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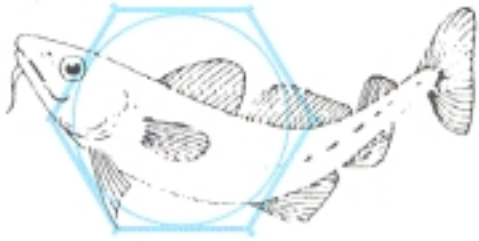

Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment and Activities Regulating the Disposal of Wastes at Sea, 1991

MAFF

Directorate of Fisheries Research

Lowestoft, 1993

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD
DIRECTORATE OF FISHERIES RESEARCH

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LOWESTOFT
1993

This report has been compiled by A. Franklin, B.Sc. and J. Jones of the MAFF Fisheries Laboratory, Burnham-on-Crouch, Essex, CM0 8HA, from whom further copies can be obtained.

Staff responsible for the projects described in this report are listed in Appendix 1.

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FOREWORD

In 1990, MAFF published the first of its reports on monitoring of non-radioactive contaminants under its new title 'Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment'. That report related to data collected in the years 1984-1987 and the intention of the new series was to make available to interested parties, on a regular basis, in a single report, information on all major on-going projects undertaken in this area of monitoring work. The report parallels that on monitoring of radioactivity in the aquatic environment, which is also published by MAFF in its Aquatic Environment Monitoring Report Series. The first report was followed in 1991 with a second report which covered work conducted in 1988 and 1989. The intention was noted in that report that future publication would be on an annual basis. Accordingly, the third report covered work carried out in 1990. This, the fourth report in this series, covers work carried out in 1991.

As with the second and third reports in the series, in addition to a general coverage of monitoring programmes, this report includes information, previously published separately, on MAFF's activities in the field of licensing of sea disposal carried out under Part II of the Food and Environment Protection Act 1985 (FEPA II). This is presented as Section 19 of the present report. Section 20 presents information on work carried out by MAFF in pursuit of its responsibilities as a statutory consultee in the consenting of pipeline discharges under the Water Act 1989.

Part of the work for Section 19 was carried out by Headquarters staff and Sea Fisheries Inspectorate staff; the remaining work was all undertaken by staff of the Directorate of Fisheries Research (DFR) Aquatic Environment Protection Division 2, at Burnham-on-Crouch.

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BACKGROUND TO THE WORK

The Fisheries Laboratory at Burnham-on-Crouch forms a Section of the Aquatic Environment Protection Division of the Directorate of Fisheries Research within the Ministry of Agriculture, Fisheries and Food (MAFF). The Laboratory is responsible for most aspects of investigation into non-radioactive contamination of the marine environment. This responsibility stems from MAFF's involvement in fish stock management. As an essential adjunct to the management of stocks *per se* it is clearly necessary, firstly to establish that the product, fish or shellfish, is safe to eat, and secondly to ensure that the viability of the stocks is not being affected by pollution.

Stemming from its interests in the well-being of fish and shellfish stocks, MAFF has a number of specific monitoring responsibilities under various Acts of Parliament. The most direct of these is a duty to license and control the disposal of land-generated wastes from ships at sea. The statutory requirements for this are laid down in Part II of the Food and Environment Protection Act 1985 (Great Britain - Parliament, 1985(a)). Among the duties specified by this Act is a requirement to monitor the quality of disposal sites to ensure that the disposal activity is not having an unacceptable impact on the marine environment of that location. This work also enables the UK to demonstrate compliance with the requirements of the Oslo and London Dumping Conventions (Great Britain - Parliament, 1972(a,b)) which regulate disposal of wastes to sea at an international level. Some examples of these monitoring investigations are described in this report.

Less direct responsibilities stem from the Water Resources Act 1991 (Great Britain - Parliament, 1991) and the various Acts and Regulations relating to the conduct of the offshore oil and gas industry and the shipping industry. In order to service its own direct needs and those of other Government Departments in relation to these Acts, the laboratory conducts a programme of more general environmental quality monitoring. This programme also produces valuable background information on the quality of the marine environment, against which the quality of disposal sites can be assessed. Such studies cover sea water and sediments as well as marine organisms, and a wide range of chemical determinands are measured each year in a very wide range of samples. A large proportion of this information is collected in accordance with procedures agreed under the auspices of organisations such as the International Council for the Exploration of the Sea, the Joint Monitoring Group of the Oslo and Paris Commissions and the North Sea Task Force. As such, they contribute to the database on the status of the marine environment around north-western Europe and much of the data referred to in this report is for example being used in the Quality Status Report currently being finalised by the North Sea Task Force.

In relation to particular measures of control, the Burnham Laboratory undertakes monitoring to demonstrate compliance with the European Community's Directive on mercury discharges, and similar requirements under the Paris Commission. It also undertakes work to assess the effectiveness of measures taken, under Part III of the Food and Environment Protection Act (Great Britain - Parliament, 1985(a)), to regulate the use of anti-fouling paints containing tributyltin (TBT).

1. MONITORING OF CONTAMINANTS IN MARINE FISH AND SHELLFISH

1.1 Introduction

In 1990, a survey of contaminants in fish from offshore sites in the North Sea was carried out to supplement an earlier baseline survey for the Joint Monitoring Group (JMG) of the Oslo and Paris Commissions (OSPARCOM). The results of this offshore survey were presented in an earlier Report in this series (MAFF, 1992(a)). At that time no data were available for the area to the north and east of the Humber, as the rough nature of the seabed makes it difficult to fish. The survey was continued in 1991 and efforts were made to improve the coverage of this area; a number of samples of dab (*Limanda limanda*) were obtained and analysed for heavy metals.

Samples of mussels (*Mytilus edulis*), from along the east coast of England, were obtained by staff from the Plymouth Marine Laboratory during 1990, for use in biological effects studies (scope for growth). These samples were analysed for contaminants by the Burnham-on-Crouch Laboratory. The results for heavy metals were included in MAFF 1992(a), the results for organochlorine pesticides and PCBs are now available and are included here.

In 1991, a number of *ad hoc* samples of cockles (*Cerastoderma edule*) were obtained from around the coast of England and Wales and analysed for heavy metal contamination, to provide baseline information for the forthcoming Shellfish Hygiene Directive.

Regular monitoring of metal levels in oysters (*Ostrea edulis*) from the River Fal in Cornwall commenced in late 1991, prior to anticipated possible contamination from discharges from the abandoned Wheal Jane mine. The results from the first year of monitoring are reported here.

Locations of all fish and shellfish sampling sites are shown in Figure 1.

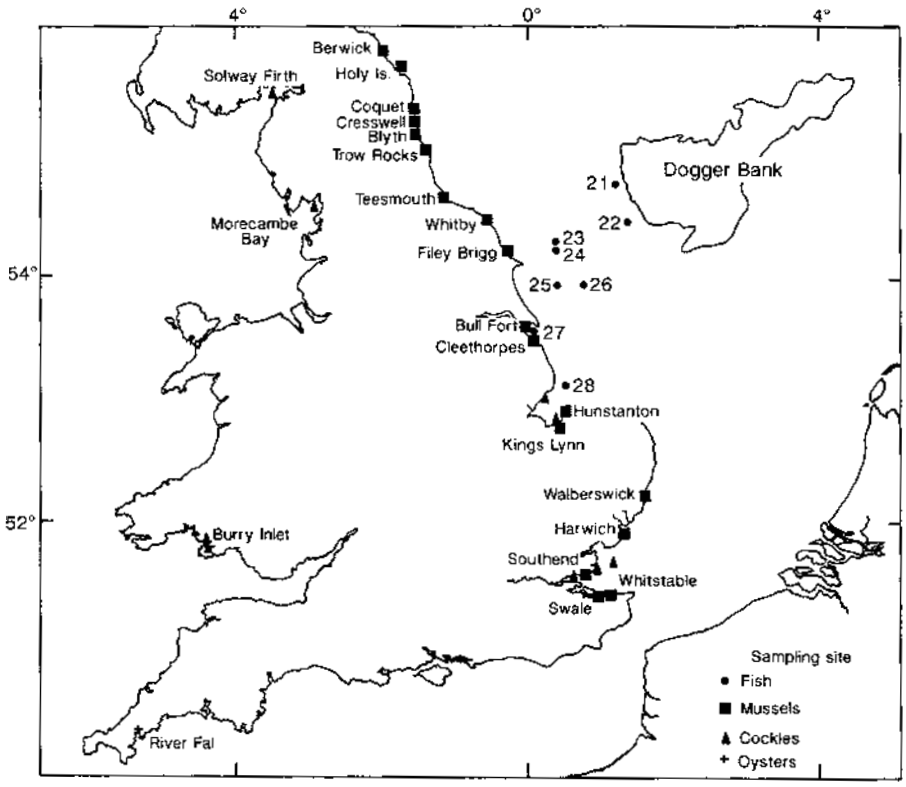

Figure 1. Locations of fish and shellfish sampling sites

1.2 Methods

1.2.1 Sampling

For the JMG work, samples of 25 dab and 50 mussels were obtained where possible within the length ranges recommended for these species by JMG – 20-30 cm for dab and 3-5 cm for mussels. Samples of cockles and oysters consisted of 50 and 10 individuals respectively, wherever possible.

Duplicate analyses were carried out on bulked tissue. Both dab muscle and liver were analysed for metals. Whole soft shellfish body tissue was analysed - for organochlorine pesticides in mussels and metals in cockles and oysters.

1.2.2 Analysis

Concentrations of copper, zinc, cadmium and lead were determined by conventional flame atomic absorption spectrophotometry (AAS) of a solution prepared from a nitric acid digest. Mercury was determined by an atomic fluorescence spectrophotometry technique.

Analyses were made of the following organochlorine residues: HCB, α-HCH, γ-HCH, dieldrin, ppDDE, ppTDE, ppDDT and PCBs. Following n-hexane soxhlet extraction of the fish and shellfish tissues, residues were determined by capillary gas chromotography, using electron capture detection after alumina and silica column 'clean up' and separation. For consistency with the earlier surveys, PCBs have been quantified using the formulation Aroclor 1254 as the standard; it is planned to reanalyse some of the samples to provide data on individual chlorinated biphenyls, commencing with the seven on an International Council for the Exploration of the Seas (ICES) Primary list (ICES, 1986). Full details of the methods employed by the Burnham-on-Crouch Laboratory are given in Harper *et al*. (1989) for metals and Allchin *et al*. (1989) for organochlorine pesticides and PCBs.

1.3 Results

The concentrations of heavy metals found in dab tissue are listed in Table 1, organochlorine pesticides in mussels in Table 2 and heavy metals in cockles and oysters in Tables 3 and 4 respectively. All concentrations are expressed on a wet weight basis; where metals analysis was carried out, the percentage of dry matter has been included to allow conversion of the data to a dry weight basis. In Table 1, for consistency with the 1990 work, results from the duplicate bulked analysis are given, in addition to the mean concentration, in brackets underneath.

Table 1. Concentrations of metals in dab (expressed as the mean, with the results from duplicate bulked analysis shown in brackets underneath)

Station / Area	Date of capture	No. of fish	Mean length (cm)	Tissue	Concentration (mg kg^{-1} wet weight) Hg	Cu	Zn	Cd	Pb	Dry matter (%)
21 (West edge of Dogger) 20	Oct-91	25	22.6	M	0.08 (0.08, 0.07)	0.18 (0.19, 0.16)	3.8, 3.6)	3.7	-	-
				L	0.09 (0.09, 0.09)	6.8 (6.8, 6.8)	28 (28, 28)	0.42 (0.44, 0.40)	<0.6 (<0.6, <0.6)	34
22 (Between Dogger and Flamborough)	Oct-91	25	22.9	M	0.1 (0.10, 0.09)	0.22 (0.29, 0.15)	3.3 (3.4, 3.2)	-	-	19
				L	0.10 (0.10, 0.10)	6.1 (5.8, 6.4)	29 (27, 30)	0.35 (0.34, 0.35)	<0.6 (<0.6, <0.6)	37
23 (Off Flamborough Head)	Apr-91	25	21.4	M	0.10 (0.09, 0.10)	0.31 (0.30, 0.32)	4.9 (4.8, 5)	-	-	18
				L	0.13 (0.11, 0.15)	6.0 (5.9, 6.1)	30 (29, 30)	0.66 (0.64, 0.68)	<1.0 + (<0.9, <1.0)	24
24 (NE Flamborough)	Oct-91	25	25.2	M	0.09 (0.09, 0.08)	0.23 (0.21, 0.25)	3.3 (3.3, 3.3)	-	-	19
				L	0.14 (0.14, 0.14)	9.8 (9.7, 9.8)	29 (30, 27)	0.37 (0.35, 0.38)	<0.6 (<0.6, <0.6)	36
25 (NE Humber)	Oct-91	25	23.7	M	0.12 (0.12, 0.12)	0.17 (0.16, 0.18)	3.5 (3.5, 3.5)	-	-	19
				L	0.15 (0.15, 0.14)	5.2 (5.3, 5.1)	25 (26, 24)	0.52 (0.53, 0.50)	<0.6 (<0.6, <0.6)	39
26 (NE of Humber)	Apr-91	25	23.1	M	0.12 (0.12, 0.11)	0.20 (0.20, 0.19)	4.4 (4.3, 4.4)	-	-	18
				L	0.10 (0.09, 0.10)	3.8 (3.8, 3.7)	29 (29, 29)	0.96 (0.92, 1.0)	<0.6 (<0.6, <0.6)	26
27 (Humber)	Feb-91	6	26.6	M	0.17 *(0.05-0.33)	0.23 (0.22, 0.23)	4.1 (4.0, 4.1)	-	-	17
23				L	0.04 (0.03, 0.04)	2.7 (2.6, 2.7)	29 (28, 29)	0.26 (0.26, 0.26)	<0.8 + (<0.7, <0.8)	
27 (Humber)	Jun-92	18	23.5	M	0.14 (0.14, 0.13)	-	-	-	-	23
				L	0.12 (0.11, 0.13)	5.1 (4.5, 5.7)	21 (21, 21)	0.20 (0.19, 0.21)	<1.7+ (<1.7, <1.6)	34

* = *range given, individuals analysed*
\+ = *higher detection limit due to small amount of tissue available*

M = muscle
L = liver

Table 2. Inshore 'organics' mussel samples collected in July 1990

Area of Capture	Number of mussels	Mean length (cm)	Concentration (mg kg^{-1} wet weight)								Lipid (%)
			HCB	α-HCH	γ -HCH	Dieldrin	ppDDE	ppTDE	ppDDT	PCB	
Berwick	50	4.1	<0.001	<0.001	<0.001	0.002	0.002	<0.001	<0.001	<0.001	1
Holy Island	50	3.9	<0.001	<0.001	<0.001	0.001	nr	<0.001	<0.001	<0.001	1
Coquet Estuary	50	4.3	nr	<0.001	<0.001	0.001	0.001	<0.001	<0.001	<0.001	2
Cresswell	50	3.8	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.005	2
Blyth	50	4.0	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.032	1
Trow Rocks	50	3.9	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	0.004	2
Tees Mouth	50	3.8	0.001	<0.001	<0.001	0.001	0.003	<0.001	<0.001	0.006	2
Whitby	50	3.9	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	2
Filey Brigg	50	3.9	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	<0.001	1
Bull Fort (Humber)	50	4.2	<0.001	<0.001	<0.001	0.003	0.002	0.004	<0.001	0.021	2
Cleethorpes	50	3.9	<0.001	<0.001	0.012	0.009	0.002	0.011	<0.001	0.009	1
Wash (Kings Lynn)	50	4.0	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	<0.001	<1
Hunstanton	50	3.8	<0.001	<0.001	<0.001	0.004	0.002	0.003	<0.001	<0.001	1
Walberswick	50	3.8	<0.001	<0.001	<0.001	0.002	0.002	0.002	<0.001	<0.001	<1
Harwich	50	4.0	<0.001	<0.001	<0.001	0.002	0.002	0.002	<0.001	0.009	<1
Creeksea	50	3.9	<0.001	<0.001	0.001	0.004	0.004	0.003	<0.001	0.037	2
Southend	49	4.0	<0.001	<0.001	<0.001	0.003	0.002	<0.001	<0.001	0.037	<1
Swale	50	3.9	<0.001	<0.001	0.001	0.006	0.004	0.004	<0.001	0.095	2
Whitstable	48	3.8	<0.001	<0.001	<0.001	0.002	0.002	<0.001	<0.001	0.031	1

nr = no result

Table 3. Concentrations of metals in cockles, 1991

Station	Area	Mean length (cm)	Concentration (mg kg^{-1} wet weight)					Dry weight (%)
			Cd	Hg	Pb	Cu	Zn	
1	Wainfleet sands, Wash	2.4	0.05	0.01	<0.6	1.3	12	15
2	Kings Lynn, Wash	2.7	0.10	0.01	<0.6	1.3	13	18
3	Off Southend	2.9	0.13	0.02	<0.6	2.0	15	19
4	Off Southend	2.8	0.08	0.02	<0.6	1.3	14	16
5	Foulness sand	2.7	0.06	0.03	<0.6	2.1	9	9
6	East Barrow	3.0	<0.06	0.03	<0.6	2.0	14	19
7	Llanridian sands, Burry inlet	2.4	0.14	<0.01	<0.6	0.92	11	14
8	Llanelli sands, Burry inlet	2.4	0.09	<0.01	<0.6	0.98	12	17
9	Morecambe Bay	3.6	0.15	0.03	<0.6	1.5	12	17
10	Solway Firth	3.3	0.11	0.03	<0.6	1.3	14	18

Table 4. Fal oysters — time series of contaminants

Date sample taken	Mean length (cm)	Concentration (mg kg^{-1} wet weight)							
		Hg	Cu	Zn	Cd	Pb	Ni	Cr	Fe
Parsons Bank									
Oct-91	6.6	0.01	95	540	0.25	1.00	<0.5	0.55	83
Jan-92*	7.4	0.01	149	780	0.42	<0.6	<0.5	<0.5	63
Feb-92*	7.6	0.02	156	612	0.48	<0.6	<0.5	<0.5	96
Mar-92*	7.3	0.02	158	910	0.35	<0.7	<0.6	<0.6	96
Apr-92	7.3	0.02	133	758	0.38	<0.7	<0.6	<0.6	51
May-92	7.6	<0.01	164	938	0.47	<0.6	1.0	<0.7	54
Jun-92	6.8	<0.01	129	1006	0.42	<0.7	<0.6	0.7	46
Jul-92	6.9	<0.01	115	948	0.42	<0.7	<0.6	0.7	81
Aug-92	7.3	<0.01	131	1030	0.47	<0.7	<0.6	<0.6	78
Sep-92*	6.8	<0.03	68	662	0.39	<0.7	<0.6	<0.6	44
Oct-92	6.4	<0.01	61	623	0.28	<0.6	<0.6	0.9	66
Oct-92D	6.8	<0.01	54	641	0.31	<0.6	<0.5	<0.5	36
Messack point									
Oct-91	6.1	0.02	119	584	0.24	<0.6	<0.5	0.98	102
Jan-92*	6.7	0.01	106	603	0.42	<0.6	<0.5	<0.6	36
Feb-92*	7.0	0.02	115	610	0.46	<0.6	<0.5	<0.5	52
Mar-92*	7.1	<0.02	102	804	0.35	<0.7	<0.7	<0.6	67
Apr-92	7.2	0.02	107	704	0.34	<0.7	<0.6	<0.6	43
May-92	6.7	<0.01	129	897	0.50	<0.7	<0.6	<0.6	45
Jun-92	6.7	<0.01	136	987	0.47	<0.7	<0.6	<0.6	33
Jul-92	7.1	<0.01	126	991	0.42	<0.7	<0.6	<0.6	39
Aug-92	7.0	<0.01	133	1098	0.47	<0.6	<0.6	<0.6	59
Sep-92*	7.0	<0.03	85	787	0.41	<0.6	<0.5	<0.5	48
Oct-92	6.5	<0.01	67	645	0.30	<0.6	<0.5	<0.5	44
Oct-92D	6.4	<0.01	48	632	0.43	<0.6	<0.5	<0.5	40

** = mean of 2 or more samples*
D = depurated sample

The JMG has developed guidelines for 'lower', 'medium' and 'upper' concentration ranges for some contaminants, based on the results submitted by all of the countries participating in its monitoring programmes. These, together with other standards and guidelines which apply in England and Wales for contaminants in fish and shellfish are summarised in Appendix 2.

There are as yet no guidelines for ranges of metal contaminants in fish liver, since this is a recent addition to the JMG's programmes.

1.3.1 Heavy metals in dab tissue

Mercury
Concentrations of mercury in dab muscle were essentially in the 'medium' JMG category; the highest value (0.17 mg kg^{-1}) was found in the Humber Estuary. The concentrations found in dab were generally higher than those found in 1990 in offshore adjacent areas (Figure 2) reported in MAFF, 1992(a) but were consistent with earlier work - concentrations of 0.22 mg kg^{-1} in dab from the Humber Estuary were recorded in 1986 (MAFF, 1990).

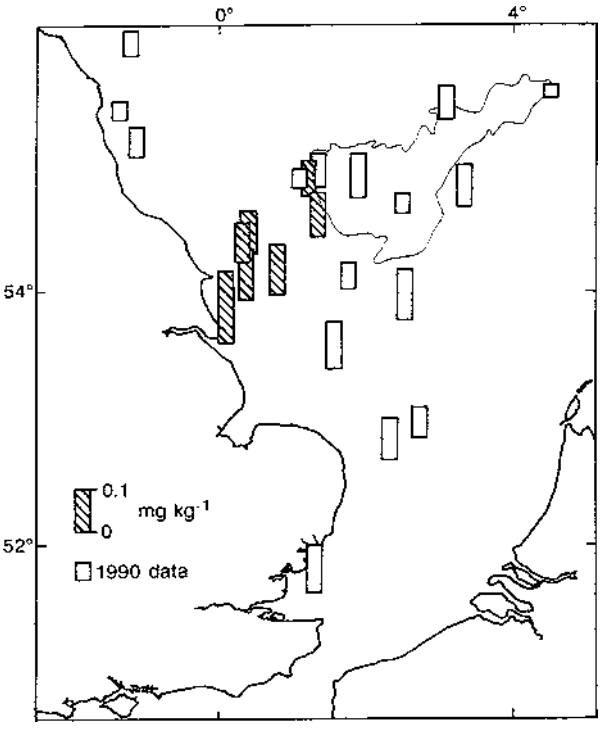

Figure 2. Concentration of mercury (mg kg^{-1}) in dab muscle

Copper
Concentrations of copper in dab muscle were all within the expected range of up to 0.6 mg kg^{-1} and well below the Food Standard Committee's recommended limit of 20 mg kg^{-1} wet weight. Concentrations in fish liver were much higher than those in muscle (by a factor of >10).

Levels were generally similar to those from adjacent offshore areas reported in MAFF, 1992(a), although the concentration in dab livers from north east of Flamborough, at 9.8 mg kg^{-1}, was considerably higher than the other stations.

Zinc
There was no obvious spatial pattern for zinc. Concentrations in fish muscle were in the expected range, up to 6.0 mg kg^{-1}, and well below the Food Standard Committee's recommended limit of 50 mg kg^{-1} wet weight. As with copper, concentrations in liver were around 10 times higher.

Cadmium
Relatively high levels of cadmium >0.4 mg kg^{-1} wet weight in whiting and dab liver, were reported in MAFF, 1992(a) from stations on, and to the south, of the Dogger Bank. In the present survey, levels of cadmium in dab liver from the west edge of the Dogger Bank were similar to those reported in MAFF, 1992(a). However, north east of the Humber, higher concentrations of up to 0.96 mg kg^{-1} were found (Figure 3). Dab are relatively scarce in the Humber estuary itself, only 6 fish were caught in this survey. The concentration in the livers from this sample was relatively low at 0.26 mg kg^{-1}. A further attempt was made to obtain dab from the Humber in 1992. More fish (18) were obtained; the concentration of cadmium in the liver tissue was again low, at 0.20 mg kg^{-1} wet weight (Table 1).

Cadmium is normally present in very low levels in fish muscle and is normally not analysed for in this matrix.

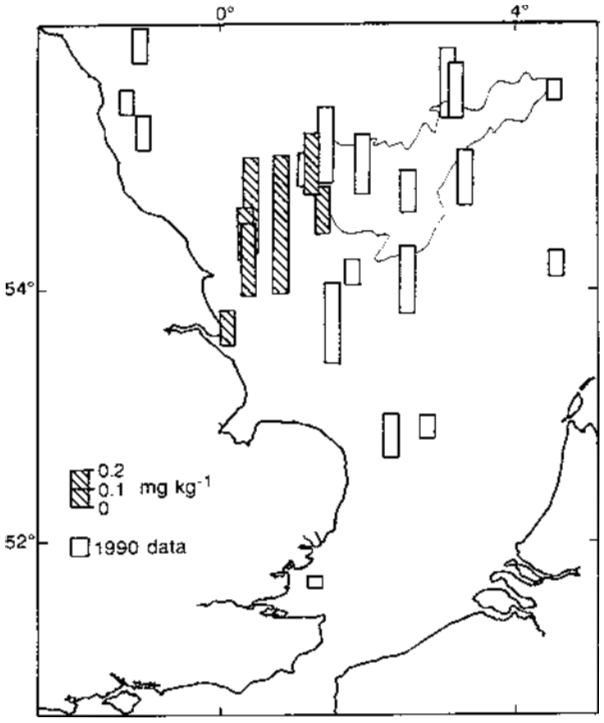

Figure 3. Concentration of cadmium (mg kg^{-1}) in dab liver

In this study, for those fish found to have relatively high concentrations of cadmium in the liver, concentrations in muscle samples were measured using graphite furnace atomic absorption spectrometry (GFAAS); these were only ~0.005 mg kg^{-1} wet weight.

Lead
Levels of lead in fish liver were below the limit of detection for the analytical method used, i.e. <0.6 mg kg^{-1}. Levels of lead are not normally determined in fish muscle as concentrations are known to be very low (~0.005 mg kg^{-1} wet weight).

1.3.2 Organochlorine pesticides in mussels

Concentrations of organochlorine pesticides in mus-sels were generally low (<0.001-0.012 mg kg^{-1}) and well within 'expected' values (see Appendix 2). No general spatial pattern was detected in the results; the highest values of γ-HCH (0.012 mg kg^{-1}) and dieldrin (0.009 mg kg^{-1}) were recorded in the sample from Cleethorpes and dieldrin (0.006 mg kg^{-1}) in the sample from the Swale. Concentrations of γ-HCH and dieldrin were considerably lower (0.002 and 0.003 mg kg^{-1} wet weight, respectively) in a sample of mussels collected from Cleethorpes in Feburary 1992.

1.3.3 PCBs in mussels

PCB concentrations in mussels were generally in the 'lower' JMG category for molluscs, i.e. <0.02 mg kg^{-1} wet weight. Concentrations in the 'medium' category, 0.02-0.10 mg kg^{-1} wet weight, were found in the Thames Estuary region, off Blyth and also in the Humber. In earlier work (MAFF, 1992(a)) the highest PCB concentrations in fish liver were also found in the Thames area, tending to confirm that this area is more contaminated than most by these compounds.

1.3.4 Heavy metals in cockles

Concentrations of heavy metals in cockles were within the 'expected' values for shellfish and well below any standards/guideline values (see Appendix 2).

Differences between samples were generally not great; slightly higher levels of cadmium were found in samples from Southend, the Burry Inlet and Morecambe Bay.

1.3.5 Heavy metals in oysters from the River Fal

An increased input of heavy metals into the Restronguet Creek area of the River Fal commenced around November 1991, when contaminated water from the abandoned Wheal Jane mine began to overflow into the estuary. The area has commercial shellfisheries and a sampling programme was implemented by MAFF to monitor any possible uptake of these metals by shellfish.

The sampling programme concentrated on oysters, the main commercially produced shellfish in the area. Samples were taken throughout from two main stock areas Parsons Bank and Messack Point, although the number of sampling points and the frequency of sampling increased from January to March 1992 following a major increase in inputs, due to a large escape of contaminated water. The time series of results from the two main stock areas are shown in Table 4.

Metal accumulation into the tissues is a fairly slow process, especially from suspended particles, and at winter functioning rates was not expected to be rapid. Monitoring continued over the summer close season to establish whether the oysters would accumulate higher body burdens of metals as the water temperatures rose though it was not considered likely that they would do so to the extent that human consumers would be placed at risk in the 1992/1993 oyster season.

Levels of Cr, Ni, Hg and Pb were generally all below the limits of detection of the method of analysis used. Levels of Cu and Fe showed some fluctuations, but there appeared to be no significant increase compared with levels prior to the event. Concentrations of cadmium increased at the beginning of 1992 to 0.4-0.5 mg kg^{-1} wet weight and remained around this level, with some fluctuations throughout the summer period (Figure 4). In September levels began to decrease and by October they had fallen to ~0.3 mg kg^{-1}

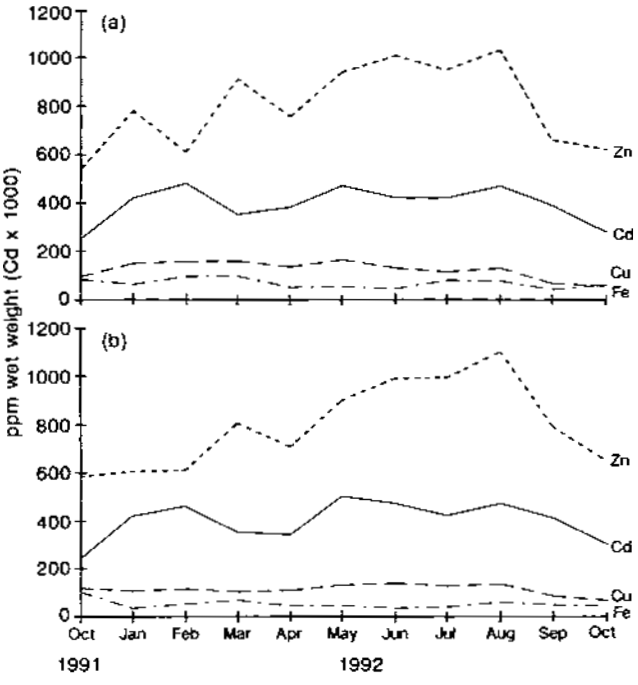

Figure 4. Time series of concentrations of metals in oysters from the Fal Estuary: (a) Parsons Bank; (b) Messack Point

wet weight. Concentrations of zinc also increased at the beginning of 1992. They continued to rise over the summer reaching a level of ~1000 mg kg^{-1} wet weight by August 1992. As with cadmium, levels began to decrease in September falling to ~650 mg kg^{-1} by October.

Oysters taken from the Fal often undergo depuration 'cleansing' in tanks for 24-48 hours prior to marketing. Additional samples of oysters were taken in October 1992 so that some could be analysed following the commercial depuration process. There did not appear to be a consistent trend in the generally small differences in contaminant concentrations observed following depuration (see Table 4).

1.4 Conclusions

Work carried out in 1991 has provided information on contaminants in dab from off the Humber region, from mussels taken from off the North Sea coast of England, from cockles taken from a number of sites around England and Wales, and from oysters from the River Fal in Cornwall.

Levels of mercury in dab from, within and to the north east of the Humber Estuary, were generally higher than those recorded from offshore adjacent areas sampled in 1990 (MAFF, 1992(a)), but did not exceed the JMP 'medium' category for mercury of 0.1-0.3 mg kg^{-1} wet weight.

There was some evidence of cadmium contamination of the livers of dab taken from the area extending from the Humber to the Dogger Bank. The source of this contamination is not clear, since concentrations in dab taken from the Humber Estuary itself were found to be relatively low.

Levels of organochlorine pesticides in mussels were low with little indication of any spatial pattern. Data for PCBs indicated concentrations generally in the 'lower' JMP category but with areas of relatively higher contamination in the Thames Estuary, off Blyth and in the Humber.

Heavy metal concentrations in cockles were all relatively low with no obvious general spatial patterns.

