1948 –
Gonyaulax spp	Unspecified toxicity.	1965 –
Noctiluca scintillans	Discolouration and hypoxia/anoxia.	1981 –
Phaeocystis spp	Production of foam and mucilage. Hypoxia/anoxia.	1946 – (presence/absence)
Prorocentrum micans	Diarrhectic shellfish poisoning (DSP). Discolouration and hypoxia/anoxia	1948 –
Pseudo-nitzschia spp	Amnesic shellfish poisoning (ASP)	1948 –
Nitzschia closterium (now *Cylindrotheca*)	Production of foam and mucilage.	1948 –
Chaetoceros spp	Gill clogging	1948 –
Skeletonema costatum	Gill clogging	1948 –

harmful and detrimental phytoplankton taxa, listed in Table 1. One of the most investigated is the foam alga *Phaeocystis* that blooms massively in the southern North Sea area, whose long-term monthly variability of *Phaeocystis* is shown in Figure 5. Whereas *Phaeocystis* was particularly common in the North Sea in the 1950s it declined through the 1960s and 1970s. Since the mid 1980s however, it has increased and been recorded frequently. 1999 in particular was an exceptional year with a number of large blooms.

It has been suggested that increases in *Phaeocystis* could be attributed to increased nitrogen and phosphorus inputs (Lancelot *et al.*, 1987). However, similar patterns were found in other non-eutrophic regions of the north-east Atlantic, suggesting the patterns were caused by climate change, mirroring the patterns of phytoplankton biomass. Without inter-regional comparisons, many blooms in the past have been wrongly attributed to eutrophication.

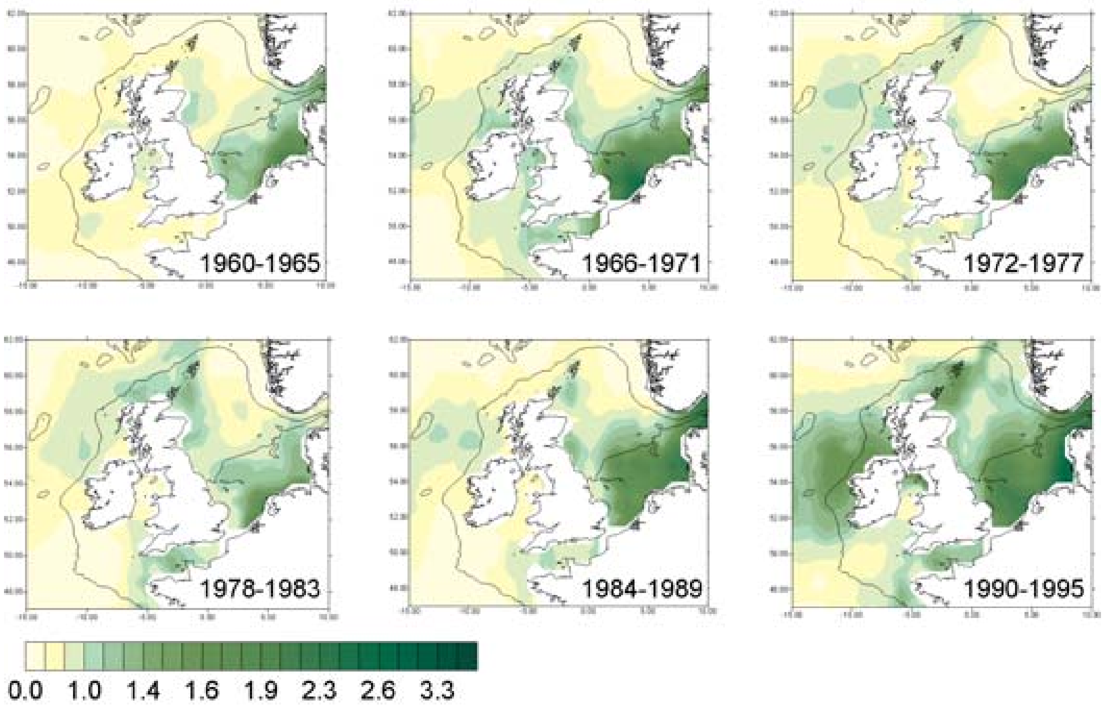


Figure 4. Changes in phytoplankton colour index in the North Sea

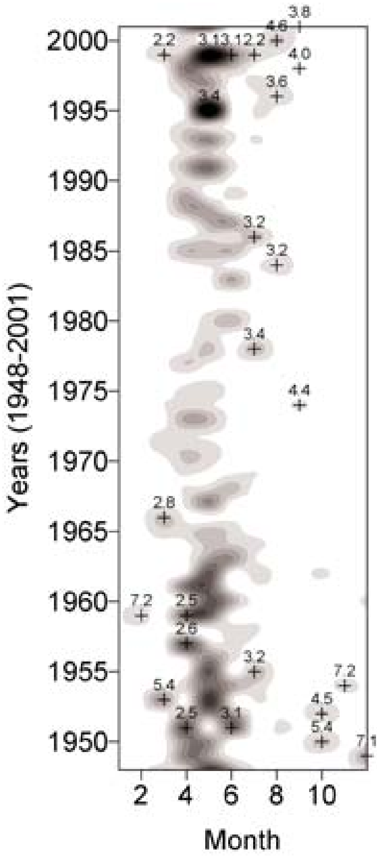


Figure 5. Long term monthly variability of Phaeocystis

Summary

Major biological and climatic changes have happened over the last few decades in water around the British Isles. It is clear that hydro-climatic variability to some extent drives ecological changes. Whether it dominates depends on the scale of study and the importance of localised or climatic variability. Indeed, eutrophication in European regional seas cannot be assessed without taking into account the wider hydro-climatic influences (Edwards and Reid, 2001; Edwards *et al.*, 2002). It is only when these wider-scale Atlantic influences are considered that we may have confidence in the assessments of anthropogenic impacts on the ecological status of UK coastal waters.

References

Allen, J.R., Slinn, D.J., Shammon, T.M. and Hartnoll, R.G., 1998. Evidence of eutrophication of the Irish Sea over four decades. Limnology and Oceanography, 43, 1970-1974.

Beaugrand, G., Reid, P.C., Ibanez, F., Lindley, J.A. and Edwards, M., 2002. Reorganisation of North Atlantic Marine Copepod Biodiversity and Climate. Science, 296, 1692-1694.

Edwards, M., 2000. Large-scale temporal and spatial patterns of marine phytoplankton and climate variability in the North Atlantic. Ph. D. thesis, University of Plymouth, 243pp.

Edwards, M. and Reid, P.C., 2001. Implications of wider Atlantic influences on regional seas with particular reference to phytoplankton populations and eutrophication. Contract report for DEFRA, 10pp.

Edwards, M., Reid, P.C. and Planque, B., 2001a. Long-term and regional variability of phytoplankton biomass in the north-east Atlantic (1960-1995). ICES Journal of Marine Science, 58, 39-49.

Edwards, M., Beaugrand, G., Reid, P.C., Rowden, A.A. and Jones, M.B., 2002. Ocean climate anomalies and the ecology of the North Sea. Marine Ecology Progress Series, 239, 1-10.

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 nutrient trends in nutrient concentrations in coastal shelf seas: a case study in the Irish Sea. Estuarine, Coastal and Shelf Sciences, 54, 927-939.

Lancelot, C., Billen, G., Sournia, A., Weisse, T., Colijn, F., Veldhuis, M.J.W., Davies, A. and Wassman, P., 1987. *Phaeocystis* blooms and nutrient enrichment in the continental coastal zones of the North Sea. Ambio, 16, 38-46.

North Sea Task Force (NSTF), 1993. North Sea Quality Status Report 1993. - London (Oslo and Paris Commissions) & Fredensborg, Denmark (Olsen & Olsen).

OSPAR Commission, 2000. Quality Status Report 2000, region ii – greater North Sea. – 136pp. London (OSPAR commission).

Reid, P. C., Edwards, M. , Hunt, H.G. and Warner, A.J., 1998. Phytoplankton change in the North Atlantic. Nature, 391, 546.

4. ASSESSMENT OF EUTROPHICATION IN THE SOLWAY FIRTH

The eutrophication status of the Solway Firth has been assessed for the requirements of the Urban Waste Water Treatment Directive and of the Oslo and Paris Commission (OSPAR) Comprehensive Procedure (CSTT, 1993, 1997; EEC, 1991; Jak, 2001).

Pressure

The Solway Firth catchment is mainly rural, with intensive farming. Agriculture accounts for 70% of land use, mainly mixed livestock with dairy, beef and sheep. In 1991, the resident population was about 415,000. The most densely populated area lies around the three industrial and port towns of Whitehaven, Workington and Maryport. Discharges of domestic sewage and industrial effluent are small, but rivers are subject to localised diffuse sources of pollution, mainly from farming

Most nutrients derive from 14 rivers (Table 2) with the Annan, Nith and Eden as the main sources (Table 3 and Figure 6).

State

Degree of Nutrient Enrichment

Winter nutrient concentrations were measured in 2001 (Figure 7), 2002 and 2003. Estuarine nutrients and salinity were correlated (Figure 8): nitrate concentrations were highest at low salinity, reflecting river input at the head; the phosphate

Table 2. Nutrient Inputs to the Solway Estuary and Firth

Source	N-NH3 (t yr^{-1})	TON (t yr^{-1})	TN (t yr^{-1})	P-DRP (t yr^{-1})	P-TP (t yr^{-1})	Mean Flow (m^3 s^{-1})
Scottish Rivers	365	6,865	10,068	214	428	192.7
English Rivers	48.8	4,527	N/A	354	N/A	19.5
Sewage Works	92	61	200	52	63	N/A
Industry	6	5	89	25	45	3.2
Totals	511.8	11,414	N/A	645	N/A	N/A

Table 3. Loadings from Individual Rivers

Scottish Rivers

River	N-NH3 (t yr^{-1})	TON (t yr^{-1})	TN (t yr^{-1})	P-DRP (t yr^{-1})	P-TP (t yr^{-1})	Mean Flow (m^3 s^{-1})
Annan	95	2,593	3,208	89	161	41.3
Nith	87	2,216	2,884	74	91	38.7
Esk	34	416	625	9	28	20.9
Dee	55	259	782	8	46	42.5
Cree	22	194	424	5	14	14.6
Bladnoch	13	142	339	4	11	11.3
Liddle	19	224	543	11	40	9.6
Luce	12	47	209	2	7	6.2
Urr	18	444	529	4	9	5.1
Kirtle	6	198	352	7	18	1.6
Piltanton	4	132	172	1	3	0.9
Totals	365	6,865	10,063	214	428	192.7

English Rivers

River	N-NH3 (t yr^{-1})	TON (t yr^{-1})	TN (t yr^{-1})	P-DRP (t yr^{-1})	P-TP (t yr^{-1})	Mean Flow (m^3 s^{-1})
Eden	44	4,027	N/A	333	N/A	18.0
Wampool	2.45	301	N/A	17	N/A	0.7
Waver	2.35	199	N/A	4	N/A	0.8
Totals	48.8	4,527	N/A	354	N/A	19.5

Note: total N and total P are measured on unfiltered water samples; na indicates that data are not available

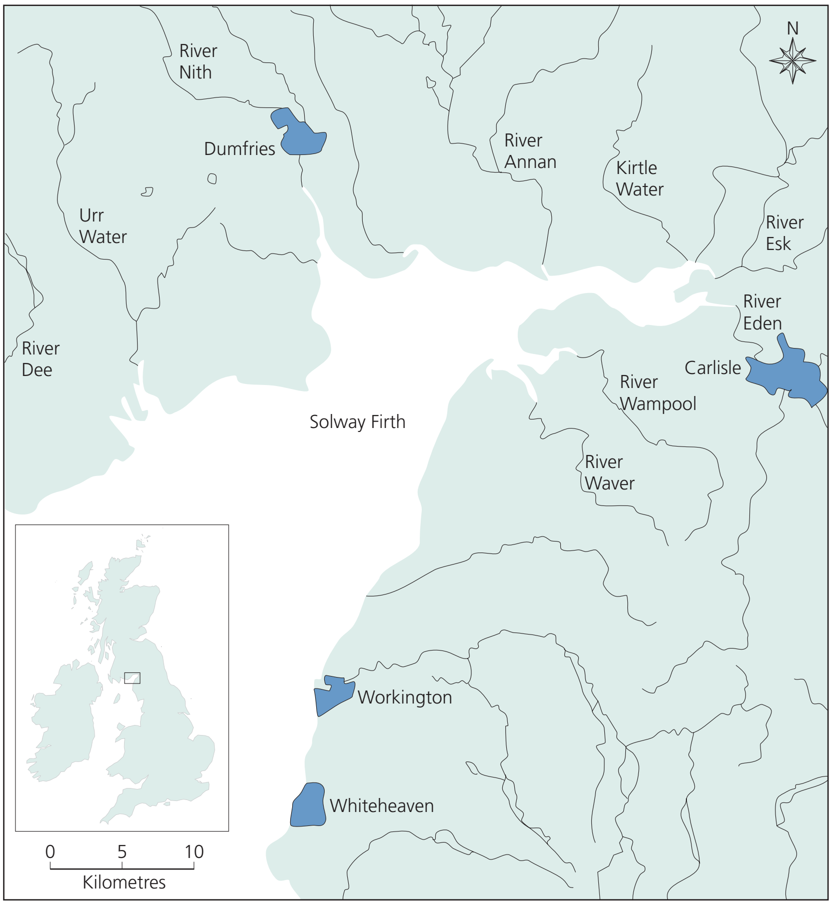


Figure 6. Location map and major river inputs

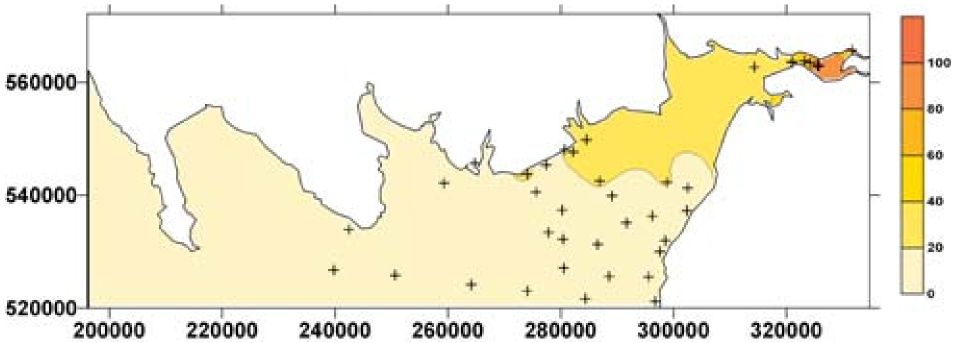


Figure 7. Winter TON concentrations (µmol l^{-1}), February 2001

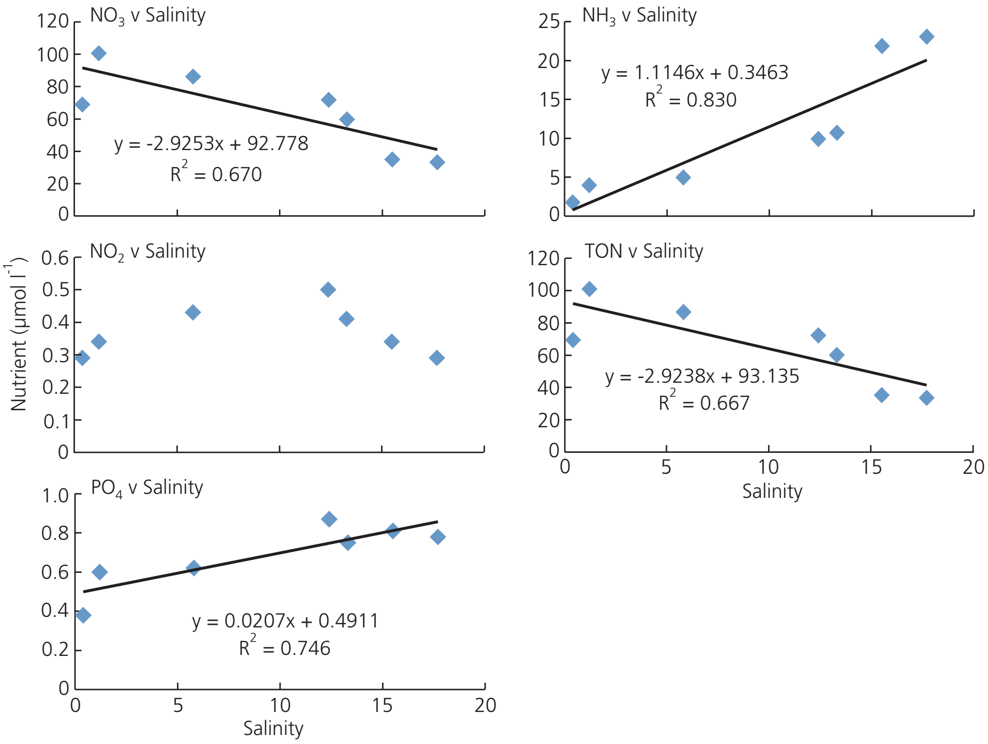


Figure 8. Nutrient-salinity plots for the Solway Estuary, February 2001

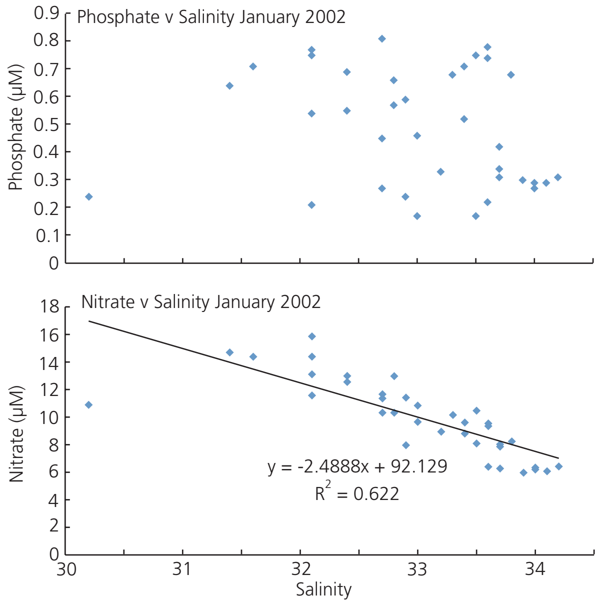


Figure 9. Nutrient-salinity plots for the Solway Firth

distribution suggested higher concentrations from the River Annan. In January 2003, salinity ranged from 9.2-21.2 and nitrate concentrations from 27-39 µM. Nutrient-salinity plots are shown in Figure 9 for the January 2002 survey. A significant correlation with salinity was observed for nitrate, but not for phosphate. Nitrate concentrations were in the range 6-16 µM and phosphate from 0.17-0.81 µM.

On the basis of these limited data, it seemed that elevated winter nutrient concentrations were associated with low salinity and high river inputs.

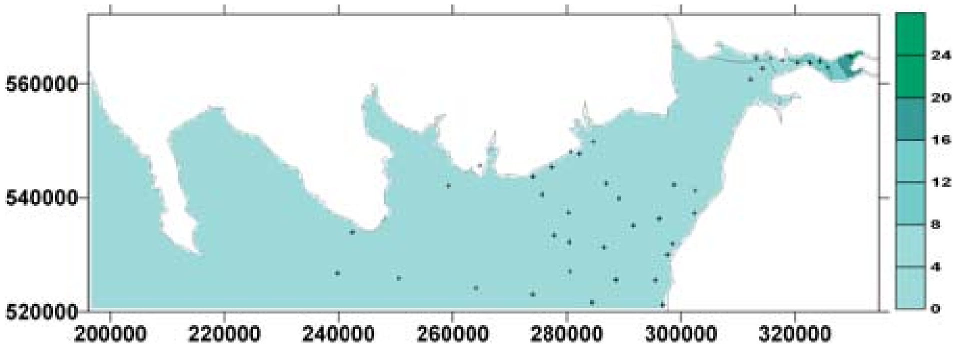


Figure 10. Summer chlorophyll α concentrations (μg l^{-1}), depth 1 metre

Impact

Chlorophyll α

Estuarine surveys in July 2001, August 2001, and July 2002 measured chlorophyll α concentrations. They ranged from 2.0-21 mg m^{-3} in July 2001 and 2.58-16.0 mg m^{-3} in August 2001. In the firth, concentrations typically were below 2.5 mg m^{-3} in July 2001 and below 4.0 mg m^{-3} in July 2002. Figure 10 shows the distribution of chlorophyll α.

On the basis of these limited data it was concluded that the Solway Firth waters were not eutrophic.

Intertidal macroalgal densities

To judge if the waters were eutrophic due to accelerated growth of algae and higher forms of plant life caused by the enrichment, intertidal macroalgal densities were looked at. At four Scottish shore sites between Gretna and Annan in August 2001, macroalgal populations were limited by the availability of stony sediment on which to settle and grow. Where stony sediment occurred, algal populations were natural.

Phytoplankton

Marine phytoplankton were collected at two stations in July 2001. The estimated total cell density at the innermost station was 840,000 cells l^{-1}, below 'bloom' levels - usually taken as exceeding 1 million cells l^{-1}. The dominant taxa were the diatoms *Asterionella* sp. and *Chaetoceros* sp., with smaller numbers of other diatom species, including *Thalassiosira* sp., *Rhizosolenia* sp., *Cerataulina* sp. and *Pseudonitzschia* sp. The only dinoflaggellate species were *Noctiluca scintillans* at an estimated density of 450 cells l^{-1}. At the outermost station there were fewer plankton, with total density of only 6,000 cells l^{-1}. The dominant taxa were the diatoms *Rhizosolenia* sp. and *Cerataulina* sp., with much smaller numbers of dinoflagellates *Protoperidium* sp. and *Prorocentrum* sp. Neither the abundance nor the species composition of these samples was a cause for concern.

Oxygen deficiency

In February (Winter), May (Spring) and July (Summer) 2001, surface and deeper waters were well oxygenated. Dissolved oxygen ranged from 8.6 to 10.6 mg l^{-1} in February, 8.7 to 11.6 mg l^{-1} in May, and 7.3 and 8.8 mg l^{-1} in July. This pattern was repeated in 2002 and 2003, with no evidence of oxygen depletion due to decaying algae.

Response

Trends cannot be examined with these limited data sets. Agricultural and some small sewage treatment nutrient inputs are not yet quantified. Nevertheless, a SEPA strategy to reduce nutrient inputs now deals with diffuse pollution - including agricultural run-off and seepage of nutrients into groundwater - by partnerships to advise farmers of best practice in activities such as slurry spreading and fertiliser use.

North West Water and Scottish Water manage public sewage discharges to the Solway Firth catchment and are upgrading their works to improve water quality in line with the Bathing Waters Directive and UWWTD. North West Water is undertaking new works at Silloth, Allonby, Workington, Seascale and St. Bees. Work is under way at Workington, Whitehaven and Egremont, and is planned for Annan to meet the needs of the UWTT Directive by 2005.

References

CSTT, 1993. Comprehensive Studies for the Purposes of Article 6 of Dir 91/271/EEC, the Urban Waste Water Treatment Directive. Report of the Comprehensive Studies Task Team of the Group Co-ordinating Sea Disposal Monitoring, for the Marine Pollution Monitoring Management Group. Edinburgh.

CSTT, 1997. Comprehensive Studies for the Purposes of Article 6 of Dir 91/271/EEC, the Urban Waste Water Treatment Directive. Second Edition of the Report of the Comprehensive Studies Task Team of the Group Co-ordinating Sea Disposal Monitoring, for the Marine Pollution Monitoring Management Group. Edinburgh.

EEC, 1991. Council of the European Communities Directive concerning urban waste water treatment (91/271/EEC). Official Journal L135/40.

Jak, R.G., 2001. Further Elaboration of Ecological Quality Objectives for the North Sea with regard to Nutrients and Eutrophication Effects.

5. METALS IN SEAWEED

Wherever mussels were not available for monitoring metals bioaccumulation, the NMMP specified brown seaweed (*Fucus vesiculosus*) as the organism of second choice. Quality assurance was not available in the same way for metals in seaweed as for the mussels, and the seaweed data are therefore included here as a Case Study.

Brown algae are commonly used in marine monitoring programmes in the UK (see Humber estuary Case Study). However, there are currently no BRCs, EACs or EQSs available for metals in seaweed.

The geographical spread of results was restricted to the northeast and south coast of England, parts of the Welsh coast, and the Severn Estuary (see Figure 11).

Median mercury concentrations in the range 4.3 - 39.6 µg kg^{-1} wet weight were reported, with the lowest value in the Tamar and the highest at Seaham on the Durham coast (Figure 11). Low levels in the Tamar may reflect low ambient mercury concentrations, or suppression of mercury uptake by competition from very high levels of other metals, associated with local mineworkings.

Median cadmium concentrations in the range 80-728 µg kg^{-1} wet weight were reported, with the lowest value in the Tamar and the highest at Bedwin on the Severn. These compared to 100-420 µg kg^{-1} wet weight for West Greenland - regarded as natural background (Riget *et al.*, 1997) - and 600 µg kg^{-1} wet weight for the Humber estuary. Concentrations were in the ranges expected for uncontaminated areas and for areas with known industrial discharges.

Lead concentrations were in the range 280-29,400 µg kg^{-1} wet weight, with the lowest value at Bedwin on the Severn. The highest concentrations were on the Wear, Tyne and Tees, possibly reflecting high sediment concentrations or contaminated suspended material in these industrialised estuaries.

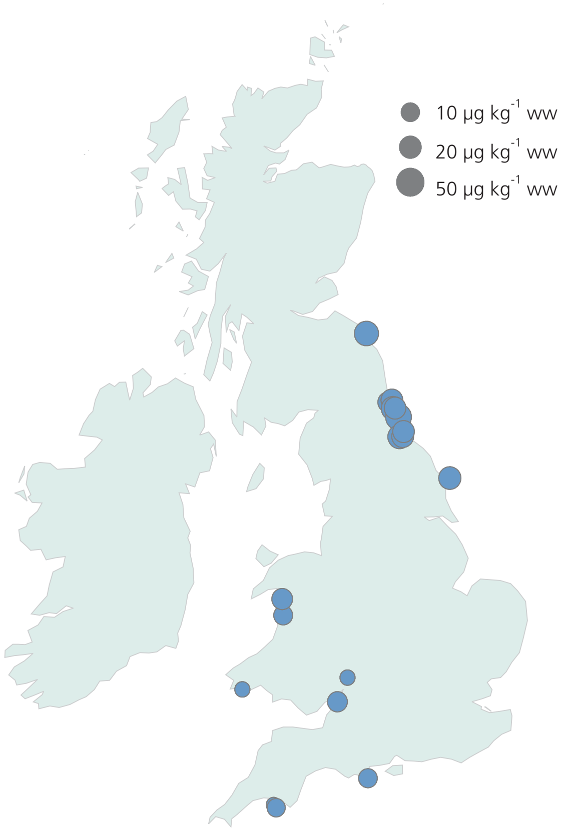


Figure 11. Median mercury concentrations in Fucoid algae

The minimum and maximum median concentrations of other metals are listed in Table 4.

Table 4. Seaweed metals concentrations

Metal	Minimum Concentration (µg kg^{-1} wet weight)	Site	Maximum Concentration (µg kg^{-1} wet weight)	Site
Arsenic	3,092	Severn Estuary	15,600	Seaham, Durham Coast
Chromium	129	Tamar	2,758	Tees
Copper	1,080	Poole Harbour	12,800	Tyne
Nickel	434	Dovey Estuary (mid-Wales)	11,200	Seaham, Durham Coast
Selenium	63	Mawddach Estuary (mid-Wales)	2,091	Tees
Silver	71	Tees	379	Seaham, Durham Coast
Zinc	17,400	Milford Haven	164,000	Wear

Summary

The limited dataset for metals in seaweed showed highest concentrations to be in the industrialised Tyne, Tees and Wear estuaries, and at Seaham off the Durham coast.

Reference

Riget, F., Kohanssen, P. and Asmund, G., 1997. Baseline Levels and natural Variability of Elements in Three Seaweed Species from West Greenland. Marine Pollution Bulletin, 34(3): 171-176.

6. MONITORING OF HEAVY METALS IN THE HUMBER ESTUARY USING *FUCUS VESICULOSUS*

Driving force

The Humber is an important estuary with a catchment area about 1/5 of England. The main inputs of pollutant are from rivers and direct discharges of sewage and industrial effluent. The locations of significant effluent discharges of metals to the Humber between 1980-90 are shown in Figure 12. The load of discharged metals is much greater from industrial effluents than from sewage effluents.

Pressure

It is usually the case that a single discharge is responsible for a high percentage of any particular metal. For example, a non-ferrous smelter on the north bank contributed practically all the arsenic and most of the cadmium load, with significant amounts of other metals. This process ceased in 1991 and the site was restored by 1994, resulting in a marked decrease in arsenic and cadmium concentrations in the estuary. A single chemical discharge on the south bank was responsible for nearly 80% of zinc discharged to the estuary but the metal recovery process was improved during the 1990s.

As direct industrial inputs fell over the years, diffuse inputs from the rivers became more prominent. This is particularly noticeable in load data for zinc and cadmium (Figure 13) in wetter years with high river flows.

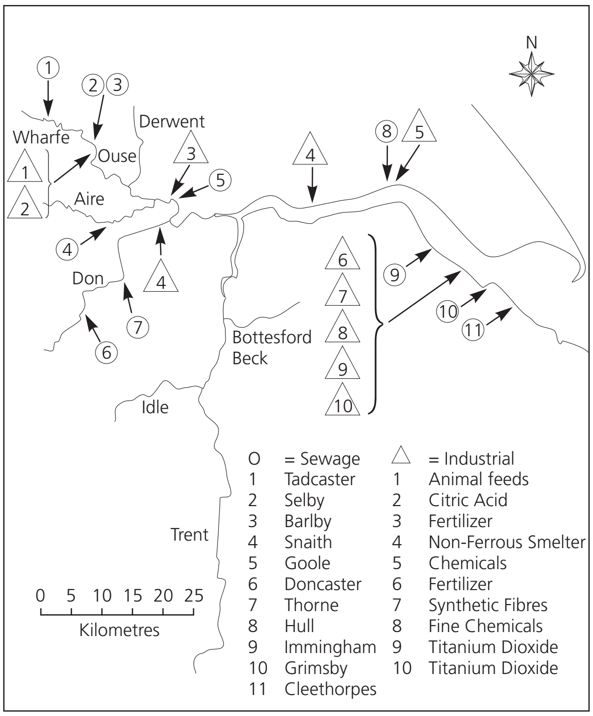


Figure 12. The locations of significant effluent discharges of metals to the Humber between 1980-90

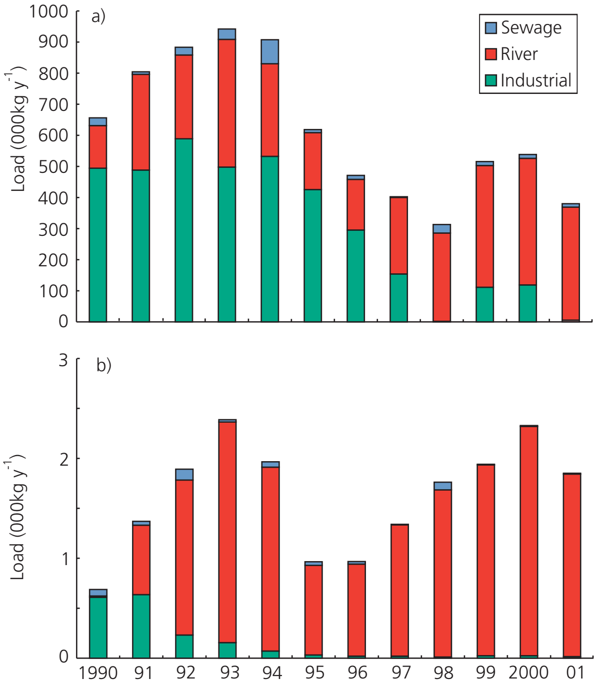


Figure 13. Loads of (a) zinc and (b) cadmium to the Humber

State

Early in 1961 it was decided to start a regular chemical survey of the Humber estuary. Detailed biological work started in 1971 and later included bioaccumulation. Monitoring of heavy metals using *Fucus vesiculosus* has been very successful. The first survey was in September 1981, and results appeared during the eighties (Barnett and Ashcroft, 1985; Barnett *et al.*, 1989). This case study considers the long term trends for iron, zinc and cadmium over the two decades of Fucoid bioaccumulation monitoring, one of the longest unbroken bioaccumulation time series.

Seaweed samples were collected at ten sites along the south bank, from approximately 3 km upstream of the Humber Bridge (the low salinity limit of *Fucus* distribution), to Cleethorpes at the seaward end (Figure 14).

A synoptic view is provided by single values for each survey of the average tissue concentration from all (ten) sites. Changes recognisable at this level represent major changes in metal inputs to the system.

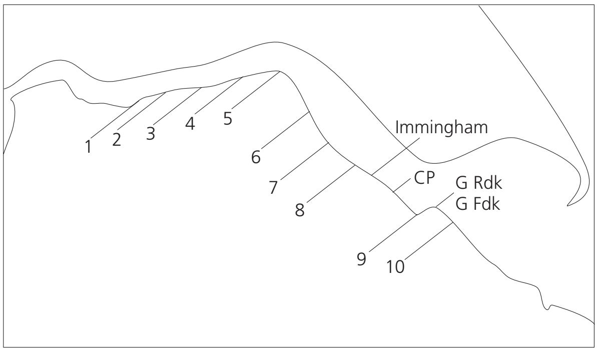


Figure 14. Fucus sampling sites

Iron

Iron does not normally accumulate in *Fucus* because it locks in complexes with the various ions in sea-water. However, the Humber received two large discharges of acid-iron waste whose nature causes the iron to remain in solution for extended periods and thus to be available for uptake.

Whole estuary average concentrations were from 1000-2000 mg kg^{-1} throughout the eighties but in 1988 new discharge arrangements started and the values declined generally to below 500 mg kg^{-1} throughout the 1990s (Figure 15).

There was an even more marked difference between the two periods for the two sites closest to the discharges (Figure 16). Before 1988 concentrations were seldom lower than 2000 mg kg^{-1} and regularly exceeded 4000 mg kg^{-1}. From 1989 onwards, values rarely exceeded 1000 mg kg^{-1} and throughout the mid- and late-nineties were typically around 500 mg kg^{-1} or less. This very clear demonstration of reduced iron in *Fucus* mirrors the new discharge arrangements.

Zinc

The lower part of the Humber was the recipient of one of the largest discharges of zinc in the UK. During the mid 1990s the discharger was required to compose a reduction programme. The whole estuary average plot (Figure 17) shows zinc to have been lower in the late nineties than in previous years, but the pattern is not entirely distinct. On

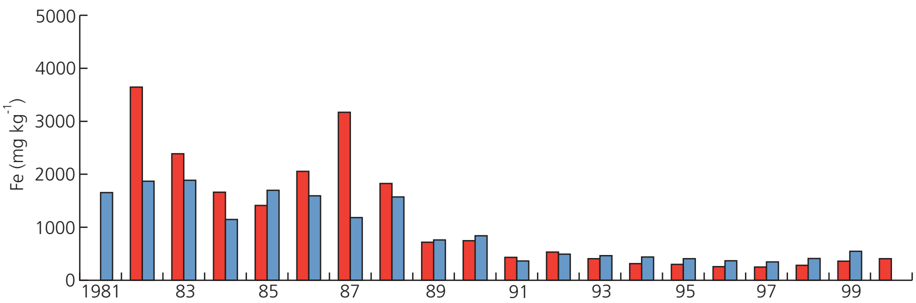


Figure 15. September iron levels in Fucus (mg kg^{-1}), dry weight

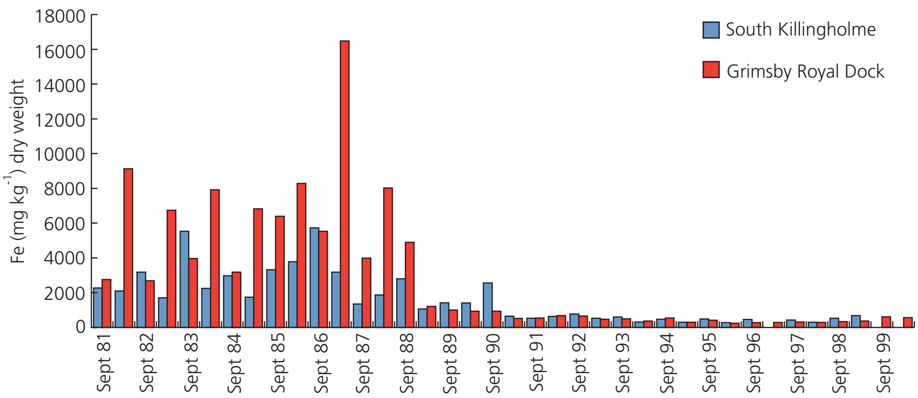


Figure 16. Whole estuary average iron concentrations in Fucus (mg kg^{-1}) dry weight (1981-1999)

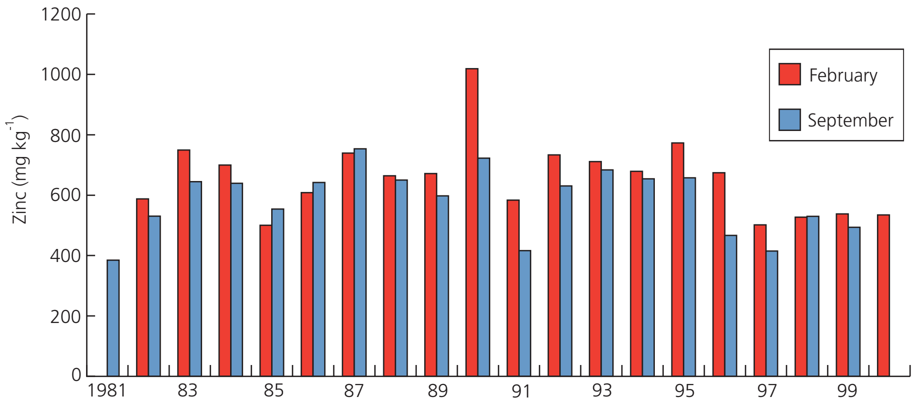


Figure 17. Whole estuary: average zinc concentrations in Fucus, dry weight (1981-2000)

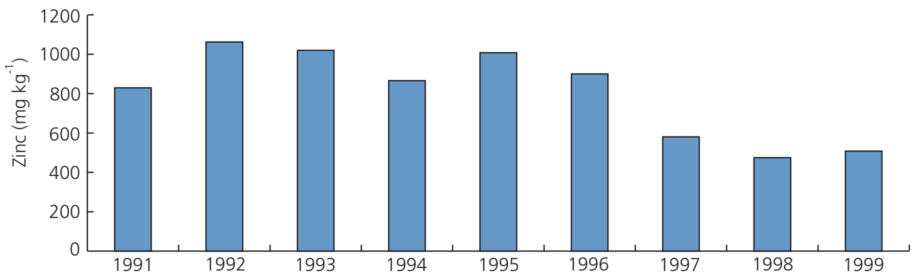


Figure 18. Zinc levels in Fucus. Average at four lower estuary sites, February

the other hand, after 1997 averaged values of zinc did not exceed 600 mg kg^{-1} at four lower estuary sites that in 1991-96 had shown high levels of 800 to 1000 mg kg^{-1} (Figure 18). The zinc reduction programme has clearly been successful, although confirmatory monitoring will be needed.

Cadmium

Unlike iron and zinc, direct inputs of cadmium have not been previously identified. Early published work recognised that elevated concentrations of cadmium in the Upper Estuary might reflect inputs via the tidal rivers Trent and Ouse, and discharge from Capper Pass smelting works in the upper reaches of the North bank of the Estuary. Whole estuary average values suggest that tissue concentrations of cadmium have been lower during the 1990s than in the preceding decade (Figure 19).

Further insight comes from looking at the distribution pattern for the ten routine sites in the late eighties, and contrasting it with the most recent pattern (1999) (Figure 20). The 1987 pattern represents that seen throughout the decade, with higher upstream concentrations and a decline seawards, in line with classical conservative mixing behaviour of riverine contaminants. By the end of the 1990s, this was no longer particularly evident: the concentrations were lower throughout the Estuary but greatest reductions were seen the upper estuary. These changes confirm that the principal sources of cadmium were in the upper estuary and tidal rivers, and that there have been considerable reductions in inputs.

Impacts

The iron acid waste from the titanium dioxide plants tended to stain the shore red and there was also a noticeable lack of algae. The discharges were moved to deeper water and the waste recovery process was improved. After several years the shoreline returned to a more normal colour and some algae returned.

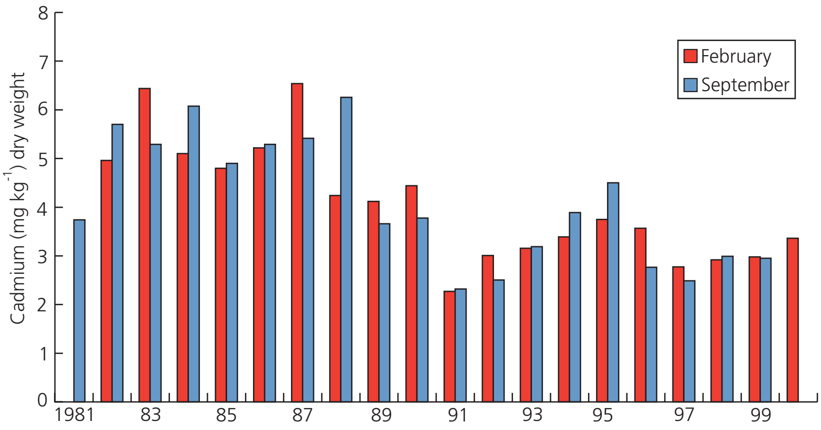


Figure 19. Whole estuary: average cadmium concentrations dry weight (1981-2000)

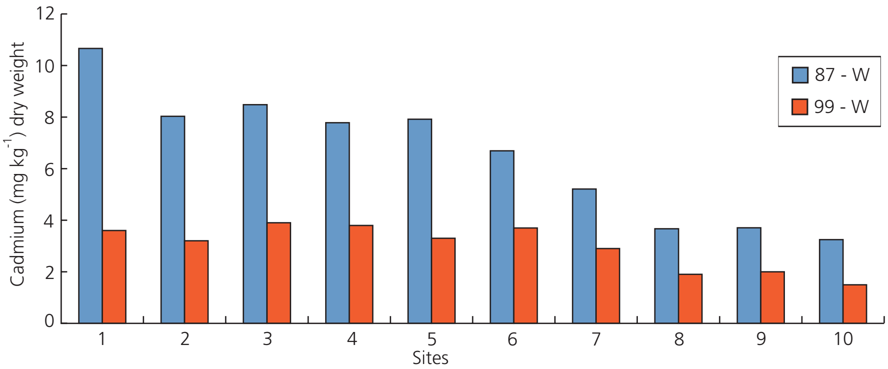


Figure 20. Cadmium levels in Fucus, Humber South Bank February 1987 and 1999

Summary

Long-term monitoring of heavy metals in the Humber with fucoid algae has been an effective method of demonstrating environmental improvements resulting from reductions in discharge of certain heavy metals. The effectiveness of the technique in revealing trends justifies its continued use in long-term monitoring programmes.

References

Barnett, B.E. and Ashcroft C.R., 1985. Heavy metals in Fucus Vesiculosus in the Humber Estuary. Environmental Pollution, B 9, 193 – 213.

Barnett, B.E., Forbes, S. and Ashcroft, C.R., 1989. Heavy Metals on the South Bank of the Humber Estuary. Marine Pollution Bulletin, 20, 17-21.

7. LONG TERM TRENDS IN MERCURY, CADMIUM AND LEAD IN THE FORTH ESTUARY

Driving force

The Forth estuary and its catchment in east Scotland have been a focus for industrial and commercial activity for many decades. The estuary receives discharges from the manufacture of yeast, paper, chemical and petrochemical industries, with waste water from about one quarter of the Scottish population (Figure 21). Many discharges remained untreated until environmental legislation was introduced in the mid 1980s. At the second Intergovernmental Conference in 1987 the UK Government agreed to reduce inputs of certain substances, including cadmium, mercury and lead, to the North Sea by half between 1985 and 1995. This led to the reduction and minimisation of point source discharges of these substances.

Pressure

Annual monitoring of discharges of cadmium, mercury and lead to tidal waters from riverine,

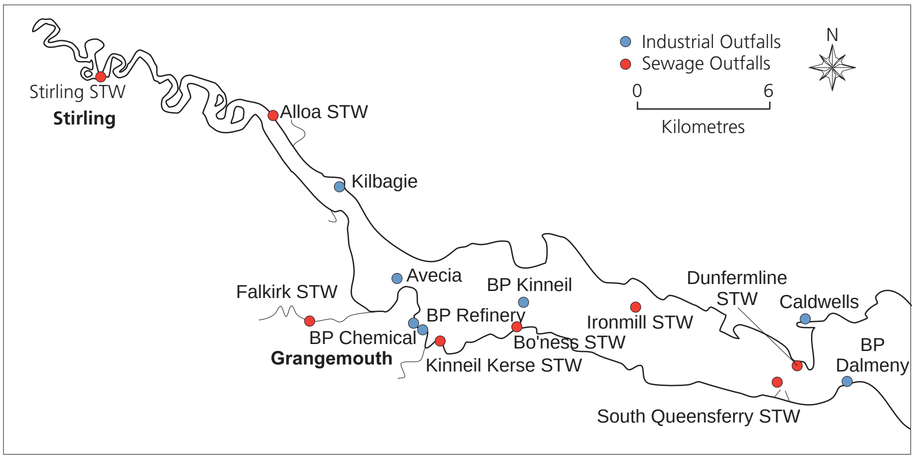


Figure 21. Major discharges to the Forth estuary

Table 5. Annual inputs of trace metals to the Forth estuary

Year	Mercury (kg y^{-1})	Cadmium (kg y^{-1})	Lead (tonne y^{-1})
1985	1,730	3,700	
1990	118	580	22
1991	305	860	11
1992	68	560	15
1993	67	630	17
1994	48	390	14
1995	43	380	15
1996	44	396	15
1997	40	289	7
1998	49	240	20
1999	53	190	9
2000	34	198	13

sewage and industrial discharges commenced in 1990. Annual inputs are presented in Table 5 with 1985 baseline data for comparison.

State

The Forth is a turbid estuary (Webb and Metcalfe, 1987) and discharged particle-reactive metals are adsorbed onto its particulate matter (Balls *et al.*, 1997). Mercury and lead are particle-reactive and are retained on the sediments whereas cadmium is more likely to remain in solution.

Contaminants in sediments

More than 90% of the discharged mercury may have been retained on the sediments of the estuary (Elliott and Griffiths, 1986). Routine monitoring of the mercury and lead content of the sublittoral sediments showed no reduction consistent with the reduction in the discharge. Cadmium, on the contrary, is less particle reactive and stays in solution. The cadmium content of the sediments is relatively low and there are no trends in the data (Figure 22). Mercury concentrations are above the upper EAC (0.5 mg kg^{-1}), lead is around the upper EAC (50 mg kg^{-1}) and cadmium is below the upper EAC (1 mg kg^{-1}). The data cannot be compared to BRCs because aluminium was not determined but Balls *et al*. (1997) reported metal-aluminium ratios for the Forth sediments above background.

Contaminants in biota

Long-term data are available for mercury in fish muscle and mussels from the Forth estuary (Figure 23). They reveal a decrease in mercury content, consistent with the reductions in the industrial discharge.

The concentration of mercury was above the EAC (1 mg kg^{-1} dry weight) in mussels until the 1990s whereas the concentration in fish fell below the EAC (0.3 mg kg^{-1} ww) in the early 1980s.

The cadmium content of mussels in the Forth estuary is below the EAC (5 mg kg^{-1} dry weight) and appears to be declining. (Figure 24). In recent years it has been close to the BRC (1.4 mg kg^{-1} dry weight).

The lead content of the mussels is substantially above the upper EAC (5 mg kg^{-1} dw) and recent results indicate less variability in the data (Figure 25).

Table 6. Trends in sources of trace metals

Year	Rivers		Sewage		Industry	
	1990	1999	1990	1999	1990	1999
Cadmium (kg y^{-1})	400	150	125	22	55	8
Mercury (kg y^{-1})	26	16	19	16	73	4.5
Lead (t y^{-1})	11	9	7	4.4	0.7	0.3

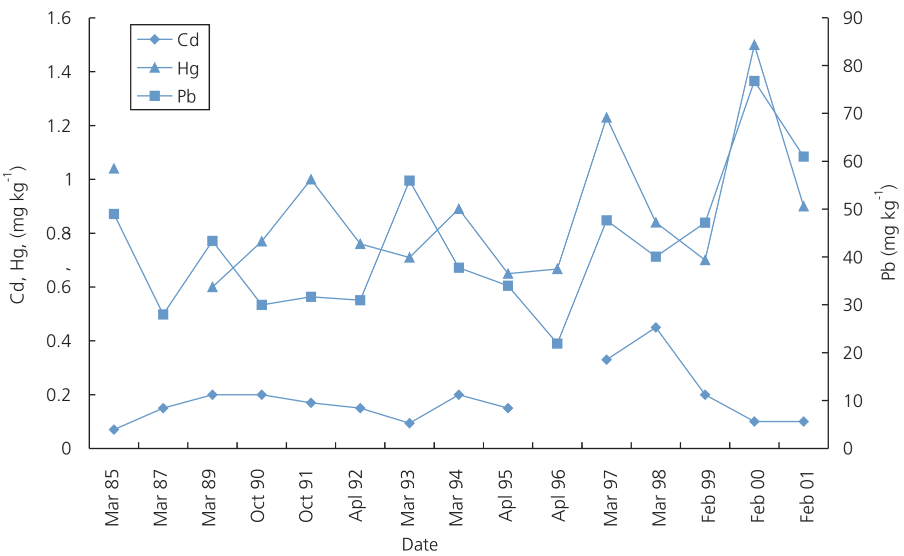


Figure 22. Mercury, cadmium and lead (mg kg^{-1}) at site RA in the Forth estuary

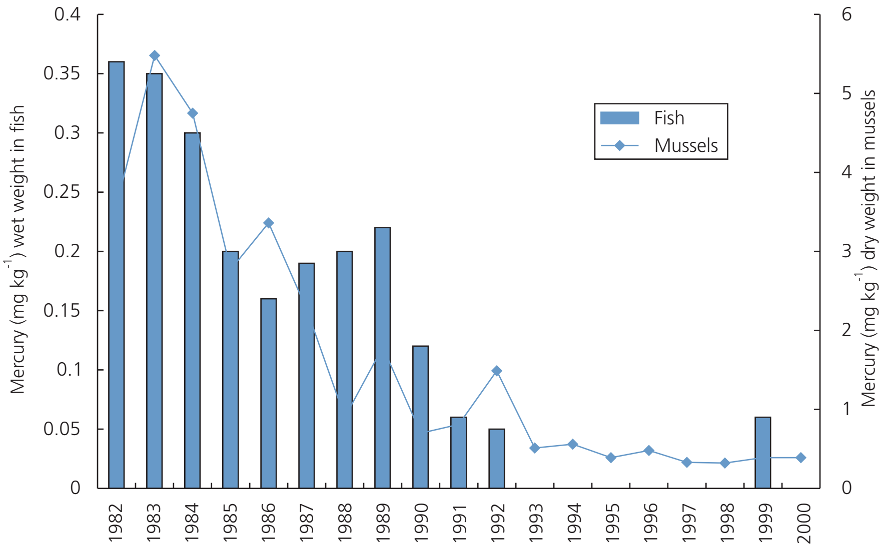


Figure 23. Mercury in fish and mussels in the Forth estuary

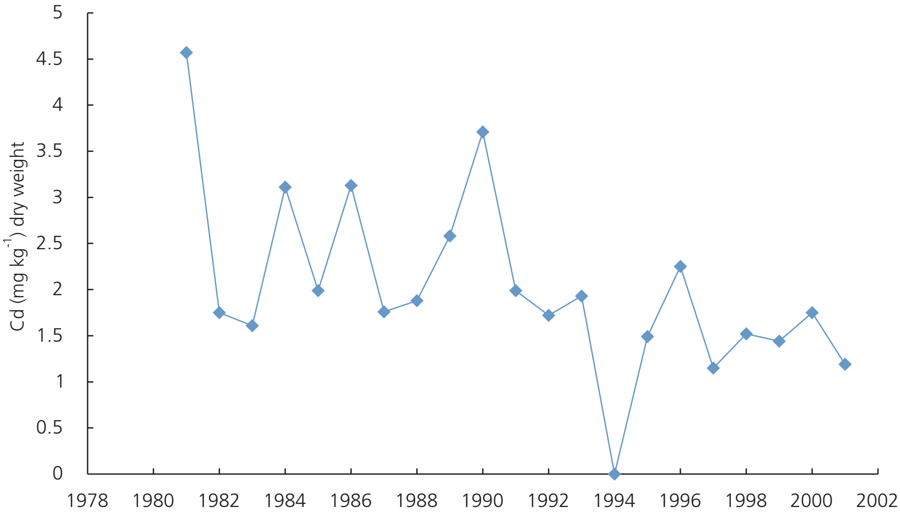


Figure 24. Cadmium (mg kg^{-1} dry weight) in mussels in the Forth estuary

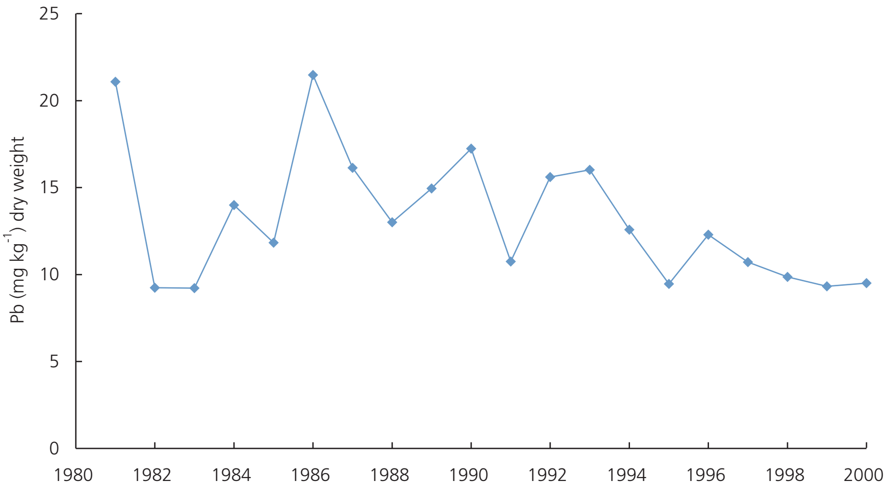


Figure 25. Lead (mg kg^{-1} dry weight) in mussels in the Forth estuary

Figure 26. Dissolved trace metals in the Forth estuary

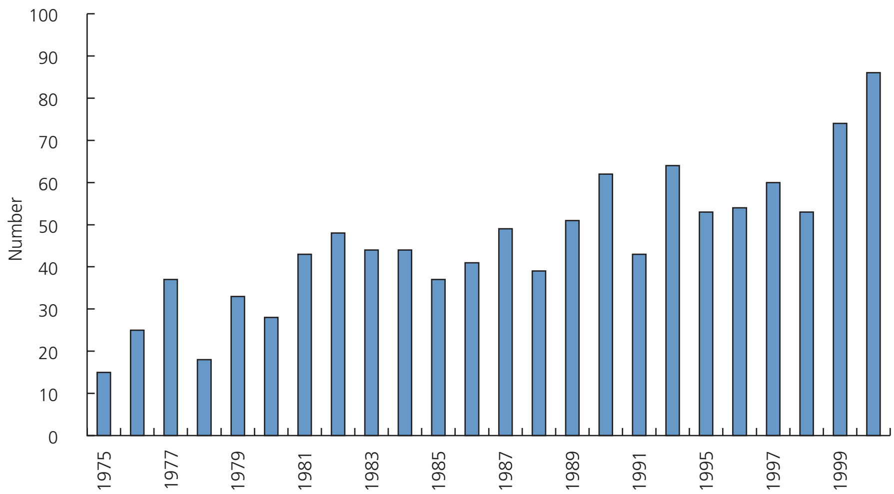


Figure 27. Total Number of taxa in main channel sediments

Contaminants in water

Concentrations of dissolved mercury, cadmium and lead in the water were always substantially below Environmental Quality Standards and close to the limit of detection of the analysis. There are no trends in the data (Figure 26)

Impact

The total number of taxa in the main channel of the estuary at Grangemouth has increased since 1975 (Figure 27). The samples were collected in the vicinity of an industrial discharge, the major point source of mercury.

Response

The discharge of all metals has fallen as a result of effluent treatment and minimisation of point source discharges (Table 6). Water companies are actively trying to reduce the discharge of mercury from dentistry.

Summary

The concentrations of dissolved metals in the Forth estuary are low as the Forth is a turbid estuary and particle reactive metals are absorbed onto the particulate phase. The sediments of the Forth are enriched with cadmium, lead and mercury relative to background levels. The enrichment factor is greatest for mercury and higher for lead than cadmium.

A substantial amount of the discharged mercury has been retained on estuarine sediments. Loss of mercury from the sediments is expected to take decades because its flux out of the estuary on particulate matter is quite slow (Davies *et al.*, 1986). Fish and shellfish ingest contaminants from sediments as well as from the overlying water column. Following the point source input reductions, mercury contamination in fish and shellfish reduced gradually, whereas the cadmium content of mussels reduced rapidly. The lead content of mussels was highly variable and showed no distinct trend.

In the macrofaunal community in the sediments of the Forth there was an increase in the number of recorded taxa that may be related to the decreased discharge of contaminants to this part of the estuary.

References

Balls, P.W., Owens, R.E and Muller, F.L.L., 1997. Dissolved trace metals in the Clyde, Forth and Tay estuaries. Coastal Zone Topics, 3, 46-56

Balls, P.W., Hull, S., Miller, B.S., Pirie, J.M. and Proctor, W., 1997. Trace Metal in Scottish Estuarine and Coastal Sediments. Marine Pollution Bulletin, 34, 42-50

Davies, I.M., Griffiths, A.H., Leatherland, T.M. and Metcalfe, A.P., 1986. Particulate mercury fluxes in the Forth estuary, Scotland. Rapports et Proces-Verbaux des Reunions du conseil International pour l'Exploration de la Mer, 186, 301-305.

Elliott, M. and Griffiths, A.H., 1986. Mercury contamination in components of an estuarine ecosystem. Water Science and Technology, 18, 161-170.

Webb, A.J. and Metcalfe, A.P., 1987. Physical aspects, water movements and modelling studies of the Forth estuary, Scotland. Proceedings of the Royal Society of Edinburgh, 93B, 259-27

8. TRENDS IN THE BIOLOGICAL EFFECT OF TBT IN SULLOM VOE

Driving force

Sullom Voe is a large fjordic inlet on the mainland of Shetland. The mouth of the Voe is about 5 km wide, and the Voe extends approximately 13 km southwards. A large oil terminal on the promontory of Calback Ness (Figure 28) was opened in November 1978. The tonnage and number of crude and gas tankers visiting the terminal peaked in 1984.

Pressure

TBT contamination arises from tankers and, up until 1986, from TBT antifoulants used on towing vessels, navigational buoys and harbour craft. There was a general movement in the shipping industry away from high leach rate free association TBT paints towards copolymer paints around this time. The tonnage and numbers of tankers fell until 1990 then levelled out at about half their peaks.