Levels of cadmium and zinc in oysters from near the Wheal Jane overflow showed some increase during the early part of 1992 and subsequently fell back in the autumn of 1992. Whether this was a result of contamination, or due to an increase in activity over the summer period, is as yet unclear and further data will be required before any firm conclusions can be drawn. Throughout 1992, MAFF were able to advise that there was no cause for concern with regard to the consumption of oysters, but monitoring will continue.

1.5 Monitoring to indicate compliance with quality standards for mercury

The Liverpool and Morecambe Bay areas are subject to discharges from chloralkali plants and thus to provisions of EC Council Directive 82/176 EEC (see Appendix 2). Compliance has also to be demonstrated with agreed Paris Commission Environmental Quality Standards (EQSs). For this purpose, data have been submitted annually since 1982, to indicate compliance with an EQS of 0.30 mg mercury per kg of representative wet fish flesh.

The weighted mean mercury concentrations in fish taken from Liverpool and Morecambe Bays in 1990 were well below the standard at 0.17 and 0.21 mg kg^{-1} wet weight respectively. In presenting these results to the Commission's meetings in 1991, the UK pointed out that the standard had been met over the last decade; it was therefore proposed that monitoring should now be reduced to once every second (even) year. No objections were received to this proposal and the UK has therefore moved to a biennial monitoring programme. The next set of results will relate to samples collected in 1992 and will be included in a future report in this series.

2. THE USE OF BIOASSAYS IN MEASURING BIOLOGICAL EFFECTS

2.1 General introduction

The 1990 North Sea Task Force Monitoring Master Plan (NSTFMMP, 1990) required data to be obtained from two bioassays for selected stations in the North Sea and English Channel. The bioassays were the oyster embryo bioassay, a well established technique for measuring water quality and an enzyme bioassay, mixed function oxidase (ethoxyresorufin-*O*-de-ethylase; EROD) induction in dab (*Limanda limanda*), a technique recognised as an indicator of environmental contamination, frequently attributed to exposure of the fish to polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs). The bioassays were deployed on the UK continental shelf on research cruises in 1990 and 1991 and the data sets submitted to the International Council for the Exploration of the Sea (ICES) for inclusion in the North Sea Quality Status Report. The results are presented here together with the results of an algal growth bioassay also used on these research cruises.

2.2 Oyster embryo bioassay

2.2.1 Introduction

MAFF have used this bioassay over a number of years for monitoring biological water quality in estuarine and

coastal waters (Thain and Watts, 1984; Byrne *et al*., 1988; Stebbing *et al*., 1991; Thain, 1991). In July 1990, a survey was carried out in five north-east coast estuaries, coastal and offshore waters out to the Dogger Bank and in the English Channel; eleven NSTFMMP stations were included. The survey was repeated in July 1991.

2.2.2 Methods

The oyster embryo bioassay method used, was that recommended for the NSTFMMP (see Thain, 1991). Adult oysters (*Crassostrea gigas*) are conditioned for spawning in the laboratory and taken to sea where the bioassay is deployed on-board a MAFF research vessel. For each assay, eggs and sperm are stripped from the oysters and the eggs artificially fertilised. Developing embryos are added to the test samples where they remain for 24 hours. During this time the embryos develop into D-shaped larvae. Failure to develop in this way implies that the water quality is poor.

For water samples, 2.5l of water is taken from 0.5 to 1.0 m below the surface; from this at least 4 x 30 ml replicate sub-samples are tested. Sediments are tested by mixing 200 ml of sediment with 500 ml of water in a 1 litre container at 100 revolutions min^{-1} on an orbital shaker. After 3 hours the slurry is filtered through Whatman GFC filter paper and the resultant filtrate assayed as described for water samples.

2.2.3 Results

The locations of the sampling sites are given in Table 5: Latitude and longitude positions are shown since 'upper' and 'lower' estuarine sites refer to relative station positions and do not indicate precise location. Samples taken as part of the NSTFMMP are indicated accordingly. Each result is expressed as a Percent Net Response (PNR),

$$\text{i.e. PNR} = \frac{(\%\ \text{test abnormal} - \%\ \text{control abnormal})}{100 - \%\ \text{control abnormal}} \times 100.$$

A value of 0 or close to 0 (including negative values) indicates that the measured response of the sample is similar to that of the control reference water. Poor water quality is defined when PNR values (usually >10) are statistically different from the reference sea water.

Table 5. Oyster embryo bioassay results for water and sediment samples collected in 1990 and 1991

Position	Location	NSTF no.	Oyster Bioassay PNR			
			1990		1991	
			Water	Sediment	Water	Sediment
55° 45.7'N 1° 57.85'W	Tweed/Mouth		-10.0	-0.6	3.2	-0.7
55° 45.83'N 1° 59.58'W	Tweed/Lower estuary		1.2	-20.0	13.3	-13.6
55° 46.12'N 2° 0.35'W	Tweed/Upper estuary		-7.6	3.9	7.5	8.2
55° 0.41'N 1° 23.76'W	Tyne/Mouth		4.0	3.0	8.2	-2.5
55° 0.63'N 1° 24.98'W	Tyne/Lower estuary		1.2	5.3	10.4	-2.2
54° 59.17'N 1° 27.86'W	Tyne/Middle estuary		3.2	100.0	11.8	30.8
54° 58.92'N 1° 31.82'W	Tyne/Middle estuary		10.0	81.4	54.5	ns
54° 58.09'N 1° 36.25'W	Tyne/Upper estuary		-9.2	100.0	71.0	ns
55° 0.46'N 1° 7.76'W	Tyne/Intermediate	14	-5.2	0.2	-0.7	-1.4
54° 54.88'N 1° 20.3'W	Wear/Mouth		3.2	0.2	12.2	6.5
54° 55.04'N 1° 21.62'W	Wear/Lower estuary		-2.0	-8.4	-6.5	1.4
54° 54.58'N 1° 22.87'W	Wear/Upper estuary		9.2	-5.5	0.4	27.6
54° 44.04'N 0° 52.84'W	Tees/Intermediate		15	14.4	-15.5	1.8 - 7.5
54° 38.93'N 1° 7.35'W	Tees/Mouth		-7.2	-12.1	9.0	-6.5
54° 38.52'N 1° 8.66'W	Tees/Lower estuary		3.6	11.0	6.5	2.9
54° 37.12'N 1° 9.32'W	Tees/Middle estuary		71.0	66.1	71.0	100.0
54° 36.22'N 1° 9.9'W	Tees/Upper estuary		100.0	ns	35.8	11.5
54° 50.1'N 1° 20.24'E	Offshore Tyne/Tees	43	-4.0	-8.4	-3.2	-5.4
55° 10.1'N 3° 6.3'E	Dogger Bank	47	-4.2	-12.0	6.1	-2.9
53° 59.94'N 2° 0.21'E	Silver Pit	53	0.0	-4.5	12.2	-12.9
53° 32.41'N 0° 20.12'E	Humber/Intermediate	16	-2.8	ns	-0.7	ns
53° 33.33'N 0° 6.37'E	Humber/Mouth		4.0	-1.7	8.6	ns
53° 37.1'N 0° 2.82'W	Humber/Lower estuary		10.4	7.0	ns	ns
53° 36.0'N 0° 3.6'W	Humber/Lower estuary		ns	ns	2.2	ns
53° 36.1'N 0° 5.9'W	Humber/Middle estuary		ns	ns	15.8	ns
53° 38.6'N 0° 11.0'W	Humber/Upper estuary		ns	ns	-1.1	ns
53° 36.0'N 0° 3.0'W	Humber/Upper estuary		-4.0	13.5	ns	ns
53° 3.65'N 0° 28.88'E	Wash	17	9.6	-5.0	2.5	-15.3
52° 50.4'N 2° 50.1'E	E. Smiths Knoll	18	-1.6	-11.2	-6.8	4.5
51° 59.97'N 2° 19.95'E	Gabbard	25	-2.4	-9.3	ns	ns
51° 30.34'N 0° 57.88'E	Thames Barrow	19	-0.1	11.6	ns	ns
50° 56.1'N 1° 17.05'E	E. English Channel	69	10.8	5.9	ns	ns

PNR = Percent Net Response
ns = no sample
NSTF = North Sea Task Force Monitoring Master Plan Station

A PNR value of 100 occurs when the sample is highly toxic and all embryos have either died or shown abnormal development over the 24 h exposure period.

Of the 32 stations, 11 were coastal or offshore and designated NSTFMMP. Poor water quality or toxic sediment elutriates were not detected at any of the NSTF sites in either 1990 or 1991.

Of the 21 estuarine sites, including those at estuary mouths, only two were found to have measurably poor water quality in 1990; the middle and upper Tees with PNR values of 71 and 100 respectively. In 1991 the same two sites had PNR values of 71 and 36 and additionally, poor water quality was measured at two sites on the River Tyne (PNR values of 54 and 71 respectively).

In 1990, sediment elutriates were found to be toxic to oyster embryos at three locations on the River Tyne; PNR values for the two middle and upper estuarine sites were 100, 81 and 100 respectively and on the River Tees a PNR of 66 was measured at a middle estuarine site. In 1991, a PNR of 31 was found for one of the middle estuarine sites on the Tyne, but due to adverse weather conditions no samples were collected from the other two sites found to have toxic sediments in 1990. On the River Tees, a sediment elutriate from the middle estuarine site was again found to be toxic in 1991 with a PNR of 100. In addition, a sediment elutriate from the upper estuary of the River Wear showed some toxicity, with a PNR of 28.

2.2.4 Conclusion and discussion

The results presented here show the oyster embryo has enough sensitivity for use in monitoring water quality in industrialised estuaries. High PNR values, indicating poor water quality, were found in the Tyne and Tees estuary. It is also of interest that samples from the same sites on the Tees estuary gave a similar bioassay response in both 1990 and 1991. Few analyses of water samples for contaminants were made in 1990. Although the highest levels of total hydrocarbons (THC) were measured in the Tees estuarine samples, the THC concentrations on their own were insufficient to cause the observed toxicity. A similar observation was made in 1991. Data are also available in 1991 for a number of metals (see Section 6). However, none of the concentrations exceeded those that would be expected to cause mortality or sublethal effects in oyster embryos in a 24 h exposure.

As with the water column results, sediment elutriate toxicity was only observed in the estuaries of the industrialised Rivers Tees, Tyne and also the Wear. It is also noteworthy that sediment elutriates from one site on the Tyne and one site on the Tees exhibited a similar bioassay response on both sampling occasions. No toxicity was observed in any of the sediment elutriates from NSTF stations. Very few contaminant data are available for sediments in 1990 and 1991 and the elutriation technique does not allow a direct comparison to be made between toxicity and contaminant loading.

It should be borne in mind that sediment elutriation is not the preferred approach for measuring sediment toxicity, but at the time of the initiation of this study it was the best practical option. Therefore, the results presented here should be interpreted with caution; contaminants eluted off sediments and presented to water column organisms do not necessarily represent the exposure of those same contaminants in sediments to sediment dwelling organisms. Future surveys will endeavour to evaluate whole sediment bioassays.

The results presented here clearly show that poor water quality and potentially toxic sediments do exist in some UK estuaries and indicate that further work should be carried out with this bioassay and other techniques. The geographical spread of sampling sites was limited on some estuaries e.g. Humber and in further investigations this will be extended to include sampling sites nearer to the tidal limit. Offshore surveying using the oyster bioassay has little value, except for site specific surveys near disposal grounds or oil platforms for example.

Finally, it is worth noting that the bioassay approach to marine monitoring is able to give a warning of environmental problems in situations where traditional analytical chemistry may be ineffective. The strength of bioassays is that they permit a more targeted and cost-effective use of expensive chemical analysis.

2.3 EROD measurement in dab

2.3.1 Introduction

The NSTFMMP chose EROD induction in dab as one of four biological effects monitoring techniques (see review by Payne *et al*., 1987). The mixed function oxidase system, of which EROD is an isozyme, catalyses the degradation of certain lipophilic compounds e.g. PCBs and PAHs to more polar and water soluble compounds which are more easily excreted or bound by the animal. It is present in all animals at a very low activity, but its activity increases dramatically in fish exposed to compounds such as PAHs. However, interpretation of EROD data is not easy because the enzyme may be induced by natural factors. In addition North Sea dab populations are known to be mobile, making site specific relationships of EROD activity to contamination difficult.

The bioassay was carried out on fish trawled on two research cruises in April and October 1991. The fish were caught at, or as near as possible to, NSTFMMP-designated stations and only dab were used for the study.

2.3.2 Methods

The bioassay procedure used follows that described by Galgani and Payne (1991) with modifications resulting from the ICES intercalibration workshop held in Aberdeen in September 1991 (R. Stagg, pers.comm.).

Dab for the assay were caught using a Granton or Dutch beam trawl. At each station an attempt was made to sample twelve males and twelve females within the size range 15-25 cm. However, this was not always possible as a result of a poor catch rate. Liver samples, taken from the fish as soon as they had been caught, were immediately preserved in liquid nitrogen. The EROD assay was then performed on these samples back at the laboratory.

2.3.3 Results

The results are given in Table 6. At five of the six stations in the North Sea, EROD activity was greater in males than in females and the highest activity was measured in male fish during the April sampling. Female fish did not show such a marked seasonality in their response, although there was some evidence from the Smiths Knoll and North Dogger stations that activity was indeed higher in fish sampled in April.

At the seven stations sampled on the south and west coasts of England and Wales there was little difference in EROD activity between males and females, with the exception of Lyme Bay where female activity was much higher.

Of the eleven stations sampled in October, the highest activity was observed in male fish from the inner and middle Liverpool Bay, and in female fish from Lyme Bay.

Of the five stations sampled in April, highest activity was measured in males from North Dogger and females from Smiths Knoll.

2.3.4 Conclusion and discussion

It is difficult to interpret these results directly in terms of levels of PAHs and PCBs since no chemical data are available on the fish sampled. However, MAFF routinely measures concentrations of PCBs and other organics (HCB, α–HCH, γ–HCH, dieldrin, ppDDE, ppTDE and ppDDT) in dab liver and other species and the residue levels reported (MAFF, 1991 and 1992(a)) are not thought to be high enough to cause biological effects, such as the induction of EROD. Furthermore, the several confounding factors mentioned in sub-section 2.3.1 and the lack of dose-response relationships between contaminant loading and EROD induction in dab makes it even more difficult to interpret this type of data.

Clearly the results obtained from this study indicate that there are differences between the response in males and females and also that EROD activity is seasonal and may be related to the breeding activity of the fish. No dab were obtained from estuarine or coastal sites. In the future it may be necessary to use flounder at these sites in order to determine the impact of habitats that are generally more contaminated than the open sea.

If this technique is to be adopted for monitoring purposes, it is essential that a greater effort is made to provide data with which to interpret the results. This should include laboratory studies, chemical residue analysis for specific contaminants and fish histopathological observations.

Table 6. Mean EROD levels (pmols (mg protein)$^{-1}$ minute^{-1}) in livers of dab (Limanda limanda) from the North Sea

Date	Station	NSTF no.	Coordinates	EROD (females)		EROD (males)	
				Mean	s.e.	Mean	s.e.
Apr-91	Smiths Knoll	18	52° 45'N2° 42'E	1247	342	2047	704
Oct-91	Smiths Knoll	18	52° 45'N2° 42'E	105	26	i.a.	i.a.
Apr-91	North of Spurn Head		53° 57'N0° 48'E	48	26	623	129
Oct-91	North of Spurn Head		53° 54'N0° 56'E	461	120	314	89
Oct-91	North of Flamborough Head		54° 30'N0° 39'E	856	324	409	77
Apr-91	North Dogger	47	55° 24'N3° 10'E	313	105	3607	475
Oct-91	North Dogger	47	55° 07'N3° 15'E	90	22	188	45
Apr-91	Oyster Grounds	50	54° 06'N4° 19'E	278	109	1333	419
Apr-91	Silver Pit	53	54° 07'N2° 27'E	131	62	783	210
Oct-91	Rye Bay	69	51° 33'N4° 32'E	161	64	73	25
Oct-91	Lyme Bay		50° 30'N3° 07'W	1013	460	225	73
Oct-91	Liverpool Bay (inner)		53° 27'N3° 21'W	474	227	483	196
Oct-91	Liverpool Bay (middle)		53° 26'N3° 32'W	427	138	473	117
Oct-91	Liverpool Bay (outer)		53° 29'N3° 38'W	149	47	47	15
Oct-91	Morecambe Bay		53° 57'N3° 20'W	347	166	322	125
Oct-91	Cardigan Bay		52° 19'N4° 14'W	133	23	205	52

i.a. = insufficient animals for analysis
s.e. = standard error

An overall assessment of this work and that of other NSTF member states will be included in the North Sea Quality Status Report in 1993.

2.4 Algal growth bioassay

2.4.1 Introduction

The algal growth bioassay is a five day test to assess the potential for algal growth in natural seawater samples. In samples in which there is poor growth, but the essential nutrients are not limiting, phytotoxic chemicals may be present. Therefore, the algal growth bioassay represents a means of integrating the effects of all the factors present in a sample upon algal growth and in this way makes it possible to determine which samples may require more detailed analysis for contaminants.

2.4.2 Methods

The basic test methodology has been described in the report on studies carried out in 1990 (MAFF, 1992(a)). The main change in the current bioassay procedure is that the algal inoculum is cultured in smaller seawater sample volumes (30 ml instead of 125 ml), which are incubated in 50 ml, as opposed to 125 ml, conical flasks. This change was introduced to allow a larger number of samples to be cultured at one time. Two species of algae, the brown diatom *Thalassiosira pseudonana* and the green flagellate *Tetraselmis suecica* were individually inoculated into seawater samples, taken from 0.5-1 m below the surface, at a concentration of 5 x 10^4 cells ml^{-1}. After 5 days incubation on a shaking table in constant light conditions, the algal cell densities were estimated spectrophotometrically. The nutrient levels were determined in sub-samples of each fresh sample using the autoanalyser method of Kirkwood (1989).

2.4.3 Results and discussion

Figure 5 shows the cell growth for *T. suecica* in 1990 and 1991 in samples from the estuaries of the Rivers Tweed, Tyne, Wear, Tees and Humber, from upper estuary to open sea locations out to the Dogger Bank (from left to right respectively). Figure 6 shows cell growth for *T. pseudonana*. Both figures also show the concentrations of phosphate, silicate and nitrate present at each sample site.

For all the sites in both 1990 and 1991 the concentrations of nutrients are of a similar order of magnitude. They exhibit a similar pattern of decline from sites higher up the estuaries to those offshore.

The growth of *T. suecica* shows a similar pattern in each year, with the highest growth occurring in the nutrient rich higher estuary samples, in particular the Tyne, Wear and Tees.

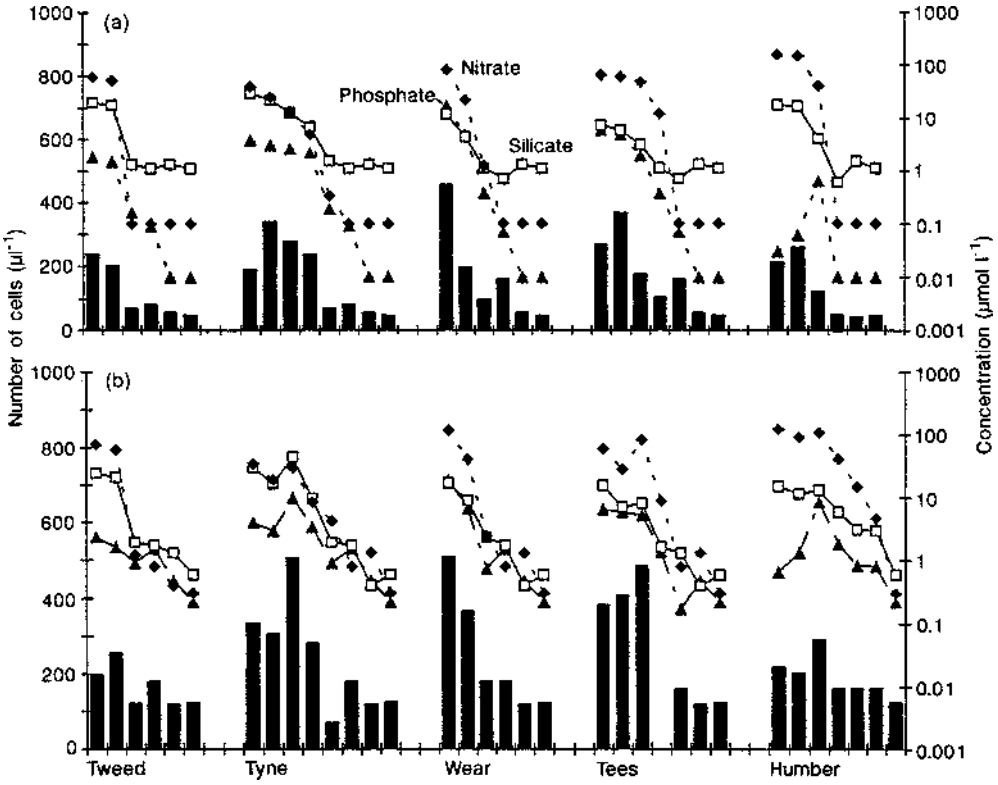

Figure 5. Growth of Tetraselmis suecica in water samples taken from five estuaries on the north-east coast of England during: (a) 1990 and (b) 1991. (Data points left to right, represent upper estuary to open sea locations out to the Dogger Bank)

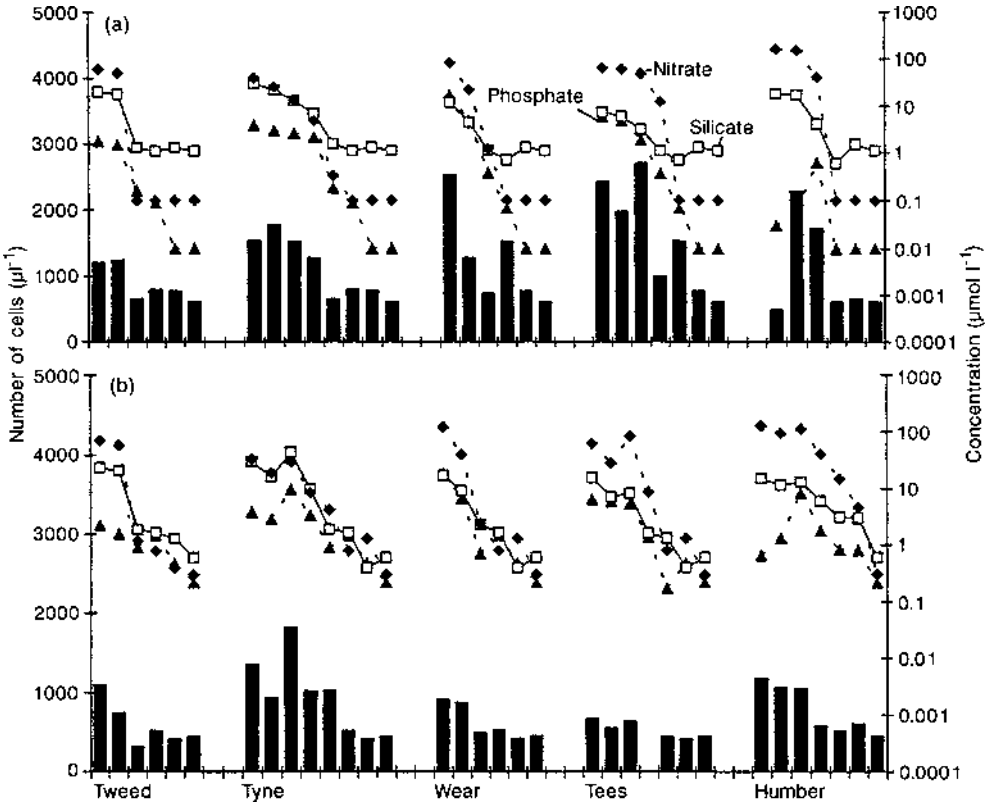

Figure 6. ***Growth of Thalassiosira pseudonana in water samples taken from five estuaries on the north-east coast of England during: (a) 1990 and (b) 1991 (in 1990 the detection limit for nitrate and silicate was 0.1 mol l^{-1} and for phosphate 0.01 mol l^{-1})***

The pattern of growth shown by *T. pseudonana* is broadly similar to that shown by *T. suecica*. The major exception is the lower growth in the higher estuary samples from the Wear and the Tees in 1991, compared to 1990.

In the past chemical analysis has often been used to describe the nutrient status of marine waters. The use of this bioassay complements such nutrient analysis and gives an indication of the biological significance of the nutrients present. In one case (upper Humber, 1990), the relatively poor growth of *T. pseudonana* could not be entirely explained by the nutrient status, indicating the possible presence of phytotoxic material.

3. SURVEYS OF CONTAMINANTS IN MARINE MAMMALS

During 1991-92, analyses of marine mammal tissues for trace metals and organochlorines were carried out as part of a Department of the Environment (DoE) co-ordinated programme, which also integrates sightings and stranding information and post-mortem studies with information on contaminant levels. This dataset is not yet fully complete and a more comprehensive assessment will be provided in a further report in this series.

Samples of liver from forty-two marine mammals, of six species, found on the coast of Wales and the Irish Sea in 1989-91 were analysed for a range of trace metals and the data reported (Law *et al*., 1992). The animals sampled comprised eleven seals, twenty-two porpoises, eight dolphins and one Minke whale. The ranges of concentrations obtained are given in Table 7. High concentrations of mercury and lead were found in a number of animals from the Liverpool Bay area of the eastern Irish Sea, confirming earlier indications of a 'hot-spot' in this area.

Table 7. ***Ranges of concentrations of trace metals in livers of marine mammals (g g^{-1} wet weight)***

Element	Range of concentration
Chromium	< 0.5 to 2.1
Nickel	< 0.5 to 2.1
Copper	2.2 to 79
Zinc	15 to 150
Cadmium	< 0.06 to 2.2
Mercury	0.5 to 280
Lead	0.05 to 7.0

At least seven of the thirty-nine cetaceans obtained from south and west Wales died in fishing nets, and it is clear that the increasing use of fixed fishing nets, set at or near the seabed, poses a threat to such animals which breathe at the sea surface. Research in Denmark and Canada on the effects of bycatch on harbour

porpoise (*Phocoena phocoena*) populations (Clausen and Andersen, 1988; Read and Gaskin, 1990) has shown signs of over exploitation (reduced age of sexual maturity, reduction in the proportion of large animals in the population). Modelling the potential rate of increase of the population in the Gulf of Maine and the Bay of Fundy, indicated that porpoises have only a limited capacity for population increase and that they are unlikely to sustain even moderate levels of incidental mortality (4% of the population per year) and led to the recommendation that immediate action should be taken to reduce the level of incidental catches in local gillnet fisheries (Woodley and Read, 1991). The limited capacity of porpoises for population increase is due to their comparatively late maturation and short lifespan; most porpoises die before the age of nine (Martin, 1990) and very few survive for more than twelve years. Better estimates of mortality of marine mammals in UK fisheries are needed in order to assess the current status of local populations, in particular of small cetaceans (porpoises and dolphins) whose populations may be declining (Evans and Scanlan, 1988).

Polycyclic aromatic hydrocarbons (PAH) some of which are carcinogenic, are ubiquitous environmental contaminants; they arise mainly from the combustion of fossil fuels and discharges containing mineral oils. However, few data are available on levels of PAH in marine mammals. Samples of muscle tissue from twenty-six porpoises sampled around the UK during the period 1988-91, were analysed for total PAH using fluorescence spectrometry and, for a range of specific PAH using gas chromatography/mass spectrometry (GC/MS) (Law and Whinnet, 1992). The concentrations of total PAH found were low and ranged from 0.47 to 2.4 g g^{-1} wet weight Ekofisk crude oil equivalents (0.11 to 0.56 g g^{-1} as chrysene equivalents), with no indication of accumulation with age (Table 8). Low but detectable concentrations of 2 - 4 ring PAH were found using GC/MS. These data are very similar to those reported earlier for marine mammals from Canada (Hellou *et al*., 1990 and 1991). Although it seems unlikely that PAH affect porpoises or other marine mammals, except in cases where extreme exposure occurs, further information on the occurence and effects of PAH is necessary before a full assessment of their significance can be made. Studies of PAH in the coastal margins of England and Wales will be conducted during the next few years under the newly established UK National Monitoring Programme.

Table 8. Concentrations of total PAH in porpoise muscle (g g^{-1} wet weight)

Location	Date	Sex	Length (cm)	Weight (kg)	Age ¶	Ekofisk equivalents	Chrysene equivalents	
Shetland	2.3.89	F	135	38.8	3	1.1	0.25	§
Shetland	7.3.89	F	¢	¢	1	1.0	0.23	§
Shetland	26.7.91	F	145	42.5	8	0.95	0.22	
Shetland	30.9.91	F	143	43	1	0.72	0.17	$
Shetland	8.10.91	F	158	42.6	4	0.65	0.15	
Blyth	22.6.91	M	102	20	1	1.4	0.33	
Saltburn	3.10.91	M	130	28.8	3	1.3	0.3	
Whitby	8.8.88	M	102	20	1	0.88	0.2	§$
Bridlington	9.3.91	M	94	19	1	1.2	0.29	
Bridlington	9.3.91	M	111	21	2	1.2	0.27	
Bridlington	9.3.91	M	103	20.5	< 1	0.99	0.23	
Bridlington	11.3.91	F	110	23	< 1	0.96	0.22	
Skegness	12.6.91	F	76	6.8	< 1	0.77	0.18	¥
Norfolk	12.6.91	M	72	5.5	< 1	1.1	0.26	¥
Lowestoft	1.8.91	M	87	8.5	< 1	0.96	0.22	¥
Holland on Sea	1.8.91	M	97	13	< 1	0.88	0.2	
Folkestone	17.2.89	F	125	32.8	1	0.47	0.11	§$
Folkestone	17.2.89	M	135	36.7	6	0.84	0.19	§$
Gwynedd	15.7.91	M	78	7.0	< 1	1.0	0.23	¥
Isle of Man	1.6.88	F	86	7.7	< 1	1.0	0.23	§
Isle of Man	27.10.88	M	130	24.8	1	0.56	0.13	§
Isle of Man	¢	M	107	20	< 1	2.4	0.56	
Isle of Man	¢	M	91	12	#	0.99	0.23	
Islay	Jun-88	M	70	6.4	< 1	1.0	0.23	§
¢	¢	F	145	41.4	5	1.6	0.36	
¢	¢	F	129	25	1	0.8	0.19	

§ = concentrations of metals in liver tissue from these sample are given in Table 3 of Law et al. (1991)
$ = these animals were entangled in fishing nets
¥ = neonatal animals
¶ = age in years estimated by counting tooth growth layer groups
 # = juvenile: no teeth obtained
¢ = no information available

4. INTRODUCTION: GENERAL SAMPLING PROCEDURES

Most of the work reported in this section was carried out on *RV CIROLANA*, during cruise 6, 22 June - 14 July 1991. The cruise track is illustrated in Figure 7. In order to avoid duplication, the basic information about this cruise will be given here and referred to again in later sections. Table 9 gives a list of station numbers, sampling dates and positions, with a descriptive indication of the location of the sampling points taken in and close to estuaries. Some of the monitoring carried out in 1991 was undertaken to provide information for the NSTFMMP, and so eventually for inclusion in the 1993 Quality Status Report of the North Sea to be presented to the Fourth Ministerial North Sea Conference. Where designated NSTFMMP stations were visited, their allotted number is also indicated in Table 9. Table 10 lists the water column data (depth, salinity, concentrations of nutrients, surface water temperature and sediment type) for each of the stations, where available.