State

The Voe has been surveyed almost biennially since 1985, providing a valuable time series more than 15 years long. In each survey, about 40 adult dogwhelks were collected from 20 sites in Sullom Voe and Yell Sound (Figure 28) and analysed by

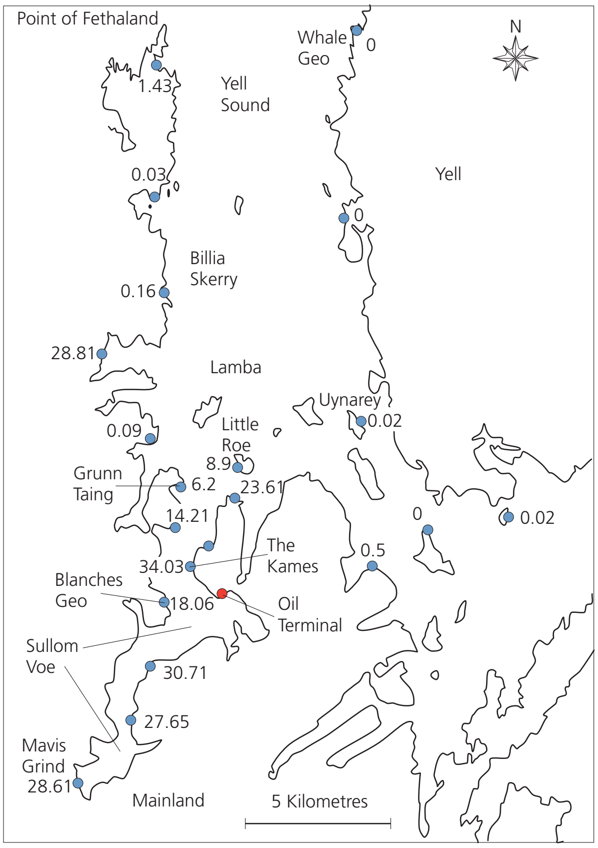


Figure 29. RPSI values in adult dog-whelk populations from the sampling sites in 2001

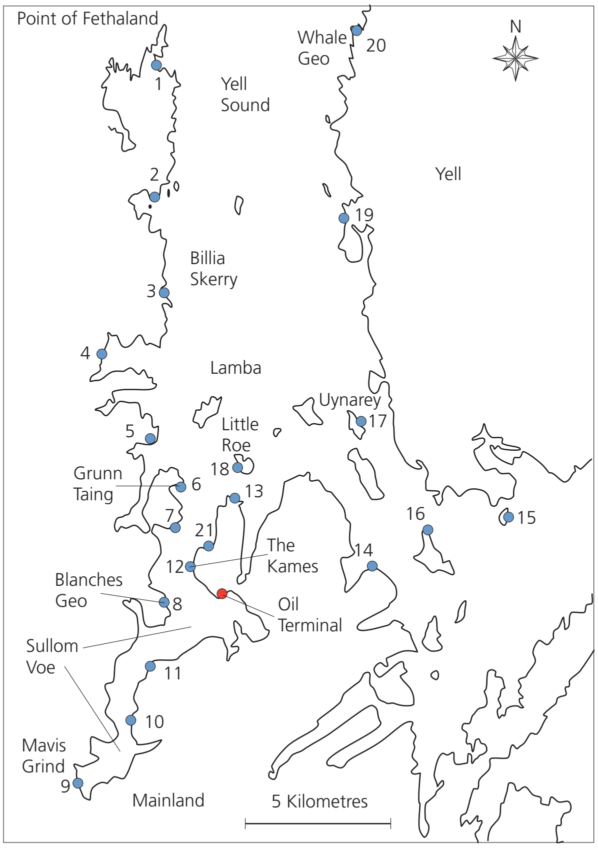


Figure 28. Map of Sullom Voe and Yell Sound showing the sampling sites

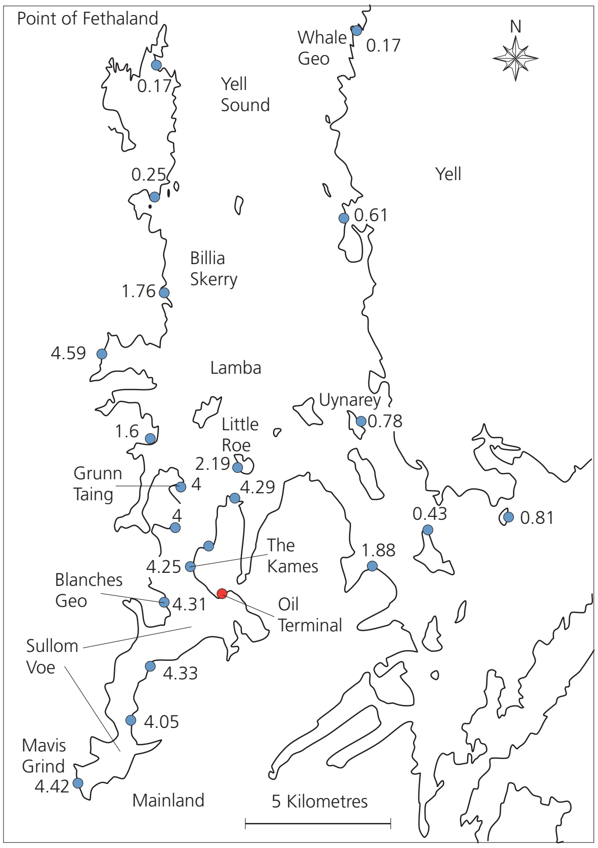


Figure 30. VDSI values in adult dog-whelk populations from the sampling sites in 2001

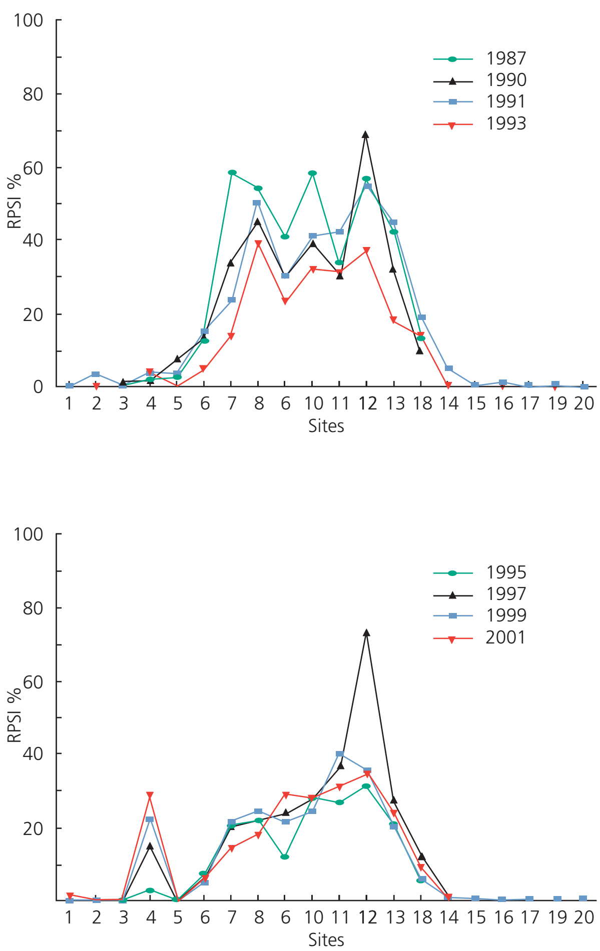


Figure 31. RPSI values for populations in the surveys from 1987-2001

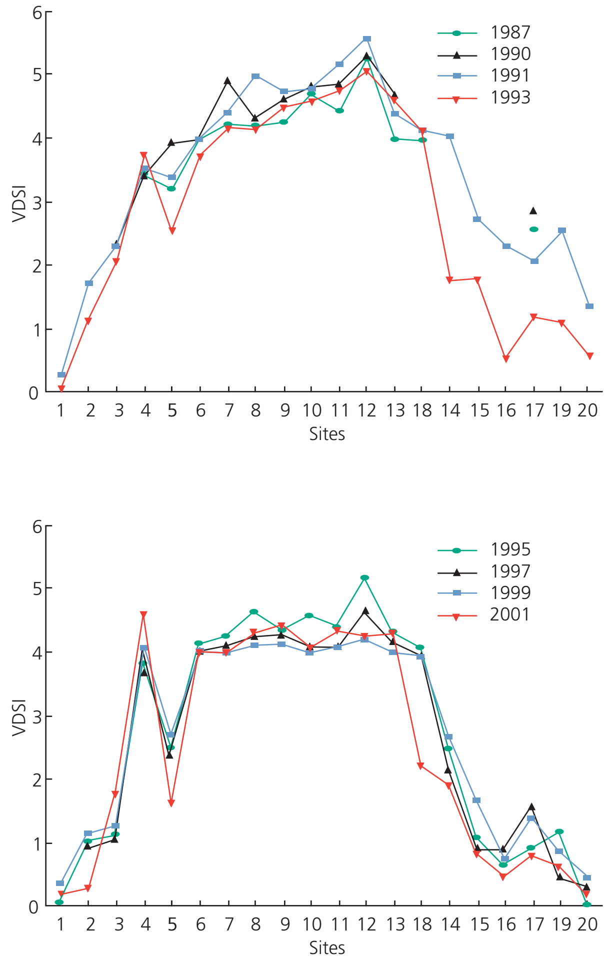


Figure 32. VDSI values for populations in the surveys from 1987-2001

standard procedures. Typical data are shown in Figures 29 and 30: high values for RPSI and VDSI in the Voe and around the oil terminal jetties are clearly differentiated from low values in the open waters of Yell Sound.

The highest RPSI value was found near the oil terminal. The RPSI values from populations in Yell Sound were much lower (0.00%-1.43%) than those in Sullom Voe (14.21%-34.03%). The low RPSI values (<1%) in Yell Sound indicated that the sites were not significantly impacted by TBT.

The VDSI values followed a similar distribution to RPSI. Populations within Sullom Voe showed values of 4.00-4.42. Most of the populations within the Yell Sound had low VDSI values (0.17-1.88).

The RPSI and VDSI values for the populations in Sullom Voe have generally decreased with time (Figures 31 and 32) and the largest decrease preceded the 1993 survey. Since then, the degree of imposex in the toothed adults from Sullom Voe declined more slowly.

Impact

VDSI values above 4.0 indicate that the reproductive potential of populations may have been impaired. In some early surveys, large proportions of the females from the immediate vicinity of the loading jetties were sterile. Exposure to TBT from tankers would have been greatest at these sites. There are no longer populations at the jetties, but that at site 12 appears to be recovering.

Response

The gradual decrease in imposex at many sites during the study is consistent with changes in shipping activity and the restrictions on use of TBT. Forthcoming restrictions on the use of TBT paints in 2003-2008 are expected to result in further improvements.

9. POLYCYCLIC AROMATIC HYDROCARBON (PAH) CONCENTRATION AND COMPOSITION IN FARMED BLUE MUSSELS (*MYTILUS EDULIS*), OTHER BIOTA AND SEDIMENT FROM LOCH LEVEN

Driving force

According to surveys for the Food Standards Agency Scotland (FSAS) between 1999 and 2002, concentrations of polycyclic aromatic hydrocarbons (PAH) in wild and farmed mussels from Loch Leven, Argyll, Scotland, were considerably greater than elsewhere around the Scottish mainland. High concentrations of PAH have been reported in discharges from aluminium smelters employing similar processes to those at Kinlochleven (Naes *et al.*, 1995 and 1999), believed to be the source of contamination.

Pressure

The smelter discharged untreated effluent into Loch Leven until March 2000 when a filter was installed on the fume treatment plant (FTP) water discharge outlet. The smelter closed and the FTP discharge ceased in June 2000. The smelter discharged into waters used for shellfish growing (Figures 33 and 34) and fish farming. As a precaution, the FSAS and the shellfish farmers agreed to voluntary closure of the commercial shellfish growing sites in September 1999.

State/Impact

Polycyclic aromatic hydrocarbon (PAH) concentration and composition were measured in farmed blue mussels (*Mytilus edulis*) from both Loch Leven and Loch Etive (Table 7, Figure 35). The monitoring covered three periods: April 1999 to the smelter closure in June 2000; June 2000 to the re-opening of the Ballachulish farm in February 2001; and February 2001 to March 2002. The total PAH concentration in mussels from the Kinlochleven, upper loch, shellfish farm, ranged from 8210 ng g^{-1} in September 1999 to 693 ng g^{-1} wet weight tissue in May 2000, while the PAH concentration in mussels from the Ballachulish, lower loch, shellfish farm ranged from 5323 ng g^{-1} in July 1999 to 206 ng/g in May 2000. PAH concentrations from both farms decreased after the smelter closed. The total measured PAH concentration from the Ballachulish shellfish farm in February 2001 was 388.6 ng g^{-1}.

Although there are no PAH standards or guideline values in current UK legislation, the UK Food Standards Agency, advised by the Committee on Toxicity of Chemicals in Food, Consumer Products and the Environment (CoT), adopted an interim limit of 15 ng g^{-1} wet weight for any of three named

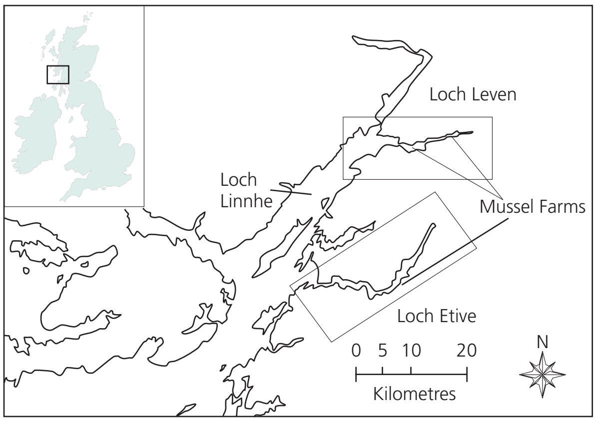


Figure 33. Shellfish growing areas

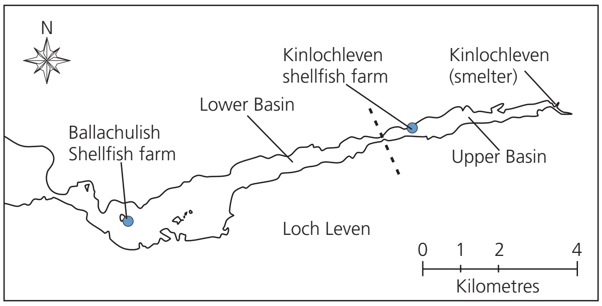


Figure 34. Sites of smelter and adjacent shellfish farms

PAHs of greatest concern: (benz[*a*]anthracene, benzo[*a*]pyrene and dibenz[*a,h*]anthracene). None of the samples from the Ballachulish shellfish farm from September 2000 to February 2001 exceeded this limit.

From September 2001 to March 2002, PAH concentrations varied seasonally at both sites (Webster *et al.*, 1997). The maximum concentration (685 ng g^{-1} wet weight PAH) occurred in January 2002. The PAH concentrations were consistently higher in mussels from the Kinlochleven farm, with the five-ring PAH dominating the PAH profiles.

Sediment

The PAH concentration and composition were determined in sediments from upper and lower Loch Leven in August 2001 (Table 8). Concentrations were consistently higher in the upper basin. Five-ring PAH dominated PAH profiles of the sediments from both basins.

Table 7. PAH concentrations in mussels from Loch Leven and the Loch Etive reference location

Site	Date	Total PAH (ng g^{-1} wet weight) (1)	% 5-ring PAH (2)	B[a]P (3) (ng g^{-1} wet weight)	B[a]PE (4) (ng g^{-1} wet weight)
Kinlochleven	04/99	7,169.3 (5)	55.1	445.9	862.4
Ballachulish	05/99	4,370.6	56.7	108.5	341.5
Loch Etive	06/99	17.0	22.4	TR	0.2
Kinlochleven	02/00	7,565.5	43.5	418.1	821.9
Ballachulish	02/00	3,046.4	48.0	118.3	295.2
Kinlochleven	10/00	300.6	61.8	14.0	45.7
Ballachulish	10/00	114.5	49.1	3.6	13.0
Loch Etive	10/00	43.3	27.0	1.4	2.2
Kinlochleven	02/01	541.1	59.1	21.1	70.0
Ballachulish (6)	02/01	388.6	41.2	8.0	23.5
Loch Etive	02/01	148.1	28.2	2.3	6.7
Kinlochleven (7)	08/01	215.0	54.4	9.9	29.9
Ballachulish	08/01	68.3	30.6	1.4	4.4
Loch Etive	08/01	56.1	7.1	TR	0.2
Kinlochleven	02/02	284.6 (5)	59.9	16.2	47.3
Loch Etive	02/02	73.6	19.9	1.3	4.6

(1) Total PAH = 2- to 6-ring parent and branched PAH including acenaphthylene, acenaphthene, fluorene and dibenz[a,h]anthracene
(2) percentage of total PAH; 5-ring PAH including dibenz[a,h]anthracene
(3) B[a]P = benzo[a]pyrene concentration
(4) B[a]PE = benzo[a]pyrene equivalent value
(5) mean of two samples
(6) relaxation of voluntary closure
(7) relaxation of voluntary closure
TR = trace (0.04 – 0.14 ng g^{-1})

Table 8. Summary of PAH concentrations in sediment in Loch Leven, August 2001

Sampling site Loch Leven	Total measured PAH (ng g^{-1} dry weight)	% 4- to 6-ring PAH	% 5-ring PAH
Lower Basin	Range:109 – 22,805 Mean:13,705 SD: 8,351	Mean: 93.7 SD: 1.3	Mean: 46.2 SD: 1.8
Upper Basin	Range:94,464 - 336,543 Mean:151,185 SD: 69,579	Mean: 95.8 SD: 0.96	Mean: 46.9 SD: 3.9

Water

Concentrations of PAH in the water at both sites in August 2001 (Table 9) were generally greater than the ranges of OSPAR background/reference concentrations (BRC) for offshore waters but were less than ranges suggested as ecotoxicological assessment criteria (EAC) (OSPAR – ASMO, 2003). The mean total concentration of PAH in the water column at the Ballachulish shellfish farm site was 9.1 ng l^{-1} and the mean total concentration at the Kinlochleven site was 25.2 ng l^{-1}.

Fish

When monitoring started in 1999, the Atlantic salmon (*Salmo salar*) fish farm within Loch Leven was fallow. The farm was restocked in March/April 2000. The total measured PAH concentrations from Loch Leven on four occasions between May and October 2000 were all less than 40 ng g^{-1} and similar to those in salmon from Kingairloch. The PAHs in salmon from both farms were dominated by low molecular weight compounds (naphthalenes and phenanthrenes/anthracenes).

Response

Discharges of PAH to Loch Leven reduced after the smelter closed. Thereafter, PAH concentration in mussels rapidly declined although the concentrations at Kinlochleven remained greater than those at the Loch Etive reference site when monitoring stopped in 2002. Contamination in the upper basin sediments is not expected to reduce in the short term.

Table 9. PAH concentrations in Loch Leven water and OSPAR ASMO BRC and EAC values

PAH compound	Loch Leven	OSPAR ASMO	
		BRC	EAC
Naphthalene	0.6 – 1.7 ng l^{-1}	0.1 – 2.7 ng l^{-1}	5000 – 50000 ng l^{-1}
Acenaphthene	TR – 0.3 ng l^{-1}	0.02 – 0.05 ng l^{-1}	ND
Fluorene	0.2 – 0.5 ng l^{-1}	0.02 – 0.81 ng l^{-1}	ND
Phenanthrene	0.8 – 1.3 ng l^{-1}	0.1 – 0.6 ng l^{-1}	500 – 5000 ng l^{-1}
Anthracene	TR	0.001 – 0.004 ng l^{-1}	1 – 100 ng l^{-1}
Fluoranthene	1.0 – 1.4 ng l^{-1}	0.04 – 0.26 ng l^{-1}	10 – 100 ng l^{-1}
Pyrene	0.8 – 1.0 ng l^{-1}	0.02 – 0.05 ng l^{-1}	50 – 500 ng l^{-1}
Benzo[*a*]anthracene	TR – 0.3 ng l^{-1}	0.001 – 0.006 ng l^{-1}	ND
Chrysene/Triphenylene	0.2 – 1.0 ng l^{-1}	0.006 – 0.057 ng l^{-1}	ND
Benzo[*b*]fluoranthene	0.2 – 1.4 ng l^{-1}	0.001 – 0.017 ng l^{-1}	ND
Benzo[*k*]fluoranthene	0.2 – 0.8 ng l^{-1}	0.001 – 0.003 ng l^{-1}	ND
Benzo[*a*]pyrene	0.6 – 0.7 ng l^{-1}	0.001 – 0.005 ng l^{-1}	10 – 100 ng l^{-1}
Indenopyrene	TR – 0.8 ng l^{-1}	0.001 – 0.017 ng l^{-1}	ND
Benzoperylene	TR – 1.1 ng l^{-1}	0.001 – 0.01 ng l^{-1}	ND
Dibenz[*a,h*]anthracene	TR – 0.3 ng l^{-1}	<0.001 ng l^{-1}	ND

TR = trace (0.04 – 0.14 ng l^{-1}); ND = no data or insufficient data

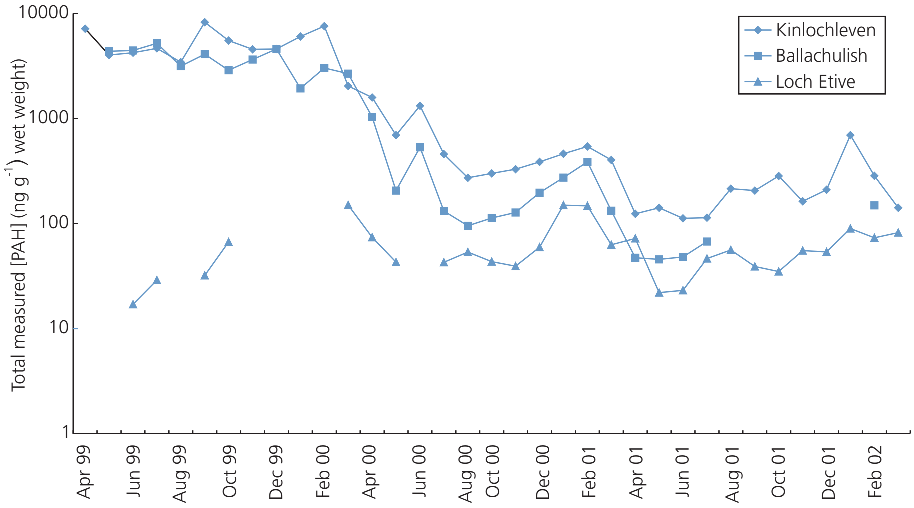


Figure 35. Variation in PAH Concentration (ng g^{-1} wet weight) in mussels (Mytilus edulis) from Loch Leven and Loch Etive, April 1999 - March 2002

In February 2001, in view of the monitoring programme results, voluntary closure was relaxed to allow commercial harvesting of mussels at the Ballachulish shellfish farm. With continued monitoring, in September 2001 the voluntary closure at the Kinlochleven shellfish farm was also relaxed.

References

Naes, K., Knutzen, J. and Berglind, L., 1995. Occurrence of PAH in marine organisms and sediments from smelter discharge in Norway. The Science of the Total Environment, 163, 93-106.

Naes, K, Hylland, K, Oug, E., Forlin, L. and Ericson, G., 1999. Accumulation and effects of aluminium-smelter generated PAHs on soft-bottom invertebrates and fish. Environmental Toxicology and Chemistry, 18, 10, 2205-2216.

OSPAR Convention for the protection of the Marine Environment of the North East Atlantic. Meeting of the Assessment and Monitoring Committee (ASMO). April 2003.

Webster, L., Angus, A., Topping, G., Dalgarno, E. J. and Moffat, C.F., 1997. Long-term Monitoring of Polycyclic Aromatic Hydrocarbons in Mussels (*Mytilus edulis*) Following the *Braer* Oil Spill. Analyst, December 1997, Vol. 122 (1491-1495).

10. MONITORING THE BIOLOGICAL EFFECTS OF POLYNUCLEAR AROMATIC HYDROCARBONS (PAH) ON FLATFISH IN THE FORTH AND CLYDE ESTUARIES

Driving force

The Forth Estuary on Scotland's East coast has been a centre for petrochemical and refining industries for many years. Refineries at Grangemouth, the coal-fired power station at Longannet and inputs from municipal discharges are all likely sources of PAH input to sediments in the Forth, and warrant monitoring of these substances and their effects in this sea area.

The Clyde Estuary and Firth of Clyde on Scotland's West coast have for decades received direct and indirect sewage and industrial discharges from activities associated with half of Scotland's population. Consequently, there are many potential sources of PAH including military, domestic, municipal and industrial effluents, oil from recreational, commercial and naval shipping, sewage and dredge spoil dump sites and a power station at Irvine Bay.

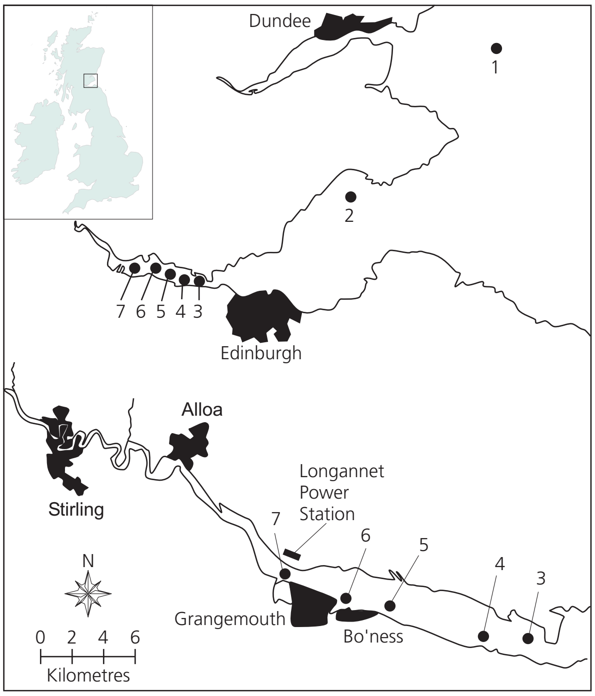


Figure 36. Sampling sites in the Forth Estuary and Firth of Forth and St. Andrews Bay (reference) for 1999 and 2001 PAH specific biological effects survey in flounder (Platichthys flesus). 1) St. Andrews Bay, 2) Kingston Hudds, 3) Port Edgar, 4) Blackness, 5) Tancred, 6) Bo'ness, 7) Longannet

Pressure

Inputs of PAH have not been quantified but are connected with the burning of fossil fuels. Inputs derive from point source discharges (e.g. processing of oil and gas) and diffuse sources (e.g. wash out of discharges to the atmosphere).

State/Impact

Forth

Flounder (*Platichthys flesus*) from stations in the Forth estuary, Firth of Forth and St. Andrews Bay were sampled for PAH specific biological effects in 1999 and 2001 and the results were compared to sediment chemistry (Figure 36).

EROD activity in livers of flounder in 1999 and 2001 was elevated from both Forth estuary and Firth

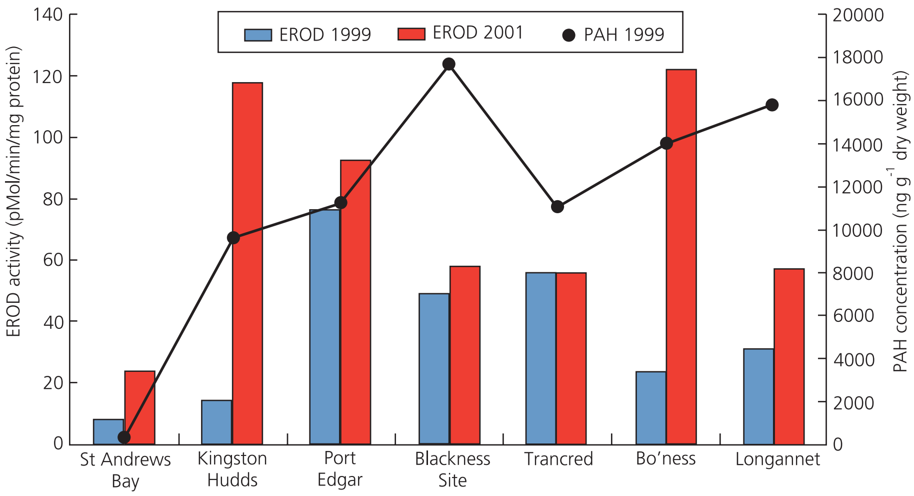


Figure 37. Hepatic EROD activity of flounder (Platichthys flesus) sampled from 7 stations in the Forth during surveys in 1999 and 2001. Total sediment PAH burden from 1999 is included for interpretation

of Forth sites compared to the control site at St. Andrews Bay (Figure 37). There were no significant differences in EROD activity between trawl stations within the estuary for either survey. Sediment PAH levels in 1999 reflected the general trends recorded for EROD activity, with the lowest levels at the control site outside of the Firth of Forth and higher but variable levels in sediments from the estuary and firth (Figure 37).

Bile metabolite analysis from fish caught in 1999 detected three metabolites (1-OH pyrene, 2-OH naphthalene and 1-OH phenanthrene). 1-OH pyrene, a metabolite commonly present at high levels in environmental samples, dominated and increased up the estuary, with fish from Longannet containing significantly greater levels of pyrene metabolites in bile samples than fish from stations further out. Other metabolites were present at much lower levels (Figure 38).

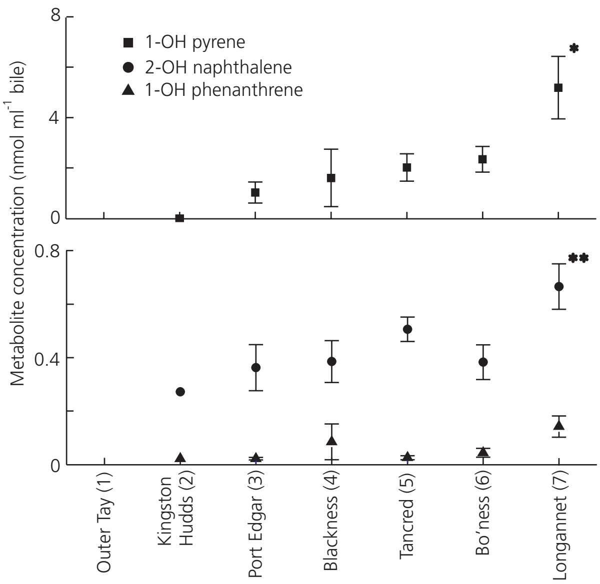


*Figure 38. Mean (+/- SEM) biliary metabolite concentrations in male flounder (Platichthys flesus) from six sites in the Forth, September, 1999. * , ** Denotes significant differences from Port Edgar*

Clyde

Plaice (*Pleuronectes platessa*) were taken from the Firth of Clyde in 1999 and 2000, as they were more readily available than flounder yet respond similarly to PAH. Fish were assessed for EROD activity and, in 2000, bile metabolites were measured and sediment samples were analysed for total PAH content so as to provide background chemistry. Fish from within the Firth of Clyde were compared to a reference station at Broad Bay on Lewis (Figure 39).

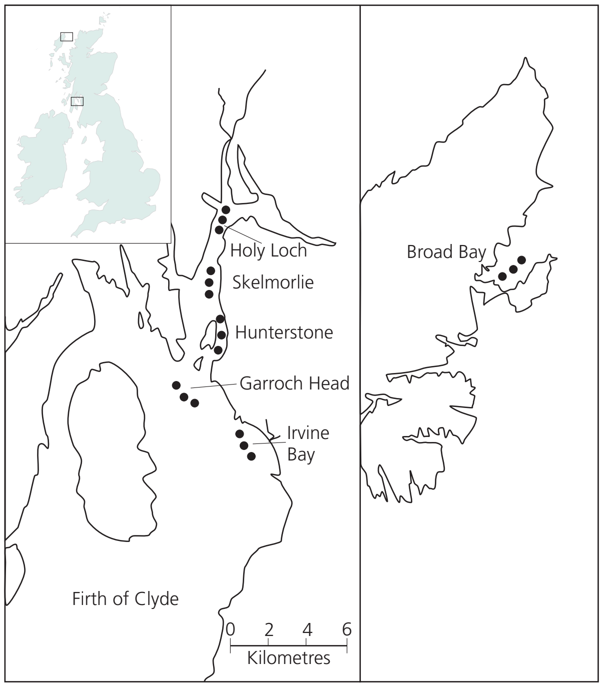


Figure 39. Sampling sites in the Firth of Clyde and Broad Bay (reference) for 1999 and 2000 PAH specific biological effects survey in plaice (Pleuronectes platessa)

In 2000, EROD activity in plaice liver from Garroch Head (a sewage sludge disposal site) and Holy Loch in the upper Clyde was elevated in comparison to the reference site at Broad Bay. Total sediment PAH concentrations followed a similar trend with the highest levels reported at Holy Loch and Garroch Head (Figure 40).

All four PAH metabolites were present in plaice bile from these areas, with higher concentrations generally found from the Firth of Clyde compared to the reference site at Broad Bay. Particularly elevated concentrations of naphthalene metabolites were present in fish from the Irvine Bay and Upper Firth stations, suggesting oil exposure (Figure 41). Fish from Irvine Bay also contained the highest levels of phenanthrene and pyrene metabolites, while fish from the Garroch Head dump site presented significantly higher levels of benzo[*a*]pyrene metabolites than any other station.

These biological effects indicators have demonstrated exposure of marine flatfish to PAH within Scottish Firths affected by industrial activity. Flatfish from these areas often showed elevated levels of EROD activity, generally associated with increased total sediment PAH. Biliary metabolite analysis confirmed that exposure to PAH had occurred and it identified specific compounds of concern.

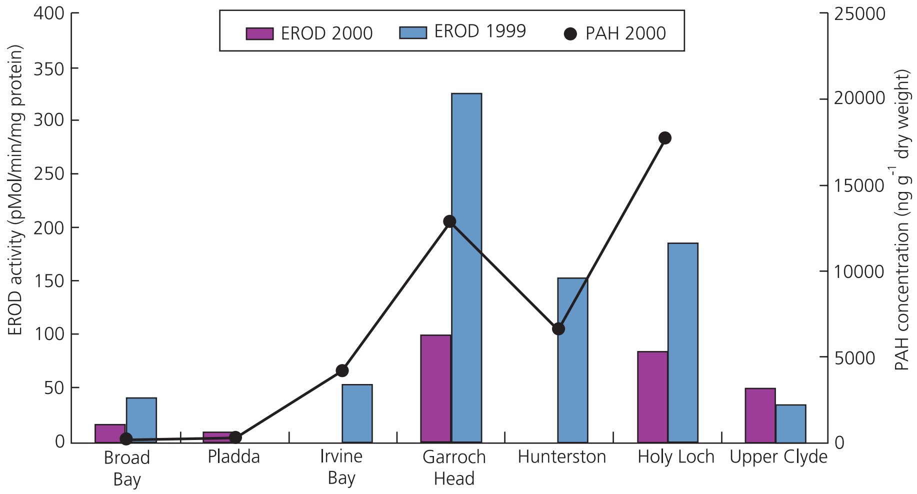


Figure 40. EROD activity in liver of plaice (Pleuronectes platessa) from the Firth of Clyde in 1999 and 2000. Total sediment PAH burden from 2000 is included for interpretation

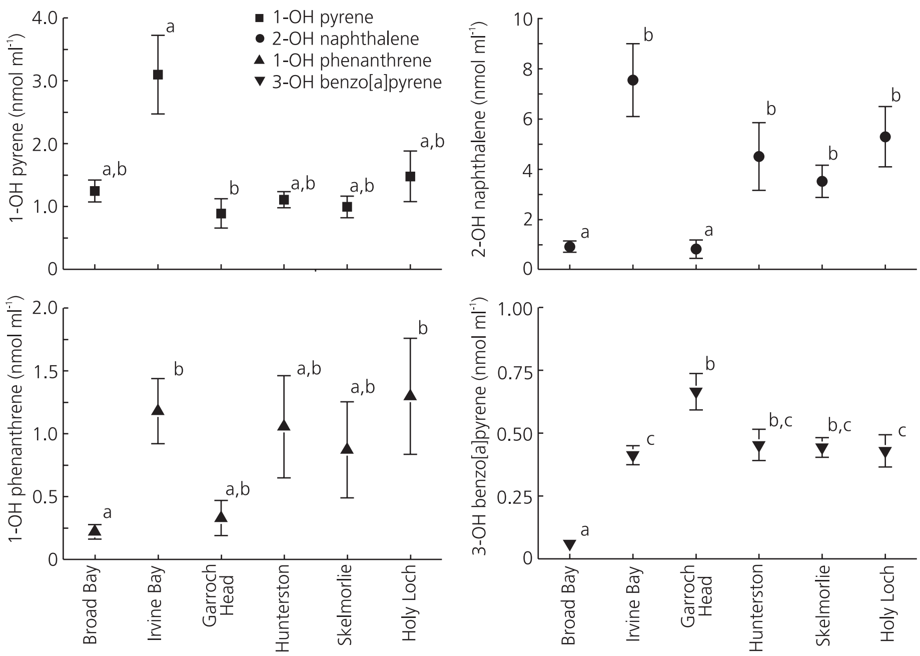


Figure 41. Biliary PAH metabolite concentrations from plaice (Pleuronectes platessa) sampled from the Firth of Clyde, 2000. Data are mean +/- SEM, while stations with different superscripts (a,b) are significantly different from Broad Bay (P<0.05)

Response

Historical point sources have reduced because of effluent treatment improved to comply with UK legislation (e.g. at the Grangemouth oil refinery). Diffuse PAH inputs are being quantified by UK regulatory bodies. Further controls will be implemented through the Water Framework Directive. Environmental monitoring will continue so as to detect trends arising from changes in inputs.

11. TRENDS IN ORGANOCHLORINE RESIDUES IN MUSSELS (*MYTILUS EDULIS*) FROM THE MERSEY ESTUARY

Driving force

The Mersey Estuary was abused and neglected since the Industrial Revolution began. For over 150 years, manufacturing effluents and burgeoning centres of population created an unenviable reputation as one of the most polluted European rivers. It was believed the large volume and the strong tides could receive almost unlimited untreated effluents. Accompanying an increase in manufacturing, many industries required large volumes of water for processes such as cooling, bleaching, dying, tanning and metalworking.

An industrial resurgence followed the Second World War, particularly in petrochemicals and heavy chemicals based upon the chloralkali industry. Although organochlorines were not produced on the scale of heavy chemicals, there was concern because of faunal impact at very low concentrations. Moreover, many such substances are not readily biodegradable and consequently remain in the environment for prolonged periods.

Blue mussels (*Mytilus edulis*) were used to determine spatial and temporal trends in the concentrations of a number of organochlorine compounds. Mussels were collected from a number of sites in February 1994 and 1998 (Figure 42) (Connor *et al*., 2001). Concentrations of all determinands except PCBs and dieldrin were highest in the estuary and decreased downstream.

Pressure

Input data were available from the tidal limit of three rivers draining into the Mersey for g-HCH, DDT and PCB. During this time inputs of PCBs decreased but there were no trends in inputs of g-HCH and DDT.

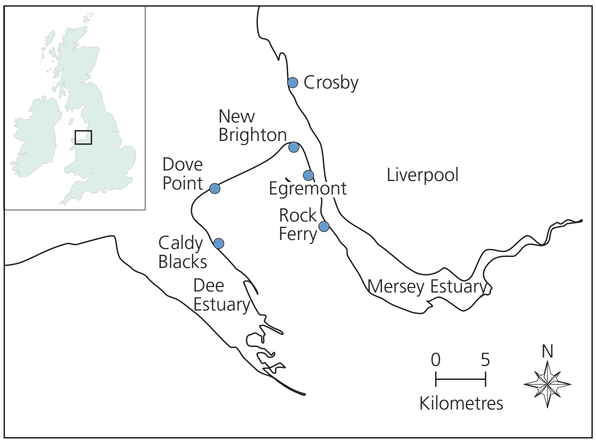


Figure 42. Mussels sampling sites

State

α-MHCH

This methyl derivative of HCH (hexachorocyclo-hexane) showed a significant 4 to 10-fold reduction in mussel tissue (Figure 43). The reduction from 1994 to 1998 was much greater than that for other organochlorines. The source was located to a chemical plant in the inner estuary. Since MHCH contamination was first reported in the early 1990s, the company commissioned a new chlorination plant in 1994 and changed the operating procedures, including treatment and disposal of chemical waste in pursuit of waste minimisation. Such a big difference in the observed concentration of this compound over only 4 years is remarkable when other organochlorines such as pp'–DDT persist for decades after termination of their use by law.

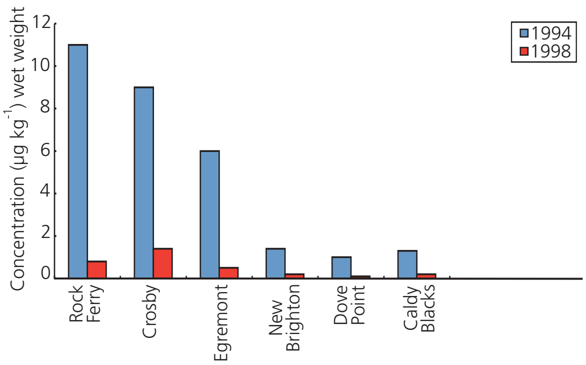


Figure 43. α-MH6CH in mussels from the Mersey Estuary (1994 vs 1998)

There is no evidence of historical contamination of MHCHs in dated sediments from other Mersey cores, in contrast to high concentrations reported for pp'-DDT and its metabolites. With reduced contemporary discharge of MHCHs, no historical sediment sink to recycle, and with short biological half-life, MHCHs could become almost undetectable within a decade.

PCBs

Although there has been little change (Figure 44) in PCB concentration when presented as ΣICES7 (summation of congeners 28, 52, 101, 118, 138, 153 and 180), there have been changes in individual congeners. For example, between 1994 and 1998 congeners 28 and 101 reduced at Rock Ferry while others remained static (Connor *et al*., 2001).

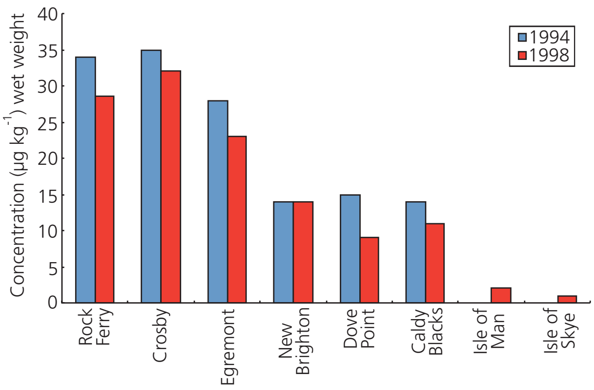


Figure 44. ΣICES 7 PCB Congeners in mussels from the Mersey Estuary (1994 vs 1998)

As inputs from point sources declined, concentrations of congeners 28 and 101 fell. Higher chlorinated congeners such as 153 and 138, predominantly attached to particulate material, may be trapped in sediment perhaps to be remobilised in response to salt marsh accretion and erosion. The slow rate of decline in ΣICES7 between 1994 and 1998 suggests it will be decades before the estuary loses its legacy of PCB contamination

DDT

DDT is presented (Figure 45) as the summation of the *pp'*–DDT, *pp'* –DDD and the *pp'*–DDE concentrations. Change was less pronounced than that for α-MHCH but more than that for PCBs, with a 25-60% reduction in the Mersey sites. In 1998, Rock Ferry, Egremont and Crosby remained the most contaminated but the difference between the two inner estuary sites had gone. The reason follows from sedimentary analyses (Connor *et al.*, 2001). In 1994, mussels had a profile similar to the 1990 nearby sediment, except for pp'-DDE, which was much lower (Figure 48), because it converts rapidly in sediment to pp'-DDD. The 1994 mussel monitoring preceded the full benefit of investments to improve water quality. Relative abundance of pp'-DDT in both the 1994 mussels from Rock Ferry and the 1990 sediment indicates that, although long since banned, pp'-DDT was still being discharged or remobilised from sediment. The 1998 data reveal a reduction in both pp'-DDE and pp'-DDT concentrations after cessation of untreated sewage discharge and increasing effectiveness of the DDT ban.

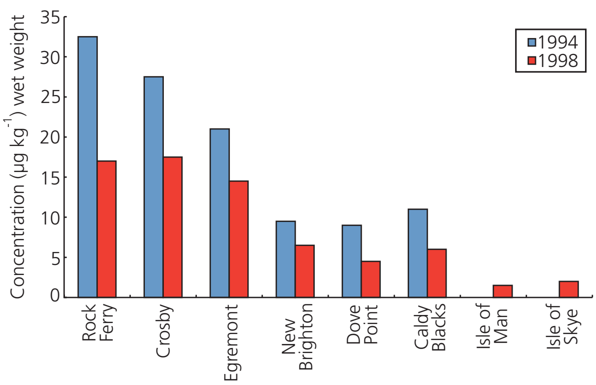


Figure 45. ΣDDT in mussels from the Mersey Estuary (1994 vs 1998)

Dieldrin

The difference in dieldrin concentrations between 1994 and 1998 (Figure 46) follows a trend similar to that for ΣDDT. Correlation between the Dieldrin and ΣDDT concentrations for the 1998 samples probably reflects similar physicochemical properties and sources.

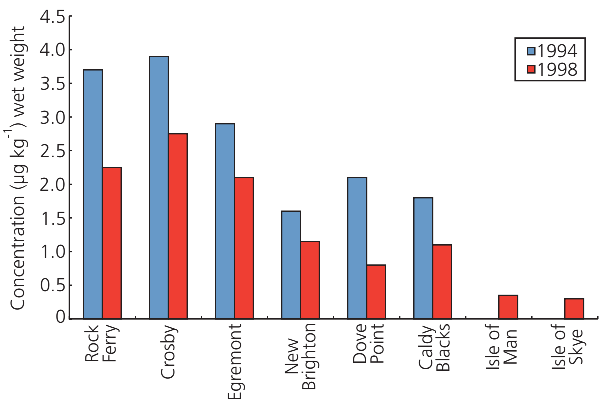


Figure 46. Dieldrin in mussels from the Mersey Estuary (1994 vs 1998)

HCB (hexachlorobenzene)

HCB levels likewise reduced over the 4 years (Figure 47), approaching those at uncontaminated locations near the Isle of Man and Isle of Skye. The change is statistically significant and may be a response to better sewage treatment and falling industrial loads. HCB is one of the by-products of the century-old UK chloralkali industry, which has responded to regulatory pressures by investing methods to reduce HCB discharges.

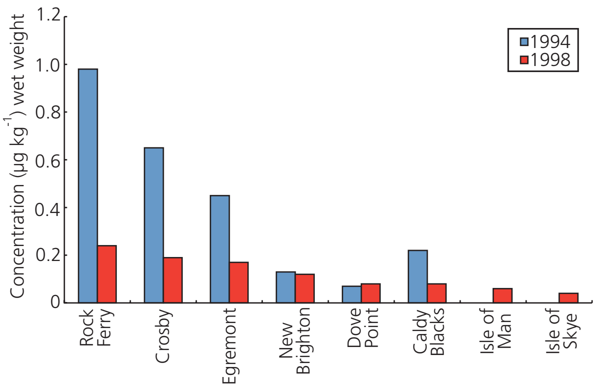


Figure 47. Hexachlorobenzene (HCB) in mussels from the Mersey Estuary (1994 vs 1998)

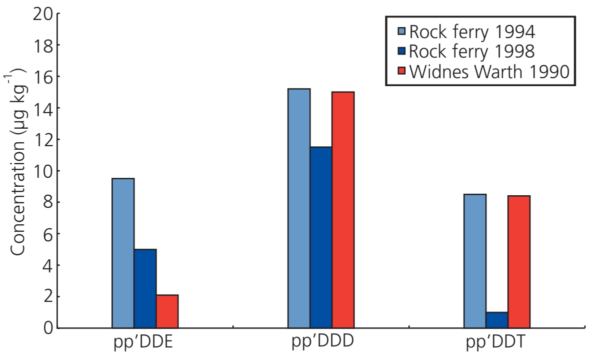


Figure 48. DDT Components of Rock Ferry mussels (1994 vs 1998) and sediment from Widnes Warth (1990)

Impact

Many years of large discharges of untreated domestic sewage and industrial effluents have seriously affected the estuary in three principal ways: fouling of beaches with sewage debris; summer dissolved oxygen loss from long reaches in the upper estuary; and accumulation of dangerous substances (heavy metals and persistent organics) in the sediments and biota.

Published work over thirty years has revealed elevated concentrations of these materials in water, fish, shellfish and sediments. In 1992, MAFF advised consumers not to eat fish caught in such industrialised estuaries.

Recent biomarker research on the stress response syndrome has raised continuing concern about the status of the estuary. Tests such as hepatic ethoxyresorufin O-deethylase (EROD) and induction of the yolk protein vitellogenin have been used to appraise sub-lethal contamination in fish from several industrialised UK estuaries. The highest mean EROD activity was in flounder (*Platichthys flesus*) from the Mersey. The intersex condition of fish based on the vitellogenin levels and oocytes in their testes underpinned these anxieties about environmental quality.

Many, if not all, of these biological effects have been linked with the concentration of PAHs and PCBs. These and other persistent organics are ubiquitous in the Mersey.

Response

In the last twenty years, numerous initiatives have tackled the problem, the most important being the Mersey Basin Campaign. This 25-year government-supported partnership between local authorities, industry, voluntary organisations and government agencies delivers water quality improvements and waterside regeneration. Continuous discharge of raw sewage from a large part of the conurbation has ceased and interceptor sewers now lead to a new wastewater treatment plant at Sandon Dock in Liverpool. The chemical industry has also changed, with continuing commercial pressure to reduce costs by increasing process efficiency, reducing waste, and cheaper treatment. Collectively, these measures have improved water quality significantly and there is now unequivocal evidence that the large capital investment is succeeding (Jones, 2000).

References

Jones, P.D., 2000. The Mersey Estuary – back from the dead? Solving a 150-year old problem. Journal of the Chartered Institute of Water Engineering Management, 14, 124-130.

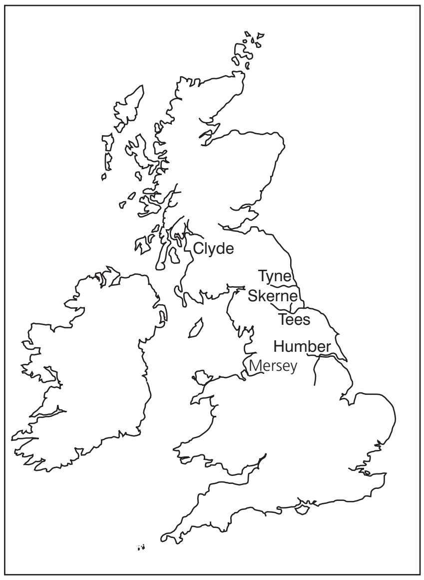
Connor, L., Johnson, M.S., Copplestone, D. and Leah, R.T., 2001. Recent trends in organochlorine residues in mussels (*Mytilus edulis*) from the Mersey Estuary. Marine Environmental Research, 52, 397-411.

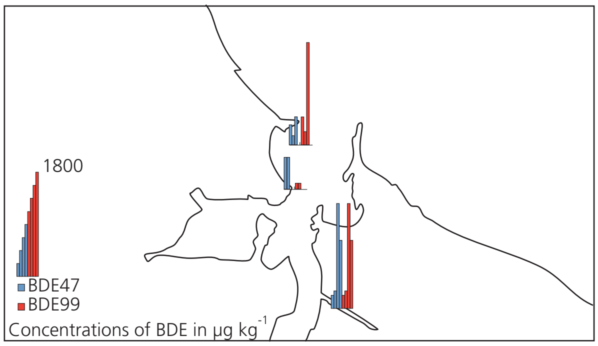
12. SPATIAL SURVEY OF BROMINATED FLAME RETARDANTS

Brominated flame retardants (BFRs) comprise a diverse group of about 75 commercially produced brominated organic compounds. Polybrominated diphenyl ethers (PBDEs), hexabromocyclohexane (HBCD) and tetrabromo bisphenol – A (TBBP-A) are used in numerous commercial and domestic applications where flame retarding and fire prevention matters. They are used in building and in common products such as carpet backing, textiles, printed circuit boards, expanded polystyrene, and polyvinyl chloride piping.

PBDEs are lipophilic hydrophobic compounds with bioaccumulative potential, now recognised as global contaminants. The lower brominated compounds have the potential for long-range atmospheric transport. Commercial formulations are classified according to the degree of bromination and commonly referred to as 'penta mix', 'octa mix' or 'deca mix'. In 1999 210 tonnes 'penta mix', 450 tonnes 'octa mix' and 7500 tonnes 'deca mix' were used in the EU. In addition 13,800 tonnes of TBBP-A and 3100 tonnes of HBCD were used.

Most UK aquatic environment monitoring has focussed on PBDEs, although more recently on HBCD and TBBP-A (Allchin *et al.*, 1999, 2000, 2001; Boon *et al.*, 2002; Morris *et al.*, in prep.). Much of this has concentrated on the Tees, with input from a manufacturer that used to produce PBDEs but has now switched production to HBCD.





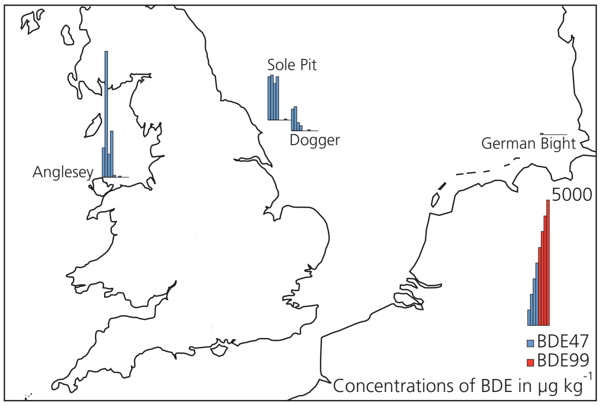


Figure 49. The concentrations of BDE in dab liver

PBDEs in sediment and biota

Concentrations of selected PBDE congeners were determined in dab liver from NMMP sites in the Tees estuary. Sediments have been sampled in the Tees at some other sites as part of a more extensive programme, see Figure 49 .

The most dominant PBDE congener found in biota is the 2,2',4,4' tetrabromodiphenyl ether, BDE47. It dominates 'penta mix' formulations, so its presence probably reflects 'penta mix' usage. PBDEs were detected in all samples. The highest total BDE concentrations outside the Tees, up to 330 μg kg^{-1} wet weight, were recorded off Anglesey : lowest levels were recorded in the German Bight.

High concentrations of BDE209, the fully brominated deca compound, have also been seen. The lack of BDE209 in biota is probably a function of its molecular weight and cross section, which prohibit bioaccumulation. With the exception of terrestrial birds of prey, (ref) BDE209 is rarely reported in biota at other than at very low levels.

The rivers Skerne and Tees had the highest PBDE concentrations but significant concentrations were also seen in the Clyde and Mersey.

Recent investigations have shown high levels of HBCD diasteroisomers in fish tissue, porpoises, cormorants and sediment from the Tees catchment (Morris *et al.*, in prep.).

Despite recent EU restrictions, PBDEs are common marine contaminants and warrant further study. TBBP-A is occasionally present in sediments and biota but at much lower concentrations than either PBDEs or HBCD. Few data are available for HBCD but recent work suggests concentrations in some locations and matrices may be higher than levels of PBDEs. Because HBCD is still unrestricted, these levels may increase.

References

Allchin, C.R., Law, R.J. and Morris, S., 1999. Polybrominated diphenyl ethers in sediments and biota downstream of potential sources in the UK. Environmental Pollution, 105:197-207.

Allchin, C.R., Morris, S., Bennett, M., Law, R.J. and Russell, I.C., 2000. Polybrominated diphenyl ether residues in cormorant (*Phalcrocorax carbo*) livers from England UK. Organohalogen Compounds, 47: 190-193.

Allchin, C.R. and de Boer, J., 2001. Results of a comprehensive survey for PBDEs in the river Tees UK. Organohalogen Compounds, 52: 30-33.

Boon, J.P., Lewis, W.E., Tjoen-A-Choy, M.R., Allchin, C.R., Law, R.J., de Boer, J., ten Halles-Tjabbes, C.C. and Zegers, B.N., 2002. Levels of polybrominated diphenyl ether (PBDE) flame retardants in animals representing different trophic levels of the North Sea food web. Environmental Science and Technology, 36: 4025-4032.

Morris, S., Allchin, C.R., Zegers, B., hafta, J., Boon, J., Belpaire, C. and de Boer, J. (in prep.). The behaviour of the brominated flame retardants and hexabromocyclododecane (HBCD) and tetrabromobisphenol-A (TBBP-A) in estuaries and marine food webs.

13. VITELLOGENIN EXPRESSION IN ESTUARINE MALE FLOUNDER (*PLATICHTHYS FLESUS*)

Some substances disrupt the normal endocrine processes of fish. The potential effects of endocrine disrupting chemicals in the sea were first recognised after freshwater research connected oestrogenic chemicals - originating from domestic sewage and industrial discharges - with various biological effects. Initial monitoring showed similar impacts within estuaries. The potency of many endocrine disrupting chemicals, their ubiquity in sewage effluents and their potential for interfering with the normal reproduction of fish have led to further research.

The most widespread evidence of endocrine disruption is the feminisation of male fish through their exposure to oestrogenic substances. The degree of feminisation is quantifiable by the presence of vitellogenin (VTG) in male blood plasma. VTG is a female-specific yolk pre-cursor protein, measured by an enzyme linked immunosorbent assay (ELISA) or radio-immuno assay (RIA) technique (Matthiessen *et al.*, 1998).

Since 1996, VTG in male flounder has been monitored in a number of UK estuaries. Decreases in VTG have been associated with improvements in effluent treatment.

Results

Plasma VTG concentrations were elevated in male flounder from a number of estuaries (Figure 50).

The highest levels (>10^7 ng ml^{-1}) were recorded in the Tees and Mersey estuaries. Occasional high levels (>10^5 ng ml^{-1}) were also observed in the Tyne, Clyde and Forth. Results for individual estuaries are discussed below.

Alde

The Alde is a small estuary in Suffolk that receives neither sewage nor industrial discharges. Plasma VTG concentrations in male flounder have remained consistently low (<200 ng ml^{-1}) since the time series began in 1996.

Mersey

VTG concentrations in male flounder from Eastham Sands decreased by an order of magnitude between 1996 and 1997. In 1996, the mean concentration was the highest ever detected in male flounder from a UK estuary (>10^7 ng ml^{-1}). However, in September 2001, mean levels were below 10^4 ng ml^{-1}. Despite this substantial reduction, the mean levels remained high compared to other estuaries, although only 30% of fish were affected.

Tyne

Different patterns were observed at each of the three sampling sites. At the most upstream (St. Anthony's), the mean concentration in September 2001 was much lower than previously, only 20% of fish showing induction. There was no clear trend in the data.

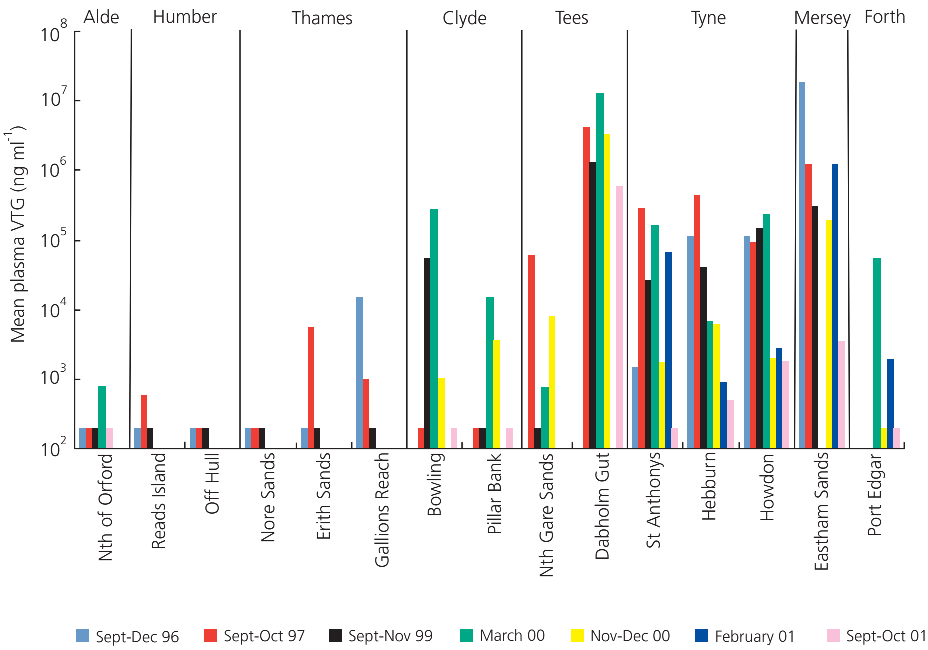


Figure 50. Mean plasma VTG (ng ml^{-1}) in male flounder from estuarine surveys between 1996 and 2001

At Hebburn, there was a clear and consistent downward trend from more than 10^5 ng ml^{-1} in 1997 to below 10^3 ng ml^{-1} in September 2001. In 1996 all fish showed evidence of VTG induction but by October 2001, levels were low (<200 ng ml^{-1}) in more than 80% of fish.

At Howdon, concentrations remained high (>10^5 ng ml^{-1}) between 1996 and March 2000. By November 2000, plasma VTG concentrations had declined to just above 10^3 ng ml^{-1} and remained there.

Tees

Concentrations at Dabholm Gut were high throughout. Between 1997 and November 2000, mean concentrations were consistently greater than 10^6 ng ml^{-1} with a peak of above 10^7 ng ml^{-1} in March 2000. A small reduction below 10^6 ng ml^{-1} was seen in September 2001, still two orders of magnitude higher than any other estuary at this time. However, the first evidence of non-induced specimens was found in November 2000 and in 2001.

Clyde

Fish were collected from two sites, Pillar Bank and Bowling. At Pillar Bank there was no evidence of VTG induction in 1997, 1999 and 2001. There was evidence of induction on both occasions in 2000. At Bowling there was low to moderate induction in 1997 and 2001, but higher levels in September 1999 and March 2000. In March 2000 the mean concentration was above 10^4 ng ml^{-1} at Pillar Bank and above 10^5 ng ml^{-1} at Bowling.

Forth

The Port Edgar site on the Forth estuary was only sampled after September 1999. Although there was clear evidence of induction in over 70% of male flounder in March 2000, with mean concentrations above 10^4 ng ml^{-1}, it was low or absent at other times.

Summary

Plasma VTG concentrations in male flounder were raised in the Tees, Mersey and Tyne, and appeared to be decreasing in the Tyne and Mersey but not the Tees.

The reduction in plasma VTG concentrations in male flounder captured near the Howdon sewage treatment outfall (Tyne) was linked to introduction of secondary treatment in 2000 (Kirby *et al*., 2003). Conversely, concentrations in plasma of male fish captured adjacent to the Dabholm Gut discharge in the Tees estuary seem not to have declined even though reduction and treatment of the effluents has also occurred there.

References

Kirby, M.F., Allen, Y.T., Dyer, R.A., Feist, S.W., Katsiadaki, I., Matthiessen, P., Scott, A.P., Smith, A., Stentiford, G., Thain, J.E., Thomas, K.V., Tolhurst, L. and Waldock, M.J., 2003. Surveys of plasma vitellogenin and intersex in male flounder (*Platichthus flesus*) as measures of endocrine disruption by oestrogenic contamination in UK estuaries: Temporal trends 1996-2001. Environmental Science and Technology, in press.

Matthiessen, P., Allen, Y.T., Allchin, C.R., Feist, S.W., Kirby, M.F., Law, R.J., Scott, A.P., Thain, J.E. and Thomas, K.V., 1998. Oestrogenic endocrine disruption in flounder (*Platichthys flesus*) from UK estuarine and marine waters. Science Series Technical Report, CEFAS Lowestoft, 107: 48pp.

14. MEIOFAUNA STUDIES IN MARINE MONITORING PROGRAMMES

Introduction

The soft-bottom benthic infauna is used frequently to monitor the effects of man-made activities on the sea bed. They are largely sedentary and so must withstand local environment extremes or perish. For convenience, most traditional biological investigations targeted larger macroinfauna (animals retained on 1000 or 500 µm sieves), whereas the smaller meiofauna (between 500 and 63 µm, Figure 51) was largely neglected. Because of high abundance, ubiquitous distribution, rapid generation times and fast metabolic rates, meiofauna play an important role in ecosystem function (Coull and Chandler, 1992) and their state may reflect the overall health of the marine benthos (Kennedy and Jacoby, 1999).

To improve understanding of meiobenthic communities' response to anthropogenic impacts and natural environmental factors, twelve locations around the UK - some of them long-term NMMP stations - were sampled (Figure 52). The specific objectives addressed in this study were: to assess the spatial pattern inshore (1998/99) and offshore (1997); to determine the temporal stability of the spatial patterns (between 1997 and 1999) at four selected offshore sites; to relate observed patterns to measured environmental variables; and to relate meiofauna distributions to the sedimentary concentration of trace metals.

Sediment samples were collected with the Bowers and Connelly multiple mini-corer and analysed for meiofauna (nematode and harpacticoid copepod assemblages), particle size distributions and trace metals concentrations. The data were analysed using univariate and multivariate statistical techniques (Clarke and Warwick, 1994).

All sediments were muddy sands or sandy muds. The silt and clay content was lowest inshore off the Tyne (17%) and the Thames (19%) and highest in Dundrum Bay (87%). The total organic carbon content, ranging from 1.3% to 3.2%, was highest off the Tyne and in the Burbo Bight, and lowest off Plymouth. The concentrations of most trace metals were generally low. Copper and mercury in particular, were higher inshore than offshore.