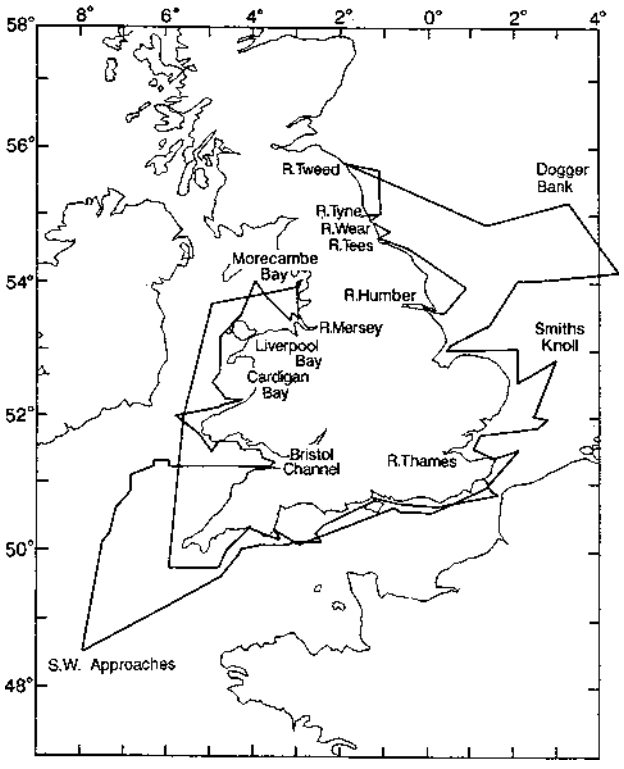

Figure 7. Cruise track of CIROLANA 6, 22 June-14 July 1991

As outlined previously (MAFF, 1992(a)), the strategy adopted for inshore sampling was to anchor the ship close to an estuary mouth and to collect samples both from the ship and from a small inflatable boat which could operate in shallow water around and within the estuary. Where the small boat was used, sampling took place at pre-arranged sites adjacent to named or numbered buoys, quays, jetties, etc., which could be readily identified, so that the sampling position could be determined accurately. Whenever possible, sampling from the small boat was carried out around low water, whilst samples were being collected over longer time periods from the ship. Water samples taken from the small boat, generally had a minimum salinity in the range of 20-30 (outer estuarine conditions), but in the River Tweed the minimum salinity observed was 1.6 (Table 10, Station 26).

The Humber, Mersey, Tees, Tyne and Wear are industrialised estuaries; the Tweed has no major industries and was included for comparative purposes.

5. NUTRIENTS

Concentrations of nutrients were at a maximum in the estuarine samples (see Table 10), the highest values observed being: phosphate, 19 mol l^{-1} (River Wear - Wearmouth Bridge; Station 17); silicate, 44 mol l^{-1} (River Tyne - Jarrow Slake; Station 20); nitrate + nitrite, 147 mol l^{-1} (River Humber - off Immingham; Station 2). These were similar to the maximum values recorded in 1990 (MAFF, 1992(a)), which were: phosphate, 17 mol l^{-1} (River Wear - Wearmouth Bridge); silicate, 30 mol l^{-1} (River Tyne - Tyne Bridge); and nitrate, 160 mol l^{-1} (River Humber - off Immingham). In offshore waters concentrations of nutrients were low, as would be expected during the summer months (Table 10). At the MAFF reference station in the Western Approaches (Station 66) concentrations of nutrients were low (<1.5 μmol l^{-1}) in the upper part of the water column (to 20 m depth), but higher at 100 and 150 m depth (1-10 μmol l^{-1}). More detailed studies of the fate of nutrients discharged to estuaries are currently being undertaken by staff at the Lowestoft Laboratory, in collaboration with agencies outside MAFF, under the Joint Nutrient Study (JONUS).

Table 9. Stations sampled during CIROLANA 6, 22 June-14 July 1991

Station no.	Date	Position		Location	NSTFMMP
1	22-Jun	53° 33.33' N	0° 6.05' E	Humber/mouth	
2		53° 38.6' N	0° 11.0' W	No. 11A buoy	
3		53° 37.4' N	0° 8.7' W	No. 10A buoy	
4		53° 36.1' N	0° 5.9' W	Diffuser	
5		53° 36.6' N	0° 4.6' W	No. 6B buoy	
6		53° 36.0' N	0° 3.6' W	Humber/upper estuary (Pyewipe outfall)	
7	23-Jun	53°33.26' N	0° 6.57' E	Humber/mouth	
8		53° 31.9' N	0° 20.16' E	Humber/intermediate	16
9		53° 55.06' N	0° 50.18' E		
10	24-Jun	54° 39.05' N	1° 7.28' W	Tees/mouth	
11		54° 38.52' N	1° 8.66' W	Tees/lower estuary (No. 8 buoy)	
12		54° 37.12' N	1° 9.32' W	Tees/middle estuary (No. 15 buoy)	
13		54° 36.22' N	1° 9.9' W	Tees/upper estuary (ICI No. 4 buoy)	
14		54° 44.06' N	0° 53.03' W	Tees/intermediate	15
15	25-Jun	54° 54.95' N	1° 20.27' W	Wear/mouth	
16		54° 55.01' N	1° 21.78' W	Wear/lower estuary (North Dock Basin)	
17		54° 54.58' N	1° 22.87' W	Wear/upper estuary (Wearmouth Bridge)	
18		55° 00.35' N	1° 23.69' W	Tyne/mouth	
19		55° 0.63' N	1° 24.98' W	Tyne/lower estuary (No. 2 buoy)	
20		54° 59.17' N	1° 27.86' W	Tyne/middle estuary (Jarrow Slake)	
21		54° 58.92' N	1° 31.82' W	Tyne/middle estuary (Swan Hunter)	
22		54° 58.09' N	1° 36.25' W	Tyne/upper estuary (Tyne Bridge)	
23		55° 0.51' N	1° 8.61' W	Tyne/intermediate	14
24	26-Jun	55° 45.76' N	1° 57.72' W	Tweed/mouth	
25		55° 45.83' N	1° 59.58' W	Tweed/lower estuary (Spittal Point)	
26		55° 46.12' N	2° 0.35' W	Tweed/upper estuary (Tweed bridges)	
27		54° 50.13' N	1° 20.07' E	Offshore Tyne/Tees	43
28	27-Jun	55° 10.2' N	3° 9.13' E	Dogger Bank	47
29		54° 8.66' N	4° 20.1' E		50
30		54° 1.37' N	2° 40.0 4' E		
31		54° 1.3' N	2° 35.97' E		
32		54° 0.02' N	1° 59.79' E	Silver Pit	53
33	28-Jun	53° 20.04' N	1° 20.27' E		16 (benthos)
34		53° 3.52' N	0° 28.54' E	Wash	17
35		52° 58.08' N	0° 21.86' E		17 (benthos)
36		52° 29.91' N	1° 59.97' E		
37	29-Jun	52° 49.91' N	2° 49.83' E	Smiths Knoll	18
38		52° 0.02' N	2° 20.37' E	Outer Gabbard	25
39		51° 57.3' N	2° 40.31' E		B/NL Area 4
40		51° 49.8' N	2° 26.04' E		
41		51° 43.4' N	1° 7.84' E		
42		51° 30.84' N	0° 57.87' E	Thames Barrow	19
43	30-Jun	51° 22.53' N	1° 29.94' E	JMG307	
44		51° 29.97' N	2° 0.37' E	JMG319	
45		51° 22.17' N	1° 49.81' E	JMG306	
46		51° 14.79' N	1° 40.01' E	JMG295	
47		50° 56.29' N	1° 16.51' E	E. English Channel (South Varne)	69
48		50° 33.18' N	0° 0.0'	Perintis 1	
49		50° 34.83' N	0° 19.9' W	Perintis 2	
50		50° 35.03' N	0° 39.93' W	Perintis 3	
51		50° 38.96' N	0° 49.63' W	Isle of Wight/Perintis 4	70
52	1-Jul	50° 29.88' N	1° 19.84' W	Perintis 5	
53		50° 24.99' N	1° 40.17' W	Perintis 6	
54		50° 20.0' N	2° 0.15' W	Perintis 7	
55		50° 15.04' N	2° 20.2' W	Perintis 8	
56		50° 10.05' N	2° 39.86' W	Perintis 9	
57		50° 5.03' N	3° 0.26' W	Central Channel/Perintis 10	72
58		50° 7.49' N	3° 9.64' W		72 (benthos)
59		50° 5.13' N	3° 19.69' W	Perintis 11	
60		50° 5.1' N	3° 39.95' W	Perintis 12	
61		50° 4.96' N	4° 0.03' W	Perintis 13	
62		50° 1.91' N	4° 22.04' W	Western Channel/Perintis 14	73 (MBA E1)
63		49° 52.48' N	4° 30.07' W	JMG125	
64		49° 45.08' N	4° 39.68' W	JMG111	
65		49° 37.52' N	4° 49.48' W	JMG 90	
66	2-Jul	48° 30.17' N	7° 59.73' W	Western Approaches (MAFF reference stn.)	
67		50° 7.56' N	7° 29.91' W	JMG168	
68		50° 15.07' N	7° 19.94' W	JMG754	
69	3-Jul	50° 37.53' N	7° 10.25' W	JMG251	
70		50° 44.97' N	7° 0.14' W	JMG760	
71		50° 52.53' N	6° 50.05' W	JMG276	
72		51° 7.45' N	6° 50.06' W	JMG294	
73		51° 15.06' N	6° 20.11' W	JMG305	
74		51° 19.98' N	6° 19.93' W	Celtic Deep	

Table 9. Continued

Station no.	Date	Position	Location	NSTFMMP
75	3-Jul	51° 20.18' N 6° 0.17' W		
76		51° 15.09' N 6° 0.19' W	JMG304	
77		51° 15.07' N 5° 40.09' W	JMG303	
78		51° 14.81' N 5° 20.11' W	JMG302	
79		51° 14.96' N 5° 0.04' W	JMG301	
80		51° 15.18' N 4° 40.0' W	JMG300	
81		51° 15.13' N 4° 19.88' W	JMG299	
82		51° 15.09' N 3° 59.75' W	JMG298	
83		51° 14.99' N 3° 39.8' W	JMG297	
84		51° 19.0' N 3° 33.62' W	Bristol Channel	
85		51° 22.51' N 3° 50.08' W	JMG310	
86	4-Jul	51° 29.67' N 3° 54.92' W	Swansea Bay	
87		51° 29.93' N 4° 00.0' W	JMG324	
88		51° 29.74' N 4° 20.08' W	JMG325	
89		51° 36.64' N 4° 25.74' W	Carmarthen Bay	
90		51° 37.55' N 4° 29.89' W	JMG336	
91		51° 37.49' N 4° 50.01' W	JMG337	
92		51° 37.41' N 5° 10.3' W	JMG338	
93		51° 45.18' N 5° 20.01' W	Skomer Island/JMG347	
94		51° 59.96' N 5° 20.0' W	JMG361	
95		52° 7.38' N 4° 50.22' W	JMG366	
96		52° 10.03' N 4° 44.93' W	Cardigan	
97		52° 13.92' N 4° 21.43' W	New Quay	
98		52° 15.04' N 4° 19.93' W	JMG373	
99		52° 15.12' N 4° 40.0' W	JMG374	
100	5-Jul	52° 22.51' N 4° 50.02' W	JMG383	
101		52° 29.75' N 4° 59.93' W	Cardigan Bay	
102		52° 37.55' N 4° 50.01' W	JMG400	
103		52° 52.51' N 4° 50.06' W	JMG420	
104		53° 7.5' N 4° 50.2' W	Caernarvon Bay/JMG443	
105		53° 29.95' N 4° 19.55' W	JMG480	
106		53° 59.91' N 4° 0.24' W	Irish Sea offshore	
107		53° 44.95' N 3° 45.07' W	Irish Sea offshore	
108	6-Jul	53° 29.39' N 3° 17.14' W	River Mersey	
109		53° 25.0' N 3° 12.1' W	Dee/mouth (Hilbre Swash)	
110		53° 31.9' N 3° 9.3' W	Mersey (Alpha buoy)	
111		53° 30.5' N 3° 5.6' W	Mersey (C12 buoy)	
112		53° 28.0' N 3° 2.9' W	Mersey (C20 buoy)	
113		53° 26.4' N 3° 0.7' W	Mersey (Canada buoy)	
114		53° 58.66' N 3° 3.37' W	Morecambe Bay	
115		54° 1.1' N 2° 56.3' W	Morecambe Bay (No. 7 buoy)	
116		53° 58.6' N 3° 0.0' W	Morecambe Bay (River Lune)	
117		53° 55.65' N 3° 0.1' W	Morecambe Bay (River Wyre)	
118	10-Jun	50° 0.07' N 4° 39.75' W	JMG146	
119		50° 20.89' N 4° 9.01' W	Plymouth Sound	
120		50° 21.55' N 4° 8.28' W	West Mallard buoy	
121		50° 21.45' N 4° 9.9' W	West Vanguard buoy	
122		50° 21.85' N 4° 11.03' W	South Rubble buoy	
123	11-Jul	50° 20.89' N 4° 9.01' W	Plymouth Sound	
124		50° 24.37' N 4° 12.05' W	Royal Albert Bridge	
125		50° 21.85' N 4° 11.03' W	South Rubble buoy	
126		50° 21.45' N 4° 9.9' W	West Vanguard buoy	
127		50° 21.55' N 4° 8.28' W	West Mallard buoy	
128		50° 18.99' N 4° 9.27' W		
129	12-Jul	50° 21.75' N 3° 34.9' W	Dartmouth College	
130		50° 23.2' N 3° 35.5' W	Dittisham	
131		50° 18.68' N 3° 34.72' W	River Dart	
132		50° 7.38' N 3° 10.68' W	JMG157	
133		50° 7.56' N 2° 50.63' W	JMG156	
134		50° 7.4' N 2° 30.68' W	JMG155	
135		50° 14.98' N 2° 39.81' W	JMG180	
136		50° 22.37' N 2° 29.7' W	JMG201	
137	13-Jul	50° 45.47' N 1° 10.38' W	Mother Bank (Solent)	
138		50° 52.28' N 1° 22.68' W	NW Netley buoy	
139		50° 51.05' N 1° 20.33' W	Cadland buoy	
140		50° 50.12' N 1° 18.62' W	Hamble Point buoy	
141		50° 48.38' N 1° 17.0' W	Calshot buoy	
142	14-Jul		Fluxmanche 6	
143			Fluxmanche 5	
144			Fluxmanche 3	
145			Fluxmanche 4	
146			Fluxmanche 2	
147			Fluxmanche 1	

JMG = Joint Monitoring Group
B/NL = Belgium/Netherlands, Joint Station
MBA = Marine Biological Association

Table 10. Water column data for stations sampled during CIROLANA 6, 22 June-14 July 1991

Station	Depth (m)	Salinity	Temperature °C	Concentrations (mol l^{-1})			Sediment type ¶
				Phosphate	Silicate	Nitrate + Nitrite	
1	16	32.9	12.6	2.0	5.4	39	sand and clay
2		28.5		1.9	18.0	147	
3		28.6		0.56	17.3	134	
4		29.3		0.66	15.1	125	
5		30.6		1.3	11.5	93	
6		29.9		8.4	13.1	110	
7	15	30.4	13.2				
8	15	34.1	11.0	0.84	3.1	14.8	
9	48	34.5	10.8	0.81	3.0	4.6	fine sand
10	12	34.0	12.3	1.2	2.0	16.3	fine sand
11		31.6		5.2	8.0	82	sand
12		31.9		5.8	7.0	28	anoxic mud
13		26.9		6.4	15.6	61	anoxic sand
14	52	34.5	12.2	0.17	1.3	0.8	mud and sand
15	14	34.2	11.2	0.76	2.4	2.4	stones and sand
16		29.4		6.7	9.0	40	sand
17		18.1		19	17	118	anoxic mud
18	15	34.4	11.2	0.91	1.9	4.2	fine sand
19		ns		3.4	9.6	8.4	fine sand
20		23.8		9.8	44	30	anoxic mud
21		26.7		2.9	16.5	19.2	
22		18.6		3.9	30	34	
23	63	34.3		1.5	1.7	0.8	mud
24	17	33.8	10.6	0.92	1.9	1.2	fine sand
25		8.5		1.6	21	58	sand
26		1.6		2.3	24	70	mud
27	28	34.8	12.4	0.48	1.3	0.4	fine sand
28	30	35	12.7	0.22	0.6	0.3	fine sand
29	48	34.5	13.4	1.1	0.9	0.4	mud
30	64	ns					mud and sand
31	67	ns					mud and sand
32	72	34.6	12.5	1.2	0.9	0.4	sand and mud
33	24	34.0	12.2	1.8	2.8	16	sand and stones
34	32	33.8	13.3	1.0	2.4	17.5	stones, sand and shell
35	26	33.4	13.6	1.7	2.2	18.6	sand and stones
36	30	ns					
37	35	34.8	11.8	0.52	1.7	1.7	fine sand
38	48	35.0	12.6	0.38	1.0	1.4	sand and shell
39	45	34.8	12.8	0.52	1.0	1.3	sand
40	43	ns					
41	12	ns					mud
42	15	33.8	15.1	2.4	1.4	17.8	fine sand
43	15	ns					
44	43	ns					
45	40	ns					
46	47	ns					stones and sand
47	35	35.2	12.7	0.64	1.2	0.8	
48	61	35.2	12.7				
49	65	35.1	13.0				
50	24	35.1	13.3				
51	17	34.7	14.9	0.02	2.0	< 0.2	
52	39	35.0	13.6				
53	40	35.1	13.3				
54	52	34.8	12.7				
55	57	35.3	12.7				
56	58	35.2	12.6				
57	66	35.2	12.5	0.55	1.3	1.4	
58	66	35.3	12.8	3.1	2.1	3.3	sand and shell
59	68	35.3	12.8				
60	70	35.3	13.1				
61	71	35.1	14.1				
62	73	35.2	14.5	0.56	2	0.6	fine sand
63	79	ns					sand and shell
64	83	35.3	14.6				shell
65	88	35.3	14.8				shell
66	168 (at 2m)	35.5	14.5	0.6	0.7	0.4	sand and shell
	(at 20m)	35.5	14.3	0.85	1.1	1.0	
	(at 100m)	35.6	11.0	1.5	5.3	9.1	
	(at 150m)	35.6	11.0	0.7	4.8	8.0	
67	115	ns					mud
68	110	ns					mud and sand
69	102	ns					mud
70	101	ns					fine sand
71	96	ns					
72	96	ns					sand and mud
73	107	ns					mud
74	116	34.8	14.0	1.1	0.8	0.8	mud

Table 10. Continued

Station	Depth (m)	Salinity	Temperature °C	Concentrations (mol l^{-1})			Sediment type ¶
				Phosphate	Silicate	Nitrate + Nitrite	
75	100	34.9	14.9	0.13	0.7	0.3	mud
76	95	ns					mud
77	80	ns					fine sand
78	73	ns					fine sand
79	64	ns					fine sand
80	51	ns					
81	42	ns					stones and sand
82	32	ns					
83	24	ns					
84	30	32.1	15.0	2.5	6.8	43	
85	36	ns					
86	20	33.0	14.9	0.94	1.1	12.8	
87	30	ns					mud and stones
88	43	ns		0.19	1.6	2.2	sand and stones
89	27	34.0	14.6				sand and shell
90	28	ns					sand and mud
91	24	ns					mud and sand
92	51	ns					fine sand
93	55	ns					
94	60	ns					
95	42	ns					sand and mud
96	38	34.6	14.3	0.26	2.2	0.8	sand and shell
97	19	34.2	15.2	1.1	4.1	0.6	mud
98	19	ns					mud
99	34	ns					sand and stones
100	47	ns					sand and stones
101	78	34.8	12.6	0.21	1.1	1.0	fine sand
102	56	ns					sand and stones
103	59	ns					
104	48	ns					
105	47	ns					stones, sand and shell
106	46	34.0	14.2	0.73	1.1	0.6	sand and shell
107	43	33.2	15.0	0.73	1.2	0.4	sand, stones and shell
108	16	32.2	16.2	1.3	1.9	4.1	sand and shell
109		32.1		1.4	1.8	1.8	fine sand
110		30.8		4.2	4.3	33	
111		29.0		5.6	7.5	76	fine sand
112		27.8		6.6	9.1	106	mud
113		27.1		6.5	9.4	118	fine sand
114	22	32.9	17.3	1	1.4	0.6	fine sand
115		32.4		3.9	2.1	0.4	fine sand
116		32.7		1.3	1.5	0.6	fine sand
117		29.5		4.2	8.1	10.8	fine sand
118	74	ns					sand and shell
119	13	34.2	15.6	1.5	4.6	5.1	sand and shell
120		32.7		0.98	8.3	10	mud
121		33.5		0.56	4.8	6.9	
122		33.5		0.93	5.6	6.9	mud
123	10	33.7	15.6	0.76	3.2	4.0	sand
124		22.6		1.4	28	51	mud
125		31.2		1.1	8.7	17	
126		33.4		1.6	6.8	11	mud
127		33.0		0.86	6.2	9.1	mud
128	25	34.7		1.0	2.7	2.8	fine sand
129		ns					mud
130		ns					mud
131	21	35.2	13.7				sand and mud
132	61	ns					sand and shell
133	62	ns					stones and shell
134	62	ns					stones, shell and sand
135	59	ns					stones, shell and sand
136		ns					
137	22	34.3	17.3	0.35	3.3	1.3	mud and shell §
138		34.0		0.66	1.8	0.8	mud
139		33.5		0.25	4.4	1.5	mud
140		32.7		0.32	5.1	3.3	mud
141		33.8		0.25	1.8	1.9	mud
142	26	34.6	16.7	0.96	3.9	1.0	
143	53	35.2	14.0	0.18	1.0	0.6	
144	31	35.1	13.9	0.25	2.2	0.8	
145	31	35.1	13.6	0.7	1.9	0.4	
146	23	35.1	14.2	1.0	3.7	0.5	
147	20	35.1	14.6	0.42	4.6	1.0	

¶ = based on inspection of the sample at the time of sampling
§ = whole shells of Crepidula fornicata
ns = no sample

6. TRACE METALS (DISSOLVED AND ON SUSPENDED PARTICULATE MATERIAL)

6.1 Introduction

Coastal waters around England and Wales have been monitored for dissolved copper, cadmium and lead for a number of years (MAFF, 1990). During 1991-1992, analytical methods were changed to include the determination of zinc, nickel and manganese in sea water. Of these six trace elements, cadmium, copper, nickel, lead and zinc are of interest because their concentrations are affected by man's activities and manganese is significant because of its redox chemistry and its control on the geochemistry of other trace metals.

The partitioning of metals between sea water and suspended particulate matter (SPM) determines their ultimate fate in the marine environment. Therefore, concentrations of trace metals in SPM are also determined.

6.2 Methods

Samples were collected from the estuaries of the Rivers Tweed, Tyne, Wear, Tees, Humber and Mersey, and the Bristol Channel and offshore from the English Channel, the North Sea and the Irish Sea (see Section 4, Figure 7 and Table 9). They were collected from the upper 5 m of the water column using a buoyed device deployed away from the side of the vessel (Harper, 1987). The sea water was filtered (0.45 μm), and acidified (0.1% HNO_3) under clean conditions. Dissolved trace metals were determined by chelation/ solvent extraction/graphite furnace atomic absorption spectrometry (GFAAS), using a modification of the method described by Statham (1985). Suspended particulate metals were digested to dryness with hot concentrated nitric acid, taken up into 20% HNO_3, then analysed by GFAAS.

6.3 Results and discussion

6.3.1 Dissolved trace metals

The ranges and means of the concentrations of metals in each region are given in Table 11, and the data are illustrated in Figure 8. Offshore concentrations of dissolved trace metals were similar to those reported elsewhere, except for Mn where the levels were a little higher. This may be a seasonal effect; during the Natural Environment Research Council (NERC), North Sea Project it was evident that concentrations of Mn were elevated in some areas during spring and summer compared to those observed in winter, possibly due to enhanced benthic inputs.

Table 11. Ranges and means of concentrations of dissolved metals

Area	Station number	Cadmium (ng l^{-1})		Copper (g l^{-1})		Manganese (g l^{-1})	
		(range)	(mean)	(range)	(mean)	(range)	(mean)
Humber	1 to 6	49 to 221	(120)	0.75 to 2.80	(2.22)	0.8 to 55.3	(24.3)
Tees	10 to 13	20 to 42	(30)	1.30 to 10.30	(3.80)	13.4 to 56.5	(42.6)
Wear	15 to 17	13 to 25	(20)	0.33 to 1.30	(0.83)	2.5 to 85.0	(53.8)
Tyne	18 to 22	11 to 126	(80)	0.30 to 1.61	(1.03)	1.3 to 115.3	(52.0)
Tweed	24 to 26	7 to 11	(10)	0.58 to 4.67	(2.01)	5.0 to 12.9	(9.5)
North Sea	27 to 46	10 to 32	(18)	0.24 to 0.83	(0.48)	0.5 to 2.5	(1.3)
English Channel	47 to 62	11 to 18	(15)	0.20 to 0.48	(0.32)	0.8 to 1.4	(1.0)
Irish Sea	74 to 107	18 to 81	(30)	0.33 to 1.50	(0.69)	0.2 to 7.8	(2.1)
Mersey	108 to 113	9 to 52	(30)	1.40 to 3.30	(2.50)	0.1 to 10.8	(4.3)
Morecambe Bay	114 to 117	9 to 38	(22)	1.78 to 1.90	(1.83)	0.1 to 1.5	(0.5)
Overall Estuarine Mean		49		2.00		24.0	
Overall Offshore Mean		22		0.52		1.2	

Area	Station number	Nickel (g l^{-1})		Lead (ng l^{-1})		Zinc (g l^{-1})	
		(range)	(mean)	(range)	(mean)	(range)	(mean)
Humber	1 to 6	0.90 to 6.30	(3.09)	23 to 422	(210)	5.1 to 12.6	(9.3)
Tees	10 to 13	0.21 to 1.00	(0.70)	96 to 815	(460)	2.6 to 14.0	(8.4)
Wear	15 to 17	0.51 to 2.93	(1.61)	69 to 408	(290)	0.5 to 6.1	(3.6)
Tyne	18 to 22	0.36 to 2.75	(1.55)	86 to 1087	(590)	0.6 to 21.7	(14.5)
Tweed	24 to 26	0.48 to 0.81	(0.67)	96 to 169	(130)	0.6 to 1.9	(1.1)
North Sea	27 to 46	0.20 to 1.00	(0.54)	21 to 809	(103)	0.3 to 2.2	(0.8)
English Channel	47 to 62	0.22 to 0.47	(0.33)	23 to 32	(30)	0.2 to 1.0	(0.7)
Irish Sea	74 to 107	0.26 to 0.87	(0.48)	24 to 169	(70)	0.3 to 1.8	(0.8)
Mersey	108 to 113	0.80 to 9.40	(5.36)	32 to 464	(310)	1.5 to 15.6	(10.5)
Morecambe Bay	114 to 117	0.80 to 1.39	(0.98)	32 to 166	(80)	1.5 to 7.4	(3.4)
Overall Estuarine Mean		2.20		283		7.7	
Overall Offshore Mean		0.46		76		0.8	

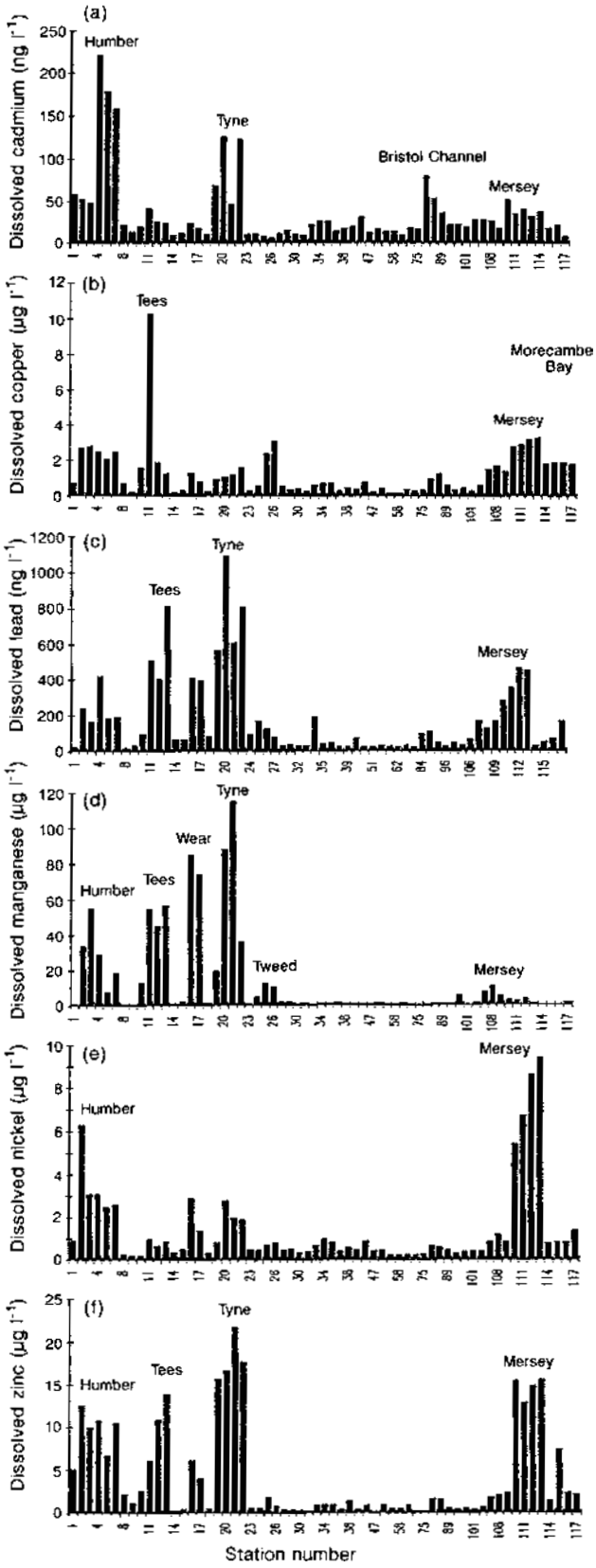

Figure 8. Concentration of dissolved trace metals by station number: (a) cadmium; (b) copper; (c) lead; (d) manganese; (e) nickel and (f) zinc

Offshore, concentrations of trace metals were generally lowest in the English Channel, as would be anticipated since a large proportion of the water passing through the Channel is relatively uncontaminated Atlantic Ocean water. The highest offshore concentrations of dissolved Mn, Cu and Cd were observed in the Irish Sea; for Pb the highest concentrations were seen in the North Sea, primarily it is thought as a result of atmospheric inputs.

Concentrations of trace metals in estuaries were very variable, but the mean values were higher than those seen offshore by a factor of 2 to 10 or more (Table 11). For all metals except Mn the lowest values were observed in the Tweed estuary; as mentioned previously unlike the other estuaries studied, this is not industrialised and was included for comparative purposes (MAFF, 1992(a)). The major sources of Mn are mineralogical, rather than anthropogenic. For all metals, the highest concentrations were seen in one or other of the industrialised estuaries: Humber (Mn and Cd), Tees (Cu and Pb), or Mersey (Ni and Zn).

In fulfilment of the EEC Dangerous Substances Directive for the reduction of water pollution, Environmental Quality Standards (EQSs) have been recommended by the Water Research Centre (WRc) for the Department of the Environment (DoE) (Gardiner and Mance, 1984).

The Environmental Quality Standard (EQS) for Zn (40 µg l^{-1}) was not exceeded in any estuary. The highest concentration observed for Pb (1.1 µg l^{-1} in the River Tyne) was only about one-tenth of the EQS (10 µg l^{-1}). For Cu, the EQS (5 µg l^{-1}) was exceeded at one of the three stations sampled in the Tees Estuary. However, since EQS values are set in terms of annual average concentrations, these data should not be regarded as implying that any EQS was breached.

6.3.2 Suspended particulate trace metals

The concentrations obtained are in general agreement with those reported elsewhere. The highest concentrations of copper and lead were observed in the Tyne Estuary, (Table 12). The mean estuarine concentrations for these two elements are much higher than the offshore means (Table 12). This is because of their anthropogenic origin, and high particle reactivity. Suspended particulate matter often remains trapped in estuaries by the water circulation pattern. This mechanism prevents the particle reactive elements from being transported to offshore areas.