Meiofauna assemblage structure

Spatial patterns

Inshore, diversity of nematode assemblages was highest at the Tees and lowest in Burbo Bight. Offshore, diversity was significantly higher in the Celtic Deep than off the Tyne and in Lyme Bay

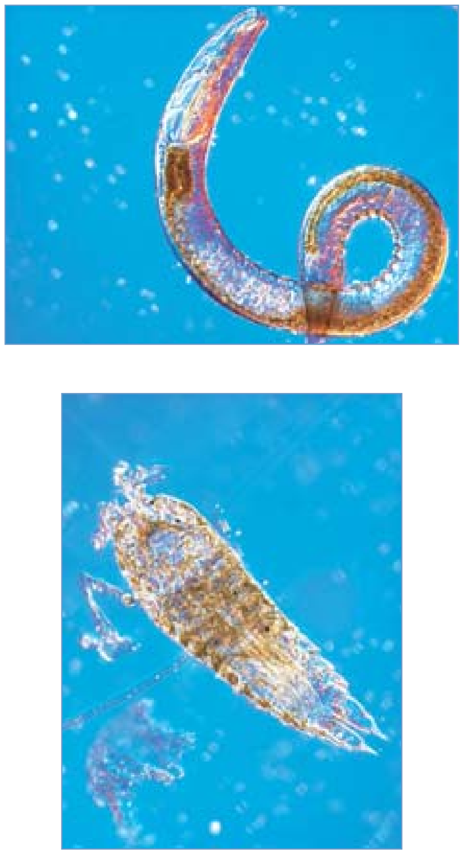


Figure 51. Dominant meiofaunal taxa - Nematode worm (top) and Harpacticoid copepod (bottom)

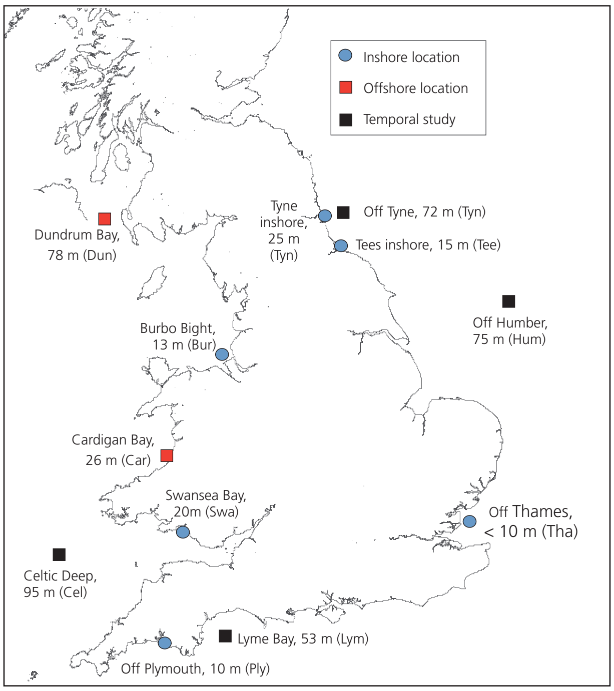
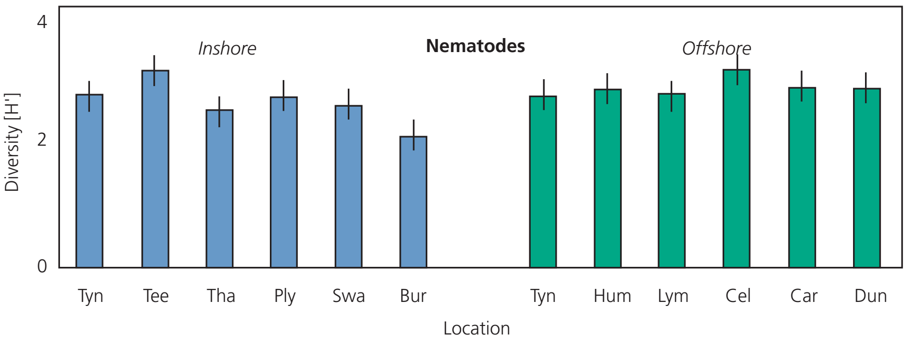


Figure 52. Sites sampled for meiofauna and environmental variables



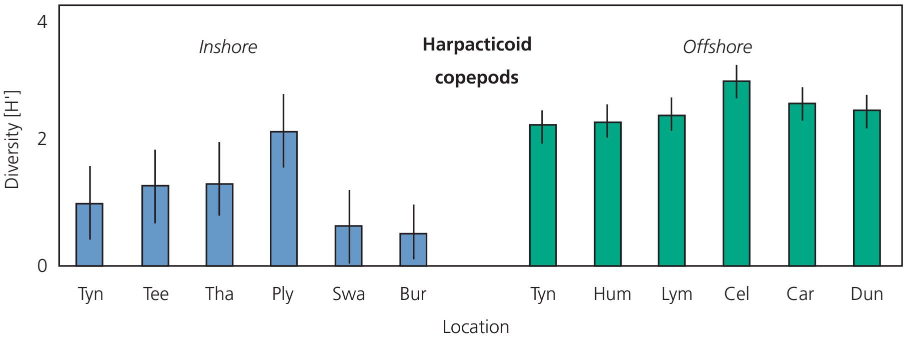


Figure 53. Mean diversity (±95% confidence) of nematode and harpacticoid copepod assemblages. Codes as in Figure 52

(Figure 53). The inshore harpacticoid copepod diversity was highest off Plymouth whereas offshore diversity was significantly higher in the Celtic Deep compared with most other stations (Figure 53).

Multivariate analysis of both the nematode (Figure 54) and harpacticoid copepod (Figure 55) components discriminated between different locations, indicating distinct differences in species composition.

Lower diversity at some inshore locations compared to offshore was clearly related to higher sedimentary concentrations of trace metals. The concentrations of most trace metals were highest in the Burbo Bight (Liverpool Bay) where the species diversity of meiofauna was lowest. The correlations between nematode and harpacticoid copepod assemblage structure and measured environmental variables were generally higher inshore than offshore.

Temporal patterns

Four NMMP offshore sites off the Humber, off the Tyne, in Lyme Bay and in the Celtic Deep were sampled for nematodes and environmental variables over three years (Figure 52).

The offshore spatial variability in terms of sediment characteristics, nematode diversity (Figure 56) and assemblage structure (Figure 57) was more pronounced than temporal differences.

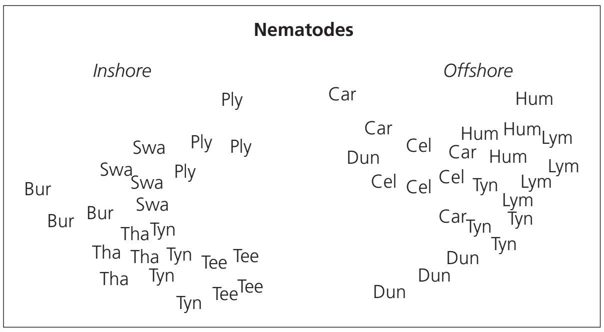


Figure 54. Non-metric multi-dimensional scaling (MDS) ordination based on relative abundance of nematodes at all sampling locations. Codes as in Figure 52

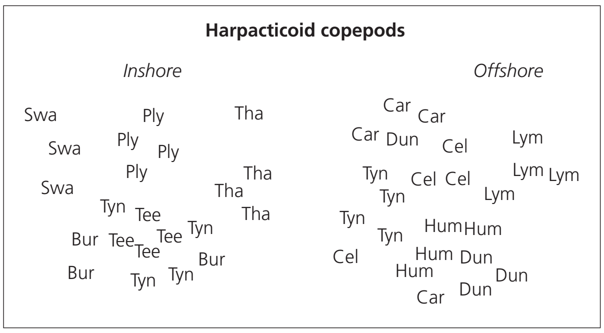


Figure 55. Non-metric multi-dimensional scaling (MDS) ordination based on relative abundance of harpacticoid copepods collected at all sampling locations. Codes as in Figure 52

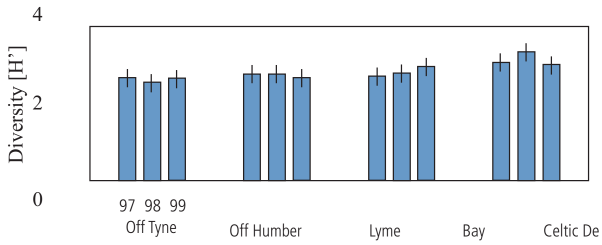


Figure 56. *Mean diversity (±95% confidence intervals) of nematode assemblages at four NMMP offshore locations over a three-year period*

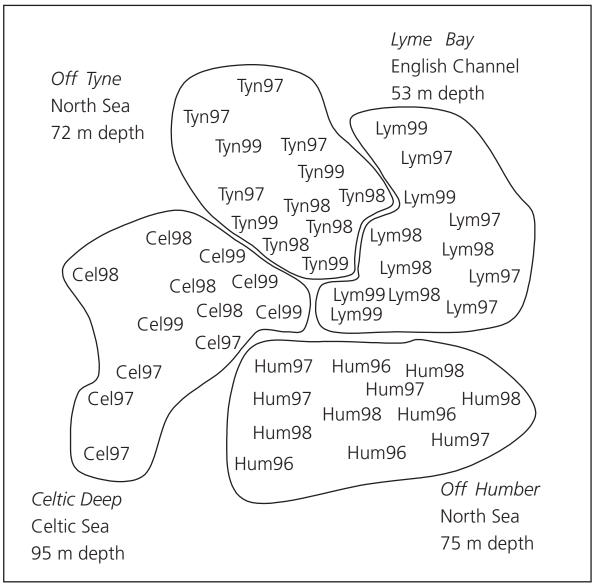


Figure 57. *Non-metric multi-dimensional scaling (MDS) ordination based on relative abundance of nematodes at four offshore locations over three years. Codes as in Figure 52*

Geographical location and water depth were major factors contributing to the establishment and maintenance of nematode distributions whereas the correlation between nematode assemblage structure and the concentrations of trace metals was low.

Summary

Distinct spatial differences in meiofauna species distributions correlate with the natural physical characteristics and sedimentary concentrations of trace metals. Inshore, elevated concentrations of most trace metals compared to offshore might indicate anthropogenic inshore impacts. The concentrations of correlated trace metals were relatively low at most stations and so were unlikely as a cause of biological effects. The diversity of meiofauna assemblages was higher offshore. Inshore, large fluctuations in physical factors including wave and tidal currents, salinity and temperature generally resulted in lower species diversity.

Univariate and multivariate analysis indicated that the spatial variability of nematode assemblage structure at four locations exceeded temporal differences. There was no evidence of any adverse effect of trace metal concentrations on meiofaunal assemblage structure .

References

Clarke, K.R., Warwick, R.M., 1994. Change in marine communities: an approach to statistical analysis and interpretation. Plymouth, Plymouth Marine Laboratory.

Coull, B.C., Chandler, G.T., 1992. Pollution and meiofauna: field, laboratory and mesocosm studies. Oceanography and Marine Biology Annual Review, 30, 191-271.

Kennedy, A.D., Jacoby, C.A., 1999. Biological indicators of marine environmental health: meiofauna - a neglected benthic component? Environmental Monitoring and Assessment, 54, 47-68.

15. HOLY LOCH, SCOTLAND: AN ASSESSMENT OF THE CONTAMINATION AND TOXICITY OF MARINE SEDIMENTS

Driving force

A United States Navy Submarine Base was located in the Holy Loch, adjacent to the Firth of Clyde, Scotland (Figure 58) for over 30 years. After the base closed in 1992, video surveys revealed considerable seabed debris, largely scrap metal. A working group was set up to examine ways of cleaning the loch. Consultants employed by the UK Ministry of Defence made an environmental survey, including assessment of the consequences of lifting the debris from the loch bed.

Pressure

The survey found sediments to be contaminated by organic compounds and metals (ERM, 1997). These are persistent, toxic and bioaccumulative substances, and concern was expressed that the debris removal operation might release sediment-bound contaminants, in turn exerting toxic effects on local flora and fauna. The Scottish Environment Protection Agency (SEPA) decided, before the debris removal, to verify sediment contaminant levels, to set these in context on a UK scale, and to assess the toxicity of sediment and water mixtures to local flora and fauna.

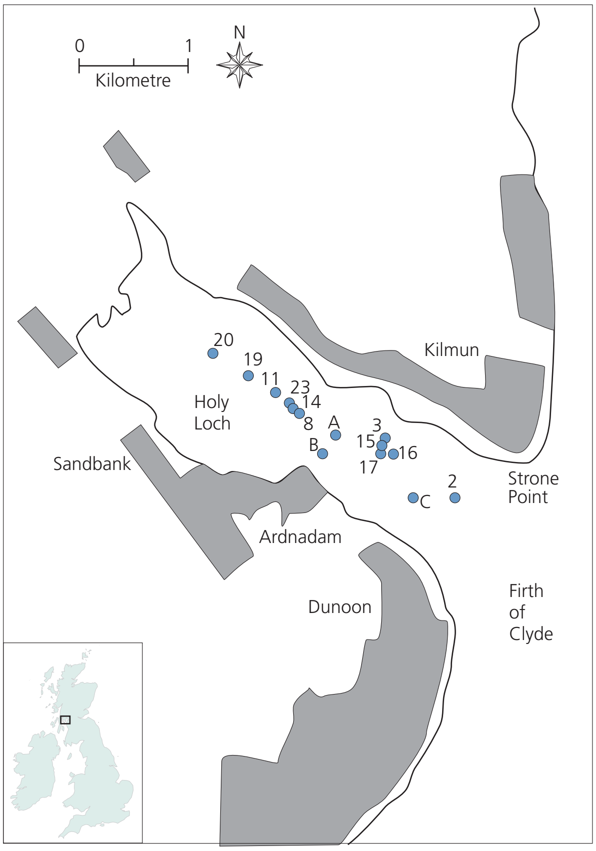


Figure 58. Location of Holy Loch within Scotland, and positions of sampling sites

Table 10. Concentration of trace metals in Holy Loch sediments (mg kg^{-1} dry weight)

Sampling Station	Depth (cm)	Cd	Cr	Cu	Mn	Ni	Pb	Zn	Hg (µg kg^{-1})
A	0-5	0.34	266	111	305	47	257	458	1,150
B	0-5	0.30	222	85	276	42	223	353	872
C	0-5	0.22	179	61	324	42	186	271	3,610
2	0-5	0.22	206	66	378	42	217	299	1,010
3	0-5	0.53	256	562	640	78	517	3,721	2,530
3	5-10	0.96	307	1,566	1,051	145	996	6,527	3,950
8	0-5	0.42	245	409	572	61	33	1,775	1,080
8	5-10	0.33	258	451	517	63	385	1,875	1,320
11	0-5	0.88	282	444	596	62	383	2,076	950
11	5-10	1.01	286	412	463	62	388	1,180	950
14	0-5	0.33	298	672	608	55	281	1,877	804
14	5-10	0.43	379	1,102	596	68	396	3,200	950
15	0-5	0.20	204	100	332	45	211	395	1,980
15	5-10	0.27	223	83	324	48	246	374	992
16	0-5	0.25	208	125	363	48	237	578	1,480
16	5-10	0.31	260	153	448	52	292	743	1,200
17	0-5	0.28	206	120	350	45	218	407	1,200
17	5-10	0.28	198	82	328	48	272	397	1,060
19	0-5	0.54	223	157	307	50	243	531	1,590
20	0-5	0.40	124	70	265	37	136	251	807
23	0-5	0.33	391	1,185	695	58	272	2,692	835
23	5-10	0.52	479	1,683	656	67	300	4,089	784

State

Metals

Sediment metal concentrations were very variable (see Table 10). The highest concentrations were at former positions of the US Navy Base, either before 1972 (station 3) or after 1972 (stations 8, 14 and 23). The pattern for copper and zinc is shown in Figure 59. Sediment contamination could not be compared to OSPAR BRCs because metal/aluminium ratios were not determined but contamination was described as severe at some sites compared to other areas (Bryan and Langston, 1992) (see Figure 60).

The results were compared to the OSPAR Ecotoxicological Assessment Criteria (EACs) and sediment quality values (SQVs) in Table 11. OSPAR EACs are provisional values designed to identify areas of potential concern. SQVs derived by Long *et al*. (1995) for the US National Oceanographic and Atmospheric Administration (NOAA) and by Environment Canada (CCME, 1998) are shown for comparison. The percentages of sites where adverse biological effects on benthic infauna were predicted as 'possible' or 'probable', on the basis of these values, are shown in parenthesis. Comparison with EACs indicates that chromium, copper, mercury and lead concentrations at 95-100% of the sites were

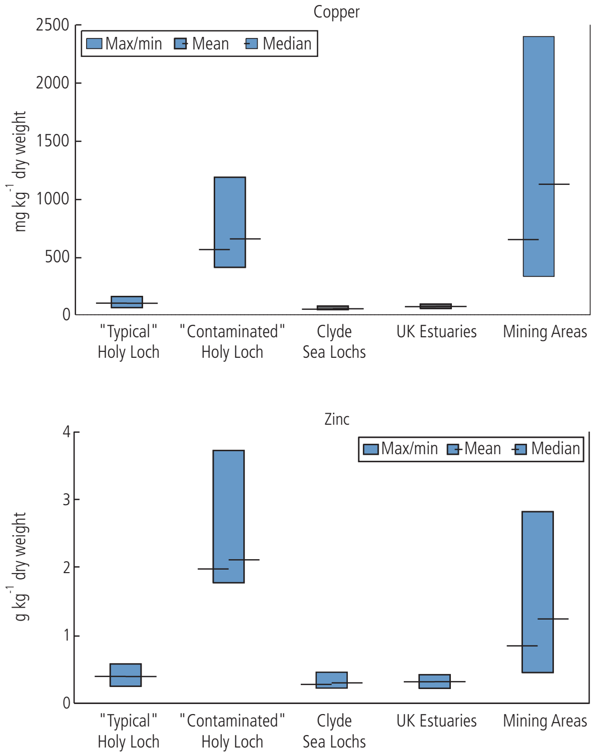


Figure 60. Comparison of sediment copper and zinc concentrations

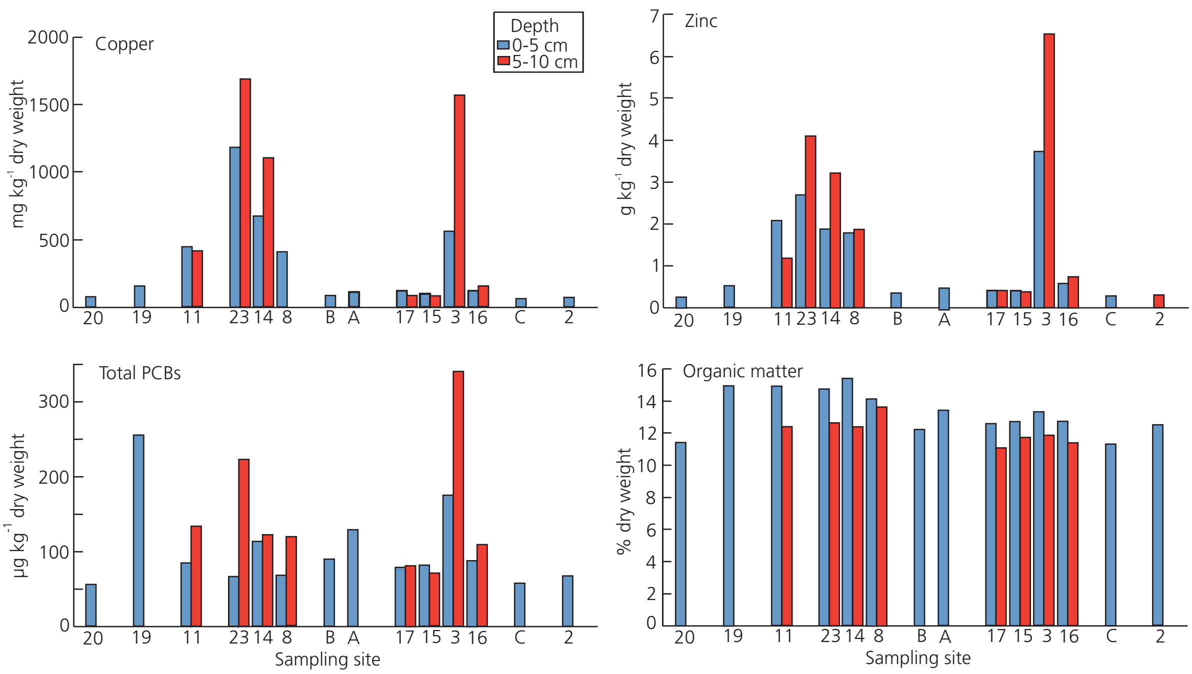


Figure 59. Concentrations of key parameters in Holy Loch sediments

Table 11. OSPAR Ecotoxicological Assessment Criteria (EACs) and US and Canadian Sediment Quality Values (SQVs)

Metal	OSPAR EACs (mg kg^{-1} dry wt) (percentage of sites above range)	US and Canada SQVs (percentage of Holy Loch sites in each category)			
		ISQG (a)	ER-L (b)	PEL (a)	ER-M (b)
		Unlikely	Possible if >ER-L	Probable if >PEL	Very Probable if >ER-M
Cadmium	0.1-1 (5)	0.7 (100)	1.2 (0)	4.2	9.6
Chromium	10-100 (100)	52.3	81 (7)	160 (86)	370 (7)
Copper	5-50 (100)	18.7	34 (36)	108 (29)	270 (36)
Mercury	0.05-0.5 (100)	0.13	0.15	0.7	0.71 (100)
Nickel	5-50 (50)		21 (64)		52 (36)
Lead	5-50 (95)	30.2 (7)	47	112 (36)	220 (57)
Zinc	50-500 (59)	124	150 (7)	271 (36)	410 (57)

Data as mg kg^{-1} dry weight from (a) CCME, 1998 and (b) Long et al.,1995.

sufficiently high to cause concern. Comparison with SQVs indicates that adverse effects were 'probable' at 86% of sites for chromium, 'very probable' at 100% of sites for mercury and at 57% of sites for lead and zinc. These considerations do not account for the additive toxic effects of metals, nor for acclimation by local species.

These results suggest that the sediments should not be dredged, since this may cause the metals to be re-mobilised.

Persistent organic pollutants

PCBs were detected in all of the sediments and their distribution is shown in Figure 59. The highest concentrations were at stations 3 and 19. Higher PCB concentrations were measured in the 5-10 cm layer than in the 0-5 cm layer. The congener patterns were similar at each site, suggesting a common origin.

Concentrations of PCBs in Holy Loch sediments were similar to those for other Clyde sea lochs such as the Gareloch and were typical of coastal waters close to an industrialised estuary (SEPA unpublished data; Kelly and Campbell, 1995; CEFAS, 1997) (see Figure 61).

The OSPAR EAC range for the sum of the ICES seven CB congeners is 0.001-0.01 mg kg^{-1} dry weight (OSPAR, 2000). Sediment Quality Values have also been developed for 'total' PCBs (2.5 x the sum of the ICES 7 congeners), and these are summarised in Table 12 (Miller *et al.*, 2000). The guidelines derived by Wells *et al.* (1989) do not account for sediment-bound organic carbon, whereas those derived by the Florida Department of Environmental Protection (FDEP, 1994) and by the Joint Monitoring Group (JMG) of the Oslo and Paris Commission (Fleming *et al.*, 1995) are normalised to 1% total organic carbon.

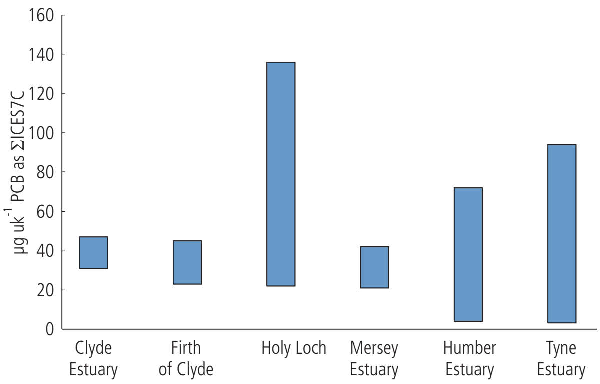


Figure 61. Comparison of PCBs in Holy loch sediments with other UK areas

The PCB concentrations in Holy Loch sediments exceeded the OSPAR EAC range at all sites. Using the guidelines (Wells *et al.*, 1989), the sediments were classed as 'contaminated' or 'heavily contaminated' by PCBs. Concentrations exceeded the FDEP threshold effects level and the JMG 'no biological effects' level, but none exceeded the FDEP probable effects level.

Table 12. Summary of Sediment Quality Values for PCBs

Origin of Sediment Guideline	Description of Sediment Guideline	PCBs Concentration ($\mu g\ kg^{-1}$ dry sediment as Arochlor 1254)
Wells *et al.*, 1989	No contamination Slightly contaminated Contaminated Heavily contaminated	<0.2 - 20 21 - 100 >100
FDEP	Threshold effects level Probable effects level	21.6 189
JMG	Concentration below which biological effects are unlikely	1-10

Impact

Sensitive site-specific Microtox and oyster embryo bioassay toxicity tests were used to assess the risk from contaminated sediments.

The Microtox test is quantitative to the toxicant concentration. Sediment elutriates were found to be non-toxic to Microtox bacteria. The results for exposure times of 5, 15 and 30 min. showed median levels of bioluminescence inhibition of only about 10% for each sediment elutriate concentration (1, 10 and 100 $g\ l^{-1}$).

The oyster embryo bioassay is sensitive and is recommended for monitoring (MPMMG, 1994). The percentage normal embryo development was very high, typically 80-90%, with median values close to 85%. Results for the Holy Loch samples were similar to embryo development in Cape Wrath and Holy Loch reference seawater, whose mean values were respectively 91% and 93%, demonstrating that the sediment elutriates were not toxic.

These tests suggest that acute water column toxicity would be unlikely were sediments to be mobilised during the lifting of debris.

Long-term effects on local flora and fauna

Bioaccumulation of contaminants was assessed by analysing batches of the common mussel *Mytilus edulis* (L.) before and after winter removal operations. Mussels were collected monthly at six stations and a control site. Metals and organics concentrations in the mussels were low, with no significant changes during the initial period of debris removal from October 1987 to March 1988 (Miller *et al.*, 2000).

Although sediment contaminants posed a risk to local fauna, toxicity tests showed that sediment elutriates were non-toxic to the test organisms. The absence of toxicity indicates that the contaminants were not released into the overlying water column when the sediments were disturbed. This was confirmed by the long term monitoring of mussels in the Loch.

Response

Debris removal was completed after the winter of 2001/2. The Holy Loch remains a scenic area suitable for tourism, and it is anticipated that trawling for commercial fish species will resume.

References

Bryan, G.W. and Langston, W.J., 1992. Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. Environmental Pollution 76, 89-131.

CCME, 1998. Canadian sediment quality guidelines for the protection of aquatic life: Introduction and Summary tables. In: Canadian sediment quality guidelines, 1998, Canadian Council of Ministers of the Environment, Winnipeg.

CEFAS, 1997. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1994. Aquatic Environmental Monitoring Report No. 47, CEFAS, Lowestoft. 59pp.

ERM, 1997. Holy Loch Environmental Assessment. Sediment Quality Investigation Interpretative Report. Project No: 08431. Final Report. January 1997.

FDEP, 1994. Approach to the Assessment of Sediment Quality in Florida Coastal Waters. Volume 1 - Development and Evaluation of Sediment Quality Assessment Guidelines. Florida Department of Environmental Protection, Tallahassee, Florida. pp 126.

Fleming, R., Johnson, I., Delaney, P. and Reynolds, P., 1995. Freshwater Sediment Assessment - Scoping Study. SNIFFER Report No.SR3931.

Kelly, A.G. and Campbell, L.A., 1995. Persistent Organochlorine Contaminants in the Firth of Clyde in Relation to Sewage Sludge Input. Marine Environmental Research, 41 (1), 99-132.

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. Environmental Management, 19, 81-97.

Miller, B.S., Pirie, D.J. and Redshaw, C.J., 2000. An Assessment of the Contamination and Toxicity of Marine Sediments in the Holy Loch, Scotland. Marine Pollution Bulletin, 40 (1), 22-35.

MPMMG, 1994. Marine Pollution Monitoring Management Group. Monitoring Co-ordination Sub-group. UK National Monitoring Plan. HMIP, London. 39pp.

OSPAR, 2000. Background Reference Concentrations (BRCs) and Ecotoxicological Assessment Criteria (EACs). MON 00/5/Info.4-E. London.

Wells, D.E., Kelly, A., Findlayson, D.M., Eaton, S., Robson, J., and Campbell, L., 1989. Report of the survey for PCB contamination following the Piper Alpha incident. SOAFD Marine Laboratory, Aberdeen. 9-10.

16. THE NMMP AND MONITORING SPECIAL AREAS OF CONSERVATION

In 1992 the European Community adopted Council Directive 92/43 EEC on the *Conservation of natural habitats and of wild fauna and flora* (the Habitats Directive) which aims to conserve Nature within all territory of the European Union. It requires that Member States designate Special Areas of Conservation (SACs) for specified habitats and habitats of specified species of plants and animals. This Directive applies to seas of the European Union out to 200 nautical miles. Figure 62 shows the proposed UK marine SACs proposed for their habitat features. SACs are also proposed for common and grey seals, and bottlenose dolphins. Work is underway to identify offshore SACs. Member States must report to Europe on the status of the designated habitats and species every six years.

SAC monitoring

The UK government's statutory nature conservation agencies (English Nature, Countryside Council for Wales, Scottish Natural Heritage, and Environment and Heritage Service Northern Ireland) have developed an integrated and consistent approach to assess designated sites. The Marine Monitoring Handbook (Davies *et al*., 2001) guides the monitoring of features within marine SACs.

The desired state is defined in terms of objectives or targets, and site monitoring assesses whether these objectives are met. Site monitoring is distinct from surveillance, which repeats surveys with a standard methodology to provide a series of observations. Surveillance may yield valuable information on trends in biodiversity and physical features but may not by itself reveal whether objectives or standards have been met. The UK National Marine Monitoring Programme is an example of a surveillance programme. Such information may form part of an assessment when combined with objective or standard setting.

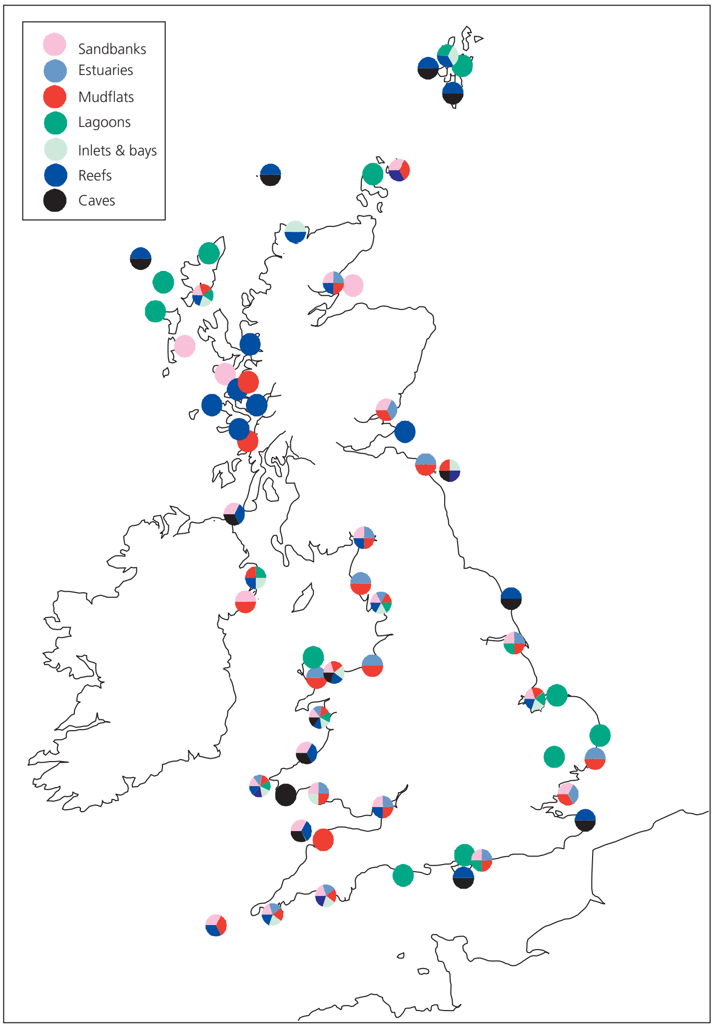


Figure 62. UK marine candidate Special Areas of Conservation with their designated features. Symbols do not represent the relative proportions of each feature

Each interest feature has measurable characteristics or attributes that indicate its condition. They may be quantitative (e.g. extent), qualitative (e.g. biotope composition) or descriptive of physical or environmental processes (e.g. salinity regime) that support the feature. Targets set for these attributes give upper or lower thresholds outside which the condition is deemed unfavourable. Routine surveillance may be compared with these targets, which are intended to trigger management action: if a target is not met, this prompts investigation of site management and whether current activities need changing. The valid interpretation of condition monitoring may also require contextual information, perhaps from a wider geographical area or longer time scales. It is important to attribute observed change correctly to local activity on a site, or to inherent variability, or to a nationwide trend caused by some other factor such as climate change.