Concentrations of cadmium in particulate matter were highest in the Tees and Tyne, presumably related to old mine workings and transport of metalliferous ores, but surprisingly low in the Humber Estuary where there are inputs of industrial origin (Table 12). Unlike lead,

cadmium is not a particle reactive element, and most of the cadmium discharged into the Humber estuary remains in the dissolved phase (Figure 8(a)). The concentrations of Cd in both the dissolved and suspended particulate phases were higher in the Irish Sea than the North Sea or English Channel. The Mersey Estuary, and the phosphate plant near St. Bees Head, north-eastern Irish Sea, are major sources of cadmium into the Irish Sea (MAFF, 1990).

Differences between estuarine and offshore concentrations were not as pronounced for nickel and zinc (Table 12) and only the values for manganese in the Tweed greatly exceeded the offshore mean. The high concentrations of particulate Mn in the Tweed are caused by oxidation/precipitation of dissolved Mn. Oxidation does not proceed to the same extent in estuaries with poorer water quality caused by higher biochemical oxygen demands. This may explain why high concentrations of dissolved Mn persist in the industrialised north east coast estuaries (Figure 8(d)). However, as solid Mn oxides/hydroxides do form, they scavenge other trace elements onto the particulate material. This is presumably why the concentrations of suspended particulate nickel are higher in the Tweed than in the industrialised estuaries (Table 12).

6.4 Conclusions

Concentrations of dissolved and suspended particulate trace metals around the coast of England and Wales were determined. Highest concentrations were observed in estuarine waters, and attributed to anthropogenic inputs from land-based sources. Concentrations of trace metals generally decreased rapidly with distance from the coast. Some of the spot sample concentrations from the industrialised estuaries were in the same range as the EQS values (set in terms of annual averages) and further monitoring effort should be deployed in these areas.

7. ALPHA- AND GAMMA-HEXACHLOROCYCLOHEXANE

Data for α- and γ-HCH in seawater samples collected in 1989 and 1990 have been reported previously (MAFF, 1991 and 1992(a)). Further samples were collected in 1991 during *RV CIROLANA* cruise 6, 22 June-14 July 1991, results are shown in Table 13 and full details of the stations occupied are given in Table 9, Section 4. A transect along the English Channel (Stations 48 to 62) was sampled as a continuation of studies undertaken following the loss of *MV PERINTIS* in March 1989, with a cargo including 5.8 tonnes of lindane (γ-HCH) (MAFF, 1991). Concentrations in this area were at background levels, generally less than 1 ng l^{-1}; although the concentration observed at Station 62 (1.8 ng l^{-1}) was somewhat higher than expected, as the station was approximately 25 miles offshore (Marine Biological Association (MBA) Station E1; NSTF73). Another series of samples were taken along a Channel transect in the summer of 1992 in order to investigate this further, and the results will be reported later.

Table 12. Ranges and means of concentrations of suspended particulate metals

Area	Station number	Cadmium (mg kg^{-1})		Copper (mg kg^{-1})		Manganese (mg kg^{-1})	
		(range)	(mean)	(range)	(mean)	(range)	(mean)
Humber	1 to 6	0.11 to 0.40	(0.29)	36.3 to 54.5	(46.1)	819 to 1424	(1140)
Tees	10 to 13	0.64 to 5.44	(2.29)	60.9 to 139.1	(106.3)	269 to 677	(389)
Wear	15 to 17	0.15 to 1.25	(0.85)	23.3 to 82.4	(55.1)	804 to 2136	(1628)
Tyne	18 to 22	0.27 to 6.05	(2.50)	27.0 to 200.0	(100.0)	228 to 895	(414)
Tweed	24 to 26	0.49 to 0.78	(0.59)	36.6 to 107.1	(71.6)	5970 to 6600	(6290)
North Sea	27 to 46	0.10 to 1.39	(0.46)	17.8 to 43.2	(27.9)	183 to 1932	(736)
English Channel	47 to 62	0.26 to 0.71	(0.49)	13.0 to 18.0	(15.5)	131 to 499	(330)
Irish Sea	74 to 107	0.43 to 1.31	(0.75)	8.0 to 84.6	(24.3)	243 to 1380	(570)
Mersey	108 to 113	0.49 to 1.20	(0.74)	24.1 to 56.0	(35.5)	282 to 972	(731)
Morecambe Bay	114 to 117	0.32 to 0.53	(0.40)	11.7 to 31.0	(21.3)	503 to 1208	(771)
Overall Estuarine Mean		1.09		57.8		1206	
Overall Offshore Mean		0.63		25.8		632	

Area	Station number	Nickel (mg kg^{-1})		Lead (mg kg^{-1})		Zinc (mg kg^{-1})	
		(range)	(mean)	(range)	(mean)	(range)	(mean)
Humber	1 to 6	42.3 to 50.8	(46.0)	140 to 189	(163)	314 to 496	(427)
Tees	10 to 13	19.3 to 28.4	(24.0)	109 to 317	(203)	188 to 1101	(494)
Wear	15 to 17	24.1 to 49.0	(36.9)	43 to 508	(306)	139 to 860	(573)
Tyne	18 to 22	26.9 to 50.3	(37.0)	44 to 653	(331)	80 to 1406	(723)
Tweed	24 to 26	45.2 to 69.2	(56.2)	51 to 118	(81)	468 to 821	(553)
North Sea	27 to 46	14.4 to 63.0	(26.1)	16 to 88	(53)	73 to 1059	(463)
English Channel	47 to 62	19.8 to 23.5	(21.7)	17 to 35	(29)	76 to 315	(180)
Irish Sea	74 to 107	12.4 to 38.1	(24.8)	19 to 101	(46)	141 to 984	(211)
Mersey	108 to 113	19.0 to 55.0	(32.0)	33 to 144	(93)	310 to 454	(418)
Morecambe Bay	114 to 117	31.0 to 62.0	(45.3)	85 to 129	(102)	304 to 351	(327)
Overall Estuarine Mean		39.0		180		496	
Overall Offshore Mean		25.4		49		326	

Table 13. Hexachlorocyclohexanes (HCH) (ng l^{-1}) in sea water (Station details are given in Table 9, Section 4)

Station	α-HCH	γ-HCH		Station	α-HCH	γ-HCH	
1	0.3	2		47	< 0.1	1.7	
2	0.4	5.3		48	< 0.1	1.0	
3	0.4	4.7		49	< 0.1	1.0	
4	0.4	4.5		50	< 0.1	1.0	
5	0.4	4.2		51	< 0.1	1.8	
6	0.3	3.8		52	< 0.1	< 0.1	
8	0.4	1.2		53	< 0.1	0.55	
9	0.3	0.7		54	< 0.1	0.45	
10	0.3 - 0.5	0.7 - 1.0	(taken during a tidal cycle)	55	< 0.1	0.24	
11	0.5	2.8		56	< 0.1	0.17	
12	0.4	1.7		57	< 0.1	0.26	
14	0.3	0.5		58	< 0.1	0.18	
15	0.3	0.7		59	< 0.1	0.26	
16	0.4	3.6		60	< 0.1	0.41	
17	0.3	7.1		61	0.17	0.96	
18	0.4	0.6		62	0.31	1.8	
19	0.4	4.8		74	< 0.1	0.29	
20	0.4	16		84	< 0.1	0.33	
21	0.17	2.2		86	< 0.1	0.24	
22	0.19	3.5		89	< 0.1	0.27	
23	< 0.1	0.4		96	< 0.1	0.31	
24	0.2	0.4		97	< 0.1	0.32	
25	0.3	2.1		101	< 0.1	0.38	
26	0.2	2.2		106	< 0.1	0.25	
27	0.2	0.6		107	< 0.1	0.41	
28	0.2	0.4		108	< 0.1 - 0.33	0.29 - 1.7	(taken during a tidal cycle)
29	0.2	1.0		109	0.26	1.4	
32	0.2	0.7		110	1.2	6.9	
33	0.2	1.0		111	2.2	10	
34	0.2	1.3		112	2.7	12	
35	0.2	1.9		113	2	7.9	
37	< 0.1	0.9		114	0.16	0.72	
38	< 0.1	1.9		115	< 0.1	0.54	
39	< 0.1	2.0		116	< 0.1	0.95	
42	< 0.1	2.8		117	< 0.1	1.8	

Occasional high values were seen in estuaries; 16 ng l^{-1} in the Tyne and 10 and 12 ng l^{-1} in the Mersey. Mean values for both α- and γ-HCH in estuarine areas were higher in 1991 than in 1990, although none breached the EQS value for estuarine and sea water of 20 ng l^{-1} (ENDS, 1992). However, in 1991 concentrations of lindane in three samples did exceed the 10 ng l^{-1} concentration, below which there is unlikely to be a threat to most species of marine life (Portmann, 1982).

8. TRIAZINE HERBICIDES (SIMAZINE AND ATRAZINE)

8.1 Background

Simazine (6-chloro-N^2,N^4-diethyl-1,3,5-triazine-2,4-diamine) and atrazine (6-chloro-N^2-ethyl-N^4-isopropyl-1,3,5-triazine-2,4-diamine) (Chemical Abstracts Service (CAS) registry numbers 122-34-9 and 1912-24-9 respectively) have been widely used as pre- and post-emergent herbicides for both agricultural and non-agricultural use. The non-agricultural use of these chemicals has recently declined and with effect from 1 September 1992, MAFF has revoked approval for this use under the Food and Environment Protection Act (FEPA), 1985 (Great Britain - Parliament, 1985(a)), Control of Pesticides Regulations 1986.

Both atrazine and simazine are on the UK Red List (Anon., 1990) of substances for which the UK Government is committed to reduce inputs to the marine environment by 50%, by the year 1992. Triazines are also included in the list of additional determinands of the Marine Pollution Monitoring Management Group (MPMMG) UK National Monitoring Plan (NMP) and of the North Sea Task Force (NSTF) for study in marine waters. Residual concentrations of both simazine and atrazine are commonly found in raw and potable drinking water in the UK, and have occasionally broken the 0.1 μg l^{-1} Maximum Admissible Concentration (MAC) as specified in the Council of European Communities Directive relating to the quality of water intended for human consumption (80/778/EEC) (Friends of the Earth, 1988). As no baseline data currently existed for these

compounds in the coastal waters of England and Wales, it was decided to conduct a pilot survey. Samples were collected during three research cruises in 1990-92, and the results for the first two surveys in 1990 and 1991 are presented here.

8.2 Analytical methods

The full method will be published elsewhere. Briefly, 2.7 l samples of sea water were collected from 1 m below the surface and extracted with two 50 ml aliquots of dichloromethane. These were pooled and dried and the solvent exchanged for 20% diethyl ether in n-hexane. The extracts were then cleaned-up and fractionated on a column of 10% deactivated alumina. Internal standards were added and residues of simazine, atrazine and the dealkylated breakdown products (des-ethyl and des-isopropyl atrazine) were determined using capillary gas chromatography/mass spectrometry on a Finnigan ITS-40 Ion Trap instrument. The limit of detection was 0.6 ng l^{-1} for simazine and atrazine, and 1 ng l^{-1} for both des-ethyl and des-isopropyl atrazine.

8.3 Results and discussion

The results from the first two surveys undertaken on *RV CIROLANA* during cruise 6, 5-27 June 1990 and cruise 6, 22 June-14 July 1991 are presented in Tables 14 and 15.

RV CIROLANA cruise 6/90

Thirty-four samples were analysed (Table 14); the dealkylated breakdown products were not detected in any of the samples. Simazine was detected in 29 of the samples. The maximum concentration recorded was 34 ng l^{-1} in a sample taken at the upstream sampling station in the River Wear. This declined to 1 ng l^{-1} at the mouth of the estuary. Similar patterns were seen for the Rivers Tyne (maximum concentration 19 ng l^{-1}), Tees (4.7 ng l^{-1}), Humber (30 ng l^{-1}), Mersey (18 ng l^{-1}) and Tweed (5.1 ng l^{-1}). Samples taken offshore were generally close to or below the limit of detection. Atrazine was generally found at higher concentrations than simazine but a similar distribution pattern was seen, with the highest concentrations observed at the upstream stations in the estuaries. The maximum concentrations seen were: River Tweed (21 ng l^{-1}), Tyne (67 ng l^{-1}), Wear (33 ng l^{-1}), Tees (5.7 ng l^{-1}), Humber (37 ng l^{-1}) and Mersey (18 ng l^{-1}). As for simazine, concentrations of atrazine declined rapidly with distance offshore, but it was detectable at all but one station in the Dover Strait (Station 30; NSTF Station 69).

RV CIROLANA cruise 6/91

Thirty-seven samples were analysed. The results show a similar pattern to those obtained in 1990. Although the sampling sites were not identical in the two surveys, and a number of other factors need to be taken into account, it is interesting to note that generally concentrations were lower in 1991 than 1990. For example, simazine was not detected in samples from the River Tees, which included a 13 h tidal station near the estuary mouth. This apparent decline in concentrations was not so marked for atrazine, which as in 1990, was detected at all but one station. However, concentrations were again generally lower.

8.4 Conclusions

To the best of our knowledge, these are the first reported data for simazine and atrazine in estuarine and full salinity water samples from the UK. Therefore, no comparable data are available. Much higher concentrations of dissolved atrazine have been reported from Chesapeake Bay in the USA, with 480 g l^{-1} observed in runoff (Forney, 1980) and concentrations up to 46 g l^{-1} in estuarine water (Kemp *et al*., 1985), and up to 15 g l^{-1} in the estuary of the Wye River in Maryland (Glotfelty *et al*., 1984). However, agricultural usage of atrazine is much higher in the USA than in the UK. Tronczynski (1991) in an overview of atrazine in the estuarine environment, also commented on the paucity of data available, but reported levels of 10-60 ng l^{-1} of atrazine for the Rhone estuary in France, very similar to the range reported here for estuaries in England and Wales.

The UK is obliged under the EC 'dangerous substances Directive' (European Communities, 1976) to set environmental quality standards (EQSs) for selected substances and to derive EQS values for Red List substances. The proposed EQS for the protection of saltwater life for combined atrazine and simazine is 2 µg l^{-1} expressed as an annual average value. A maximum allowable concentration of 10 µg l^{-1} has also been proposed. Clearly the results of these two surveys indicate that although simazine and atrazine can be detected in full salinity samples, the concentrations found are well below the proposed EQS. Therefore, it is unlikely that atrazine and simazine would have any biological impact in estuarine and coastal waters of England and Wales. Given the recent restrictions on usage it is thought that concentrations are unlikely to increase and indeed should decrease in the future.

Table 14. Stations sampled for simazine and atrazine during CIROLANA 6, 5-27 June 1990

Station	Date	Position	Location	NSTF no.	Concentration (ng l^{-1})			
					Simazine	Atrazine	Des-ethyl Atrazine	Des-isopropyl Atrazine
1	6-Jun	55° 45.7' N 1° 57.85' W	Tweed/mouth		0.6	1	<1	<1
2		55° 45.83' N 1° 59.58' W	Tweed/lower estuary		4.1	20	<1	<1
3		55° 46.12' N 2° 0.35' W	Tweed/upper estuary		5.1	21	<1	<1
5	7-Jun	55° 0.41' N 1° 23.76' W	Tyne/mouth		<0.6	1.5	<1	<1
6		55° 0.63' N 1° 24.98' W	Tyne/lower estuary		9.7	42	<1	<1
7		54° 59.17' N 1° 27.86' W	Tyne/middle estuary		14	41	<1	<1
8		54° 58.92' N 1° 31.82' W	Tyne/middle estuary		19	51	<1	<1
9		54° 58.09' N 1° 36.25' W	Tyne/upper estuary		18	67	<1	<1
11	8-Jun	54° 54.88' N 1° 20.3' W	Wear/mouth		1	2.1	<1	<1
12		54° 55.04' N 1° 21.62' W	Wear/lower estuary		10	11	<1	<1
13		54° 54.58' N 1° 22.87' W	Wear/upper estuary		34	33	<1	<1
15	9-Jun	54° 38.93' N 1° 7.35' W	Tees/mouth		1.4	2.1	<1	<1
16		54° 38.52' N 1° 8.66' W	Tees/lower estuary		<0.6	3.8	<1	<1
17		54° 37.12' N 1° 9.32' W	Tees/middle estuary		4.7	4.2	<1	<1
18		54° 36.22' N 1° 9.9' W	Tees/upper estuary		<0.6	5.7	<1	<1
23	11-Jun	53° 33.33' N 0° 6.37' E	Humber/mouth		14	21	<1	<1
24		53° 37.1' N 0° 2.82' W	Humber/lower estuary		30	37	<1	<1
25		53° 36.0' N 0° 3.0' W	Humber/upper estuary		29	30	<1	<1
26		53° 3.65' N 0° 28.88' E	Wash	17	1.5	2.9	<1	<1
27	12-Jun	52° 50.4' N 2° 50.1' E	E. Smiths Knoll	18	<0.6	<0.6	<1	<1
29		51° 30.34' N 0° 57.88' E	Thames Barrow	19	2.1	2.2	<1	<1
30		50° 56.1' N 1° 17.05' E	E. English Channel	69	<0.6	<0.6	<1	<1
46	15-Jun	51° 18.87' N 3° 33.14' W	Bristol Channel		3	4.6	<1	<1
47		51° 29.72' N 3° 54.89' W	Bristol Channel		1.9	3.8	<1	<1
48		51° 36.56' N 4° 25.77' W	Bristol Channel		1.1	2	<1	<1
49		51° 38.7' N 5° 7.6' W	Bristol Channel		1.1	1.9	<1	<1
50	16-Jun	52° 10.13' N 4° 44.7' W	Cardigan Bay		1	1.9	<1	<1
51		52° 13.92' N 4° 21.48' W	Cardigan Bay		1.3	2	<1	<1
53		52° 40.9' N 4° 25.9' W	Cardigan Bay		1.1	2.2	<1	<1
54	17-Jun	53° 32.7' N 3° 15.3' W	Mersey		18	18	<1	<1
64		53° 55.6' N 3° 12.4' W	Morecambe Bay		3.1	4.8	<1	<1
66		54° 1.1' N 2° 56.26' W	Morecambe Bay		5.4	7.3	<1	<1
67		53° 58.6' N 3° 0.0' W	Morecambe Bay		4.8	6	<1	<1
68		53° 55.68' N 3° 0.2' W	Morecambe Bay		3.2	8.6	<1	<1

Table 15. Stations sampled for simazine and atrazine during CIROLANA 6, 22 June -14 July 1991

Station	Date	Time	Position	Location	NSTF no.	Concentration (ng l^{-1})			
						Simazine	Atrazine	Des-ethyl Atrazine	Des-isopropyl Atrazine
1	22-Jun		53° 33.33' N 0° 6.05' E	Humber/mouth		7.2	11	<1	<1
2			53° 38.6' N 0° 11.0' W	No. 11A buoy		9.8	25	<1	<1
3			53° 37.4' N 0° 8.7' W	No. 10A buoy		9.6	25	<1	<1
4			53° 36.1' N 0° 5.9' W	Diffuser		9.2	26	<1	<1
5			53° 36.6' N 0° 4.6' W	No. 6B buoy		9.3	21	<1	<1
6			53° 36.0' N 0° 3.6' W	Humber/upper estuary		8	24	<1	<1
8			53° 31.9' N 0° 20.16' E	Humber/intermediate	16	4.2	3.9	<1	<1
9			53° 55.06' N 0° 50.18' E	Off the Humber		<0.6	2.5	<1	<1
10	24-Jun	7:30	54° 39.05' N 1° 7.28' W	Tees/mouth		<0.6	6.1	<1	<1
		8:30				<0.6	2.4	<1	<1
		9:30				<0.6	2.2	<1	<1
		10:30				<0.6	7.6	<1	<1
		11:30				<0.6	9	<1	<1
		12:30				<0.6	7.3	<1	<1
		13:30				<0.6	5.9	<1	<1
		14:30				<0.6	4.8	<1	<1
		15:30				<0.6	5.4	<1	<1
		16:30				<0.6	6.9	<1	<1
		17:30				<0.6	4.8	<1	<1
		18:30				<0.6	4.8	<1	<1
12			54° 37.12' N 1° 9.32' W	Tees/middle estuary		<0.6	12	<1	<1
14			54° 44.06' N 0° 53.03' W	Tees/intermediate	15	5.1	3.2	<1	<1
15	25-Jun		54° 54.95' N 1° 20.27' W	Wear/mouth		5.6	9.6	<1	<1
18			55° 00.35' N 1° 23.69' W	Tyne/mouth		5.5	7.3	<1	<1
21			54° 58.92' N 1° 31.82' W	Tyne/middle estuary		<0.6	5.7	<1	<1
23			54° 0.51' N 1° 8.61' W	Tyne/intermediate	14	10	37	<1	<1
24	26-Jun		55° 45.76' N 1° 57.72' W	Tweed/mouth		5.4	3.5	<1	<1
86	4-Jul		51° 29.67' N 3° 54.92' W	Swansea Bay		0.8	1.6	<1	<1
87			51° 29.93' N 4° 00.0' W	JMG324		0.7	1.1	<1	<1
88			51° 29.74' N 4° 20.08' W	JMG325		0.93	<0.6	<1	<1
108	6-Jul		53° 29.39' N 3° 17.14' W	River Mersey		6.8	6.3	<1	<1
109			53° 25.0' N 3° 12.1' W	Dee/mouth		8.7	11	<1	<1
110			53° 31.9' N 3° 9.3' W	Mersey (Alpha buoy)		11	17	<1	<1
111			53° 30.5' N 3° 5.6' W	Mersey (C12 buoy)		12	30	<1	<1
112			53° 28.0' N 3° 2.9' W	Mersey (C20 buoy)		11	23	<1	<1
113			53° 26.4' N 3° 0.7' W	Mersey (Canada buoy)		12	29	<1	<1
114			53° 58.66' N 3° 3.37' W	Morecambe Bay		5.1	2.6	<1	<1

9. HYDROCARBONS

Concentrations of total hydrocarbons (THCs) were determined in subsurface (1 m) water samples using ultra-violet fluorescence spectrometry, as described previously (Law *et al*., 1988). Samples were quantified against Ekofisk crude oil at excitation and emission wavelengths of 310 and 360 nm respectively, and synchronous excitation-emission spectra ($\Delta\lambda$ = 25 nm) were recorded for all samples. A number of the extracts yielding the highest THCs were also analysed by gas chromatography. Low THCs were recorded offshore and in coastal areas away from estuaries (Table 16), often below the detection limit of 0.2 or 0.3 g l^{-1}. The maximum THCs observed in the estuaries visited were : River Humber, 9.3 g l^{-1} (Station 2); River Mersey, 11 g l^{-1} (Station 112); River Tees, 48 g l^{-1} (Station 13); River Tyne, 31 g l^{-1} (Station 20); River Wear, 13 g l^{-1} (Station 17). These concentrations are very similar to those reported previously (MAFF, 1992(a)); Table 17 shows the data for the Rivers Tyne and Mersey, in which the same stations were sampled in 1990 and 1991 (see Figures 9 and 10 for locations). Extracts analysed by gas chromatography were those from Stations 12, 13, 15, 19, 20, 21, 22 and 113. None of these resembled fresh oil — a typical chromatogram is

Table 16. Total hydrocarbon concentrations in subsurface water (g l^{-1} Ekofisk crude oil equivalents) (see Table 9, Section 4 for station details)

Station	Type §	THC
2	E	9.3
3	E	6.7
4	E	6.7
5	E	3.8
6	E	3.8
8	O	< 0.3
9	O	< 0.3
10	E	2.8
11	E	18
12	E	19
13	E	48
14	O	< 0.3
15	E	10
16	E	ns
17	E	13
18	E	0.5
19	E	20
20	E	31
21	E	15
22	E	28
23	O	0.6
24	E	< 0.3
42	O	1.2
51	O	0.4
57	O	< 0.2
58	O	< 0.2
62	O	0.4
74	O	1.4
84	O	3.6
86	O	1.7
89	O	0.7
96	O	< 0.2
97	O	0.7
101	O	0.3
106	O	0.4
107	O	0.7
108	E	1.3
109	E	2.1
110	E	6.1
111	E	8.3
112	E	11
113	E	9.8

§ = estuarine / offshore
ns = no sample

Table 17. Concentrations of total hydrocarbons in subsurface water (g l^{-1} Ekofisk crude oil equivalents) in the Rivers Tyne and Mersey, 1990/91.

Location		Sampling date			
		7 Jun.90	17 Jun.90	25 Jun.91	6 Jul.91
River Tyne:	mouth	0.7		0.5	
	no. 2 buoy	16		20	
	Jarrow Slake	17		31	
	Wallsend	18		15	
	Tyne Bridge	22		28	
River Mersey:	mouth		2.1		1.3
	alpha buoy		7.3		6.1
	C12 buoy		8.4		8.3
	C20 buoy		10		11
	Canada buoy		12		9.8

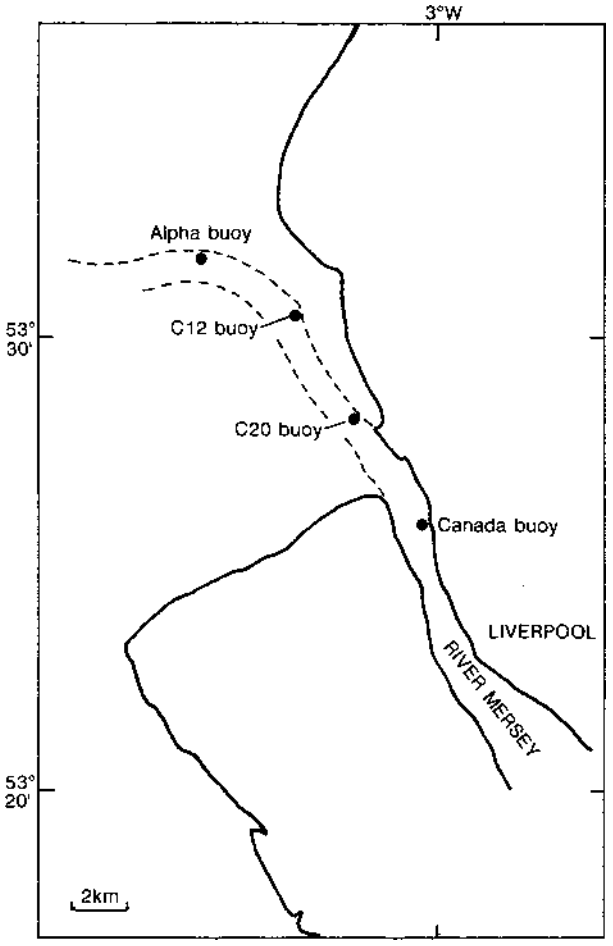

Figure 9. Sampling stations in the River Mersey

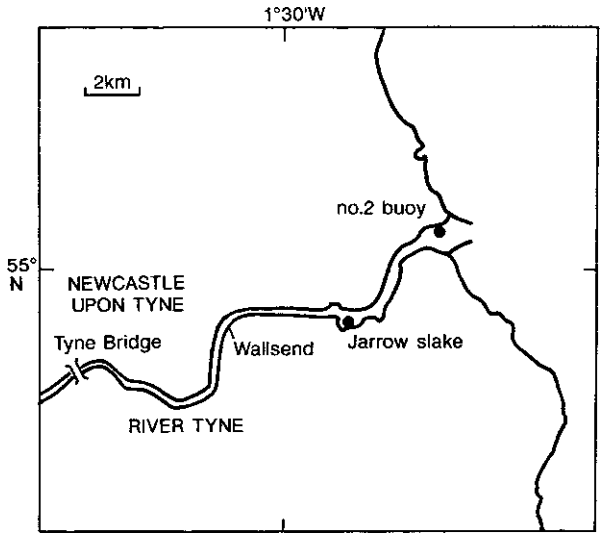

Figure 10. Sampling stations in the River Tyne

shown at Figure 11(a), from Station 113 (Canada buoy, River Mersey). The lack of a prominent series of n-alkane peaks (as seen in Figure 11(b) — a chromatogram of fresh diesel oil) suggests that the source oil has been considerably degraded since discharge. Such consistency in hydrocarbon concentrations and lack of evidence of fresh oil inputs suggests that these concentrations arise largely as a result of chronic, low-level inputs rather than episodic spillages.

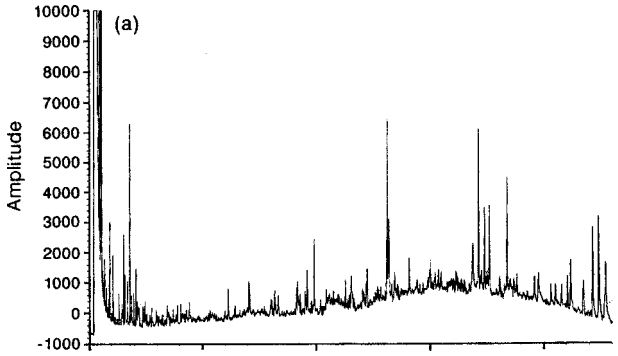

Figure 11(a). Gas chromatogram of a sub-surface water sample from Station 113 (Canada buoy, River Mersey)

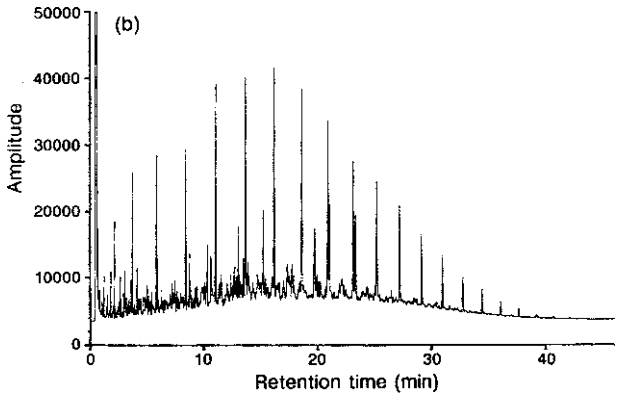

Figure 11(b). Gas chromatogram of fresh diesel oil

Concentrations of a range of polycyclic aromatic hydrocarbons (PAH) have been determined in water samples taken at two depths (1 m subsurface and 3 m above the seabed), at each of the six stations established in the Dover Strait under the EC funded FLUXMANCHE programme (Figure 12). Sampling was

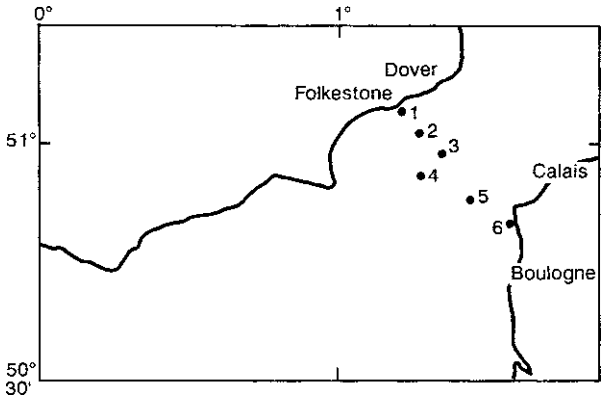

Figure 12. Fluxmanche station positions

carried out on three occasions; in June and October 1990, and July 1991. Analyses were conducted using coupled capillary gas chromatography/mass spectrometry in the multiple-ion detection mode, utilising the parent ions of each of the PAH studied (Law and Whinnett, in press). Naphthalene and phenanthrene were detected in all samples analysed, at concentrations between 0.7 and 18; and 0.9 and 9.3 ng l^{-1} respectively. The larger PAHs analysed (compounds with molecular weights of 228, 252 and 276) were found in only four samples (three of them from near the seabed) and probably reflect a high load of suspended particulates, as these compounds are easily adsorbed to particles. In all, concentrations of individual PAHs and groups of alkylated PAHs ranged from not detected (limits of detection in the range 0.2 to 2 ng l^{-1}) to 410 ng l^{-1}; with alkylated naphthalenes predominating in the majority of samples. During sampling surface oil sheens and small slicks were often apparent, and capillary GC/FID traces of the samples after GC/MS analysis were often similar to that in Figure 13. This shows a chromatogram of the subsurface sample from Fluxmanche Station 4 (July 1991) and clearly indicates

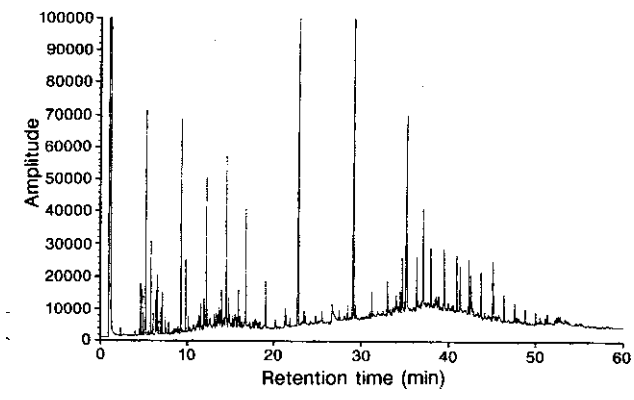

Figure 13. Gas chromatogram of a sub-surface water sample from Fluxmanche Station 4

the presence of petroleum-derived hydro-carbons, in the boiling ranges nC_{11} to nC_{18} and nC_{21} to nC_{36} (NB: the three large peaks in the centre of the chromatogram are the deuterated internal standards added to aid GC/MS quatification). The latter group (nC_{21} to nC_{36}) probably arise from medium or heavy fuel oils as are used in ships; the former, lighter, group lie within the range of kerosenes and light diesels, and may derive from the smaller ferries operating across the Channel (such as hovercraft and jet-foils).Results therefore indicate local inputs of oil from passing shipping in the Straits of Dover. This is one of the world's busiest shipping areas, with major traffic lanes running between the southwest and the northeast connecting the North Atlantic with the main European harbours such as Europoort, and a thriving ferry traffic across the English Channel between ports including Dover, Ramsgate, Calais and Boulogne. It is estimated that, on average, 500 ships pass through this stretch of water each day: 300 through and 200 across the Strait (Captain WJM Hargreaves, Department of Transport; pers. comm.). Therefore, the concentrations of PAH determined in this study presumably reflect those present in English Channel water, with an additional component derived from inputs of oil from ships operating in the Dover Strait.