Marine SAC features need frequent surveillance in future to reveal their inherent variation, to detect important changes, to judge better the aptness of existing targets, or to define new target values for those attributes lacking many data. Existing surveillance programmes, such as the NMMP, thus contribute significantly to the SAC programme, providing data where sites fall within the SAC boundary and also wider contextual information.

The NMMP assesses the distribution and biological impact of contaminants. A large proportion of sites lies in contaminated estuaries with some reference sites in uncontaminated areas. Benthic faunal data are the only directly relevant biological community information collected by the NMMP but water quality, bioaccumulation and biological effects data help the wider contextual description. Seventeen percent of NMMP stations fall within a marine SAC.

Uses of NMMP data

The use of data for SAC monitoring depends on the location of the NMMP station in relation to a SAC. There are three possibilities:

NMMP sediment station within a SAC with sediment features

Where a NMMP biological station lies within a sediment feature of a SAC, the NMMP data may be used directly in the assessment of feature condition. Historical data may inform target values for attributes measuring the species composition, or may reveal the acceptable range of historical variability. Future NMMP data may also illuminate changes or trends within the SAC interest feature.

NMMP stations within SACs

NMMP benthic community stations within a SAC not designated for a sediment feature do not contribute directly to feature assessment but nevertheless provide a valuable context. For example, changes observed by the NMMP of sediment habitats within an SAC designated only for its reef feature might mirror changes seen on the reef feature. The NMMP might thus inform or explain indirectly changes or trends within the interest features or it may provide an early warning in condition assessment that the condition has changed. NMMP stations measuring chemical and biological effects parameters within a SAC contribute data on anthropogenic pressures and any of their trends. Adverse trends in these variables may help explain any failure of a designated feature to meet its target, and provide information to identify appropriate management action.

NMMP stations outside SACs

Data from NMMP stations outside existing SACs provide a valuable context of regional or national biological or chemical trends that may lie behind changes within a single SAC, or in a feature common to several SACs: for example if a sediment feature in all SACs in a region were to have failed to meet its target. Alternatively, the overall status of a feature at UK level may be compared with national trends in the NMMP data to help ensure a consistency of judgements at the national level.

Reporting

Regular NMMP reporting provides an early warning mechanism for the planning of SAC monitoring. It helps identify any local trends within the six-yearly SAC monitoring cycle, enabling precautionary management.

Conclusion

The NMMP makes a useful contribution to the monitoring of SACs under the Habitats Directive directly via the provision of data, and indirectly with contextual information on local, regional and national trends in biological and chemical parameters

Contextual information from other biological surveillance programmes gives the UK conservation agencies confidence that the attributes they have identified describe the status of the feature, that the targets are appropriate and that they take account of natural variation. Contextual information also helps to compare local and national trends, to explain changes and to ensure consistency of judgements at the national level.

Reference

Davies, J., Baxter, J., Bradley, M., Connor, D., Khan, J.,Murray, E., Sanderson, W., Turnbull, C. and Vincent, M., 2001. Marine Monitoring handbook, Joint Nature Conservation Committee.

17. THE *SEA EMPRESS* OIL SPILL

Driving force

Milford Haven Waterway developed in the 1960s as a major oil terminal and processing site. The handled volume of oil and associated products rose steadily to a peak of 60 million tonnes in 1974 (Figure 63). Annual cargo passing through the waterway fell as the production of North Sea oil reached a peak in the 1980s but it still exceeded 32 million tonnes in 1990.

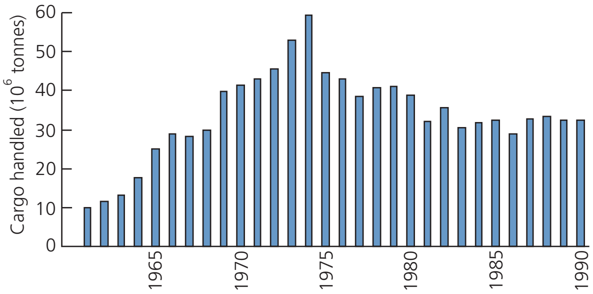


Figure 63. Oil handled in Milford Haven: 1961 to 1990 (Milford Port Authority)

Pressure

An inevitable hazard is the danger of oil spills. Between 1961 and 1990 the annual average was 39 spills, although most were small usually with less than 50 tonnes spilled (Hobbs and Morgan, 1992). The mean annual quantity spilled from tankers is estimated to be about 14.5 tonnes, compared with 180 tonnes from other effluents. Overall, no more than an estimated average 240 tonnes of oil enters the waterway annually, most well dispersed in water and already associated with suspended particles (Little *et al.*, 1987).

In February 1996, 72000 tonnes of crude oil and 480 tonnes of heavy fuel oil were released when the *Sea Empress* ran aground at the mouth of the Milford Haven Waterway. Most of the oil was released over four days, generally on ebbing tides and in northerly winds. These combined to carry the oil offshore into deeper water where over 400 tonnes of chemical dispersants were used to disperse the oil. After a further five days the wind veered to the Southwest and the remaining slick was driven on to the coast. In all, oil contaminated about 200 km of Pembrokeshire coastline from Skomer Island to Pendine Sands in Carmarthen Bay and penetrated the Haven as far as Pembroke (Figure 64).

It is estimated that up to 40% of the oil evaporated soon after release and that approximately 50% dispersed into the water column as a result of natural mixing and the spraying of dispersants. A further 2% was recovered from the sea surface. Of the 7% of the oil that came ashore much was removed by clean-up and natural remediation.

State

Water

Approximately 38% of the spilled oil dispersed as a result of dispersants, while a further 14% dispersed naturally (SEEEC, 1998). Following the spill a maximum total hydrocarbon concentration of 10 mg l^{-1} was recorded in the water near the grounding site compared with pre-spill levels of 5–53 µg l^{-1} within the Haven. Within six weeks

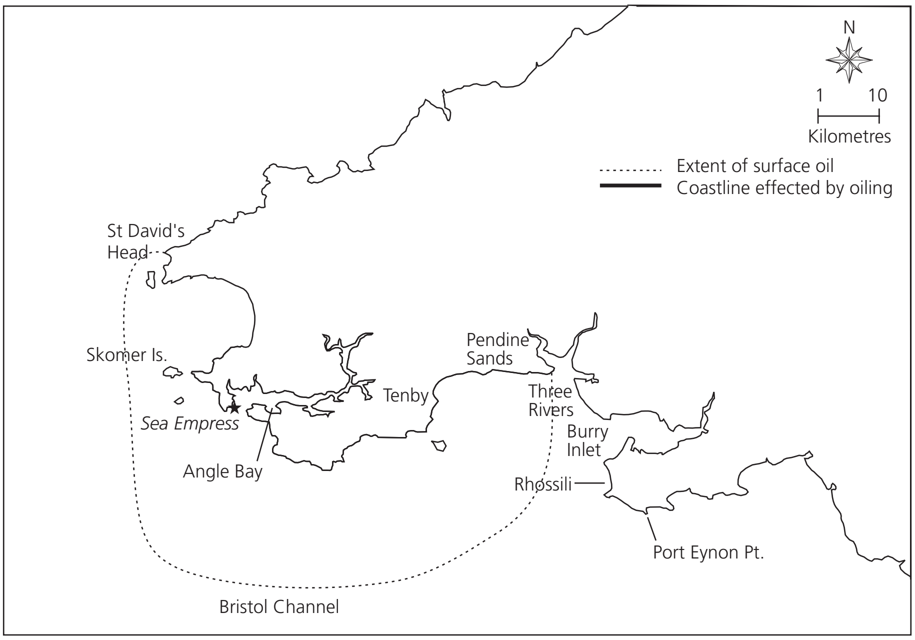


Figure 64. Extent of the oil at sea and the coastline affected by oiling. Variability in severity of the slick and shoreline contamination means this map is merely a rough guide

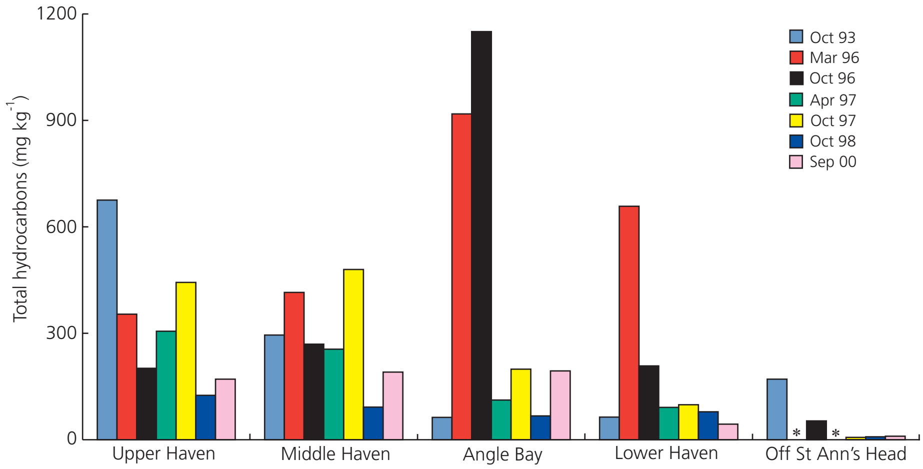


Figure 65. Average total hydrocarbon concentrations in sediments from the Milford Haven Waterway between October 1993 and September 2000. (= no samples)*

levels had fallen below 100 μg l^{-1} over most of the affected area and by June 1996 had returned to background levels of between 1 and 10 μg l^{-1} (SEEEC, 1998).

Sediment

Pre-spill hydrocarbon concentrations increased into the Haven (Rostron, *et al.*, 1986; McLaren and Little, 1987; Little and McLaren, 1989) as a result of the dominant flood tide carrying fine particles and associated contaminants up the waterway (McLaren and Little, 1987). After the *Sea Empress* spill, hydrocarbon contamination was evident in the lower reaches of the waterway and Angle Bay with order of magnitude increases in sediments one month afterwards (Figure 65). No appreciable differences in hydrocarbon levels before and after the spill were apparent elsewhere, although in March 1996 there was a sheen of oil on sediments collected from all sites other than the uppermost part of the Haven (Levell *et al.*, 1997). By Spring 1997 concentrations were similar to those before the spill, with degraded and combustion-derived hydrocarbons predominating over those derived from crude oil (Hobbs, 1998).

Biota

Hydrocarbon concentrations in mussels in the most impacted areas rose rapidly to approximately 2500 mg kg^{-1} total hydrocarbon content (THC) and 1000 mg kg^{-1} polycyclic aromatic hydrocarbons (PAH) one week after the grounding (Dyrynda *et al.*, 1997). Within three months they had reduced - especially PAH concentrations which fell by over 90%. THC concentrations in whelks taken in Carmarthen Bay were up to 40000 μg kg^{-1} and 3800 μg kg^{-1} PAH, although variable, indicating localised seabed contamination (SEEEC, 1998). PAH concentrations of 86 μg kg^{-1} were detected in limpets from the mouth of Milford Haven two weeks after the spill (Glegg *et al.*, 1999). Over the following seven months these levels fell by several orders of magnitude to near background levels.

Despite the amount of oil spilled from the *Sea Empress* and the sensitivity of the contaminated ecosystems, long-term effects appear small.

Impact

Water

There were three oyster embryo bioassays of water samples from various sites in the Haven within two weeks of the spill. There were low level toxic effects in samples from the upper Haven within one week of the spill, although none after two weeks (Environment Agency, 1997).

Sediments

The macrobenthic fauna of the Milford Haven Waterway was studied periodically until the early 1990s, with regular monitoring after the grounding of the *Sea Empress* in 1996. Generally, benthic communities showed little impact of oil contamination, although there was some change at population level (SEEEC, 1998). There was a decline in amphipod fauna throughout the Haven, particularly in the middle and lower reaches, with the genera *Ampelisca* and *Harpinia* and the family Isaeidae particularly affected. This was accompanied by increases in abundance of opportunist polychaete populations as individual species took advantage of the decline of the amphipod fauna (Figure 66). Within five years of the spill the amphipod fauna was clearly re-established and the abundance of opportunists had returned to pre-spill levels (Nikitik, 2001).

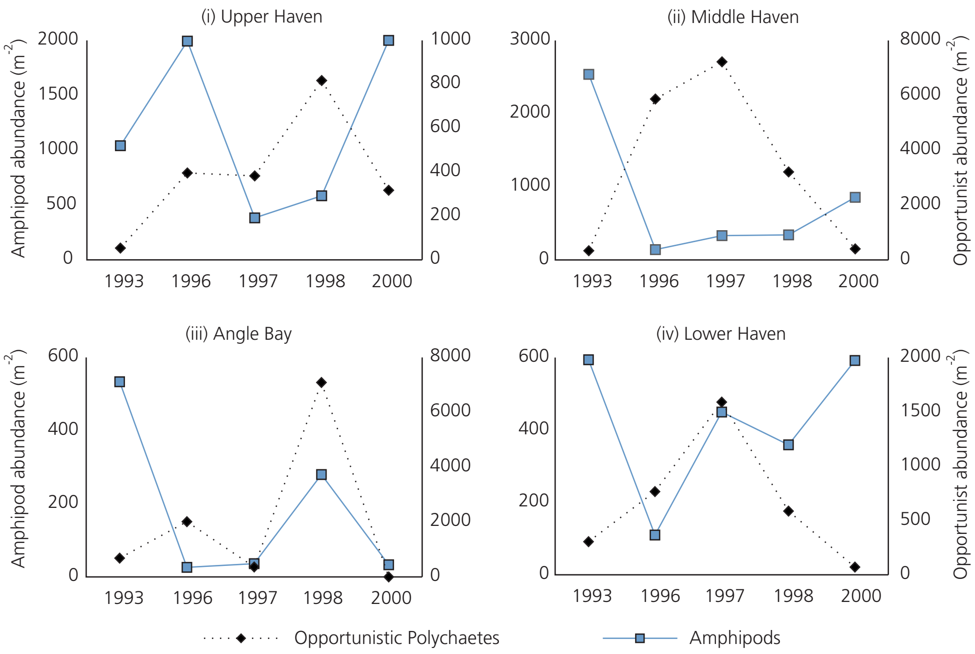


Figure 66. Abundance of amphipods and opportunist species in the Milford Haven Waterway between October 1993 and September 2000.

Biota

After the spill, there were no obvious adult fish kills, but appreciable numbers of dead or moribund sediment-dwelling animals including cockles, striped *Venus* sp. and razor shells were washed ashore in following weeks (SEEEC, 1998). Also, 7 to 10 weeks after the spill many thousands of moribund rayed trough shells stranded on the eastern side of Carmarthen Bay, outside the zone of heavy bulk oil pollution. The concentrations of hydrocarbons in some tissues indicate that these events were a consequence of the spill.

Populations of salmonids (Roberts *et al.*, 1998a & b). and seabass were not affected by the spill (Lancaster *et al.*, 1998) and there was no significant impact on plankton in the southern Irish Sea (Batten *et al.*, 1998). In the aftermath, about seven thousand oiled birds of some thirty six species were recovered, although the total number killed is probably much greater. Of these, over 90% were common scoters, razorbills and guillemots, with other diving species commonly affected. During the subsequent breeding season guillemot numbers reduced by 13% and razorbills by 7%, although by the 1997 breeding season numbers of both species had recovered significantly (SEEEC, 1998).

Response

Following the *Sea Empress* spill, clean-up involved physical removal of bulk oil from beaches, beach washing, onshore use of dispersants and sorbents, and offshore use of booms, mechanical recovery and chemical dispersants.

Immediately after the incident formal fishing controls were put in place under the Food and Environmental Protection Act (FEPA). The designated area covered about 2000 km^2 of coastal water and included all entering freshwater rivers and streams. Restrictions were removed in stages once monitoring showed that concentrations of hydrocarbons and PAH in edible tissues posed no further risk to consumers and that the relevant species were free of taint. Controls on fish, cockle beds and crustacea were removed respectively within 3, 7 and 8 months. The restrictions covering intertidal mussels in the south east area of the closure area and oysters within Milford Haven were removed relatively slowly and last, after 19 months. In contrast, some shellfisheries were closed for seven years following both the *Amoco Cadiz* and *Braer* spills.

References

Batten, S.D., Allen, R.J.S. and Wotton, C.O.M., 1998. The effects of the *Sea Empress* oil spill on the plankton of the southern Irish Sea. Marine Pollution Bulletin, 36(10), 764-774.

Dyrynda, E.A., Law, R.J., Dyrynda, P.E.J., Kelly, C.A., Pipe, R.K., Graham, K.L., and Ratcliffe, N.A., 1997. Modulation in cell-mediated immunity of *Mytilus edulis* following the '*Sea Empress*' oil spill. Journal of the Marine Biological Association of the United Kingdom, 77 281-284.

Environment Agency, 1997. *Sea Empress* environmental monitoring. Version II. Data to May 1997. Environment Agency, Welsh Region, Haverfordwest.

Glegg, G.A., Hickman, L. and Rowland, S.J., 1999. Contamination of limpets (*Patella vulgata*) following the *Sea Empress* oil spill. Marine Pollution Bulletin, 38(2), 119-125.

Hobbs, G., 1998. Macrobenthic monitoring in Milford Haven and the adjacent coastal waters following the *Sea Empress* oil spill of February 1996. Report to the Environment Agency from OPRU/CORDAH. Report No. OPRU/29/97.

Hobbs, G. and Morgan, C.I., 1992. A review of the current state of environmental knowledge of the Milford Haven Waterway. A report from the Field Studies Council Research Centre commissioned by the Milford Haven Waterway Environmental Monitoring Steering Group. pp. 140. FSC Report No. FSC/RC/5/92.

Lancaster, J.E., Pawson, M.G., Pickett, G.D. and Jennings, S., 1998. The impact of the '*Sea Empress*' oil spill on seabass recruitment. Marine Pollution Bulletin, 36(9), 677-688.

Levell, D., Hobbs, G., Smith, J. and Law, R.J., 1997. The effects of the *Sea Empress* oil spill on the sub-tidal macrobenthos of the Milford Haven Waterway: a comparison of the survey data from October 1993 and October 1996. Report to the Environment Agency from OPRU/CORDAH. Report No. OPRU/22/97.

Little, D.I., Howells, S.E., Abbiss, T.P. and Rostron, D., 1987. Some factors affecting the fate of estuarine sediment hydrocarbons and trace metals in Milford Haven. In: (Eds. P.J. Coughtrey, M.H. Martin & M.H. Unsworth) Pollutant Transport and Fate in Ecosystems. Blackwell, Oxford. pp. 55-87.

Little, D.I. and McLaren, P., 1989. Sediment and Contaminant transport in Milford Haven. In: (Ed. B. Dicks) Ecological Impact of the Petroleum Industry. John Wiley & Sons, London. pp203-234.

McLaren, P. and Little, D.I., 1987. The effects of sediment transport on contaminant dispersal: an example from Milford Haven. Marine Pollution Bulletin, 18(11), 586-594.

Nikitik, C., 2001. Patterns in the benthic populations in the Milford Haven Waterway following the '*Sea Empress*' Oil Spill. Environment Agency, Welsh Region, St. Mellons. Report No. EASE/REP/01/06.

Roberts, D.E., Jones, F.H., Wyatt, R.J. and Milner, N.J., 1998a. The impact of the *Sea Empress* oil spill on the abundance of juvenile migratory salmonids in West Wales. Environment Agency, Welsh Region, Llanelli. Report No. EA/M/10/A.

Roberts, D.E., Jones, F.H., Wyatt, R.J. and Milner, N.J., 1998b. The impact of the *Sea Empress* oil spill on the abundance of adult migratory salmonids in West Wales. Environment Agency, Welsh Region, Llanelli. Report No. EA/M/8.

Rostron, D.M., Little, D.I. and Howells, S.E., 1986. A study of the sediments and communities in Milford Haven, Wales. Oil & Chemical Pollution, 3, 131-166.

SEEEC, 1998. The environmental impact of the Sea Empress Oil Spill. Final report of the *Sea Empress* Environmental Evaluation Committee. pp. 135. Stationery Office, London.

18. LONG TERM TRENDS IN THE TYNE ESTUARY

Driving force

The Tyne estuary has been heavily modified, 85% of its intertidal area has been reclaimed, the lower reaches were canalised in the 1850s and are dredged. The industrial heritage is typical of the region, with shipbuilding, mining, and discharges of waste and crude sewage from the expanding population. Commercial decline, increasing regulation and better sewage treatment have substantially reduced the discharges. The estuary is now affected by activities such as riverine regulation, wastewater discharge and fishing. Persistent pollutants such as TBT and PCBs remain, raising issues of dredging and riverside development.

Pressure

After 1980, sewage discharges were amalgamated and treatment was improved. By 2000 almost all discharges were pumped to Howdon STW in the lower estuary for secondary treatment, with UV added in 2002. These steps resulted in a smaller load from only one point. Numerous consented industrial discharges remain, but Howdon STW represents the largest organic load.

There is a coastal net fishery for salmon, although it is impossible to determine the proportion of caught salmon originating in the Tyne. A net limitation order in 1992 reduced licence numbers, but catches fluctuated. Over 20 angling clubs and associations fish the river, mainly concentrating on game fishing.

State

The estuary is stratified to an extent that depends on tidal state, season and river flow but is most marked during neap tides and low river flows. With lower summer flows, salinity increases and the freshwater layer may be warmer than deeper saline water. Dissolved oxygen (DO) concentration is highest at the mouth and at the tidal limit, sagging in the middle-upper area. The surface (freshwater) DO is relatively high with a depression in the lower reaches because of the influence of the STW (Figure 67). In warm dry conditions fish have died: the summers of 1995 and 1996 were the worst. These deaths were usually localised to a small reach.

Sedimentary organic matter derived from terrestrial plant material, decaying marine phytoplankton and anthropogenic inputs such as sewage consumes oxygen from the water via the sediment oxygen

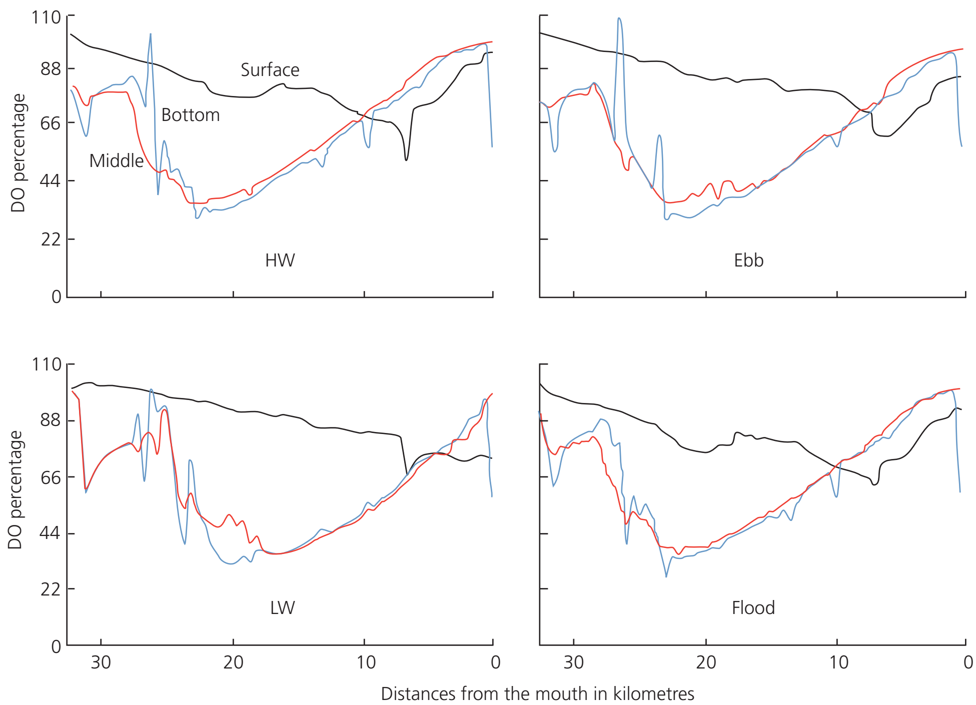


Figure 67. DO levels in August 1996

demand (SOD). In 1980-1982 the SOD was high, typically 0.5-0.9 $gO_2m^{-2}h^{-1}$ throughout the middle reaches. It dropped to <0.15 $gO_2m^{-2}h^{-1}$ in 1995, a significant improvement. The new STW reduced by half the settling organic matter, and the biochemical oxygen demand (BOD) of the effluent reduced by two thirds. In particular, secondary treatment at

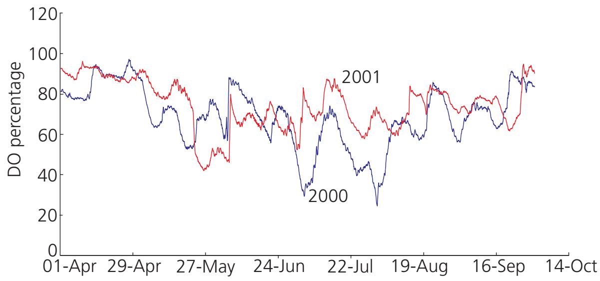


Figure 68. DO changes after installation of secondary treatment at Howdon STW in late 2000

Howdon STW in 2000 increased the DO (Figure 68). A VITOX system (NRA, 1996) operated in the mid-upper estuary in July and August of 2000 and had an impact upstream.

DO increased significantly in 2001 in the middle reaches at Newcastle, but changed little in the upper reaches. This may reflect the influence of SOD, and of VITOX oxygenation on DO in the upper estuary in 2000. STW effluent BOD would diminish during travel from Howdon to Newburn and is unlikely to have had an impact here. A decrease in estuarine SOD will eventually reduce the oxygen debt but has a time lag that cannot be predicted accurately. It is likely to be greatest in the first few years.

Impact

Benthos

Macrofaunal surveys in 1992 revealed organically enriched sediments, species poor and dominated by deposit feeding annelid worms. Between 1992 and 1995 the communities remained stable, with symptoms of enrichment. Community composition was variable between 1994 and 1998, with relative stability in 2000/1 (Figures 69(a) and (b)). Figures 70 and 71 show changes in taxa, individuals and diversity at three NMMP sites. The limited series for site NMMP1 in the upper estuary shows little change. The NMMP2 site upstream of Howdon STW shows a gradual increase in number of taxa and in diversity, with fluctuating numbers of individuals.

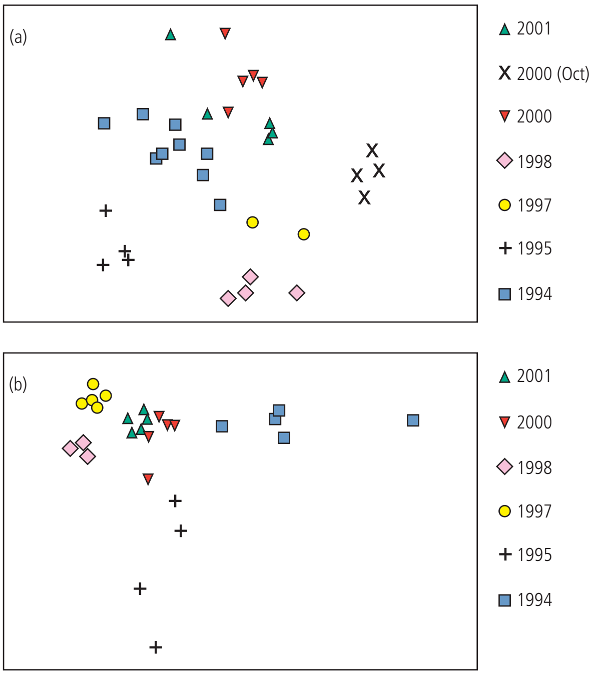


Figure 69. Numbers of macrobenthic invertebrate species and individuals at 2 sites (MDS plot). (a) Site NMMP2 in the middle estuary, (b) Site NMMP3 in the lower estuary

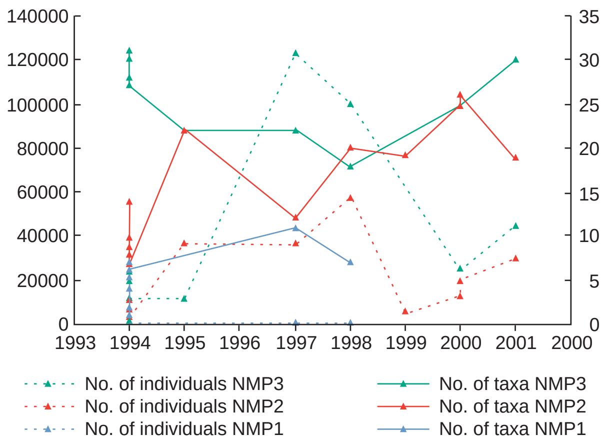


Figure 70. Number of macrobenthic invertebrate species and individuals at the 3 NMMP sites

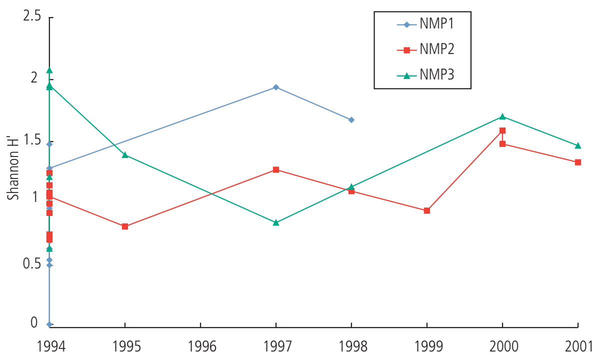


Figure 71. Diversity (Shannon H') of the macrobenthic communities at the 3 NMMP sites

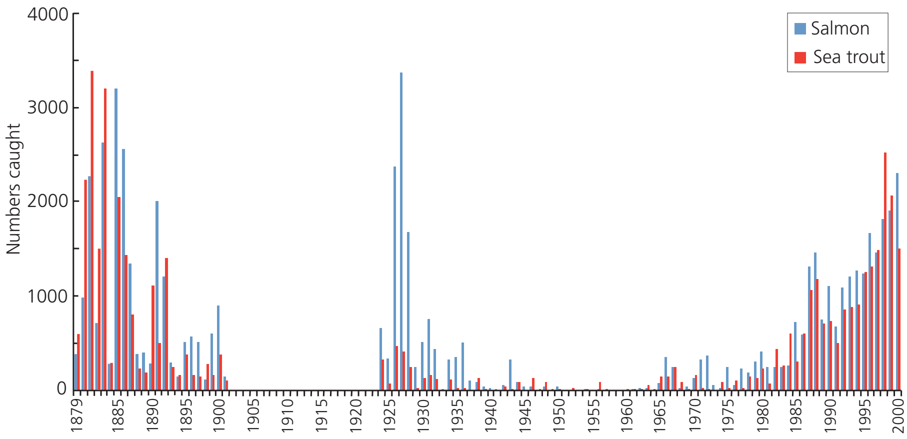


Figure 72. Declared rod catches of migratory salmon and sea trout in the Tyne catchment

NMMP3 in the lower estuary showed an insignificant increase in diversity, with a low 1997 value caused by increase in one very ephemeral and opportunistic species, *Capitella capitata*. Much variability here was associated with fluctuations in *C. capitata*, despite transformation of the data to account for this in the MDS plots of Figure 69. Decline in *Ophyrotrocha* spp. after 1994 and *Tubificoides* spp. after 1997 accounted for most of the remaining variability.