SEDIMENTS

10. BASELINE SEDIMENT SURVEY FOR THE OSPARCOM JOINT MONITORING GROUP

10.1 Introduction

As part of an international monitoring programme carried out for the Joint Monitoring Group and North Sea Task Force, seabed sediment samples were collected around the UK in 1990 and 1991. This section mainly discusses the results produced from the English sector of the North Sea but includes results from two sample stations located in Dutch waters where replicate samples were collected for intercomparison purposes.

This project was carried out by MAFF and the University of Cambridge; it was jointly funded by MAFF and DoE.

10.2 Method

Seabed sediments were sampled at 274 locations in UK waters between the Dover Straits and Berwick-on-Tweed (Figure 14). The collection and analytical methods used in this study followed ICES advice (ICES, 1987).

Most samples were collected between May 1990 and December 1991 from the MAFF ships *RV CORYSTES* and *RV CIROLANA*, using a Day grab fitted with stainless steel jaws. Where the substrate was either too hard or too gravelly for the effective use of this device, a stainless steel Shipek grab was employed instead.

Approximately 1 kg of the surface (0-1 cm) layer of sediment was taken using a polyethylene scoop and stored in a polyethylene container at -18^{o}C. On return to the laboratory the samples were defrosted, sub-sampled wet and about 150 g freeze dried, the remainder being returned to an archive freezer. Each freeze-dried sub-sample was sieved at 2 mm to remove gravel particles and any large detritus. The sub-sample was then split to yield a 30 g sub-sample which was ground to a powder using a mechanical agate mortar and pestle.

500 mg of the powder was completely digested in a microwave furnace using a mixture of hydrofluoric acid and *aqua regia* (Jones, in press). The resulting solution was treated to minimise matrix effects before being analysed for a range of elements (Al, Cd, Cr,

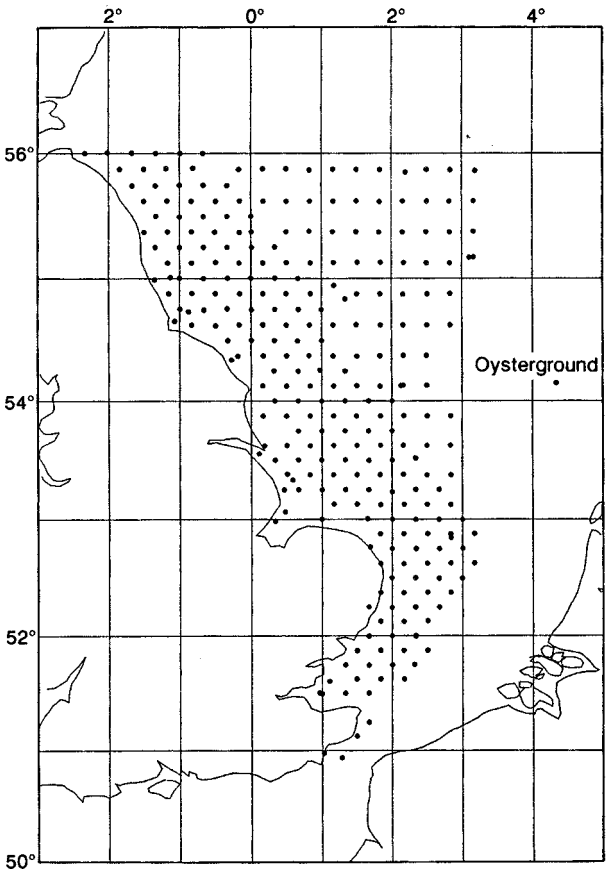

Figure 14. Sampling sites used in the JMG baseline sediment survey 1990/91

Cu, Fe, Li, Mn, Ni, Pb and Zn) using flame and furnace atomic absorption spectrophotometry. Mercury was also measured, using an atomic fluorescence technique.

Clay minerals have a relatively high natural metal content compared to sand and also a greater active surface area which may adsorb metals. This causes concentrations of sediment metals to be generally higher in areas of mud than in sandy regions. The effect is demonstrated in Figure 15, where the muddier coastal and estuarine samples can be seen to contain higher concentrations of zinc. Such mineralogical and grain-size effects can confound attempts to compare concentrations of metals throughout a given area, but may be taken into account by using a normalisation procedure (ICES, 1987, Loring, 1991).

In the present context, normalisation requires the use of a non-contaminant element associated with clay minerals to account for mineralogical variations. Figures 16 and 17 show the distributions of two potential normalising elements; aluminium and lithium (Loring, 1991). Both of these elements are present in clay minerals and may act as a surrogate for the clay size fraction. In the present study more data are available for aluminium, which is therefore used as normaliser.

Several methods of normalisation are possible; that chosen for the present study is based on a metal/ normaliser regression model with calculation of residuals about the regression line (Figure 18). The residual is the distance between a data point and the regression of variable (i.e. element of interest) and normaliser (in this case aluminium). The charts prepared from these residual data use symbols representing percentile classes with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent. This results in greater clarity and ease of identification of extreme values.

10.3 Results

Table 18 lists the basic statistical properties of the variables Al, Li, Cd, Cr, Hg, Pb and Zn.

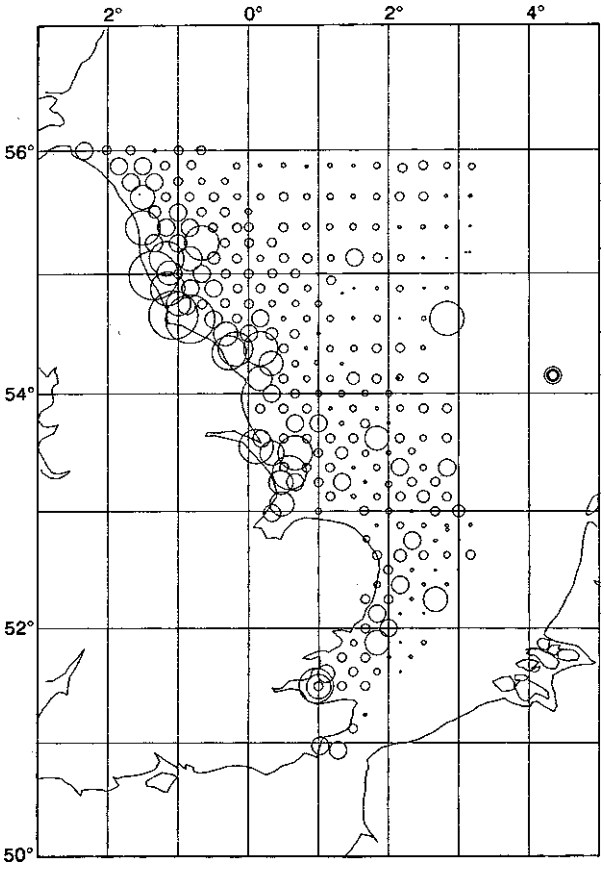

Figure 15. Distribution of concentrations of zinc in the western North Sea expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles. (Concentric circles indicate a number of samples from the same site)

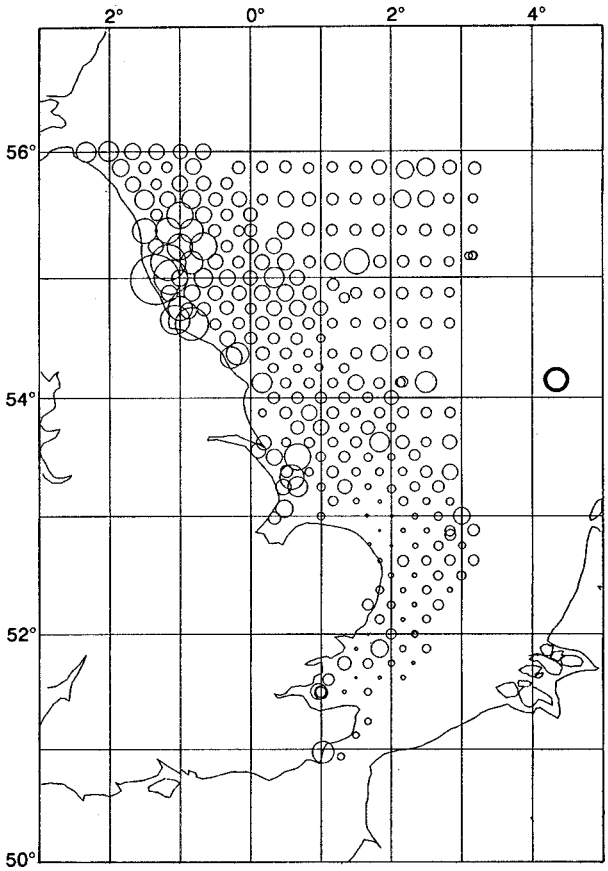

Figure 16. Distribution of concentrations of aluminium in the western North Sea expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles. (Bold circle indicates a number of samples taken from the same site, giving the same values)

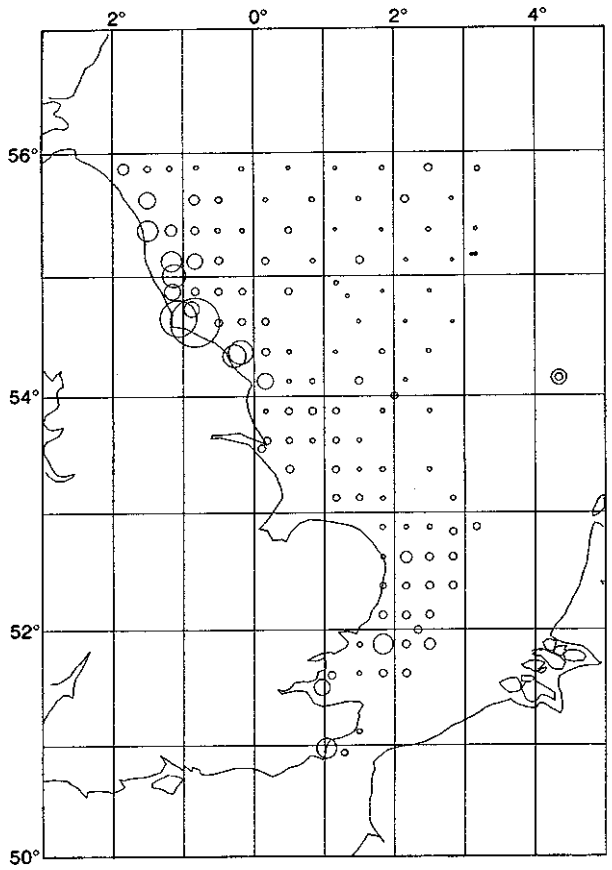

Figure 17. Distribution of concentrations of lithium in the western North Sea expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

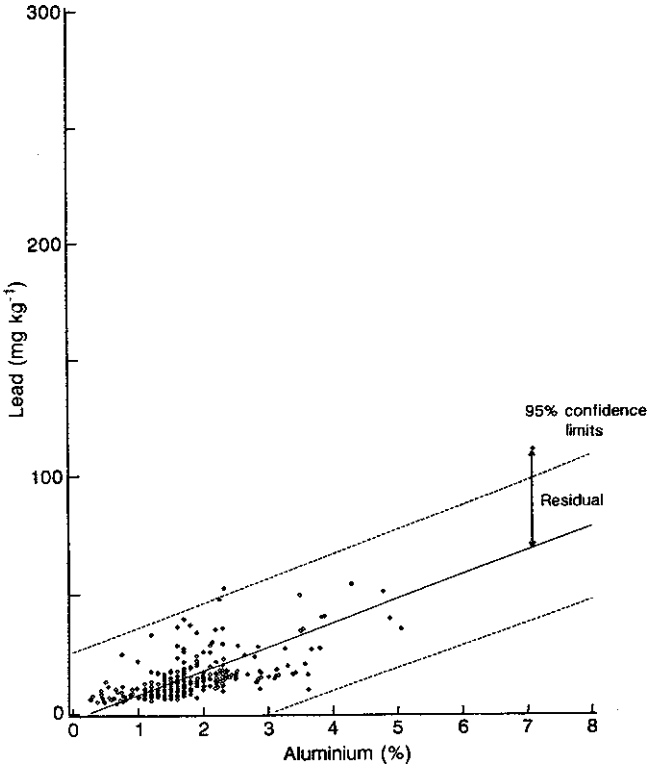

Figure 18. Relationship between concentrations of lead and aluminium in sediments from the western North Sea

hypothesis. Areas where several high samples occur close together are considered to reflect an effect. These are the Tyne, Tees, Humber, and Thames estuaries and the northern edge of the Dogger Bank.

Table 18. Basic properties of the variables measured in the JMG Baseline Sediment Survey. All concentrations as mg kg $^{-1}$ except aluminium which is in per cent and cadmium and mercury which are in g kg $^{-1}$

Parameter	Al	Li	Cd	Cr	Hg	Pb	Zn
N	288	130	154	266	213	288	288
Mean	1.79	10	57	21	85	15	19
Mode	1.7	10	22	10	13	11	13
Percentiles:							
100	7.09	64	262	100	850	234	133
75	2.16	10	56	26	68	16	24
50	1.7	8	38	20	38	12	13
25	1.3	6	26	10	22	9	10
0	0.26	<3	<20	<5	11	4	3

Figures 19-23 show the distribution of the residuals for Cd, Cr, Hg, Pb and Zn.

Normalised concentrations of cadmium (Figure 19) are highest at two offshore locations, one off East Anglia and one off Berwick. Both sites are surrounded by samples with low values and, based on this evidence, it is presumed that these isolated samples are the result of an error, possibly in sampling; future work will test this

Values for chromium (Figure 20) are also relatively high in the Tyne-Tees, Thames and Dogger areas although the high offshore values occur at the centre of the Dogger rather than at the northern edge.

Values for mercury (Figure 21) are generally higher in the coastal zone than offshore, although the highest values occur at the Oysterground, the site of one of the intercomparison samples in Dutch waters.

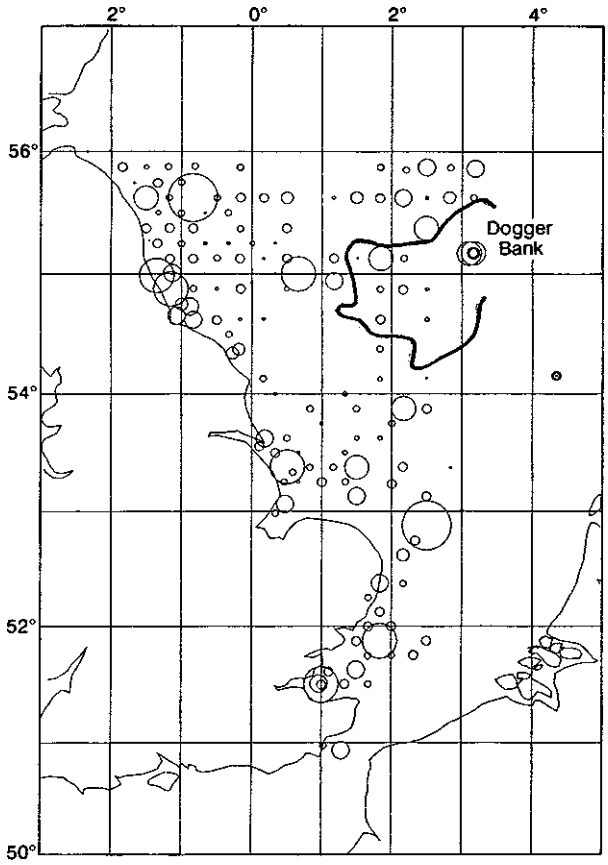

Figure 19. Residual values around the relationship between cadmium and aluminium expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

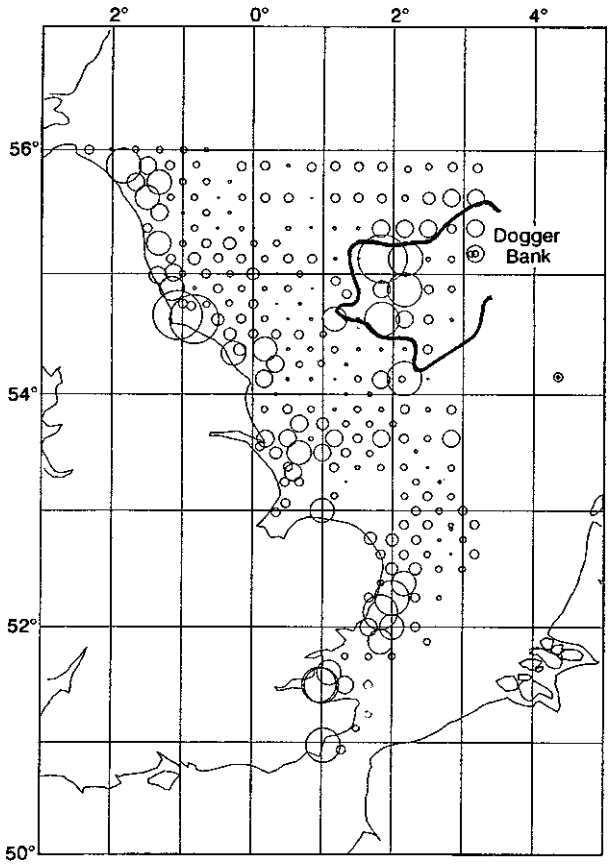

Figure 20. Residual values around the relationship between chromium and aluminium expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

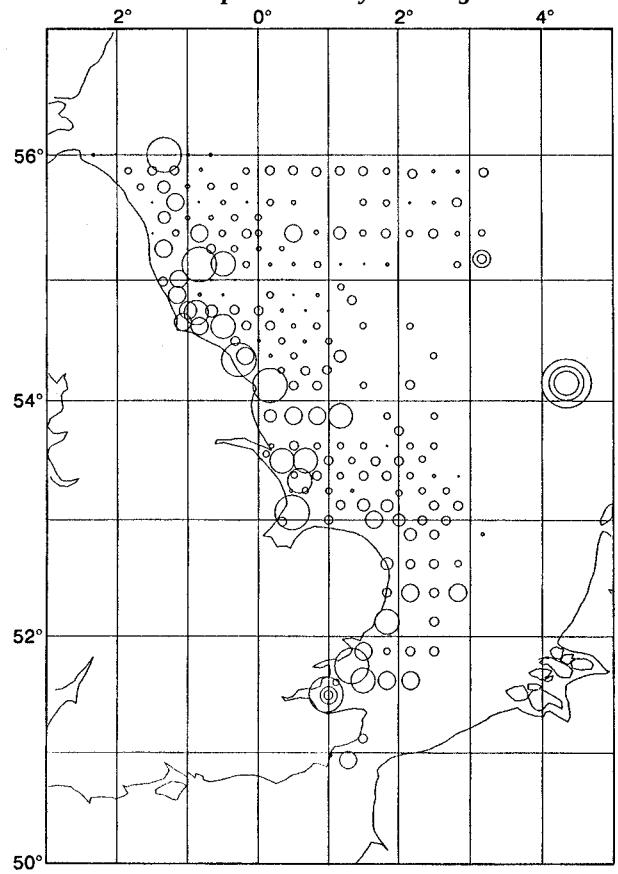

Figure 21. Residual values around the relationship between mercury and aluminium expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

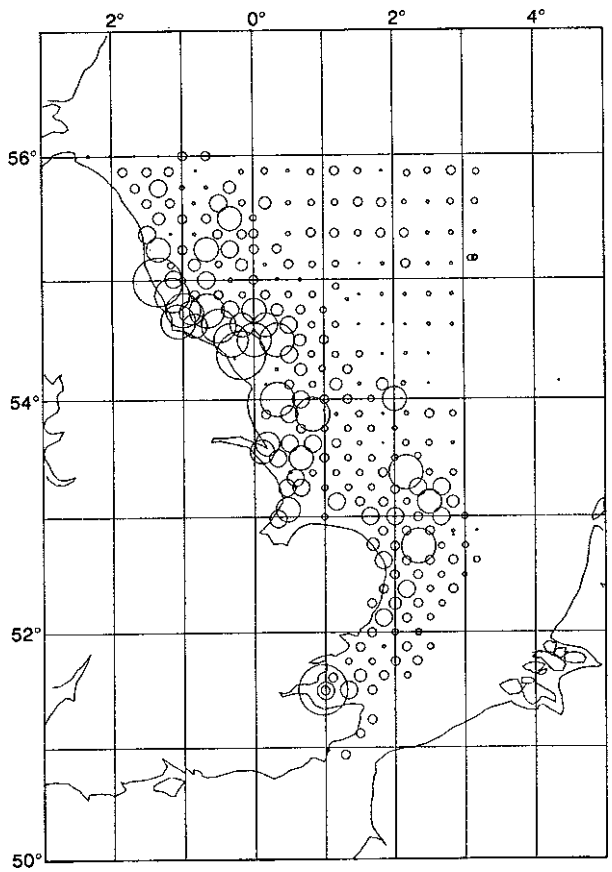

Figure 22. Residual values around the relationship between lead and aluminium expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

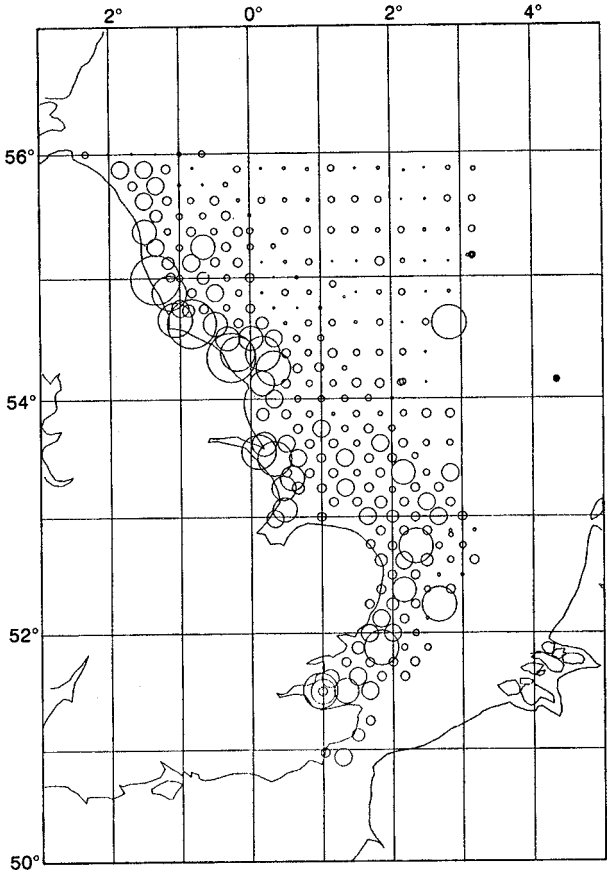

Figure 23. Residual values around the relationship between zinc and aluminium expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

Values for lead and zinc (Figures 22 and 23) are higher near the coast than offshore with the highest values occurring off the north-east coast of England. Values for lead are also high in the Thames estuary.

10.4 Conclusions

In general, normalised concentrations of sediment metals are higher near the coast and around estuaries than they are offshore, allowing the deduction to be made that riverine inputs from the land are significant sources of sediment metals. It is probable that contaminants released to the rivers as a consequence of anthropogenic activity are a significant component of these inputs, but until information on the natural (pre-industrial) concentrations of metals in sediments is obtained from core samples, it is not possible to comment on the exact extent of this effect. This problem is particularly relevant to the Tyne-Tees area, where the rivers drain naturally mineralised zones and also flow through industrialised areas. It can be seen that concentrations of sediment metals are relatively high, probably due to both natural and anthropogenic effects, but judgment of the relative significance of these effects cannot yet be made. The local sediments are largely shallow, vertically mixed sands where coring is difficult and the interpretation of downcore variations in metal content at best problematic. For this reason, core samples have been collected from muddy patches and are presently being processed. It is hoped that they will shed some light on this question.

The Dogger Bank is also an area of interest due to its relatively high normalised values of cadmium and chromium. It is not clear how this area could be influenced by anthropogenic inputs while the sediments between the coastal zone and the bank are apparently not. It is possible that the higher values of these two elements result from a natural process such as progressive erosion exposing underlying sediments richer in these elements. This question may be answered by the analysis of core samples and the mineralogical analysis of the sediments. Work is proceeding on this topic.

11. ORGANOCHLORINE CONTAMINANTS IN SEDIMENTS

11.1 Introduction

The monitoring of organic contaminants in sediment is becoming increasingly important. Sediments act as a reservoir or long term sink for many hydrophobic organic contaminants and such compounds can occur in sediments at relatively high concentrations compared to those seen in sea water. Following advice from the ICES Advisory Committee on Marine Pollution (ICES, 1989), the North Sea Task Force adopted surficial sediments as their primary matrix for the determination of chlorinated biphenyls and organochlorine pesticide residues. Surface sediment samples were collected during a MAFF research cruise on *RV CIROLANA* from 5-27 June 1990. Full details of the station positions are given in Table 19; data for nutrients and other contaminants relating to these sites were included in a previous report (MAFF, 1992(a)).

11.2 Method

The sediment samples were collected using a modified 0.1 m^2 Day grab. Samples were then dried and processed, using published methodologies (Allchin *et al.*,1989), the only variation being to replace mirex, (the internal standard orginally used) with a range of 2,4 - dichlorobenzyl alkyl ethers, which were used as chromatographic reference peaks as well as internal standards (Wells *et al.*, 1985). The sediment types (see Table 19) were determined by visual inspection of the sample at the time of collection. Certified reference materials were analysed with each batch of sediment samples, either HS-1 or HS-2 from the Canadian marine analytical standards programme. Preliminary data on chlorinated biphenyls in sediments (MAFF, 1991) were reported as concentrations expressed on a formulation basis as Aroclor 1254; in this study, data are reported for fourteen individual chlorobiphenyl congeners, as a sum of these compounds (Total CBs), and as Aroclor 1254. The fourteen congeners determined are drawn from the ICES primary and secondary lists defined for monitoring purposes (ICES, 1986). Subsequent analyses will include an even wider range of chlorinated biphenyl congeners.

11.3 Results

The results are presented in Table 19. In the study following the Piper - Alpha incident in the North Sea in 1988, Wells *et al.*, (1989) defined a series of concentration guidelines. In this report a series of concentration guidelines have also been defined in order to categorise the levels of chlorinated biphenyls in sediments. These are:-

$<0.2\,\mu g\,kg^{-1}$	contamination not detectable.
$0.2\text{-}20\,\mu g\,kg^{-1}$	slightly contaminated.
$21\text{-}100\,\mu g\,kg^{-1}$	contaminated.
$>100\,\mu g\,kg^{-1}$	heavily contaminated.

Samples from the Rivers Tyne, Wear, Mersey, and the Rame Head disposal ground near Plymouth all fall into the contaminated category, with Station 8 in the River Tyne being heavily contaminated relative to Aroclor 1254. Samples from the Rivers Humber, Ribble, Tweed, Tees, and from the sewage-sludge disposal ground in Liverpool Bay are within the slightly contaminated category; whereas samples from offshore sites (i.e. the E. English Channel, the outer Thames and the Western Approaches) are below the current limit of detection, and therefore fall into the 'contamination not detectable' category. As may be expected, concentrations of chlorinated biphenyls from coastal sites are generally higher than those seen offshore. Figure 24 shows the distribution of concentrations of Aroclor 1254 using symbols representing percentile classes with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent.

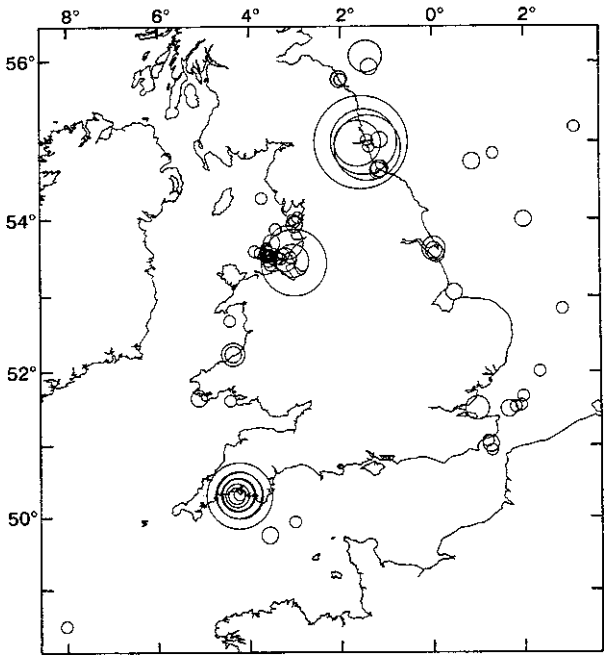

Figure 24. Distribution of concentrations of Aroclor 1254 expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20, 10, 5 and 1 per cent. The highest percentile groups are represented by the largest circles

Table 19. Stations sampled during CIROLANA 6, 5-27 June 1990 and Chlorinatedbiphenyl results in surface sediments as g kg^{-1} dry weight

Stn.	Position	Location	NSTF no.	Sediment type	1254	CB#28	CB#52	CB#101	CB#118
1	55° 45.70' N 1° 57.85' W	Tweed/mouth		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
2	55° 45.83' N 1° 59.58' W	Tweed/lower estuary		Sand	2.0	<0.2	<0.2	<0.2	<0.2
3	55° 46.12' N 2° 0.35' W	Tweed/upper estuary		Sand	2.0	<0.2	<0.2	<0.2	<0.2
5	55° 0.5' N 1° 7.6' W	Tyne/mouth		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
6	55° 0.63' N 1° 24.98' W	Tyne/lower estuary		Sand and gravel	22	0.36	0.24	0.33	0.3
7	54° 59.17' N 1° 27.86' W	Tyne/middle estuary		Mud	57.0	2	7	4	3
8	54° 58.92' N 1° 31.82' W	Tyne/middle estuary		Mud	109.0	3	6	9	6
9	54° 58.09' N 1° 36.25' W	Tyne/upper estuary		Mud	40.0	4	8	3	2
10	55° 0.46' N 1° 7.76' W	Tyne/intermediate	14	Mud	7.0	<0.2	<0.2	<0.2	<0.2
11	54° 54.88' N 1° 20.3' W	Wear/mouth		Stones and sand	4.0	<0.2	<0.2	<0.2	<0.2
12	54° 55.04' N 1° 21.62' W	Wear/lower estuary		Sand	0.6	<0.2	<0.2	<0.2	<0.2
13	54° 54.58' N 1° 22.87' W	Wear/upper estuary		Sand mud and stones	74.0	<0.2	3	6	6
14	54° 44.04' N 0° 52.84' W	Tees/intermediate	15	Sand and mud	4.0	<0.2	<0.2	<0.2	<0.2
15	54° 38.93' N 1° 7.35' W	Tees/mouth		Sand	3.0	<0.2	<0.2	<0.2	<0.2
16	54° 38.52' N 1° 8.66' W	Tees/lower estuary		Sand	2.5	<0.2	<0.2	<0.2	<0.2
17	54° 37.12' N 1° 9.32' W	Tees/middle estuary		Mud	4.3	0.65	1.6	0.64	0.4
19	54° 50.1' N 1° 20.24' E	Offshore Tyne/Tees	43	Fine sand	1.3	<0.2	<0.2	<0.2	<0.2
20	55° 10.1' N 3° 6.3' E	Dogger Bank	47	Fine sand and shell	0.6	<0.2	<0.2	<0.2	<0.2
21	53° 59.94' N 2° 0.21' E	Silver Pit	53	Fine sand	1.8	<0.2	<0.2	<0.2	<0.2
23	53° 33.33' N 0° 6.37' E	Humber/mouth	16	Fine sand	5.4	0.24	<0.2	0.31	0.31
24	53° 37.1' N 0° 2.82' W	Humber/lower estuary		Mud and sand	8.7	0.62	2.5	0.65	0.53
25	53° 36.0' N 0° 3.0' W	Humber/upper estuary		Mud	5.8	0.42	1.9	0.45	0.49
26	53° 3.65' N 0° 28.88' E	Wash	17	Stones sand and mud	2.9	<0.2	0.36	<0.2	<0.2
27	52° 50.4' N 2° 50.1' E	Smiths Knoll	18	Sand and shell	0.41	<0.2	<0.2	<0.2	<0.2
28	51° 59.97' N 2° 19.95' E	Gabbard	25	Medium sand	1.2	<0.2	<0.2	<0.2	<0.2
29	51° 30.34' N 0° 57.88' E	Thames Barrow	19	Mud	8.7	<0.2	<0.2	<0.2	0.2
30	50° 56.1' N 1° 17.05' E	E English Channel	69	Shell and sand	<0.2	<0.2	<0.2	<0.2	<0.2
42	49° 56.11' N 3° 1.81' W	W English Channel	72	Coarse sand, stones & shell	1.4	<0.2	<0.2	<0.2	<0.2
44	49° 45.7' N 3° 35.6' W	W English Channel		Sand stones and shell	1.8	<0.2	<0.2	<0.2	<0.2
45	48° 29.2' N 8° 1.29' W	SW Western Approaches		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
48	51° 36.56' N 4° 25.77' W	Bristol Channel		Sand and mud	0.69	<0.2	<0.2	0.69	<0.2
49	51° 38.7' N 5° 7.6' W	Bristol Channel		Sand and shell	3.1	<0.2	<0.2	<0.2	<0.2
50	52° 10.13' N 4° 44.7' W	Cardigan Bay		Sand and shell	1.8	<0.2	<0.2	<0.2	<0.2
51	52° 13.92' N 4° 21.48' W	Cardigan Bay		Mud and stones	11	0.6	0.3	0.25	0.3
53	52° 40.9' N 4° 25.9' W	Cardigan Bay		Shingle	<0.2	<0.2	<0.2	<0.2	<0.2
54	53° 32.7' N 3° 15.3' W	Mersey		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
55	53° 26.4' N 3° 0.7' W	Mersey		Mud	43	4	3.1	2.3	2.5
56	53° 28.0' N 3° 2.9' W	Mersey		Mud	15	0.31	0.89	0.84	1.3
57	53° 30.5' N 3° 5.6' W	Mersey		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
59	53° 30.2' N 3° 21.04' W	Mersey		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
60	53° 30.09' N 3° 29.04' W	Liverpool Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
61	53° 41.78' N 3° 29.96' W	Ribble		Mud and sand	5.0	<0.2	<0.2	0.24	0.39
62	53° 41.99' N 3° 10.43' W	Ribble		Sand and mud	23	<0.2	0.66	1.6	1.5
63	53° 51.9' N 3° 25.16' W	Morecambe Bay		Mud and sand	<0.2	<0.2	<0.2	<0.2	0.6
65	53° 59.37' N 3° 2.19' W	Morecambe Bay		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
66	54° 1.1' N 2° 56.26' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
67	53° 58.6' N 3° 0.0' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
68	53° 55.68' N 3° 0.2' W	Morecambe Bay		Mud	5.3	1	<0.2	0.28	<0.2
69	54° 16.34' N 3° 42.97' W	East of Isle of Man		Mud	<0.2	<0.2	<0.2	<0.2	<0.2
70	53° 30.0' N 3° 20.1' W	Mersey		Coarse sand mud and shell	<0.2	<0.2	<0.2	<0.2	<0.2
75	53° 32.0' N 3° 39.8' W	Sewage Sludge DS		Sand shell and mud	<0.2	<0.2	<0.2	<0.2	<0.2
76	53° 28.97' N 3° 13.49' W	Sewage Sludge DS		Mud	12	1.1	0.7	0.62	1.1
77	53° 29.66' N 3° 18.55' W	Sewage Sludge DS		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
78	53° 30.53' N 3° 23.12' W	Sewage Sludge DS		Fine sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2
79	53° 31.13' N 3° 27.43' W	Sewage Sludge DS		Fine sand and shell	1.8	<0.2	<0.2	<0.2	<0.2
80	53° 31.78' N 3° 31.56' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
81	53° 32.11' N 3° 33.79' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
82	53° 32.39' N 3° 35.36' W	Sewage Sludge DS		Sand shell and stones	<0.2	<0.2	<0.2	<0.2	<0.2
83	53° 32.7' N 3° 37.41' W	Sewage Sludge DS		Sand , shell and mud	<0.2	<0.2	<0.2	<0.2	<0.2
84	53° 33.18' N 3° 39.82' W	Sewage Sludge DS		Mud sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
85	53° 33.94' N 3° 44.94' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
86	53° 35.2' N 3° 52.57' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
87	53° 37.39' N 3° 37.02' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
88	53° 35.86' N 3° 36.55' W	Sewage Sludge DS		Mud , sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2
89	53° 34.8' N 3° 36.13' W	Sewage Sludge DS		Sand mud and stones	<0.2	<0.2	<0.2	<0.2	<0.2
90	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
90a	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and mud	2.2	<0.2	<0.2	0.2	<0.2
91	53° 29.85' N 3° 34.44' W	Sewage Sludge DS		Sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2
92	53° 27.52' N 3° 33.61' W	Sewage Sludge DS		Sand and pebbles	<0.2	<0.2	<0.2	<0.2	<0.2
93	53° 24.7' N 3° 32.5' W	Sewage Sludge DS		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
94	50° 18.3' N 4° 18.14' W	Ramehead DS		Mud	7.1	withCB#31	<0.2	0.2	0.53
95	50° 18.42' N 4° 17.53' W	Ramehead DS		Mud	7.8	withCB#31	<0.2	0.33	0.55
96	50° 18.56' N 4° 16.83' W	Ramehead DS		Mud	15	withCB#31	0.33	0.68	0.86
97	50° 18.74' N 4° 16.16' W	Ramehead DS		Mud	21	withCB#31	0.48	1	1.2
99	50° 19.29' N 4° 16.49' W	Ramehead DS		Mud	19	withCB#31	0.34	0.73	0.88
100	50° 19.52' N 4° 17.06' W	Ramehead DS		Mud	16	withCB#31	0.26	0.54	0.7
102	50° 19.25' N 4° 15.19' W	Ramehead DS		Mud	23	withCB#31	0.84	1.1	1.3
103	50° 18.85'N 4° 15.55' W	Ramehead DS		Mud	25	withCB#31	0.39	1	1.2
104	50° 19.05' N 4° 15.28' W	Ramehead DS		Mud	21	withCB#31	0.36	0.71	0.87
105	50° 19.26' N 4° 14.43' W	Ramehead DS		Medium sand	<0.2	withCB#31	<0.2	<0.2	<0.2
106	50° 18.66' N 4° 15.26' W	Ramehead DS		Mud and sand	37	<0.2	0.66	2.4	2.6
107	50° 18.32' N 4° 14.6' W	Ramehead DS		Stones and sand	72	<0.2	0.67	3.6	4
108	50° 18.14' N 4° 13.83' W	Ramehead DS		Stones	29	<0.2	0.58	1.4	1.5
117	51° 1.17' N 1° 15.48' E	Fluxmanche		Sand and shell	3.0	<0.2	<0.2	<0.2	0.32
118	51° 3.61' N 1° 11.76' E	Fluxmanche		Mud ,sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
119	51° 29.9' N 1° 1.40' E	Outer Thames Estuary		Mud	4.3	<0.2	<0.2	<0.2	0.22
120	51° 31.44' N 1° 48.85' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
121	51° 32.85' N 1° 55.51' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
122	51° 40.01' N 1° 58.34' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2

Table 19. Continued

Stn.	Position	Location	NSTF no.	Sediment type	CB#153	CB#138	CB#180	CB#31	CB#105
1	55° 45.70' N 1° 57.85' W	Tweed/mouth		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
2	55° 45.83' N 1° 59.58' W	Tweed/lower estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
3	55° 46.12' N 2° 0.35' W	Tweed/upper estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
5	55° 0.5' N 1° 7.6' W	Tyne/mouth		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
6	55° 0.63' N 1° 24.98' W	Tyne/lower estuary		Sand and gravel	1.1	1.8	0.5	<0.2	<0.2
7	54° 59.17' N 1° 27.86' W	Tyne/middle estuary			4	5	2	<0.2	<0.2
8	54° 58.92' N 1° 31.82' W	Tyne/middle estuary		Mud	12	13	8	<0.2	<0.2
9	54° 58.09' N 1° 36.25' W	Tyne/upper estuary			4	3	2	<0.2	<0.2
10	55° 0.46' N 1° 7.76' W	Tyne/intermediate	14	Mud	0.4	<0.2	<0.2	<0.2	<0.2
11	54° 54.88' N 1° 20.3' W	Wear/mouth		Stones and sand	0.41	<0.2	0.8	<0.2	<0.2
12	54° 55.04' N 1° 21.62' W	Wear/lower estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
13	54° 54.58' N 1° 22.87' W	Wear/upper estuary		Sand mud and stones	4	6	1	<0.2	<0.2
14	54° 44.04' N 0° 52.84' W	Tees/intermediate	15	Sand and mud	<0.2	0.51	<0.2	<0.2	<0.2
15	54° 38.93' N 1° 7.35' W	Tees/mouth		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
16	54° 38.52' N 1° 8.66' W	Tees/lower estuary			<0.2	<0.2	<0.2	<0.2	<0.2
17	54° 37.12' N 1° 9.32' W	Tees/middle estuary		Mud	0.41	0.35	<0.2	0.23	<0.2
19	54° 50.1' N 1° 20.24' E	Offshore Tyne/Tees	43	Fine sand	0.2	0.44	<0.2	<0.2	<0.2
20	55° 10.1' N 3° 6.3' E	Dogger Bank	47	Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
21	53° 59.94' N 2° 0.21' E	Silver Pit	53	Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
23	53° 33.33' N 0° 6.37' E	Humber/mouth	16	Fine sand	0.63	0.81	0.31	<0.2	<0.2
24	53° 37.1' N 0° 2.82' W	Humber/lower estuary		Mud and sand	0.71	0.73	0.36	0.29	0.31
25	53° 36.0' N 0° 3.0' W	Humber/upper estuary			0.45	0.46	0.2	<0.2	0.23
26	53° 3.65' N 0° 28.88' E	Wash	17	Stones sand and mud	<0.2	0.22	<0.2	<0.2	<0.2
27	52° 50.4' N 2° 50.1' E	Smiths Knoll	18	Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
28	51° 59.97' N 2° 19.95' E	Gabbard	25	Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
29	51° 30.34' N 0° 57.88' E	Thames Barrow	19	Mud	0.75	0.37	<0.2	<0.2	<0.2
30	50° 56.1' N 1° 17.05' E	E English Channel	69	Shell and sand	<0.2	<0.2	<0.2	<0.2	<0.2
42	49° 56.11' N 3° 1.81' W	W English Channel	72	Coarse sand stones & shell	<0.2	<0.2	<0.2	<0.2	<0.2
44	49° 45.7' N 3° 35.6' W	W English Channel		Sand stones and shell	<0.2	<0.2	<0.2	<0.2	<0.2
45	48° 29.2' N 8° 1.29' W	SW Western Approaches		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
48	51° 36.56' N 4° 25.77' W	Bristol Channel		Sand and mud	2	<0.2	<0.2	<0.2	<0.2
49	51° 38.7' N 5° 7.6' W	Bristol Channel		Sand and shell	0.27	<0.2	<0.2	<0.2	<0.2
50	52° 10.13' N 4° 44.7' W	Cardigan Bay		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
51	52° 13.92' N 4° 21.48' W	Cardigan Bay		Mud and stones	1.1	0.3	<0.2	0.5	<0.2
53	52° 40.9' N 4° 25.9' W	Cardigan Bay			<0.2	<0.2	<0.2	<0.2	<0.2
54	53° 32.7' N 3° 15.3' W	Mersey		Fine sand	0.2	<0.2	<0.2	0.95	<0.2
55	53° 26.4 ‘ N 3° 0.7’ W	Mersey		Mud	2.6	2.5	<0.2	5.1	0.8
56	53° 28.0' N 3° 2.9' W	Mersey		Mud	1.3	1.2	0.68	1.2	0.2
57	53° 30.5' N 3° 5.6' W	Mersey		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
59	53° 30.2' N 3° 21.04' W	Mersey		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
60	53° 30.09' N 3° 29.04' W	Liverpool Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
61	53° 41.78' N 3° 29.96' W	Ribble		Mud and sand	0.69	0.54	0.41	<0.2	<0.2
62	53° 41.99' N 3° 10.43' W	Ribble		Sand and mud	1.6	2.1	0.69	0.44	0.69
63	53° 51.9' N 3° 25.16' W	Morecambe Bay		and sand	0.5	0.37	0.2	<0.2	0.24
65	53° 59.37' N 3° 2.19' W	Morecambe Bay		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
66	54° 1.1' N 2° 56.26' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
67	53° 58.6' N 3° 0.0' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2
68	53° 55.68' N 3° 0.2' W	Morecambe Bay		Mud	0.49	0.54	0.2	1.2	0.26
69	54° 16.34' N 3° 42.97’W	East of Isle of Man		Mud	<0.2	<0.2	<0.2	<0.2	<0.2
70	53° 30.0' N 3° 20.1' W	Mersey		Coarse sand mud & shell	0.2	<0.2	<0.2	<0.2	<0.2
75	53° 32.0' N 3° 39.8' W	Sewage Sludge DS		Sand shell and mud	<0.2	<0.2	<0.2	<0.2	<0.2
76	53° 28.97' N 3° 13.49' W	Sewage Sludge DS		Mud	1.1	1	0.55	1.2	0.44
77	53° 29.66' N 3° 18.55' W	Sewage Sludge DS		Fine sand	<0.2	<0.2	<0.2	<0.2	0.33
78	53° 30.53' N 3° 23.12' W	Sewage Sludge DS		Fine sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2
79	53° 31.13' N 3° 27.43' W	Sewage Sludge DS		Fine sand and shell	0.23	0.2	0.26	0.88	<0.2
80	53° 31.78' N 3° 31.56' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
81	53° 32.11' N 3° 33.79' W	Sewage Sludge DS		Fine sand and shell	0.2	0.2	<0.2	<0.2	<0.2
82	53° 32.39' N 3° 35.36’W	Sewage Sludge DS		Sand shell and stones	<0.2	<0.2	<0.2	<0.2	<0.2
83	53° 32.7' N 3° 37.41' W	Sewage Sludge DS		Sand , shell and mud	<0.2	<0.2	<0.2	<0.2	0.4
84	53° 33.18' N 3° 39.82' W	Sewage Sludge DS		Mud sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
85	53° 33.94' N 3° 44.94' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
86	53° 35.2' N 3° 52.57' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
87	53° 37.39' N 3° 37.02' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
88	53° 35.86' N 3° 36.55' W	Sewage Sludge DS		Mud , sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2
89	53° 34.8' N 3° 36.13' W	Sewage Sludge DS		Sand mud and stones	0.2	0.2	<0.2	<0.2	<0.2
90	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
90a	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and mud	0.3	0.28	0.2	<0.2	<0.2
91	53° 29.85' N 3° 34.44' W	Sewage Sludge DS		Sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2
92	53° 27.52' N 3° 33.61' W	Sewage Sludge DS		Sand and pebbles	<0.2	<0.2	<0.2	<0.2	<0.2
93	53° 24.7' N 3° 32.5' W	Sewage Sludge DS		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2
94	50° 18.3' N 4° 18.14' W	Ramehead DS		Mud	0.35	0.54	0.28	0.89	<0.2
95	50° 18.42' N 4° 17.53' W	Ramehead DS		Mud	0.64	0.81	0.33	1.1	0.2
96	50° 18.56' N 4° 16.83' W	Ramehead DS		Mud	0.8	0.89	0.3	0.88	0.26
97	50° 18.74' N 4° 16.16' W	Ramehead DS		Mud	1.2	1.4	0.59	2.3	0.44
99	50° 19.29' N 4° 16.49' W	Ramehead DS		Mud	0.84	0.99	0.44	0.52	0.33
100	50° 19.52' N 4° 17.06' W	Ramehead DS		Mud	0.73	0.2	0.29	0.88	0.25
102	50° 19.25' N 4° 15.19' W	Ramehead DS		Mud	1.2	1.4	0.32	1.1	0.87
103	50° 18.85’N 4° 15.55' W	Ramehead DS		Mud	1.4	1.5	0.43	2	0.47
104	50° 19.05' N 4° 15.28' W	Ramehead DS		Mud	0.89	0.95	0.46	0.89	0.3
105	50° 19.26' N 4° 14.43' W	Ramehead DS		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
106	50° 18.66' N 4° 15.26' W	Ramehead DS		Mud and sand	2	2.9	0.48	0.33	0.94
107	50° 18.32' N 4° 14.6' W	Ramehead DS		and sand	3.5	5.2	0.65	0.65	1.5
108	50° 18.14' N 4° 13.83' W	Ramehead DS		Stones	1.6	2.1	0.5	<0.2	0.5
117	51° 1.17' N 1° 15.48' E	Fluxmanche		Sand and shell	<0.2	0.32	0.2	<0.2	<0.2
118	51° 3.61' N 1° 11.76' E	Fluxmanche		Mud, sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2
119	51° 29.9' N 1° 1.40'E	Outer Thames Estuary		Mud	0.28	0.31	<0.2	<0.2	<0.2
120	51°31.44'N 1° 48.85' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
121	51° 32.85' N 1° 55.51' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2
122	51° 40.01' N 1° 58.34' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2

Table 19. Continued

Stn.	Position	Location	NSTF no.	Sediment type	CB#128	CB#149	CB#170	CB#183	CB#187	Total CBs
1	55° 45.70' N 1° 57.85' W	Tweed/mouth		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
2	55° 45.83' N 1° 59.58' W	Tweed/lower estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
3	55° 46.12' N 2° 0.35' W	Tweed/upper estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
5	55° 0.5' N 1° 7.6' W	Tyne/mouth		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
6	55° 0.63' N 1° 24.98' W	Tyne/lower estuary		Sand and gravel	<0.2	0.57	0.35	0.21	0.32	6.1
7	54° 59.17' N 1° 27.86' W	Tyne/middle estuary		Mud	0.9	3	1	0.7	1	34
8	54° 58.92' N 1° 31.82' W	Tyne/middle estuary		Mud	2	11	4	3	4	81
9	54° 58.09' N 1° 36.25' W	Tyne/upper estuary		Mud	0.5	3	1	0.7	1	32
10	55° 0.46' N 1° 7.76' W	Tyne/intermediate	14	Mud	<0.2	<0.2	0.4	<0.2	<0.2	0.8
11	54° 54.88' N 1° 20.3' W	Wear/mouth		Stones and sand	<0.2	<0.2	0.27	<0.2	0.7	2.2
12	54° 55.04' N 1° 21.62' W	Wear/lower estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
13	54° 54.58' N 1° 22.87' W	Wear/upper estuary		Sand mud and stones	1	4	0.9	0.4	0.6	33
14	54° 44.04' N 0° 52.84' W	Tees/intermediate	15	Sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2	0.51
15	54° 38.93' N 1° 7.35' W	Tees/mouth		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
16	54° 38.52' N 1° 8.66' W	Tees/lower estuary		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
17	54° 37.12' N 1° 9.32' W	Tees/middle estuary		Mud	<0.2	0.35	<0.2	<0.2	<0.2	4.6
19	54° 50.1' N 1° 20.24' E	Offshore Tyne/Tees	43	Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2	0.64
20	55° 10.1' N 3° 6.3' E	Dogger Bank	47	Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
21	53° 59.94' N 2° 0.21' E	Silver Pit	53	Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
23	53° 33.33' N 0° 6.37' E	Humber/mouth	16	Fine sand	<0.2	0.42	<0.2	<0.2	<0.2	3
24	53° 37.1' N 0° 2.82' W	Humber/lower estuary		Mud and sand	<0.2	0.57	<0.2	<0.2	0.3	7.6
25	53° 36.0' N 0° 3.0' W	Humber/upper estuary		Mud	<0.2	0.36	<0.2	0.46	<0.2	5.4
26	53° 3.65' N 0° 28.88' E	Wash	17	Stones sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2	0.58
27	52° 50.4' N 2° 50.1' E	Smiths Knoll	18	Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
28	51° 59.97' N 2° 19.95' E	Gabbard	25	Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
29	51° 30.34' N 0° 57.88' E	Thames Barrow	19	Mud	<0.2	0.36	<0.2	<0.2	<0.2	1.7
30	50° 56.1' N 1° 17.05' E	E English Channel	69	Shell and sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
42	49° 56.11' N 3° 1.81' W	W English Channel	72	Coarse sand, stones & shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
44	49° 45.7' N 3° 35.6' W	W English Channel		Sand stones and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
45	48° 29.2' N 8° 1.29' W	SW West. Approaches		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
48	51° 36.56' N 4° 25.77' W	Bristol Channel		Sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2	2.7
49	51° 38.7' N 5° 7.6' W	Bristol Channel		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
50	52° 10.13' N 4° 44.7' W	Cardigan Bay		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
51	52° 13.92' N 4° 21.48' W	Cardigan Bay		Mud and stones	0.2	0.3	<0.2	<0.2	<0.2	3.9
53	52° 40.9' N 4° 25.9' W	Cardigan Bay		Shingle	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
54	53° 32.7' N 3° 15.3' W	Mersey		Fine sand	<0.2	<0.2	<0.2	<0.2	0.2	1.4
55	53° 26.4 ' N 3° 0.7' W	Mersey		Mud	0.51	2.7	<0.2	2.5	0.93	30
56	53° 28.0' N 3° 2.9' W	Mersey		Mud	0.45	1.4	0.36	0.34	0.43	11
57	53° 30.5' N 3° 5.6' W	Mersey		Sand	<0.2	<0.2	<0.2	<0.2	0.25	0.25
59	53° 30.2' N 3° 21.04' W	Mersey		Sand and shell	<0.2	<0.2	<0.2	<0.2	0.25	0.25
60	53° 30.09' N 3° 29.04' W	Liverpool Bay		Sand	<0.2	<0.2	<0.2	<0.2	0.29	0.29
61	53° 41.78' N 3° 29.96' W	Ribble		Mud and sand	0.23	0.78	0.23	<0.2	<0.2	3.5
62	53° 41.99' N 3° 10.43' W	Ribble		Sand and mud	0.38	1.3	0.25	0.37	0.51	12
63	53° 51.9' N 3° 25.16' W	Morecambe Bay		Mud and sand	0.25	0.96	<0.2	<0.2	<0.2	3.1
65	53° 59.37' N 3° 2.19' W	Morecambe Bay		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
66	54° 1.1' N 2° 56.26' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
67	53° 58.6' N 3° 0.0' W	Morecambe Bay		Sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
68	53° 55.68' N 3° 0.2' W	Morecambe Bay		Mud	<0.2	0.71	<0.2	<0.2	<0.2	4.7
69	54° 16.34' N 3° 42.97'W	East of Isle of Man		Mud	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
70	53° 30.0' N 3° 20.1' W	Mersey		Coarse sand mud and shell	0.35	0.47	<0.2	0.2	<0.2	1.2
75	53° 32.0' N 3° 39.8' W	Sewage Sludge DS		Sand shell and mud	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
76	53° 28.97' N 3° 13.49' W	Sewage Sludge DS		Mud	0.32	1.1	0.34	0.23	0.3	10
77	53° 29.66' N 3° 18.55' W	Sewage Sludge DS		Fine sand	0.29	0.35	<0.2	0.2	<0.2	1.2
78	53° 30.53' N 3° 23.12' W	Sewage Sludge DS		Fine sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
79	53° 31.13' N 3° 27.43' W	Sewage Sludge DS		Fine sand and shell	0.2	0.32	<0.2	0.26	<0.2	2.4
80	53° 31.78' N 3° 31.56' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
81	53° 32.11' N 3° 33.79' W	Sewage Sludge DS		Fine sand and shell	0.38	<0.2	<0.2	0.24	<0.2	1
82	53° 32.39' N 3° 35.36'W	Sewage Sludge DS		Sand shell and stones	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
83	53° 32.7' N 3° 37.41' W	Sewage Sludge DS		Sand , shell and mud	<0.2	<0.2	<0.2	0.31	<0.2	0.71
84	53° 33.18' N 3° 39.82' W	Sewage Sludge DS		Mud sand and shell	0.29	<0.2	<0.2	<0.2	<0.2	0.29
85	53° 33.94' N 3° 44.94' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
86	53° 35.2' N 3° 52.57' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
87	53° 37.39' N 3° 37.02' W	Sewage Sludge DS		Fine sand and shell	<0.2	<0.2	<0.2	0.2	<0.2	0.2
88	53° 35.86' N 3° 36.55' W	Sewage Sludge DS		Mud , sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
89	53° 34.8' N 3° 36.13' W	Sewage Sludge DS		Sand mud and stones	<0.2	<0.2	<0.2	0.2	<0.2	0.6
90	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and shell	<0.2	<0.2	<0.2	0.2	<0.2	0.2
90a	53° 30.89' N 3° 34.92' W	Sewage Sludge DS		Sand and mud	<0.2	<0.2	<0.2	<0.2	<0.2	0.98
91	53° 29.85' N 3° 34.44' W	Sewage Sludge DS		Sand and stones	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
92	53° 27.52' N 3° 33.61' W	Sewage Sludge DS		Sand and pebbles	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
93	53° 24.7' N 3° 32.5' W	Sewage Sludge DS		Fine sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
94	50° 18.3' N 4° 18.14' W	Ramehead DS		Mud	<0.2	<0.2	<0.2	<0.2	<0.2	2.8
95	50° 18.42' N 4° 17.53' W	Ramehead DS		Mud	<0.2	0.7	0.2	0.2	<0.2	5
96	50° 18.56' N 4° 16.83' W	Ramehead DS		Mud	<0.2	0.9	<0.2	0.2	<0.2	6.1
97	50° 18.74' N 4° 16.16' W	Ramehead DS		Mud	0.2	1.3	0.21	0.25	<0.2	11
99	50° 19.29' N 4° 16.49' W	Ramehead DS		Mud	<0.2	0.91	0.2	0.21	<0.2	6.4
100	50° 19.52' N 4° 17.06' W	Ramehead DS		Mud	<0.2	0.7	<0.2	0.2	<0.2	4.8
102	50° 19.25' N 4° 15.19' W	Ramehead DS		Mud	<0.2	1.2	0.2	0.23	<0.2	10
103	50° 18.85'N 4° 15.55' W	Ramehead DS		Mud	0.2	1.3	0.26	0.3	<0.2	11
104	50° 19.05' N 4° 15.28' W	Ramehead DS		Mud	0.2	0.87	0.2	0.2	0.25	7.2
105	50° 19.26' N 4° 14.43' W	Ramehead DS		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
106	50° 18.66' N 4° 15.26' W	Ramehead DS		Mud and sand	<0.2	1.8	0.36	0.56	0.24	15
107	50° 18.32' N 4° 14.6' W	Ramehead DS		Stones and sand	0.25	3.5	0.53	1	0.33	25
108	50° 18.14' N 4° 13.83' W	Ramehead DS		Stones	<0.2	1.4	0.34	0.44	0.26	11
117	51° 1.17' N 1° 15.48' E	Fluxmanche		Sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	0.84
118	51° 3.61' N 1° 11.76' E	Fluxmanche		Mud ,sand and shell	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
119	51° 29.9' N 1° 1.40' E	Outer Thames Estuary		Mud	<0.2	<0.2	<0.2	<0.2	<0.2	0.81
120	51° 31.44' N 1° 48.85' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
121	51° 32.85' N 1° 55.51' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
122	51° 40.01' N 1° 58.34' E	Outer Thames Estuary		Medium sand	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2

DS = disposal site

This study showed the most prevalent congeners to be CB#153 and CB#138 (present in nearly 50% of samples), with CB#101, CB#118, CB#180, CB#149, and CB#183 present in over 30% of samples (see Figure 25). CB#153 and CB#138 were also usually the congeners present at the highest concentrations.

This survey has provided some preliminary information on the levels of chlorinated biphenyls in marine sediments, and their geographical distribution around England and Wales. Further investigations will be conducted in subsequent years to provide a more complete picture of this distribution and the results will be published in future monitoring reports.