Fish

Numbers of returning salmonids have increased substantially, although in the early 1980s poor water quality still presented a partial barrier and in recent hot summers migratory fish have died in the upper reaches. Resident estuarine populations are not as diverse as similar regional estuaries.

Records of rod catches for over a century have been based on fishing effort (Figure 72) and indicate population trends. This century, salmon catches peaked in the late 1920s. The 1930s fall and persistent low catches until the late 1960s were attributed to the barrier of poor water quality. Since 1979 the population has been supported by introductions from the salmon hatchery (EA, 1998). Further fish were introduced in addition to the mitigation stocking and, more recently, to compensate for deaths in the summers of 1995 and 1996.

In the early 1970s the salmon and sea trout rod catches increased, more rapidly from 1985 to 2000, the highest since the 1920s. It is impossible to distinguish the effect of the hatchery. Many more salmon were released than sea trout, yet both catches indicated a similar increase: it is clear that the driving force has been the improvement in water quality.

Response

Annual average measurements of DO have increased but sustained low values still present problems especially in the urbanised stretches. As the SOD decreases and DO levels rise in the water, conditions are expected to improve. The decline in fish populations previously linked to poor water quality (Harwood *et al*., 1997) is expected to reverse as reduced SOD reduces the stress and allows a greater diversity.

References

Harwood, K.G., Gill, M.E. and Frid, C.L.J., 1997. Temporal changes in fish abundance, species diversity and the health of flounder, Platichthys flesus, in the Tyne Estuary: implications for water quality monitoring. Report for the Environment Agency, NE Region.

NRA, 1996. Tyne Estuary Investigation 1995: Report of the Tyne Estuary Group. National Rivers Authority, Northumbria Area, NE Region.

EA, 1998. Tyne Salmon Action Plan. Environment Agency, NE Region.

19. LONG TERM TRENDS IN THE TEES ESTUARY

Driving force

Large scale industry started in the Tees Estuary during the Industrial Revolution. Today, chemical, petrochemical and steel making industries play an important economic and social role in the area. As a consequence of industrialisation, patterns of human settlement were modified and important populations arose in the area. Today therefore, the estuary receives treated urban waste waters, nutrients of agricultural origin and industrial effluents. Industrial activity is likely to increase in the near future and pollution incidents occur frequently. The estuary is dredged regularly, with the risk of re-suspending contaminants. Nevertheless, biodiversity, aesthetics and recreational use must be preserved. The site qualifies as a Special Protection Area under the Habitats Directive (79/409/EEC) by supporting bird populations of European importance.

Pressure

The daily discharge from industrial processes and untreated domestic sewage reached more than 500 tonnes of Biochemical Oxygen Demand (BOD) in 1970. No dissolved oxygen was available for biological processes along many sections of the estuary and the lower stretch was virtually devoid of life.

Programmes to improve water quality and to encourage the return of migratory fish to the river started in 1980. They included the commissioning of sewerage schemes, sewage treatment works, a reed bed and a sulphuric acid treatment plant commissioned by ICI at their Billingham site. As a consequence of the improvements, the discharge loads in 1998 were less than a quarter of those of 1970 (Figure 73). Efforts focused on reducing BOD and ammonia inputs. Ammonia levels, though substantially reduced, remained high enough for concern (Figure 74).

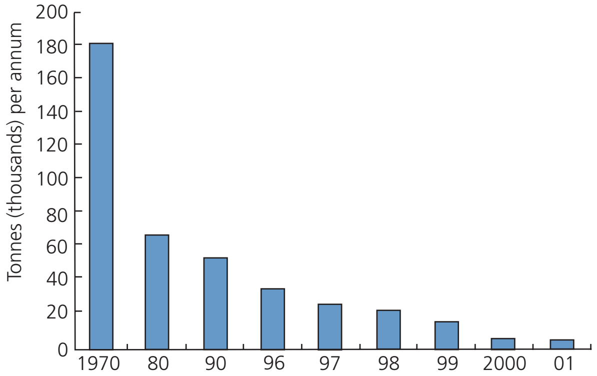


Figure 73. Biochemical Oxygen Demand (BOD) released to the estuary between 1970 and 2001

These improvements in water quality significantly affected the biological populations: diversity and abundance increased substantially from the late 1970s to 1997 (Gray, 1976).

The Tees Barrage reduced the estuarine extent in 1995 from 40 km to 17 km. It substantially reduced dynamic energy in the system and increased stratification. In consequence, oxygen input from the upper layers was reduced in some stretches, silt deposited upstream of the barrage, while downstream sediments coarsened. Salt-water residence times increased and the salt wedge has penetrated further up the estuary (Riddle, 1977; Tapp *et al.*, 1978; Riddle *et al.*, 1996; Lewis *et al.*, 1998).

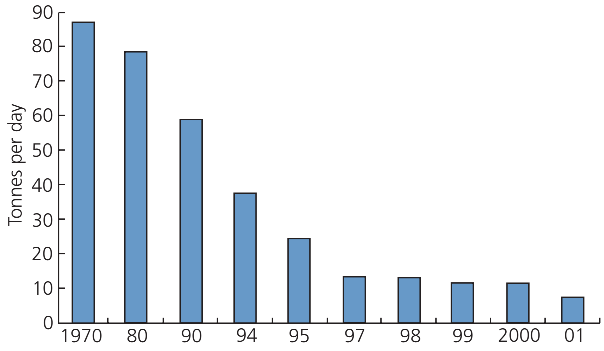


Figure 74. Average daily load of ammonia released to the estuary between 1970 and 2001

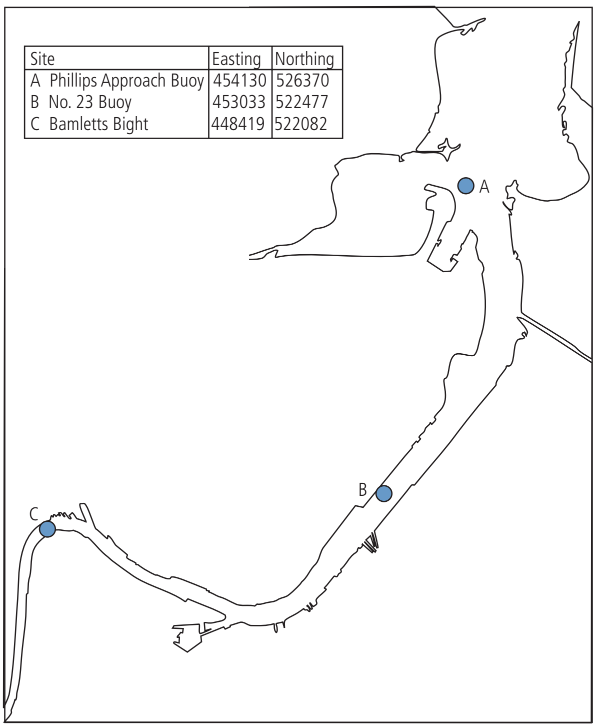


Figure 75. Sampling stations in the Tees estuary

State/Impact

Ammonia and BOD

Relocation of sewage discharges has had a major impact on water quality. Loads of organic material and associated BOD have decreased in the main channel, and dissolved oxygen levels remain mainly above 60% saturation (Figures 76 and 77). Ammonia is no threat for biota, apart from at Redcar Jetty, where concentrations of un-ionised ammonia have periodically exceeded their EQS of 21 µg l^{-1}.

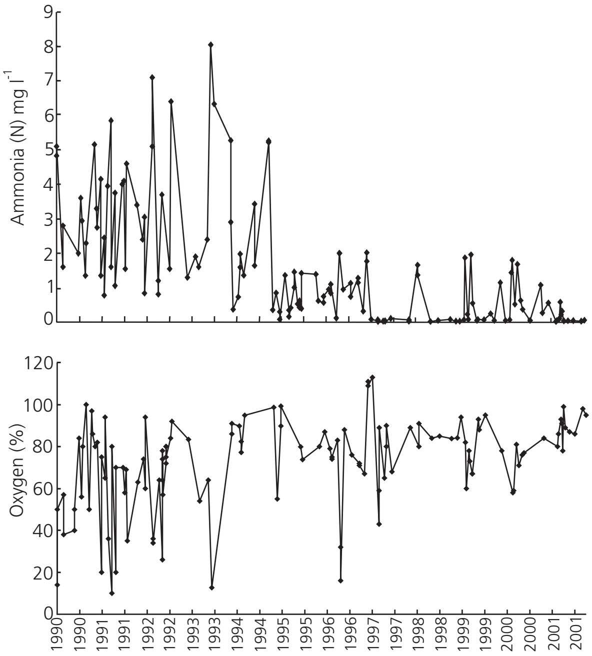


Figure 76. Ammonia and oxygen saturation, Phillips Buoy

Biological populations

Populations of migratory birds and infaunal invertebrates have declined on intertidal mudflats included within the Teesmouth and Cleveland Coast Special Protection Area since 1995 (Evans *et al.*, 2000). Coverage (extension and thickness) of filamentous green algae (*Enteromorpha* spp.) has also increased over the last few years. It is not known if these changes relate to water or sediment quality or to effects of the Barrage on water dynamics and sediment deposition patterns (Evans *et al.*, 2000).

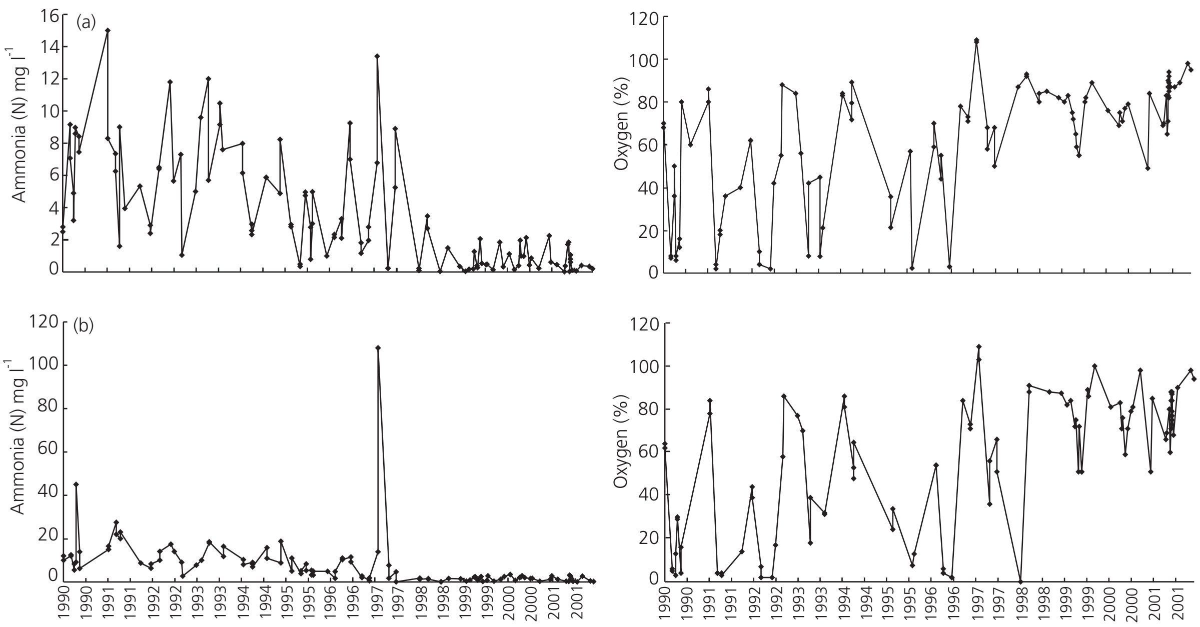


Figure 77. Ammonia and oxygen saturation in water column: (a) Phillips Buoy and (b) Buoy 23

Macrobenthos

Biological diversity of infaunal invertebrates improved steadily between 1979 and 1985. Diversity declined during 1986-1988, followed by a recovery in 1989 that continued to 1990. Water quality improvement has improved the benthic biology of the estuary (Shillabeer and Tapp, 1989, 1990; Tapp *et al.*, 1993).

Since 1992, subtidal invertebrate infaunal populations were sampled once a year at three sites (Figure 75). Total abundance of macrobenthic invertebrates, calculated on averaged replicates, at each site is graphically represented in Figures 78 and 79 Other univariate statistics - a Pielou's evenness (J) and Shannon-Wiener (H') - were calculated on combined replicates and are represented in Figure 80.

NMMP1 (Philip's Approach, Lower estuary)
Abundance declined in 1995-1996, due to reduction in opportunistic, pollution tolerant species. *Polydora ciliata* is a very resilient, opportunistic species, often found in environments with high levels of nutrients and sewage. Diversity also declined in 1995-1996, to recover by 1998 with increases in more specialised and less tolerant species.

NMMP2 (Buoy 23, Mid-estuary)
As at NMMP1, abundance declined in 1995-1996, mainly by falls in numbers of opportunistic-tolerant species (*Polydora ciliata*, *Capitella capitata*, *Tubificoides pseudogaster* and *Tubificoides benedii*). However, abundance remained high, averaging about 5000 individuals per sample. Diversity increased slightly but it was still higher than at NMMP1.

NMMP3 (Bamlett's Bight, upper estuary)
Abundance has declined at this site since 1996. Values were still relatively high, with the exception of those of 2000. Diversity increased since 1992 and to become higher than at NMMP1 and NMMP2.

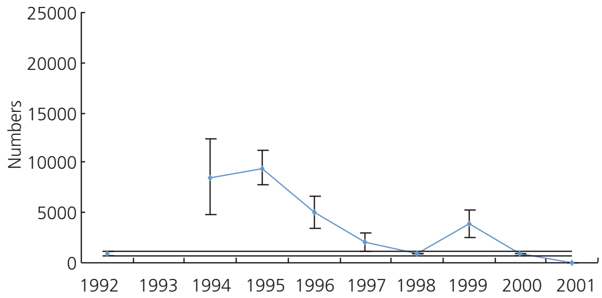


Figure 78. Abundance in NMMP1 (Philip's Approach)

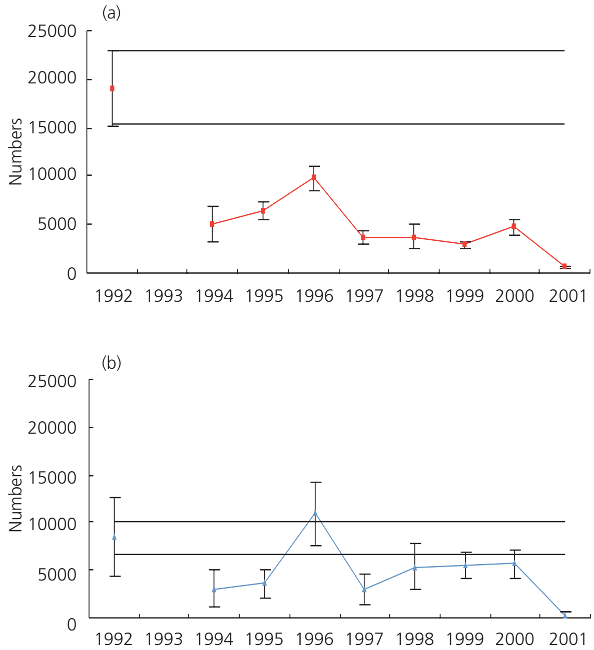


Figure 79. Abundance and diversity in (a) NMMP2 (Buoy 23) and (b) NMMP3 (Bamlett's Bight)

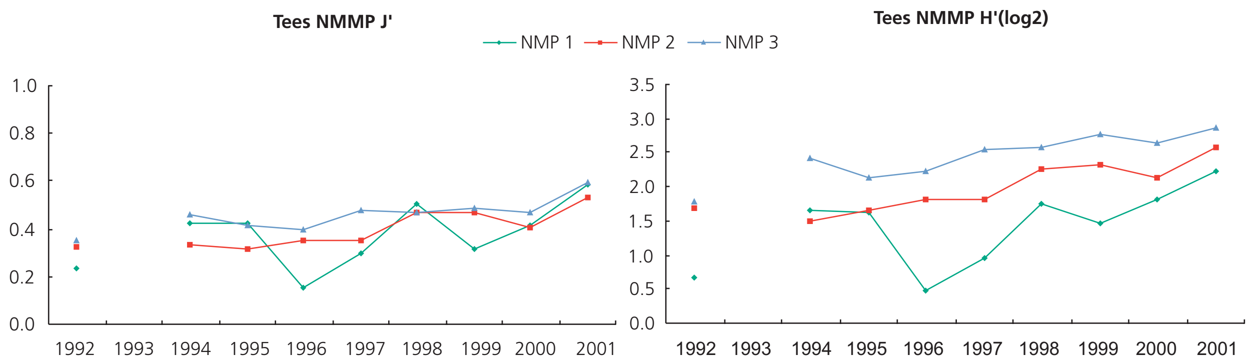


Figure 80. Shannon-Wiener and Pielou indices

Classification analysis

In view of the natural variability of salinity and temperature along the different sections of the estuary, the distribution (Figure 81) of samples in the graphical representation of Multi-dimensional Scaling (MDS) is typical of estuarine benthos.

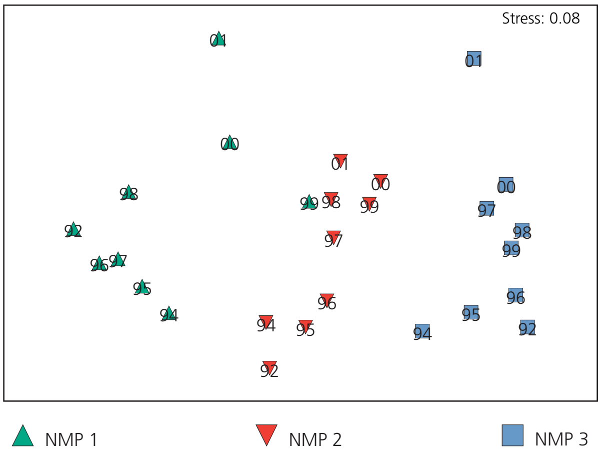


Figure 81. Tees macrobenthos 1992-2001: Multi-dimensional Scaling (MDS)

A trend may also be seen within the samples of each group. Recent samples form separate sub-groups within the MDS plot: samples taken in 2000 and 2001 are represented in the upper side of the plot, whereas the earliest ones (1992, 1994, 1995) are closer to the bottom. Therefore, the horizontal axis represents lower estuary to upper estuary geographical variation, while the vertical axis shows the temporal variation. This represents a change from communities with higher abundance and lesser diversity - a few pollution-tolerant and opportunistic species reaching high densities - to less abundant and more diverse communities, with some less tolerant species.

Response

The decline in abundance of subtidal macrobenthic populations in 1995-1996 corresponded to a reduction in abundance of species that occur in great numbers in polluted areas. Thereafter, a more diverse and relatively stable macrobenthic community was established, with an increase from 1998-1999 in less pollution-tolerant species. This change was most noticeable in the lower estuary (NMMP1), with species that tolerate slight pollution but decrease as pollution increases - like *Corophium volutator*, *Hydrobia ulvae* and *Cerastoderma edule (See also Warwick et al., 2002)*..

It is noteworthy that the polichaete *Euchone* sp. – one of the dominant species in the three sites over the last few years - is an alien, perhaps brought to the Tees with ballast water.

These changes from 1992 to 2001 may relate to general improvements in water quality over the last thirty years. However, the Tees Barrage changed estuarine circulation, current speed, sedimentation patterns and salinity. These are the most important environmental parameters to determine macrobenthic community type. They seem to have been particularly affected in the intertidal areas of the lower estuary and may explain others' observations of decline in intertidal macrobenthic populations and migratory birds.

The NMMP macrobenthic data suggest that, since 1979, the improving trend in biological diversity of subtidal infaunal invertebrates in the Tees estuary has continued between 1992 and 2001. This more diverse macrobenthic community, less dominated by opportunistic species and with some sensitive species, reflects the improvements in water quality and, possibly, some of the effects of the Tees Barrage.

References

Evans *et al.*, 2000. Preliminary Evaluation of the Results of the 1999 Monitoring Programmes related to the Nature Conservation Concerns arising from the River Tees Barrage and Crossing Act 1990.

Gray, J.S., 1976. The Fauna of the Polluted River Tees Estuary. Estuarine and Coastal Marine Science, 4, 653-676.

Lewis, R.E., Riddle, A.M. and Lewis, J.O., 1998. Effect of a tidal barrage on currents and density structure in the Tees estuary. Physics of Estuaries and Coastal Seas, Dronkers & Schefflers (eds). Balkema, Rotterdam, ISBN 90 54 10 965 3.

Riddle, A.M., 1977. Tees Bay Hydrography 1974-1976 and predictions of density circulation. ICI, Brixham Laboratory, BL/ B/1785

Riddle, A.M., Lewis, J.O. and Sharpe, A.D., 1996. Tees Estuary Survey: An assessment of water quality. Brixham Laboratory, Zeneca Ltd. BL/5650/ B.

Shillabeer, N. and Tapp, J.F., 1989. Improvements in the Benthic fauna of the Tees Estuary after a period of Reduced Pollution Loadings. Marine Pollution Bulletin Volume, 20, No. 3, pp 119-123

Shillabeer, N. and Tapp, J.F., 1990. Long Term Studies of the benthic biology of Tees Bay and the Tees Estuary. Hydrobiologia, 195: 63-78.

Tapp, J.F., Lewis, R.E., Taylor, D., 1978. Tees Bay: Hydrographical and Ecological Monitoring 1976-77. ICI, Brixham Laboratory. BL/B/1871.

Tapp, J., 1993. Continued observations of the benthic fauna of the industrialised Tees Estuary, 1979-1990. Journal of Experimental and Marine Biological Ecology, 172 67-80.

Warwick, R.M., Ashman, C.M., Brown, A.R., Clarke, K.R., Dowell, B., Hart, B., Lewis, R.E., Shillabeer, N., Sommerfield, P.J. and Tapp, J.F., 2002. Inter-annual changes in the biodiversity and community structure of the macrobenthos in Tees Bay and the Tees estuary, UK, associated with local and regional environmental events. Marine Ecology Progress Series, Vol. 234: 1-13.

20. RECOVERY OF THE THAMES ESTUARY

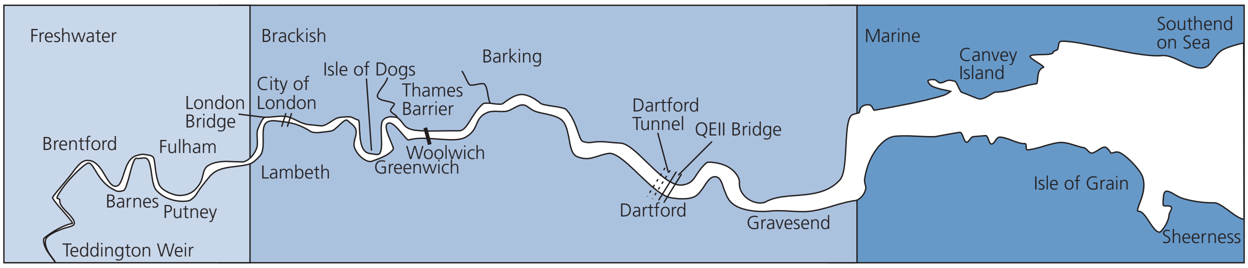


Figure 82. The Thames Estuary

Driving force

The Thames estuary extends from the upstream tidal limit at Teddington, through central London, to Shoeburyness and into the North Sea (Figure 82). The estuary receives wastewater from the large urban population of London and its industries. In 1949, the Thames Survey Committee identified the need for improvements to the main sewage treatment works, and a government study in 1951 concluded that the water quality should be improved to enable passage of migratory fish.

Pressure/State

In the nineteenth and twentieth centuries, major increases in industrialisation and urbanisation placed tremendous pressure on the water quality status of the estuary.

Prior to 1800, records show that the estuary supported a good diversity of fish species, including a number of commercial fisheries, indicating that water quality was good.

Industrialisation and urbanisation in the nineteenth and twentieth centuries increased the untreated wastewater discharge to the estuary. Development during the first half of the 20th Century raised polluting loads to the estuary dramatically. By the 1950s the combined effect of sewage and industrial effluents, thermal pollution from power stations and the introduction of non-biodegradable detergents placed tremendous pressure on the estuary so that it almost died for lack of oxygen in the water. Figure 83 illustrates the organic loading from the 1950s onwards. Between the 1950s and 1960s, the middle reaches were completely devoid of oxygen during much of the summer (Figure 84). Anaerobic conditions associated with release of hydrogen sulphide occurred frequently. In the late 1950s surveys found no established fish populations in a 70-kilometre length of river between Kew and Gravesend.

Improvements to sewage treatment, diversion of some industrial effluents, and introduction of biodegradable detergents reduced organic loading after the 1950s to the present (Figure 83). Anaerobic conditions were eliminated after completion of Crossness STW in 1964 and background dissolved oxygen levels continued to increase as Beckton STW provided full treatment. Further improvements were recorded as treatment was installed at tideway STWs. (Figure 84).

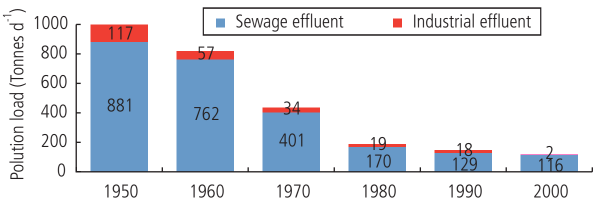


Figure 83. Temporal organic load to the Thames Estuary

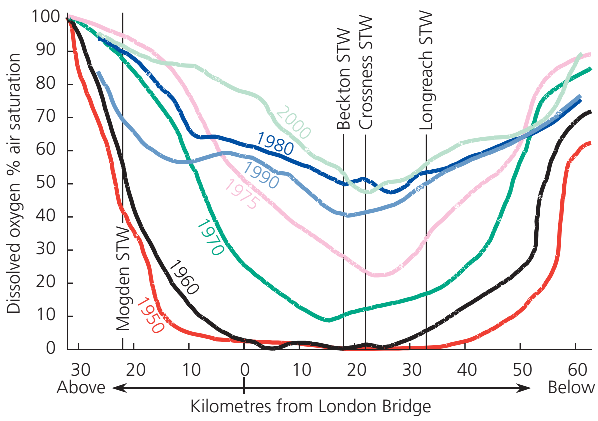


Figure 84. Temporal dissolved oxygen levels in the Thames Estuary

Impact

The serious deterioration in water quality in the nineteenth and first part of the twentieth centuries, and subsequent improvements since the 1950s have resulted in large-scale changes to the ecological status of the estuary, including its fish populations. Prior to 1800, there were substantial fisheries for a range of species across the whole estuary from Teddington to Southend and beyond. In the lower estuary below Gravesend, the major fisheries were for smelt, shrimp, whitebait, herring, flounder, sprat, sole, cockles, and oysters. Above that point there were large-scale fisheries for smelt, shad, whitebait, salmon, lamprey and flounder.

Most of the fish and associated fisheries above Gravesend disappeared by the late 1830s with the advent of sewage pollution, and the last salmon was recorded in 1833. By 1849, fish had disappeared from the London reaches of the Thames, and temporary improvements in water quality towards the end of the nineteenth century were followed by a further deterioration in the early part of the twentieth.

Fish began to appear on power station cooling water intake screens at West Thurrock and Tilbury after sewage treatment began in 1964. Early recovery was rapid with 56 species of fish by 1970 and 100 by 1980. A salmon rehabilitation scheme began in 1978, with returning adults appearing by 1983. A commercial eel fishery began below Tower Bridge in the same year. Up to 338 adult salmon have been recorded ascending the estuary in any one year. Today, the estuary supports 119 species of fish (Figure 85), representing the majority of available freshwater and marine species.

In the past two decades, the estuary has become one of the largest sole nurseries in England and Wales as well as the largest new bass nursery in the southern North Sea. Spawning species include the common goby, smelt, sand-smelt, dace and sole. These have recently been joined by the sea lamprey. River lamprey have begun to recolonize the catchment and twaite shad numbers built up steadily in the lower estuary. These last three species have important implications for the Habitats Directive. Apart from the eel fishery, there are commercial fisheries for bass and grey mullet below Woolwich and whiting, sprat, whitebait and shrimp below Gravesend. Recreational fisheries for dace, roach, perch and bream exist in the upper estuary down to Battersea and for sea fish such as bass, grey mullet, flounder, eel and sole below Woolwich.

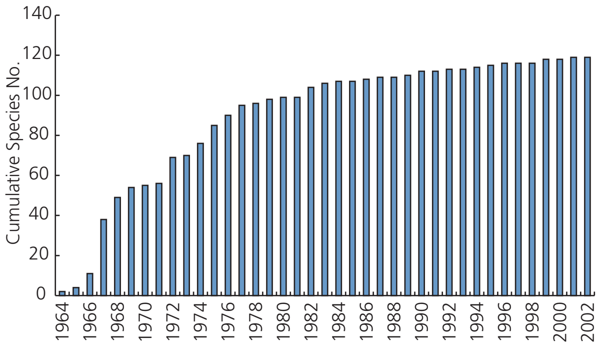


Figure 85. Temporal trend in the number of fish species found in the Thames Estuary

The improvements started in the early 1960s brought rapid results. However, the recovery continues in more subtle ways. The smelt is now rare and threatened in Europe, reflecting its nature as a fastidious species, needing good water quality and clean subtidal gravels just above the saline wedge to spawn on. Smelt spend most of the year in the lower estuary below Gravesend. In March and April, waves of spawning fish ascend to spawn just below the low tide mark in the Wandsworth area. The ova, initially attached to the bed, shear off after 48 hours and move downstream. Hatching may take up to 28 days. By the time the fry are able to breast the current at 14 days old, they will have been taken down to Millwall, Greenwich and beyond. The fry then employ selective tidal stream transport to move upstream on flooding tides. Passage through the narrow reaches in the city is particularly fraught because of the poor refuge afforded in the margins on the fast moving ebb tide. By mid June the fry may be distributed across most of the estuary above the city and by late autumn, young may be found across the whole estuary. Smelt have a short life cycle, maturing at three years of age, and living for a further year or two. This life strategy, in combination with demanding quality requirements, mean that the species is liable to large population crashes.

Response

The Thames estuary is now one of the cleanest metropolitan estuaries in Europe and, to sustain this, continuing pressures are monitored and managed interactively in an integrated way. An operating agreement between Thames Water and the Environment Agency is crucial to the management of dissolved oxygen levels in the summer. However, dissolved oxygen may still fall such as to kill fish after summer storms. To respond rapidly to poor water quality, the Agency receives real time data every 15 minutes from the critical reaches of the estuary.

The water quality management strategy for the estuary was reviewed in 1989, revealing a need for improved sewage treatment works performance in summer. It also highlighted the problem of acute de-oxygenation due to combined sewer overflows (CSOs), especially in heavy summer storms. These events may kill fish, despite the deployment of 2 oxygenation vessels and the use of hydrogen peroxide, and the Agency is now working closely with Thames Water Plc, DEFRA and the GLA to produce a new sustainable strategy for London's drainage.

21. LONG TERM TRENDS IN BELFAST LOUGH

Driving force

Belfast Lough is a shallow semi-enclosed marine bay on the east coast of Northern Ireland (Figure 86). At the head of the Lough lies the city of Belfast, which was the main centre of the industrial revolution in Ireland. In the nineteenth century Belfast became the world's largest linen producer and, between 1850 and 1900, a major industrial centre, hosting a large shipyard and ropeworks as well as whiskey distilling, tobacco rolling and service industries. In this period, many mudflats were claimed for industrial and port development, with over 1100 ha of estuarine intertidal area claimed since 1750 (Buck and Donaghy, 1996). During the 20th Century the Port of Belfast continued as a centre of industry with approximately 20,000 ship movements per annum. Other industries were ship building and repair, aircraft industry, timber treatment plants, a major fertilizer manufacturer, a power station and two major sewage treatment works. In recent years, industry declined, either reducing production or closing. In addition, tighter legislation from the EC through the Urban Waste Water Treatment Directive lead to the upgrading of major sewage works.