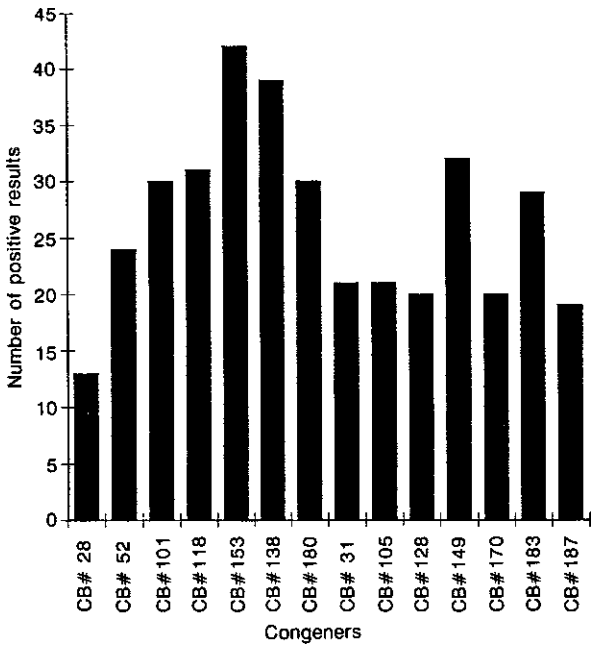

Figure 25. Frequency of congeners

GENERAL STUDIES

12. TBT AND THE MARINE ENVIRONMENT

12.1 Routine monitoring of estuaries and marinas

During 1991, MAFF continued with its programme of monitoring tributyltin at shellfish sites on the Crouch, Dart, Blackwater, Teign, Kingsbridge and Beaulieu estuaries. Concentrations of TBT in the water were generally in the range 1-6 ng l^{-1}, indicating that levels of TBT had continued to decrease at most of the sites compared with previous years (Table 20). The notable exception to this was at West Mersea, on the River Blackwater, where the TBT concentrations had persistently remained high and where the concentration in 1991 was 28 ng l^{-1}, compared to 24 ng l^{-1} in 1990. Concentrations of TBT in *Crassostrea gigas* continued to either decrease or remain the same at all of the shellfish sites, except again West Mersea which showed an increase from 1990 to 1991 (0.38-0.41 µg g^{-1} wet weight) (Table 21). The concentration of TBT in

Table 20. Mean summer (May to September) concentrations of TBT (ng l^{-1}) in the water of estuaries and marinas 1986-1991

Estuary/Location	Shellfish Sites	1986	1987	1988	1989	1990	1991
Crouch	Fambridge	15±8	33±27	21±8	13	7±5	1±1
	Bridgemarsh	22±12	17±12	3	8	4	3±2
	Creeksea	35±17	17±9	22±14	8±2	4±0.4	5±8
	Burnham	45±17	31±18	23±18	11±4	7±6	6±12
	Bush Shore	26±9	22±15	13±5	8	5	3±0.5
	Roach Mouth	26±12	18±13	15±12	8±2	3	3±3
	Holliwell Buoy	11	26±23	10±4	3±2	3±1	3±3
	Holliwell Point	16	6±5	6±5	2	1±1	2±1
Blackwater	West Mersea	38±21	36±29	76±43	25	24±17	28±29
Dart	Blackness	38±33	13±4	13±5	8±3	9±10	2±1
Kingsbridge	Frogmore	15±6	11±6	51±95[b]	5±2	5±6	1±1
Teign	Arch Brook	12±10	7±6	6±2	6±4	5±3	4±3
	Marinas/Harbours						
Plymouth	Sutton Marina	1160±84	882±323	274±79	266±134	133±79	64±58
Dartmouth	Dart Marina	95±66	85±32	21±4	16±6	6±8	3±3
Kingsbridge	Salcombe	117±84	62±71	30±18	21±15	16±11	6±5
Teign	Teignmouth	22±24	23±25	19±12	25±15	24±16	24±17
Beaulieu River	Bucklers Hard*	93±45	1090±1850[a]	82±9	25±7	25±17	5±1
Southampton Water	Hythe Marina			1960±2470[c]	93±62	103±135	66±115

** = also a shellfish site*
a = 263±130 without June value
b = 8±2 without August value
c = mean = 728 without May value
SD = only given where N ≥ 4

Mytilus edulis followed a similar pattern to that for *C. gigas*; however there was an increase in the level of TBT in mussels from Blackness on the River Dart as well as in mussels from West Mersea (Table 22). The slower decline in concentrations at West Mersea and the increase in the level of TBT contamination in 1991, was probably a result of the illegal use of TBT-based paints; in fact a chandler at this site was successfully prosecuted for illegally selling such paints.

Average summer concentrations of TBT in marinas continued to decrease in 1991 as in previous years (Table 20), although high values were still recorded in some samples from Sutton and Hythe marinas (164 and 271 ng l^{-1} respectively).

12.2 Inputs of TBT from shipping activity

Since the 1987 ban on the use of TBT-based paints on small boats (FEPA, 1985) (Great Britain - Parliament 1985(a)), MAFF and DoE have increasingly directed effort towards assessing the environmental impact of the use of TBT-based paints on ships. During 1991, surveys of the TBT contamination of major harbours and waterways was carried out, including a detailed study of the River Thames (see Figures 26 and 27). The routine monitoring of Teignmouth Harbour was also continued in 1991, the average concentration of TBT in the harbour was 24 ng l^{-1}; since monitoring of this site began in 1986, the annual average level of TBT has remained in the range 20-25 ng l^{-1} (Table 20), which reflects the use of the harbour by large ships as opposed to small boats.

In 1991, concentrations of TBT ranged from 2 to 26 ng l^{-1} in the open stretch of the River Thames, from Teddington to Beckton. In St. Katherine's Dock, which is now used by yachts and barges, the level of TBT was 165 ng l^{-1} (in 1990 it had been 1200 ng l^{-1}); there is a lock system in operation at the dock and consequently there is poor exchange of water. By contrast, in the open water at nearby Tower Bridge, the concentrations of TBT in two water samples, taken on separate occasions, were 4 and 26 ng l^{-1}. This is in accordance with previous observations that TBT levels can vary temporally as well as spatially. Water samples were also taken from Purfleet to the mouth of the Thames at Chapman Buoy, where concentrations of TBT ranged from <2 ng l^{-1} to 38 ng l^{-1}. The highest concentration (38 ng l^{-1})

Table 21. Mean summer (June-August) concentrations of TBT (g g^{-1} wet weight) in Crassostrea gigas

Estuary/Location	Shellfish site	1986	1987	1988	1989	1990	1991
Crouch	Fambridge	1.61	1.64	0.62	0.36	0.21	0.20
	Bridgemarsh	1.20	1.46	0.44	0.33	0.18	0.18
	Creeksea	1.49	1.73	0.61	0.38	0.23	0.14
	Burnham	1.24	1.57	0.50	0.45	0.27	0.18
	Bush Shore	0.74	1.26	0.34	0.31	0.15	0.13
	Roach Mouth	0.80	0.98	0.24	0.27	ns	0.09
	Holliwell Buoy	0.37	0.56	0.17	0.11	0.07	0.04
	Holliwell Point	0.18	0.28	0.08	0.08	0.05	0.04
Blackwater	West Mersea	2.26	2.18	1.34	0.65	0.38	0.41
Dart	Blackness Point	0.88	1.35	0.50	0.26	0.17	0.09
Kingsbridge	Frogmore	1.39	1.44	0.48	0.21	0.11	0.07
Teign	Arch Brook	0.30	0.49	0.25	0.13	0.09	0.06
Beaulieu	Bucklers Hard	6.35	3.65	5.60	1.28	0.40	0.27

ns = not sampled

Table 22. Summer concentrations of TBT (g g^{-1} wet weight) in Mytilus edulis

Estuary/Location	Shellfish site	1986	1987	1988	1989	1990	1991
Crouch	Fambridge	0.64	0.75	0.34	0.16	0.09	0.08
	Bridgemarsh	0.66	0.44	0.30	0.15	0.07	0.08
	Creeksea	1.01	0.61	0.46	0.29	0.10	ns
	Burnham	0.98	1.04	0.38	0.29	0.16	0.13
	Bush Shore	0.75	0.47	0.27	0.17	0.09	0.08
	Roach Mouth	0.45	0.41	0.19	0.12	ns	ns
	Holliwell Buoy	0.30	0.35	ns	0.07	0.03	0.03
	Holliwell Point	0.17	0.11	0.11	0.05	0.03	0.02
Blackwater	West Mersea	1.06	0.82	0.32	0.35	0.19	0.25
Dart	Blackness Point	0.60	0.41	0.20	0.11	0.06	0.11
Kingsbridge	Frogmore	0.93	0.58	0.19	0.10	0.07	0.05
Teign	Arch Brook	0.33	0.23	0.09	0.07	0.07	0.04
Beaulieu	Bucklers Hard	2.57	3.22	ns	0.81	0.11	ns

ns = not sampled

recorded for this stretch of the Thames was at Tilbury on the open river. Surprisingly, in the enclosed water of Tilbury main docks the level of TBT was only 12 ng l^{-1}, although in the old dock next to a disused ship the concentration of TBT was 37 ng l^{-1}. At Southend the level of TBT in the water was <2 ng l^{-1} despite the presence of many dinghies and yachts (even though TBT was banned from use in 1987 on small boats under 25 m in length, inputs may still result from them). Even at the mouth of the Thames, at Chapman Buoy, the concentration of TBT was 3 ng l^{-1}.

Sediment samples were taken in the Thames from Richmond Bridge to Shoeburyness. The concentration at Richmond was only 0.06 µg g^{-1} dry weight. From Kew to Creekmouth (just east of Beckton) the levels of TBT ranged from 0.10 to 0.24 µg g^{-1} dry weight; at Wapping, to the east of Tower Bridge, the concentration of TBT in a sample of mud was 0.22 µg g^{-1}, while in a sandy sample from the same site it was only 0.008 µg g^{-1}. In the Grays/Tilbury region concentrations of TBT were in the range 0.18-0.49 µg g^{-1} dry weight. Towards the mouth of the Thames, at Canvey, Southend and Shoeburyness, levels of TBT

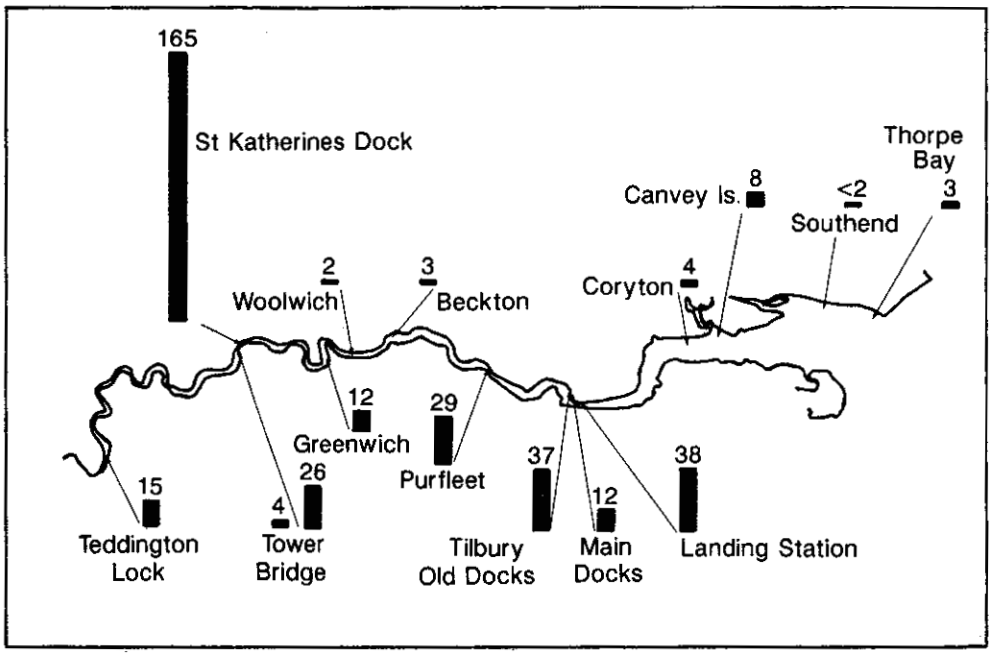

Figure 26. Concentration of TBT (ng l^{-1}) in water samples from the Thames

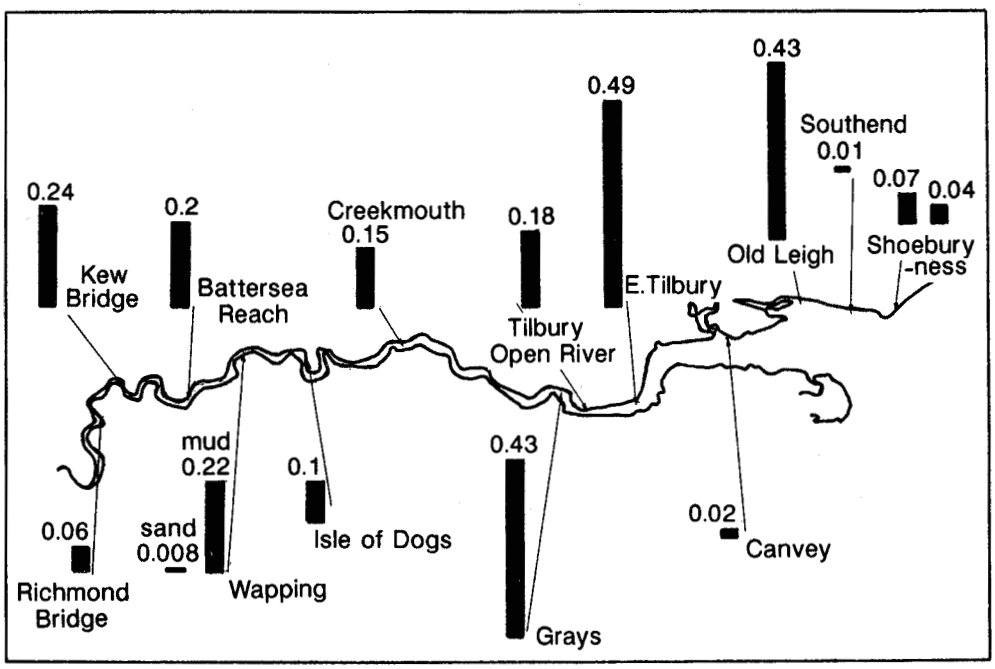

Figure 27. Concentration of TBT (g g^{-1}) in sediment samples from the Thames

were only 0.01-0.07 μg g^{-1} dry weight. However, in a sample taken at Old Leigh, in a creek between Canvey and Southend, the concentration of TBT was much higher, being 0.43 μg g^{-1} dry weight.

The inputs of TBT to the Thames may come from a variety of sources e.g. pleasure boats and barges (St. Katherine's Dock), water taxis (Greenwich), ferries (Woolwich), sewage disposal ships (Beckton and Crossness), container ships (Purfleet and Tilbury), oil tankers (Coryton and Canvey) yachts and dinghies (Southend). A potentially major input of TBT to the Thames comes from the high density of pleasure craft within the enclosed waters of St. Katherine's Dock, although inputs will be limited to the times when the lock gates are open. With successive years one would expect the inputs of TBT from St. Katherine's Dock to decrease in significance as fewer boats will be coated with the banned TBT-based paints (concentrations of TBT have dramatically decreased at this site from 1200 ng l^{-1} in 1990, to 165 ng l^{-1} in 1991). In addition to the inputs from boats and ships, the spillage of timber preservative containing TBT may also be another source to the Thames. Although such inputs are likely to be infrequent, they may contribute significantly to levels of TBT in the river at the time.

TBT is very persistent in sediment, with a half-life of several years in the anaerobic layer. Therefore sediments may form a long term reservoir for TBT, and those from areas such as Grays/Tilbury and Old Leigh may become significant sources of TBT inputs to the River Thames as other inputs reduce. In the dock area of Grays and Tilbury, the TBT concentration of 0.18-0.49 μg g^{-1} probably reflects past dry-docking activity when ships were hosed down and repainted. The fairly high level of TBT (0.43 μg g^{-1} dry weight) in the sediment at Old Leigh is probably a result of past inputs from repainting of yachts and cockle boats at the boat-repair facilities along the foreshore of the creek.

13. PESTICIDE RUN-OFF STUDIES AT ROSEMAUND EXPERIMENTAL CENTRE

13.1 Introduction

The purpose of these studies was described in full in MAFF (1992(a)). Briefly, the Burnham-on-Crouch Fisheries Laboratory provides advice under FEPA (1985) (Great Britain-Parliament, 1985(a)) to the Advisory Committee on Pesticides, on the risk posed to the aquatic environment by pesticides. An essential part of these risk assessments concerns the need to predict likely environmental concentrations of products before they come into use. The most promising new approach to this problem involves the use of computerised models of the way pesticides behave in fields and their associated drainage water. The studies at Rosemaund Farm are an attempt to provide realistic field data on environmental concentrations of pesticides, so that model predictions can be checked, and the models themselves improved, if necessary.

Previous work at Rosemaund farm was summarised in MAFF (1992(a)). Since then, further data have been reported by Hack (1992) and published by Matthiessen *et al.* (1992). In the years 1987-1990, the Rosemaund project monitored the run-off and leaching of 11 pesticides, applied according to Good Agricultural Practice at the upper end of the Rosemaund water catchment. The maximum concentrations reported were of the widely-used cereal herbicide isoproturon (maximum levels of 26.2 and 17.2 μg l^{-1} for field drains and streamwater respectively) although concentrations were generally below 0.1 μg l^{-1}. The major finding was that a small proportion (<1%) of most pesticides was able to transfer from the fields to the stream within a few hours of heavy rainfall, but that these levels peaked rapidly and then declined below detection limits once more. Computer modelling was only partially successful in predicting these events.

The purpose of this report is to describe in more detail the monitoring results obtained at Rosemaund during 1991.

13.2 Methods

These were described in MAFF (1992(a)), and consist of manual and automatic sampling of water in soil, field drains and stream. The automatic sampling is triggered by a pre-determined water flow-rate which is generated under most autumn and winter conditions by rainfall of about 10 mm in 24 h. Samples are solvent-extracted and the pesticide residues quantified by gas-liquid chromatography. Soil water samples are taken by suction samplers installed at 3 depths (0.5, 1.0 and 1.5 m) in Field 2 (see Figure 28).

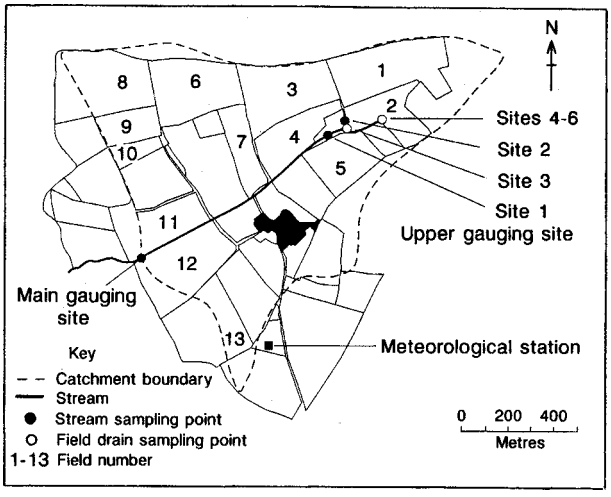

Figure 28. Location of sampling sites within Rosemaund catchment

The main reason for using the Rosemaund Farm site near Hereford is that the boundary of the farm largely follows the boundary of a small water catchment (Figure 28), thus affording excellent control over pesticide inputs. Collaborative work by the Soil Survey and Land Resource Centre (SSLRC) and the Institute of Hydrology (IoH) has shown that almost all the rain falling on the catchment that does not evaporate, passes into the stream through the field drains, mainly via cracks and other small holes in the soil matrix. Transfer of pesticides to the stream is therefore likely to be maximal, and the concentrations measured can probably be regarded as representing the upper end of the range for UK agricultural situations. This is an important consideration, given that the ultimate objective is to make realistic 'worst-case' predictions of the concentrations of pesticides to which aquatic organisms may become exposed.

The main pesticides studied in 1991 were the phenoxy herbicide MCPA and the organophosphorus insecticide oxydemeton-methyl. Both were applied to a winter barley crop on Fields 1 and 2 (Figure 28) between 28 February and 1 March, at rates of 1.68 and 0.114 kg ha^{-1} respectively (values expressed as active ingredient). Both of these substances are highly soluble in water and poorly adsorbed to soil particles, so they would be expected to leach fairly readily. However, their degradation half-lives in soil are rather short (days to weeks), a factor which would tend to mitigate the amounts washing into the Rosemaund stream.

13.3 Results

Oxydemeton-methyl was not detected (limit of detection = 0.2 µg l^{-1}) in any drain or stream samples taken between rainfall events when flows were small. Concentrations of MCPA between events were in the range 0.03-0.54 mg l^{-1} in the stream at Site 1 (Figure 28), although higher concentrations (0.02-4.75 µg l^{-1}) were seen in field drains at Sites 3-6. This effect is explained by the fact that a permanent spring at Site 2 contained almost uncontaminated water (MCPA = 0.02-0.09 µg l^{-1}) which diluted the field drain discharges just below the point where they enter the stream.

Three sets of rainfall events were monitored during March 1991. The first series occurred between 4 and 8 March when a total of 43.5 mm fell in 3 main bursts (Figure 29), although stream flow rates did not peak until after the second burst on 7 March. Initial automatic stream samples, taken about 12 h after the first burst, contained 12.4 µg l^{-1} MCPA and 0.8 µg l^{-1} oxydemeton-methyl, but these rapidly declined to about 0.5 and <0.2 µg l^{-1} respectively. It is interesting to note that the subsequent rainfall did not produce such large pesticide peaks, suggesting that most of the material then available had been flushed from the system. A similar pattern was seen in the field drain at Site 3, except that the MCPA peak was higher (18.8 µg l^{-1}). The total quantity of MCPA mobilised into the stream during 4-8 March was approximately 450 mg, or about 0.002 % of the total applied to the fields. An equivalent percentage of the oxydemeton-methyl was mobilised.

A similar picture emerged during the two subsequent rainfall events (16 March and 19 March), although by this stage the levels of oxydemeton-methyl were below the detection limit. On 16 March (Figure 30), a 10 mm rainfall event was followed within 2 hours by an MCPA peak in the stream of 12.7 µg l^{-1} which rapidly declined to about 0.3 µg l^{-1}. Figure 30 clearly shows that the pesticide peak coincided with peak water flow in the stream. Approximately 680 mg of MCPA was mobilised by this event. No samples were collected from field drains on 16 March, but after further light rainfall on 18 March, MCPA peaked in the Site 3 drain at 46.8 µg l^{-1}, declining to an average of 5 µg l^{-1} within 24 h. The total amount of MCPA estimated to have flowed down the Rosemaund stream during the period 28 February-19 March was approximately 1.3 g, or 0.005% of the total applied.

No further stream monitoring was conducted in spring 1991, but soil water sampling revealed that mean levels of MCPA dropped from 0.9 µg l^{-1} on 21 March to 0.2 µg l^{-1} on 23 April, most of this drop being attributable to degradation.

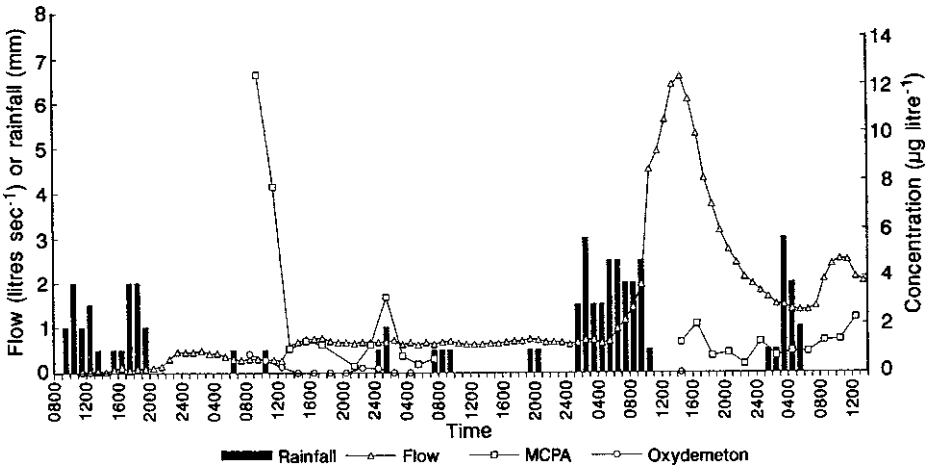

Figure 29. Rosemaund Site 1. MCPA/Oxydemeton-methyl experiment 4-8 March 1991

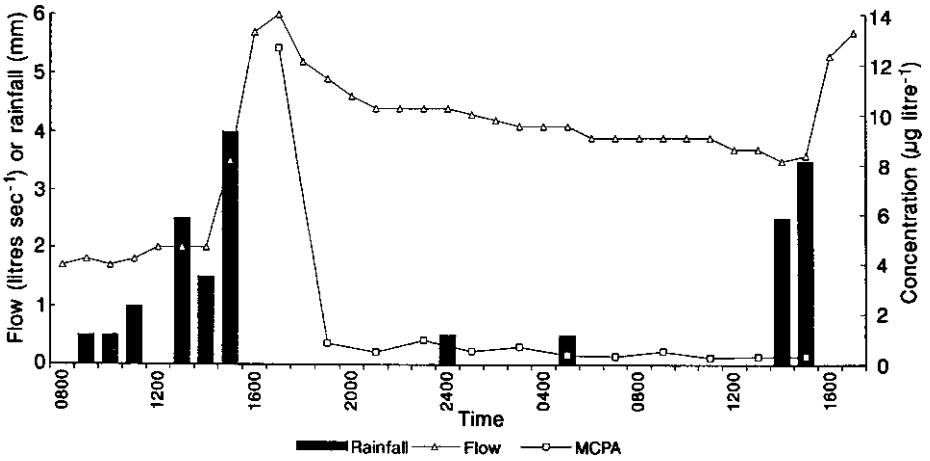

Figure 30. Rosemaund Site 1. MCPA/Oxydemeton-methyl experiment 16-17 March 1991

13.4 Conclusions

These data have not yet been fully evaluated in comparison with model predictions, but they are typical of earlier results obtained with similarly leach-prone pesticides (MAFF, 1992(a)), and confirm that some agrochemicals can rapidly translocate to streams at transiently high concentrations. However, it should be noted that less than 1% of any pesticide application has been found to transfer to the Rosemaund stream and that the stream itself is well upstream of any water abstraction points. A final season of work with leach-prone products (carbofuran, atrazine and aldicarb) in early 1992, will be reported in a future Report in this series. Subsequently, it is intended to study some highly adsorptive products (e.g. deltamethrin and trifluralin) which may, nevertheless, be transported to the stream in significant quantities associated with eroded soil particles. The monitoring work with carbofuran and deltamethrin, both of relatively high toxicity to aquatic life, will be accompanied by bioassays in order to assess the potential biological impacts of transient pesticide flushes.

14. STUDIES ON THE EFFECTS OF AGGREGATE EXTRACTION

14.1 Hastings Shingle Bank

In July 1991, a survey was conducted of the sediments and benthos present within and around a licensed aggregate extraction area at Hastings Shingle Bank in the English Channel (Figure 31(a)). This followed on from a side scan sonar survey of the same area, conducted seven months earlier, during which dredged and undredged substrates were distinguished (Figure 31(a)).

Samples were collected, using a modified Forster anchor dredge, from 16 stations in the vicinity of zone Y (see Figure 31(b)), a region of the Shingle Bank known to be subject to intensive suction trailer dredging (MAFF, 1992(a)). The anchor dredge is at best a semi-quantitative sampling device, due to uncertainties regarding its behaviour at the seabed. It is nonetheless useful for providing a qualitative description of the benthos and for showing up gross effects in the benthic communities. Half-litre and five litre sub-samples of the sediment were removed from each sample for the study of sediment

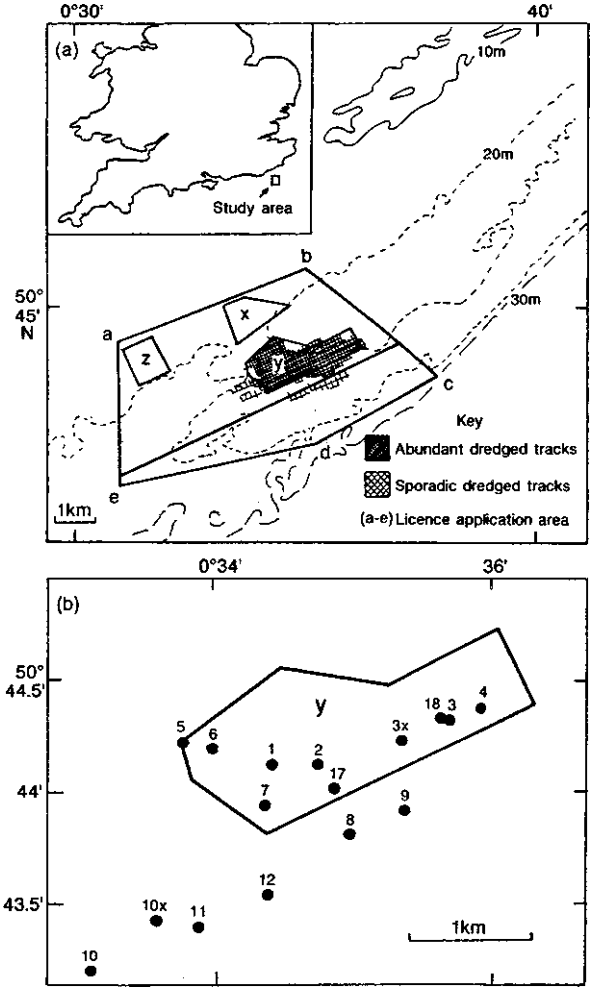

Figure 31. Location map of Hastings Shingle Bank showing (a) licensed extraction zones X, Y, Z and the distribution of dredged tracks as determined by the side-scan sonar survey of 1990; and (b) sampling stations for the July 1991 survey

particle size and fauna, in the 1-5 mm size fraction, respectively. The remaining sediment was sieved over a 5 mm mesh and all benthic species retained on this and in the 1-5 mm fraction, were identified, counted and, where necessary, preserved for later study in the laboratory. Distinction of the fauna in the 1-5 mm size fraction was considered necessary as, broadly speaking, these tend to represent the infauna in gravelly substrates such as those that occur at Hastings Shingle Bank, whereas species retained in the >5 mm fraction mostly represent the epifauna. For this purpose, the removal of a 5 l sub-sample was considered acceptable on the pragmatic grounds that it saved time during sample processing, while allowing direct comparisons to be made between anchor dredge samples of different volumes. More detailed descriptions of these procedures are presented in Lees *et al*., (1990) and Kenny *et al*., (1991). In the laboratory all animals were identified to species level, as far as possible, using a range of standard taxonomic keys.

The composition of the sediments from these stations is shown in Figure 32. Almost all may be classified as sandy gravels, containing from 40-80% gravel and less than 1% mud. Only at three stations, 3X, 10 and 18, does gravel form less than 30% of the sediment. These are also lacking in mud and are classified as gravelly sands.