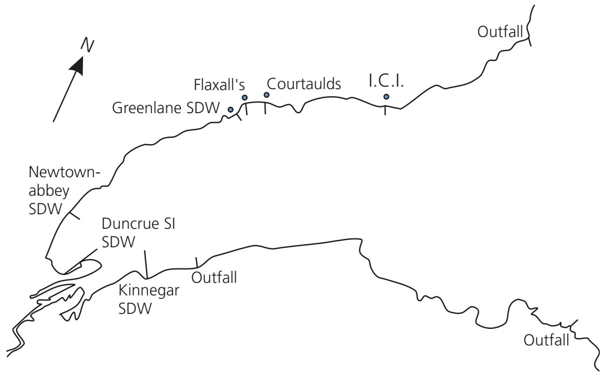


Figure 86. Belfast Lough in the late 1970s: heavy industry concentrated on the North Shore and Harbour Industry

Pressures

The catchment of Belfast Lough is highly urbanised with a total sewered population in the Lough and Lagan catchments of 750,000, approximately half of the total population of Northern Ireland (Charlesworth and Service 2000). The principal watercourse entering the Lough is the River Lagan, draining about 609 km² of agricultural land. The riverine input is small relative to the sea and therefore the Lough is fully saline for most of its length.

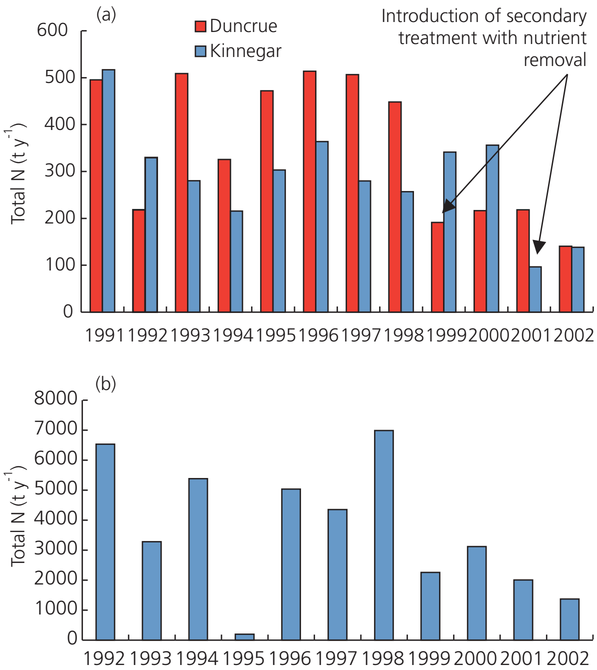


Figure 87. Total nitrogen loadings from major sources to Belfast Lough: (a) major sewage treatment works and (b) fertilizer manufacturer

Metal contamination

By the end of the 1990s, heavy industry had been largely reduced, as had heavy metal loads. For example, the annual loading of zinc in the mid seventies from one major textile plant was about 1095 t y^{-1} (DOE, 1974). By contrast, in 1992 the total annual loading from industry, sewage treatment works and rivers was estimated at 33 t y^{-1}.

Nutrients

The major nutrient inputs have been sewage treatment works and a fertilizer plant in the Inner Lough (Figure 87). In 1996, these sources supplied between 70% and 90% (it varies seasonally) of the total dissolved inorganic nitrogen loading to the Lough (Charlesworth and Service, 1999). After 1996 the effluent consent conditions for the fertilizer plant were tightened. Additionally, the two major sewage works discharging to the inner Lough were upgraded from primary treatment to full secondary treatment with nutrient removal (Figure 87). The inner Lough was designated as a sensitive area in 2000 under the Urban Waste Water Treatment Directive.

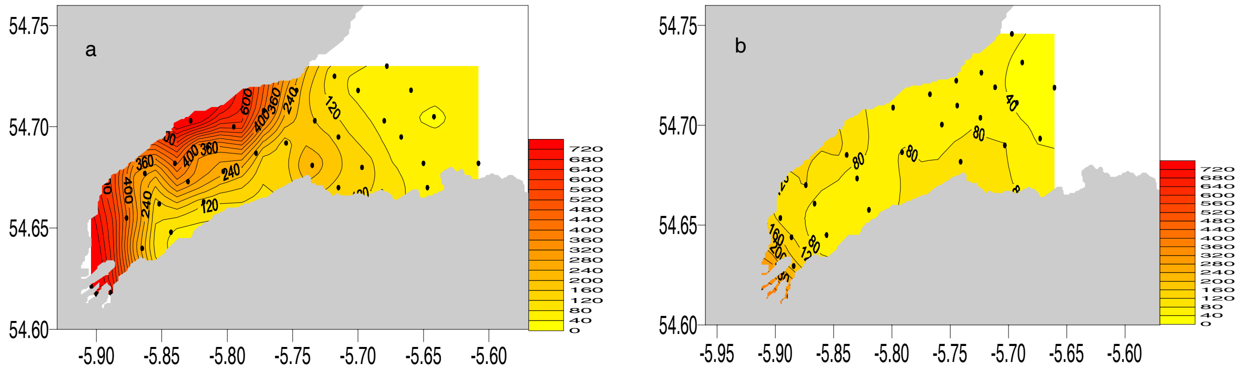


Figure 88. Zinc concentrations in surficial sediments in 1978 (re-analysed samples) (Parker, 1982) and 2000 (Gens, 2001)

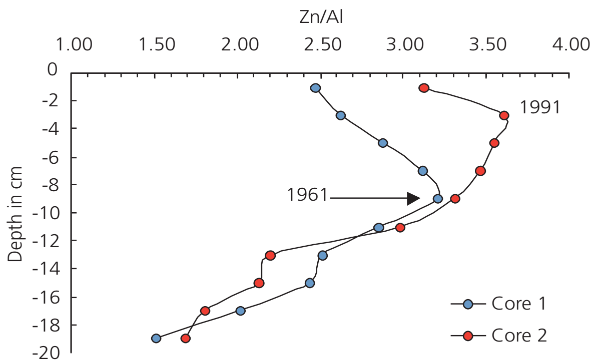


Figure 89. Zinc/aluminium ratios in a dated sediment core from Belfast Lough

State

Metal contamination

Reductions in heavy metal loading have been mirrored by a reduction in zinc in surficial sediments (Figure 88).

Cores showed the build up of sedimentary zinc concentrations from development of industry, with a sharp decline following closure of industrial plants after 1980 (Figure 89).

The reduction in zinc concentrations can also be seen in shellfish flesh (Figure 90).

Nutrient enrichment

Reductions in nutrient inputs have been mirrored by a decrease in winter nutrient concentrations both in the inner and outer Lough (Figure 91).

Impacts

Improvements in water quality have reflected in the ecology. Amphipods are sediment dwelling

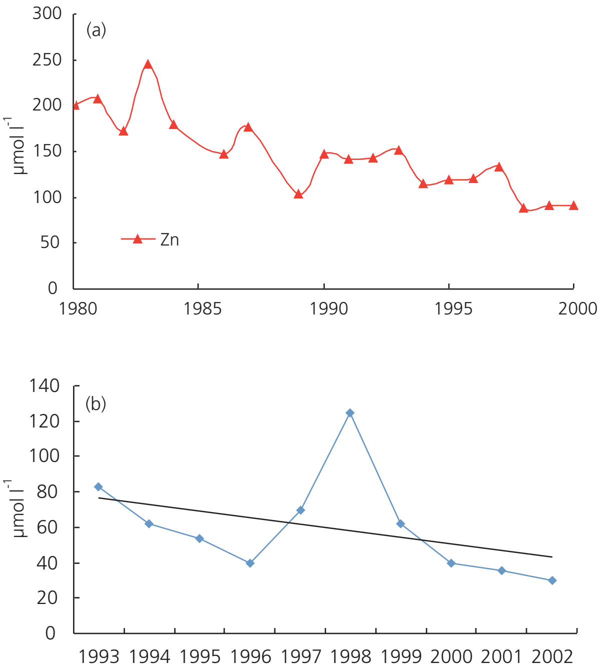


Figure 90. Zinc concentrations in shellfish (mg kg^{-1} dry weight) at (a) Bangor and (b) Belfast Lough

organisms among the most sensitive to pollution of all marine taxa (Rand and Petrocelli, 1985). A 1976 survey (Parker, 1980) showed that the amphipod and crustacean taxa increased significantly in abundance and diversity the greater the distance from Belfast and that specific species were typically absent from the polluted areas (Figure 92).

In 2001, 20 of the 42 sites surveyed in 1976 were compared. In 1976, five sites in the inner harbour sites and two sites on the North shore of the Lough were azoic (without any species). A similar picture could be seen from the wider infaunal community, clearly showing areas of severe impact with widespread impoverishment and modification of benthic community composition in Figure 93.

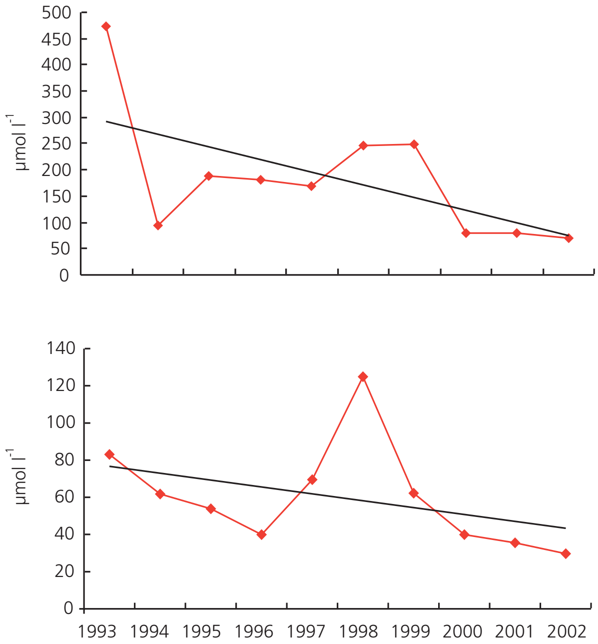


Figure 91. Mean winter dissolved inorganic nitrogen in Belfast Lough (a) Inner Lough - (BL2) and (b) Outer Lough (BL5)

In 2001, none of the sampled sites was azoic. The general picture was one of biological improvement. The inner harbour sites and North shore sites all supported a mixed fauna, including several amphipod species (*Perioculodes longimanus* and *Photis longicau*. Eleven of the sites showed an increase in diversity since 1976. Four sites showed no significant change. Five sites showed a decline in diversity, attributed to the dredging of the shipping channel at the time of the 2001 survey.

Response

Improvements in water quality are clear over the last 30 years. They have been reflected in both the sediment chemistry and biology. These improvements have owed partly to improved environmental protection, particularly in recent years when there has been clearer understanding of some of the environmental problems. However, some of the improvements also owe to the decline of industry in Northern Ireland over the last 30 years. Further improvements are necessary in some areas of the Lough and continued monitoring is essential to measure the effectiveness of management action.

References

Buck, A.L. and Donaghy, A., 1996. An Inventory of UK Estuaries – Volume 7, Northern Ireland. Joint Nature Conservation Committee.

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 for EHS.

Charlesworth, M. and Service, M., 2000. An assessment of metal contamination in Northern Irish coastal sediments. Biology and Environment: Proceedings of the Royal Irish Academy, Volume 100B, No.1, 1-12.

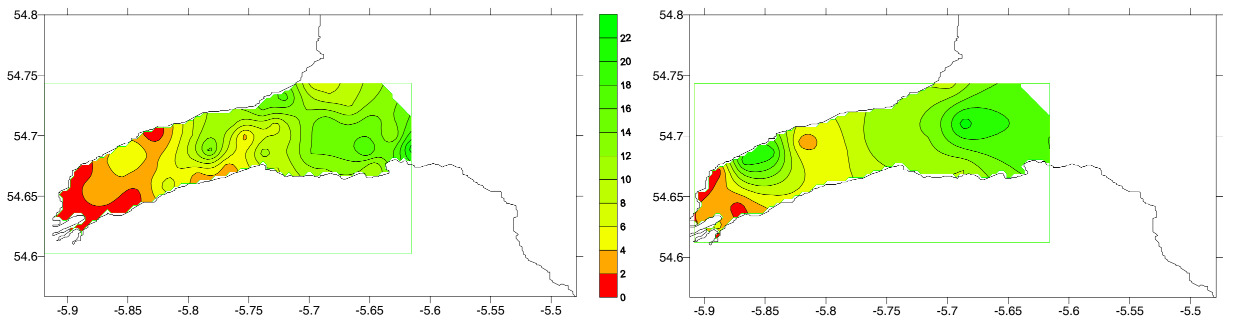


Figure 92. (a) 1976 Number of crustacean taxa (based on 42 sites) and (b) 2001 Number of crustacean taxa (based on 20 sites)

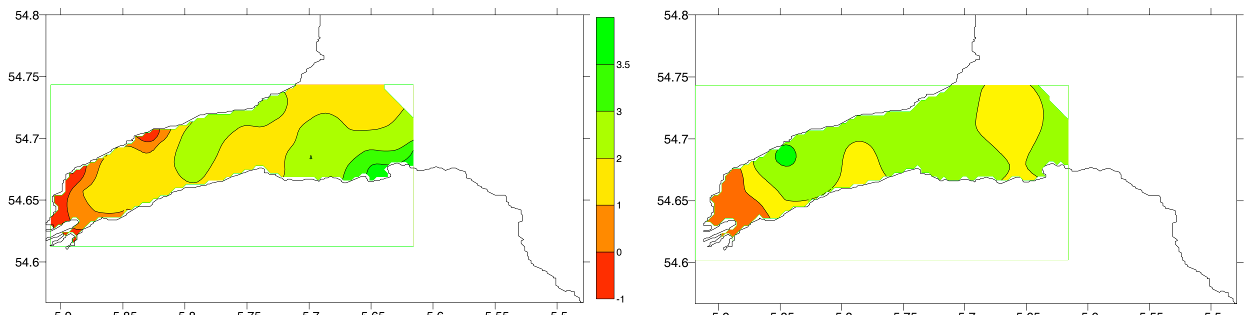


Figure 93. (a) 1976 Log_{10} crustacean abundance (Based on 42 sites) and (b) 2001 Log_{10} crustacean abundance (based on 20 sites) (Azoic sites plotted as –1)

DOE, 1974. Pollution of Belfast Lough – Interim Report. Northern Ireland Water Council.
Parker, J.G., 1982. Structure and chemistry of sediments in Belfast Lough, a semi-enclosed marine bay. Estuarine, Coastal and Shelf Science 15: 373-384.
Gens, S., 2001. A History of Metal Contamination in Belfast Lough. MSc Thesis Queen's University of Belfast.
Parker, J.G. 1980. Effects of pollution upon the benthos of Belfast Lough. Marine Pollution Bulletin. 11: 80-83.
Rand, G.M. and Petrocelli, S.R., 1985. Fundamentals of Aquatic Toxicology. Hemisphere, Washington DC.

22. MONITORING UK MARINE DREDGED MATERIAL DISPOSAL SITES

Driving force

Statutory control of the disposal of wastes to sea from ships is provided by the Food and Environment Protection Act (Great Britain - Parliament, 1985). This disposal route is now largely confined to dredged material arising from the maintenance of port/harbour facilities and their approach channels, and periodically from capital projects such as those involving channel deepening or new constructions. The effective management of the sea disposal option has strategic importance due to the continuing significance of maritime trade for the UK economy.

Through application of a screening process, applications for the disposal of excessively contaminated material are rejected at the licensing stage.

Monitoring of licensed disposal activity is conducted to ensure that:

- environmental conditions at newly designated sites are suitable for the commencement of disposal activities
- disposal operations conform with licence conditions
- predictions for established sites concerning limitations of effects are met

Monitoring outcomes contribute directly to the licensing/enforcement process, by ensuring that action is quickly taken if any evidence of unacceptable changes or practices is found.

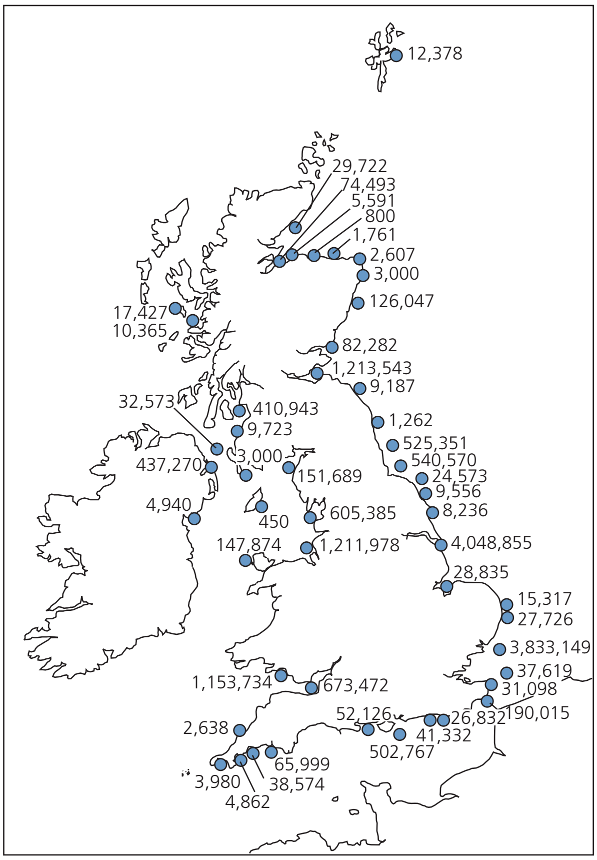


Figure 94. Amounts of dredged material deposited in tonnes wet weight in 2000. (Vivian, 2003), Closed circles represent individual sites or, where more than one is used locally, (e.g. in the Humber estuary) as an approximation to centres of activity

Pressure

Pressure arises from:

- the physico-chemical characteristics of the dredged material
- the amount and frequency of disposal
- the characteristics of the receiving area

There are over 150 sites around the UK designated for the disposal of dredged material, although not all of these are used in any one year (Figure 94). In total, approximately 40 million wet tonnes are disposed of annually. Quantities disposed of at individual sites range from a few hundred to several million tonnes. This material ranges from consolidated sediments, for example to extend harbours ('capital' works), boulders and crushed rock, through to finer material as a result of 'maintenance' dredging in estuaries.

Given the large number of disposal sites and significant variations in year-on-year usage, regular monitoring is targeted at representative sites. Other locations are also monitored where there is local concern over the potential for adverse effects of contamination, or where disposal may have implications for conservation interests.

State

Since it is the longer-term transport and fate of particulates that is of most concern and since sediments represent the eventual sink for most contaminants discharged to sea, monitoring is focused on seabed sediments and their associated fauna. Effects may arise from a combination of physico-chemical impacts and organic enrichment, which can complicate interpretation of the results (Rees *et al.*, 1991).

The diversity of approaches necessary for effective field evaluations at dredged material disposal sites are illustrated below through discussion of monitoring at three sites.

North Tyne and Souter Point Sites, off the Tyne

These sites are located at about 40 m depth, where the seabed is characterised by muddy sands (Figure 95). However, both sites have been modified by historical disposal of minewaste and fly ash from power stations, giving rise to local accumulations of coarser material (Rees and Rowlatt, 1994). Currently, the sites jointly receive about 0.5 m tonnes mainly of maintenance dredgings from the Tyne estuary each year, although these amounts vary over time due to the periodic nature of 'capital' works.

Elevated TBT concentrations in sediments were found in the area, with the highest concentrations confined to the disposal sites (Figure 95) and, in the case of Souter Point, at the western edge, indicating that disposal activity is concentrated there.

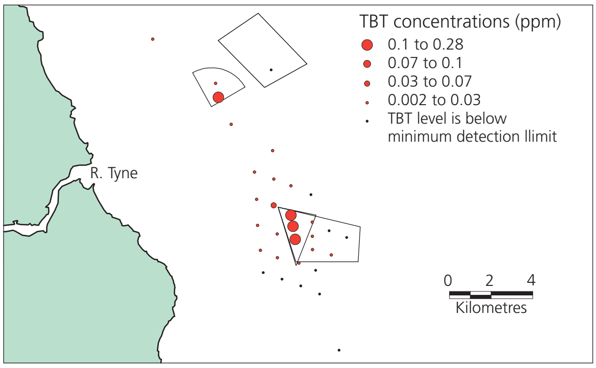


Figure 95. TBT content at the North Tyne and Souter Point dredge disposal sites off the Tyne in 1999

Site Z, in inner Liverpool Bay

This site is located at about 10 m depth in inner Liverpool Bay. Sediments are typically muddy sands which are periodically vulnerable to disturbance by wave action, and by tidal currents of up to 0.8 m s^{-1}. The site receives about 1.5 million wet tonnes of maintenance dredgings from the Mersey estuary each year, a considerable reduction over amounts in previous years (Vivian, 2003). Material disposed of ranges from sand to mud according to source, and may contain elevated levels of organic matter and trace metals (Rowlatt, 1988; Rees *et al.*, 1992; Rowlatt and Rees, 1993; Somerfield *et al.*, 1995). In the mid-1990s, there was evidence of shallowing at the centre of the site due to disposal practices.

Roughs Tower, off Harwich

Located at about 20 m depth, this site has been used for the disposal of dredged material principally from the ports at Harwich and Felixstowe and, until 1996, for the disposal of sewage sludge. The periodic disposal of large quantities of capital dredged material had previously given rise to concerns over shallowing of the site, while the dispersion of finer material was a concern for the local crustacean fishery. During 1999/2000, the site received 29 million tonnes of dredged material from Harwich. Although the site is naturally dispersive, a 'containment' approach was adopted in this instance, due to the nature and quantity of the material and the accompanying decision for its effective closure upon completion. Using a variety of survey techniques, conditions at the seabed were determined at intervals during and after disposal (Rees *et al.*, 2003).

Images obtained after cessation show that the boundary between the dredged material placement and the surrounding substrata was still intact.

Impacts

North Tyne/Souter Point, off Tyne

Earlier studies demonstrated significant localised changes in the benthic macrofauna, principally due to the physical consequences of the disposal of a combination of minewaste, fly ash and dredged material (Rees and Rowlatt, 1994). Recent studies of the effects of maintenance dredgings disposal at Souter Point identified localised modifications to nematode assemblages (Schratzberger *et al.*, 2004). Complementary laboratory microcosm studies indicated that TBT at concentrations comparable to those found at the site could be a contributory factor to observed changes.

Site Z, Liverpool Bay

Earlier surveys (e.g. Rees *et al.*, 1992) identified higher macrofaunal densities adjacent to the site in response to the disposal of dredged material (Figure 96), along with an expected reduction in most species at the centre. Species typically inhabiting estuary sediments were found, suggesting that these had survived the process of estuary dredging and disposal.

Meiofaunal studies at site Z and at other disposal sites (Somerfield et al., 1995; Boyd et al., 2000) identified species common to the site centres, even though they were widely separated and of different character. This was in contrast to the macrofaunal results where 'indicator' species, if found, were local to an area, and not universal in their response to dredged material (Anon., 1996).

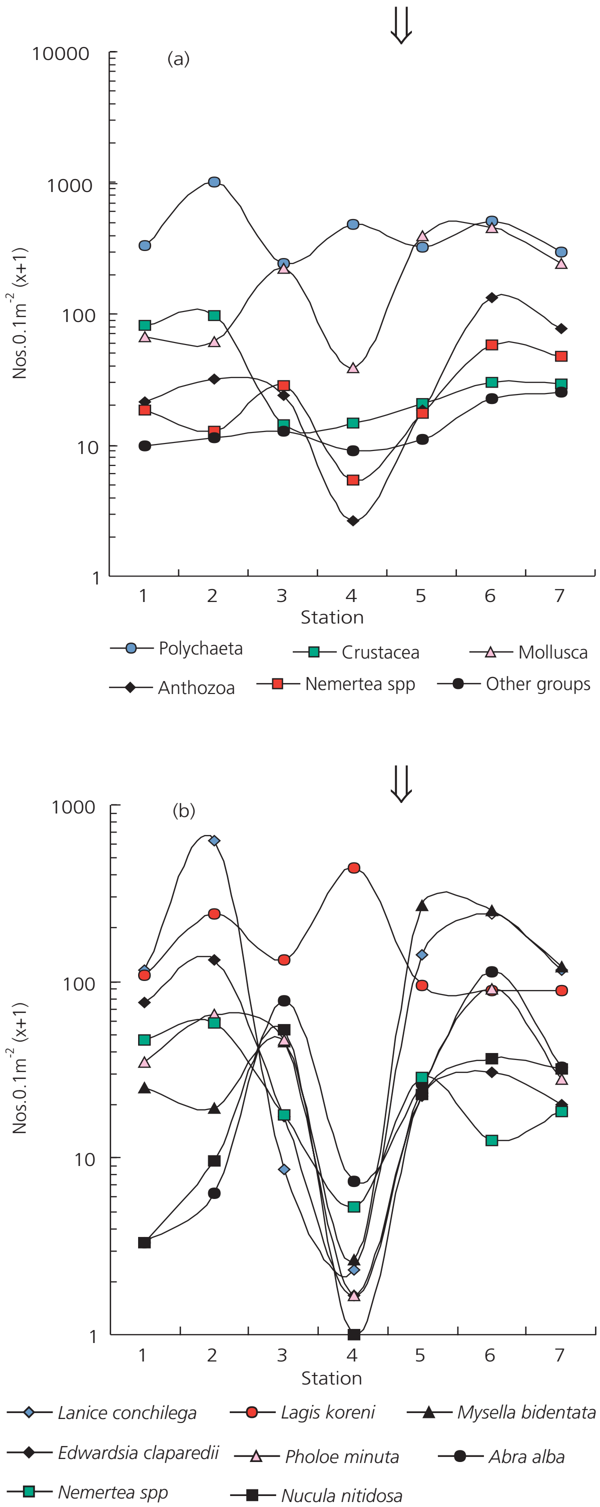


Figure 96. Densities of numerically dominant macrofaunal groups (a) and individual taxa (b) along a transect though site Z. Error bars are omitted for clarity. (Arrows indicate the location of the disposal site)

Roughs Tower, off Harwich

Monitoring results supported earlier predictions concerning containment of material, and limitation of ecological impacts to the near vicinity of disposal (Rees *et al*., 2003). Furthermore, there was evidence of significant amelioration of these impacts over time, including recolonisation by appreciable densities of juvenile crabs and lobsters.

Response

Prior to disposal, samples of dredged material are routinely screened for a range of contaminants. The range has increased in recent years in line with tighter regulatory controls, and now includes TBT originating from anti-fouling paints (Murray *et al*., 1999). Occasionally, TBT concentrations were sufficiently high (>0.1 mg kg^{-1}) to preclude sea disposal, for example, in the case of sediments near to dry-dock facilities. However, concerns remain over the potential for adverse effects arising from earlier disposal of TBT-contaminated sediments. Consequently, monitoring effort at the Tyne sites (where the highest concentrations were found) has been increased in recent years, and now includes a number of research projects aimed at improving knowledge regarding contaminant fate and effects.

The case studies revealed significant variation in ecological responses due to differences in the nature (and quantity) of deposited materials, and the nature of the receiving area. Thus, in the case of site 'Z', these were less negative than might have been anticipated. For example, there was no evidence of a widespread area of azoic sediments. This may be due to the stabilising or nutritional properties of deposited material, and might suggest a beneficial consequence through an increased food supply for fish. However, this may be offset by the risk of enhanced bioaccumulation of contaminants (Rees and Rowlatt, 1994). Therefore, controls on contaminant concentrations at source continue to be rigorously enforced. Recent concerns over shallowing at the centre of the disposal site have been addressed by extending the boundaries to the west and, to date, this strategy appears to have alleviated the problem.

Monitoring at Roughs Tower demonstrated the utility of sidescan sonar techniques as a complement to conventional sampling. An ongoing study is addressing the progress towards recovery of the benthic community (Rees *et al*., 2003). Evidence to date suggests that the adopted 'containment' strategy, accompanied by the effective closure of the site, has been successful.

Common elements to observed environmental responses to the disposal of dredged material allow hypotheses for changes to be erected prior to new or altered disposal regimes (Anon, 1996). However, monitoring the local marine environment remains essential to test for the effectiveness of controls on dredgings disposal.

While monitoring has succeeded in identifying local problems, thereby contributing to their subsequent resolution, findings at the majority of sites indicate that the option of sea disposal of dredged material is acceptable, subject to continued oversight of the activity.

References

ANON., 1996. Monitoring and assessment of the marine benthos at UK dredged material disposal sites. Scottish Fisheries Information Pamphlet, No. 21, 35pp.

Boyd, S.E., Rees, H.L. and Richardson, C.A., 2000. Nematodes as sensitive indicators of change at dredged material disposal sites. Estuarine and Coastal Shelf Science, 51: 805-819.

Great Britain - Parliament, 1985. Food and Environment Protection Act, 1985. Chapter 48. London: HMSO.

Murray, L.A., Waldock, R., Reed, J. and Jones, B., 1999. Sediment quality in dredged material disposed to sea from England and Wales. In: Proceedings of CATS 4 (Characterisation And Treatment of Sediments), Antwerp: September 15 – 17.

Rees, H.L., Heip, C., Vincx, M. and Parker, M.M., 1991. Benthic communities: use in monitoring point-source discharges. ICES Techniques in Marine Environmental Sciences, No. 16, 70pp.

Rees, H.L., Rowlatt, S.M., Limpenny, D.S., Rees, E.I.S. and Rolfe, M.S., 1992. Benthic studies at dredged material disposal sites in Liverpool Bay. Aquatic Environment Monitoring Report, MAFF Directorate of Fisheries Research, Lowestoft, No. 28, 21pp.

Rees, H.L. and Rowlatt, S.M., 1994. Studies at solid waste and dredged material disposal sites. In: Franklin, A. and Jones, J. (compilers), Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1992. Aquatic Environment Monitoring Report, MAFF Directorate of Fisheries Research, Lowestoft, No. 40, 52-67.

Rees, H.L., Murray, L.A., Waldock, R., Bolam, S.G., Limpenny, D.S. and Mason, C.E., 2003. Dredged materialfrom port developments: a case study of options for effective environmental management. In: Proceedings, 28th International Conference on Coastal Engineering. World Scientific Inc., 3616-3629.

Rowlatt, S.M., 1988. Metal contamination in sediments from Liverpool docks and the Mersey estuary. ICES C.M. 1988/E:12, 10pp (mimeo).

Rowlatt, S. and Rees, H., 1993. Studies at the Liverpool Bay site Z dredged material disposal site. In: Franklin, A. and Jones, J. (compilers), Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1991. Aquatic Environment Monitoring Report, MAFF Directorate of Fisheries Research, Lowestoft, No. 36, 59-61.

Schratzberger M., Wall C.M., Reynolds W.J., Reed J. and Waldock M.J., 2002. Effects of paint-derived tributyltin (TBT) on structure of estuarine nematode assemblages in experimental microcosms. Journal of Experimental Marine Biolology and Ecology, 272: 217-235

Somerfield, P.J., Rees, H.L. and Warwick, R.M., 1995. Interrelationships in community structure between shallow-water marine meiofauna and macrofauna in relation to dredgings disposal. Marine Ecology Progress Series, 127: 103-112.

Vivian, C., 2003. Licensing of deposits in the sea. In: Irish, R. (compiler), Monitoring of the quality of the marine environment, 1999-2000. Science Series, Aquatic Environment Monitoring Report, CEFAS, Lowestoft, No. 54, 74-80.

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