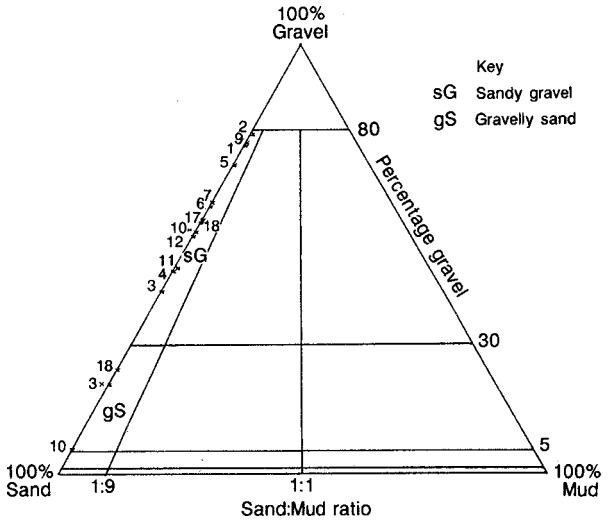

Figure 32. Hastings Shingle Bank 1991. Classification of sediments from anchor dredge samples

Approximately 150 different benthic taxa were recorded during the survey, of which up to 71 were present at each station. Figures 33 and 34 show the total numbers of taxa present in the 1-5 mm size fraction and in the >5 mm size fraction respectively, at each station. By comparison of these Figures with Figure 31(a), it is apparent that the substrates at those stations at which abundant dredged tracks were discerned, generally support far fewer benthic taxa than those at which few or

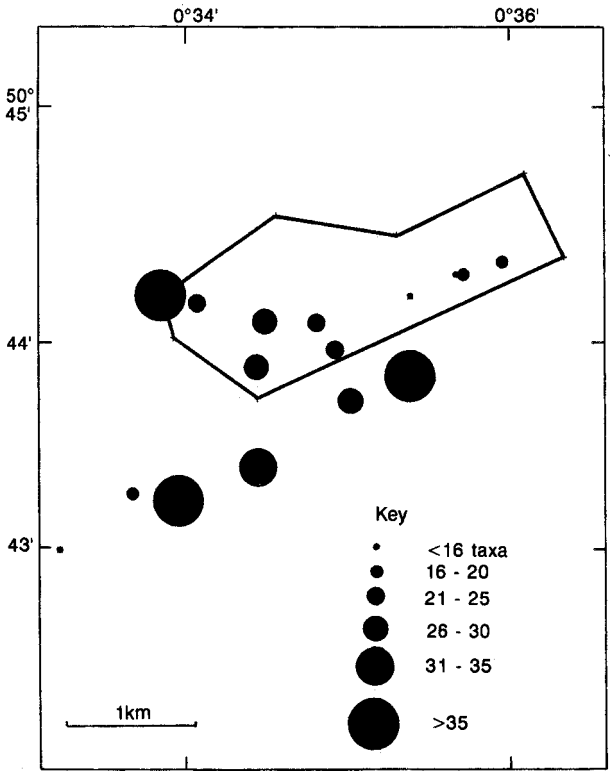

Figure 33. Hastings Shingle Bank 1991. Total number of taxa present in the 1-5 mm fraction of anchor dredge samples

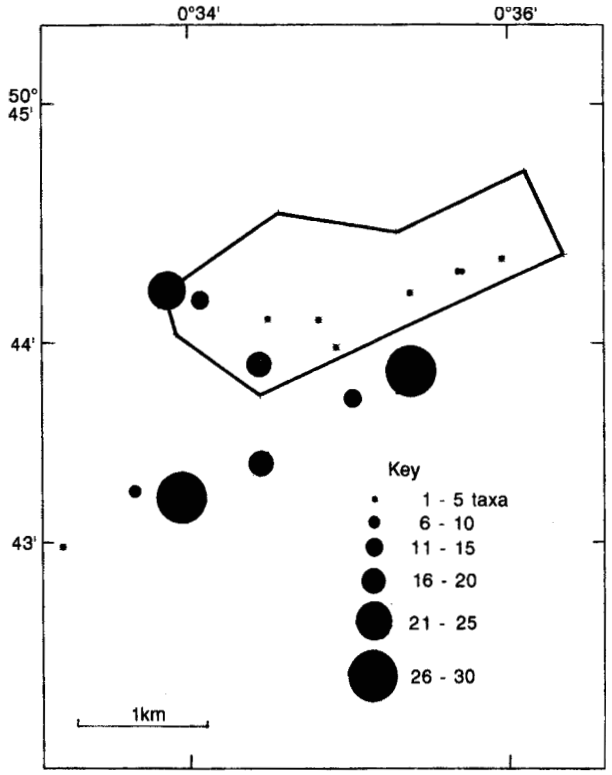

Figure 34. Hastings Shingle Bank 1991. Total number of taxa present in the >5 mm fraction of anchor dredge samples

no tracks were discerned. The one notable exception was at Station 10 where, despite being apparently undredged, the sediments supported few benthic taxa. This may however be explained by the sediment composition at this station which, as Figure 32 shows, was especially sandy in comparison to the other stations and almost devoid of gravel, suggesting a reduced diversity of habitats suitable for colonization.

Though varying greatly in abundance and diversity, the benthic communities sampled at each station were broadly similar in terms of the characterising species, being dominated by sessile epifauna attached to the pebbles, such as the barnacle *Balanus crenatus*, hydroids, serpulid worms and the bryozoan *Electra pilosa*, and by sand-dwelling species such as the polychaetes *Lagis koreni* and *Lanice conchilega*. No obvious change in the communities was evident from when they were surveyed in 1986 (Rees, 1987).

In order that the benthic communities present on the heavily dredged substrates might be compared with those from sparsely dredged or undredged substrates, the stations were split into two groups, A (dredged) and B (undredged), according to the distribution of dredged tracks shown in Figure 31(a). Stations 1, 2, 3, 3X, 4, 7, 17 and 18 may be ascribed to the former group (A) and Stations 5, 8, 9, 10, 10X, 11 and 12 to the latter (group B). Station 6 is situated on the edge of a heavily dredged region. However, it was also allocated to group B in order that each group was represented by a similar number of stations, so facilitating the comparison between the two regions.

Sixty-three taxa were identified in the 1-5 mm fraction from group A compared to 87 taxa from group B. The most abundant taxa from this fraction in both groups are listed in Table 23 and cumulative abundance curves for the non-colonial taxa are shown in Figure 35. The range of taxa and the dominant species present in both cases, were fairly similar. Although supporting a more diverse range of benthic species, overall abundance of individuals was slightly lower in group B than in group A. This may be accounted for by the higher abundance of the barnacle *Balanus crenatus* and the polychaete *Lagis koreni* in stations of group A.

The principal differences between the benthic communities on dredged and undredged substrates in this area are more evident upon examination of the fauna in the >5 mm fractions. Lists of the most abundant non-colonial taxa and of the most frequently recorded colonial taxa, are presented in Table 24. It is evident from this table that non-colonial species are generally more common, and colonial species more widely represented, on undredged and sparsely dredged substrates (group B stations) than on heavily dredged substrates (group A stations). Sixty-four taxa were recorded from stations in group B compared to less than half that number, 23, from those of group A. Benthic communities on dredged and undredged substrates

Table 23. Hastings Shingle Bank 1991. Numerically dominant non-colonial taxa from 1-5mm fraction of anchor dredge samples: Group A stations (from heavily dredged substrates); Group B stations (non-dredged and sparsely dredged substrates)

Stations	Taxon	Total abundance
Group A	*Balanus crenatus*	484
	Lagis koreni	296
	Lanice conchilega	62
	Anaitides spp	31
	Scalibregma inflatum	25
	Phaxas pellucidus	15
	Spiophanes bombyx	12
	Nemertea spp	11
	Lumbrinereis spp	10
	Abra alba	9
	Echinocyamus pusillus	9
Group B	*Balanus crenatus*	370
	Lagis koreni	174
	Spiophanes bombyx	47
	Echinocyamus pusillus	33
	Photis longicaudata	28
	Lanice conchilega	28
	Spisula elliptica	23
	Mysella bidentata	22
	Harmothoe spp	15
	Scalibregma spp	15
	Gari fervensis	15
	Phaxas pellucidus	15
	Glycera lapidum	12
	Scolelepis squamata	10

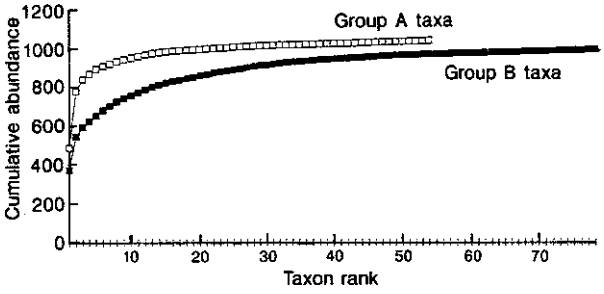

Figure 35. Hastings Shingle Bank 1991. Cumulative abundance curves for non-colonial taxa in 1-5 mm fraction

are dominated by the barnacle *Balanus crenatus*, which comprises approximately 99% of all the non-colonial individuals in the >5 mm size fraction on both types of ground. Cumulative abundance curves for the remaining non-colonial species recorded from both groups of stations show clearly the relative paucity of individuals and taxa upon the heavily dredged grounds (Figure 36).

Further studies, with scallop dredges and underwater cameras, are being conducted to verify these results.

Table 24. Hastings Shingle Bank 1991: (a) numerically dominant non-colonial taxa from >5mm fraction of anchor dredge samples; and (b) most frequently recorded colonial taxa from >5 mm fraction of anchor dredge samples; Group A stations (heavily dredged substrates); Group B stations (non-dredged and sparsely dredged substrates)

(a)

Stations	Taxon	Total abundance
Group A	*Balanus crenatus*	8431
	Lanice conchilega	15
	Lagis koreni	7
	Spisula elliptica	2
	Ophiura albida	2
Group B	*Balanus crenatus*	10142
	Pomatoceras triqueter	30
	Sabellaria spinulosa	24
	cf. Sagartia sp	11
	Lanice conchilega	11
	Dendrodoa grossularia	9
	Anthozoa sp B	7
	Lagis koreni	3
	Liocarcinus spp	3
	Onchidoris muricata	3
	Ophiura albida	3
	Tunicata sp B	3

(b)

Stations	Taxon	Count of stations at which recorded
Group A	*Electra pilosa*	8
	Halecium spp	3
	Hydrallmania falcata	2
	Sertularia cupressina	2
Group B	*Electra pilosa*	7
	Hydrallmania falcata	7
	Sertularia cupressina	6
	Halecium spp	6
	Alcyonidium diaphanum	4
	Campanularia hincksii	3

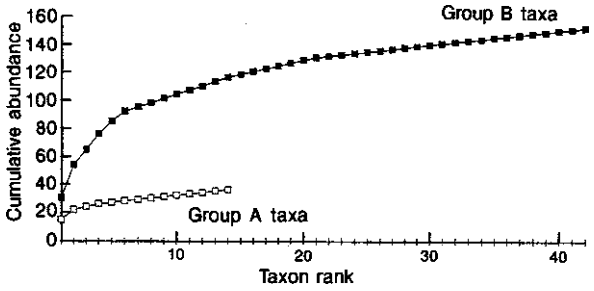

Figure 36. Hastings Shingle Bank 1991. Cumulative abundance curves for non-colonial taxa (excluding Balanus crenatus) in >5 mm fraction

14.2 Dredger Outwash Study

In October 1991, a study was conducted to ascertain the physical condition of benthic organisms being returned to the sea in the outwash of a suction trailer dredger. Through the assistance of ARC Marine Ltd, the sampling work for this study was conducted aboard the vessel *ARCO TYNE* during routine dredging operations in the 'Owers Bank' (eastern English Channel) and 'South-west Needles' (English Channel, west of the Isle of Wight) licensed aggregate extraction areas (Figure 37).

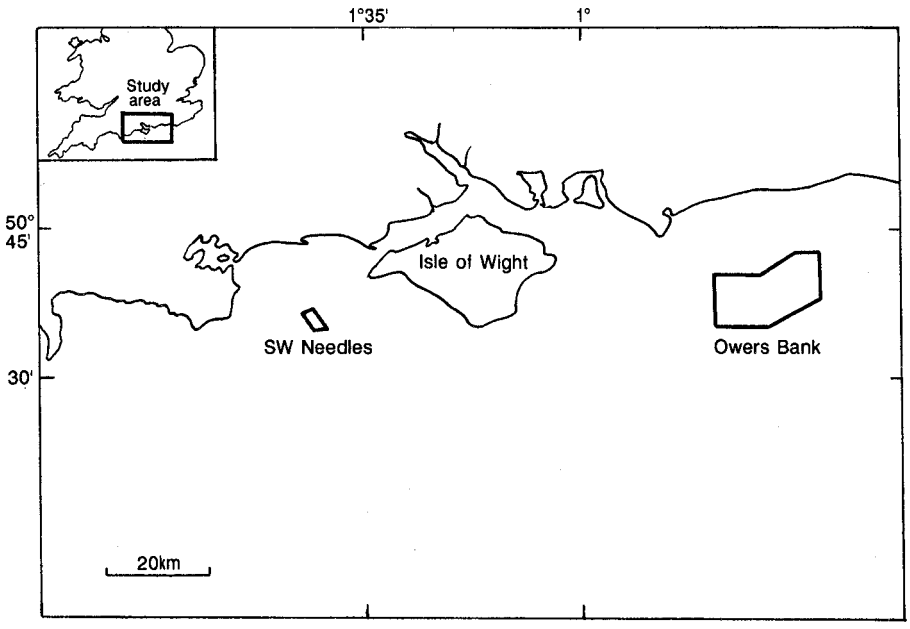

Figure 37. Dredger Outwash Experiment. Location of aggregate extraction areas dredged during the study

The *ARCO TYNE* has a hopper capacity of about 2200 m^3 (equivalent to roughly 3500 t of aggregate when fully loaded), which takes from two and a half to five hours to load. At least ten times this volume of water is also discharged into the hopper during the operation. This flows back to the sea, as outwash, via eight spillways. The fauna were sieved out of the outwash by fitting an open ended mesh basket, constructed out of 5 mm stainless steel mesh, into one of the dredger's spillways. Benthic species were also collected from the surface of the dredger's hopper after dredging had been completed, in order that their physical condition could also be assessed and a comparison made between the types of animals retained in the dredger and those that were washed back to sea. A potential disadvantage of the spillway sampling method is that injury to some individuals may occur through entrapment in the mesh basket, due to the strength of the current flowing through it, rather than as a consequence of the dredging process. Available evidence suggests that the proportion of injuries which arose in this way was slight compared to that resulting from the dredging operation; however, sampling procedures aimed at reducing this source of error are currently under investigation.

A species abundance list for all the fauna trapped by the mesh basket in the dredger spillway over a total period of 50 minutes, and for those retrieved from the hopper surface, is presented in Table 25. The number of individuals showing signs of physical damage is also indicated. The fauna collected from the spillway have been totalled separately from those collected from the hopper as the former, had they not been sampled, would have been washed back to the sea whereas the latter would have remained in the cargo.

It is apparent from this table that many species were susceptible to injury during the dredging operation, even though they were subsequently discharged to sea with the outwash. Most worms were damaged, often seriously, or, in the case of tube-dwelling species, washed out from their tubes so predisposing them to predation upon return to the sea. Many crustaceans also appeared particularly prone to injury, often through loss of limbs (e.g. spider crabs, *Macropodia* spp, and the porcelain 'crab' *P. longicornis*). The ability of the worms to regenerate or the crustaceans to grow further limbs, would vary from species to species and depend upon the extent of the injuries. However, it is likely that all damaged individuals would be susceptible to predation.

Of the fauna discharged to sea via the outwash many, potentially unharmed, individuals would have been too small to have been retained within the mesh basket and so would not have been recorded. Certain of the animals which were trapped did appear in remarkably good physical condition and may have survived upon return to the sea. These included the bulk of the fish, most amphipods, the shrimp *P. brevirostris*, the burrowing anemone *C. lloydii* and the holothurian *L. inhaerens*. Many other species also appeared generally undamaged by the dredging operation, but were recorded too infrequently for any inferences to be made concerning each species' ability to withstand the dredging process.

Certain benthic species are more likely to be retained in the hopper than discharged as outwash and so, whether damaged or not, will be lost as they are processed with the aggregate. Species from this study most susceptible to such a fate included hermit crabs (*P. bernhardus*), starfish (*A. rubens*) and virtually all fauna encrusting or attached to stones such as slipper limpets (*C. fornicata*), bryozoa, tunicates, hydroids and certain anemones and polychaete worms. Although infrequently sampled in this study, it would seem likely that heavily shelled fauna, such as certain bivalves (e.g. *G. glycymeris*), gastropods (e.g. *Buccinum*) and crustaceans (e.g. brown/edible crab, *C. pagurus*) might also suffer in this way.

It is clear from this initial study that not all benthic species are affected in the same way by suction trailer dredging and that, while some appear prone to injury or death, others may be able to survive such operations.

Table 25. Dredger outwash study 1991. List of species sampled from the dredger spillway and from the surface of hopper and a count of those showing signs of physical damage. Presence of colonial species is indicated by the letter P; no tally is given of the proportion of such species showing signs of damage due to the difficulty of making such an assessment. See text for discussion of the probable fate of such species

Species	Benthos retrieved from spillway		Benthos collected from hopper	
	Total	Damaged	Total	Damaged
CNIDARIA				
Halecium beanii	P			
Halecium halecinum	P		P	
Nemertesia sp	P			
Plumularia setacea	P		P	
Abietinaria abietina	P			
Hydrallmania falcata	P		P	
Sertularia cupressina	P		P	
Sertularia distans	P			
Anthozoa sp A	2	0		
Anthozoa sp B	1	1		
Cerianthus lloydii	25	4		
Edwardsiidae sp	4	2	1	0
NEMERTEA				
Nemertine spp	2	2		
PRIAPULIDA				
Priapulus sp	1	1		
SIPUNCULA				
Golfingia elongata			1	1
Golfingia vulgaris	1	1		
ANNELIDA				
Harmothoe sp	2	2		
Anaitides spp	4	3		
Glyceridae sp	1	1		
Glycera sp			1	1
Glycera gigantea	6	6		
Glycera lapidum	10	6		
Goniada maculata	4	2		
Syllidae sp	1	0		
Nereis sp	1	1		
Nephtys sp	1	1	3	3
Marphysa belli	5	3		
Marphysa sanguinea	1	1		
Nematonereis unicornis	1	1		
Lumbrinereis spp	39	36		
Drilonereis filum	2	1		
Orbinia latreilli	1	1		
Chaetopterus sp	1	1		
Caulleriella sp	5	4		
Notomastus latericeus	17	16		
Nicomache lumbricalis	1	1		
Scalibregma inflatum	5	5		
Owenia fusiformis	1	1		
Lagis koreni	15	11		
Sabellaria spinulosa	8	8		
Ampharete acutifrons	5	3		
Terebellides stroemi	49	26		
Terebellidae sp	1	1		
Sabellidae sp A	4	4		
Sabellidae sp B	1	1		
Sabellidae sp C	10	10		
Branchiomma bombyx	3	0		
Pomatoceros lamarcki	2	0		
Protula tubularia	2	2		
Oligochaete sp	1	1		
CHELICERATA				
Nymphon brevirostre	1	0		

Species	Benthos retrieved from spillway		Benthos collected from hopper	
	Total	Damaged	Total	Damaged
CRUSTACEA				
Scalpellum scalpellum	1	0		
Amphipoda spp	1	1		
Gammarellus homari			1	0
Ampelisca sp	1	1		
Ampelisca spinipes	32	9		
Maera othonis	7	3		
Astacilla longicornis	1	1		
Decapoda sp	1	1		
Caridea sp	1	0		
Hippolyte varians	2	1		
Pandelina brevirostris	30	5	1	0
Crangon crangon	1	0		
Pontophilus bispinosus	1	0		
Nephrops norvegicus	1	1		
Callianassa subterranea	28	28		
Upogebia deltaura	2	2		
Pagurus bernhardus			84	11
Galathea intermedia	11	11		
Galathea strigosa	1	1		
Pisidia longicornis	19	19	9	9
Inachinae sp	1	1		
Macropodia linaresi	5	5		
Macropodia rostrata	18	17		
Cancer pagurus			1	1
Liocarcinus marmoreus	1	1	1	1
Liocarcinus pusillus	1	1	2	2
MOLLUSCA				
Leptochiton asellus	1	0		
Acanthochitona sp	1	1		
Crepidula fornicata (cols.)				41
Goniodoris nodosa	3	0		
Onchidoridea sp	1	0		
Pelecypod sp A	2	2		
Pelecypod sp B	2	2		
Glycymeris glycymeris			1	0
Solenidae sp	1	1		
BRYOZOA				
Alcyonidium diaphanum	P		P	
Vesicularia spinosa	P			
Amathia lendigera	P			
Electra pilosa	P			
Flustra foliacea	P		P	
Cellaria sinuosa	P			
Bicellariella ciliata	P			
ECHINODERMATA				
Asterias rubens			9	0
Psammechinus miliaris	2	2	3	0
Holothuroidea sp	1	0		
Thyone fusus	1	0		
Leptosynapta inhaerens	12	3		
Ophiura albida	1	1		
TUNICATA				
Ascidia mentula	2	2		
Molgula manhattensis	4	0		
PISCES				
Diplecogaster bimaculata	5	2		
Aspitrigla cuculus			1	0
Callionymus lyra	4	0		
Pomatoschistus pictus	13	4		

14.3 Regional comparison of aggregate assemblages off the east and south coasts of England

14.3.1 Introduction

The purpose of this study was to describe the benthic fauna of sediments in the vicinity of areas commercially dredged for marine aggregates (Figure 38). A preliminary account was given in MAFF (1992(a)) and a full account of the objectives can be found in Kenny *et al.* (1991).

14.3.2 Methods

All samples were collected from MAFF research vessels using a modified Forster Anchor Dredge (Holme and McIntyre, 1984) and procedures for sample processing are described in Kenny *et al.* (1991). Although uncertainties exist over the performance of the Anchor Dredge as a quantitative device, it is nevertheless considered adequate for describing regional differences in the nature of gravel assemblages. Improved quantification may be achieved through the use of alternative sampling devices such as the Hamon Grab (Oele, 1978). The comparative sampling efficiency of the Hamon Grab vs Anchor Dredge is the subject of current research by MAFF.

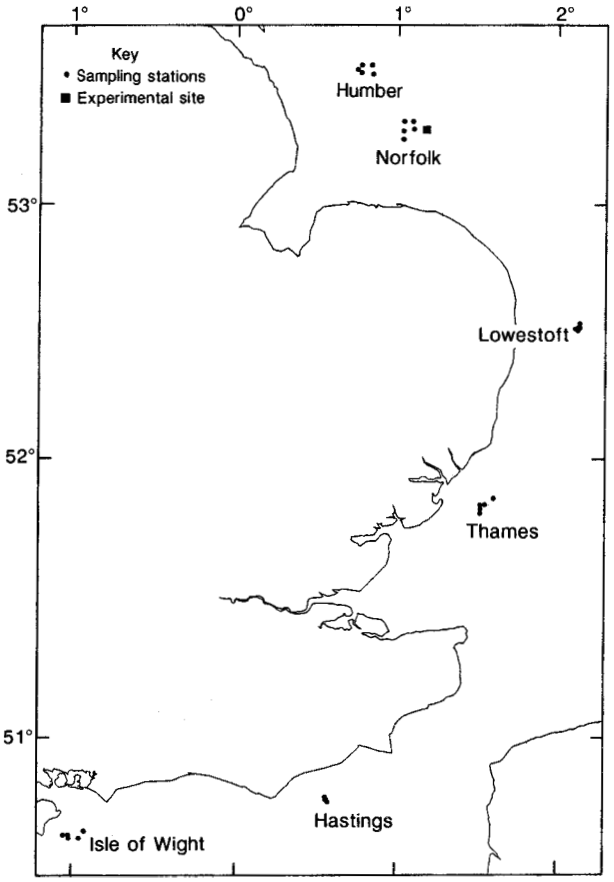

Figure 38. Sample station positions and experimental site

14.3.3 Results

The outcome of average-linkage cluster analysis (Lance and Williams, 1967) of presence-absence data is shown in Figure 39. It is clear that stations have been clustered according to their respective regions and that regions have been linked at different levels of similarity.

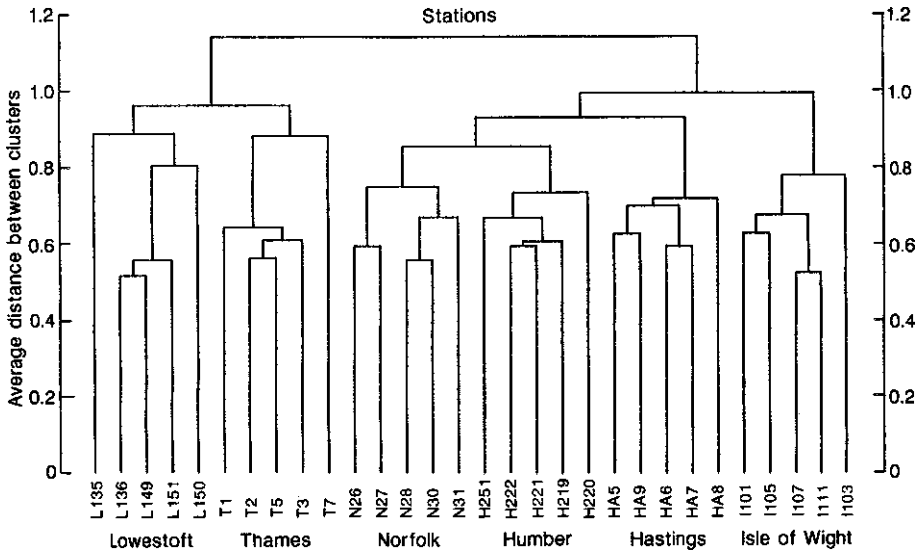

Figure 39. Dendrogram showing the outcome of cluster analysis of presence/absence data for the >5 mm fraction of the benthos from anchor dredge

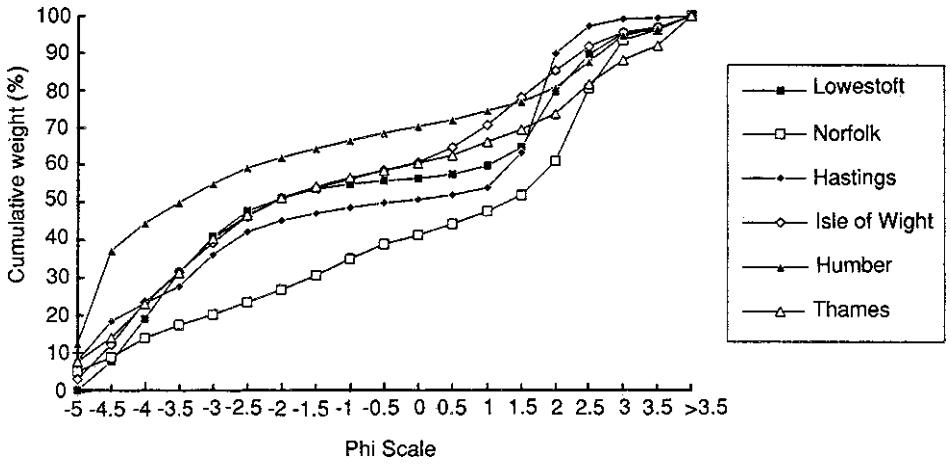

Figure 40. Average particle size distribution curves

Particle size distributions, averaged between stations for each region, show that the sediments all have a bimodal distribution with the exception of Norfolk which has a poorly defined coarse mode (Figure 40). 'Static' particle size distributions alone cannot explain the observed faunistic differences. However, dynamic properties such as the abrasive effects of sand shifting under strong tidal currents and disturbance caused during storms are likely to be particularly important in determining the assemblage types especially off Lowestoft and the outer Thames. Such effects on the benthos may account for the relatively close similarity of these two regions in the dendrogram (Figure 39) . Similarities in the faunal composition off Lowestoft and the outer Thames, such as the prominance of the purple sea urchin *Psammechinus miliaris*, are identified in Table 26.

By contrast, the Norfolk and Humber regions, which are also paired in Figure 39 appear to be more stable, as indicated by the dominance of the long-lived horse mussel *Modiolus modiolus* and associated epifauna (Table 26). Further research is being undertaken to determine the cause of such regional differences; studies include an assessment of the biogeographic range of species, analyses of tidal current data and investigations of the seabed geology.

14.4 Selection of an experimental dredging site

During September 1991, a collaborative geological and biological survey was undertaken on board the *RV CHIEFTAIN* chartered by South Coast Shipping Company Limited, with the aim of selecting an experimental site to investigate the processes of recolonisation of aggregate sediments following commercial dredging. A small area (300 x 500 metres) 17 miles north of Cromer, was successfully defined (Figure 38) and subsequently dredged in April 1992. The results of the pre- and post-dredging surveys will be reported at a later date.

Table 26. The four top-ranked taxa from combined Anchor Dredge samples taken from six locations

Humber	**Norfolk**	**Lowestoft**
Modiolus modiolus	*Modiolus modiolus*	*Ophiura albida*
Nucula nucleus	*Crepidula fornicata*	*Psammechinus miliaris*
Ocenebra erinacea	*Ophiura albida*	*Sagartia sp*
Ophiothrix fragilis	*Polycarpa sp*	*Owenia fusiformis*
Thames	**Hastings**	**Isle of Wight**
Sagartia sp	*Galathea intermedia*	*Nucula nucleus*
Psammechinus miliaris	*Sagartia sp*	*Crepidula fornicata*
Owenia fusiformis	Chitons	*Pagurus sp*
Nucula nucleus	*Psammechinus miliaris*	*Polycarpa sp*

15. TEMPORAL TRENDS IN THE CONCENTRATIONS OF METALS AT THE TYNE, THAMES AND NAB SEWAGE-SLUDGE DISPOSAL SITES

15.1 Tyne

Each year approximately 500 000 tonnes (wet) of sewage sludge are deposited at a disposal site 10 kilometres off the mouth of the River Tyne. By determining faecal bacteria in seabed sediments, Rowlatt *et al.* (1991) showed that the zone of initial sludge settlement is situated at the southern edge of the disposal site (Figure 41). They also demonstrated that, with the possible exception of chromium, there was no evidence of any accumulation of metals at the settlement area in 1988.

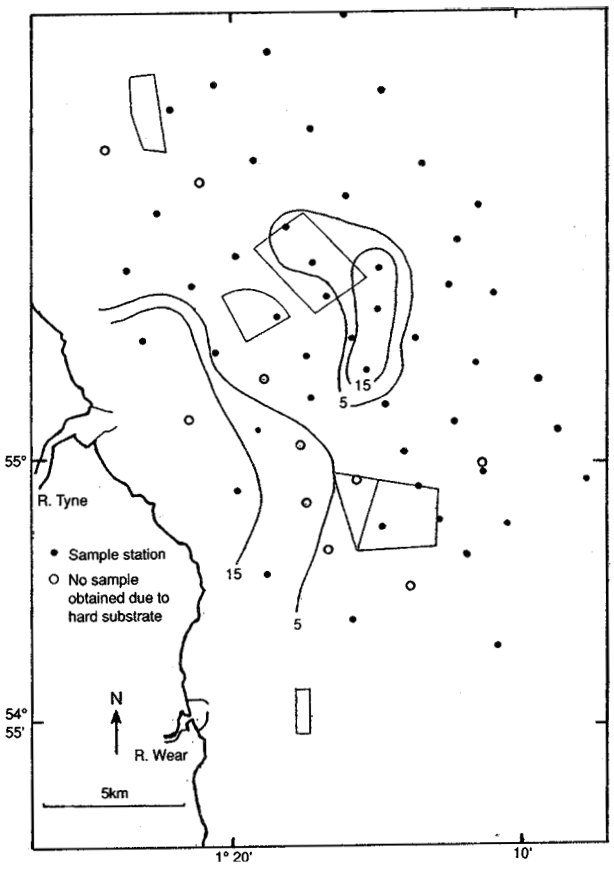

Figure 41. Distribution of faecal bacteria (E. coli) in sediment around the Tyne sewage-sludge disposal site. The samples were collected in May 1988 and the results expressed as numbers ml^{-1} of sediment

In order to assess whether settlement results in a long-term buildup of sludge-derived metals at the seabed, sediments have been collected annually in May, when weather conditions permitted. They were taken at positions located randomly in a sampling zone centred on the settlement area. The surface 1 cm of sediment was sampled and sieved at 63 µm according to the Group Coordinating Sea Disposal Monitoring (GCSDM) guidelines (MAFF, 1989). The fraction passing through the sieve was digested with *aqua regia* and analysed using atomic absorption spectrophotometry.

Figure 42 shows the mean concentrations of sediment metals with error bars of 1 standard error of the mean. This figure shows no increasing trend in concentrations of metals and in some cases (e.g. chromium and lead) suggests a decrease. From these data it may be concluded that the sewage-sludge disposal operation at the Tyne site is not causing a year-on-year deterioration in sediment quality. Taking these observations with the earlier work of Rowlatt *et al.* (1991), it may be concluded that the sludge disposal operation at the Tyne is not, as far as metals are concerned, having a detrimental effect on environmental quality either in the short- or long-term.

15.2 Thames

Sewage sludge from London has been deposited at a designated site in the Barrow Deep since 1967 (Figure 43). Water movements transport the dispersing sludge around the East Barrow Sand and initial settlement of sludge solids occurs at several locations, as has been demonstrated using radiotracers (Figure 43).

During the period 1985-1991, samples of surface sediment were collected, at least annually, from five stations in both the Barrow and Middle Deeps. On each occasion new sets of stations were located randomly within set sampling zones (Figure 43). Figure 44 shows the mean value of fines (% <63µm) at the two sites and demonstrates a high degree of synchronicity between them, suggesting that at least a component of the variations are region-wide rather than simply local. During the period 1986-1990, the fines at the two sites exhibited not only synchronicity but also generally lower concentrations in autumn, the time of greatest storm frequency, and higher in summer, a time of relatively quiescent conditions. This indicates that the fine fraction of the surface sediments is mobile, particularly during autumn and winter storms, and confirms the dispersive nature of the outer Thames Estuary.

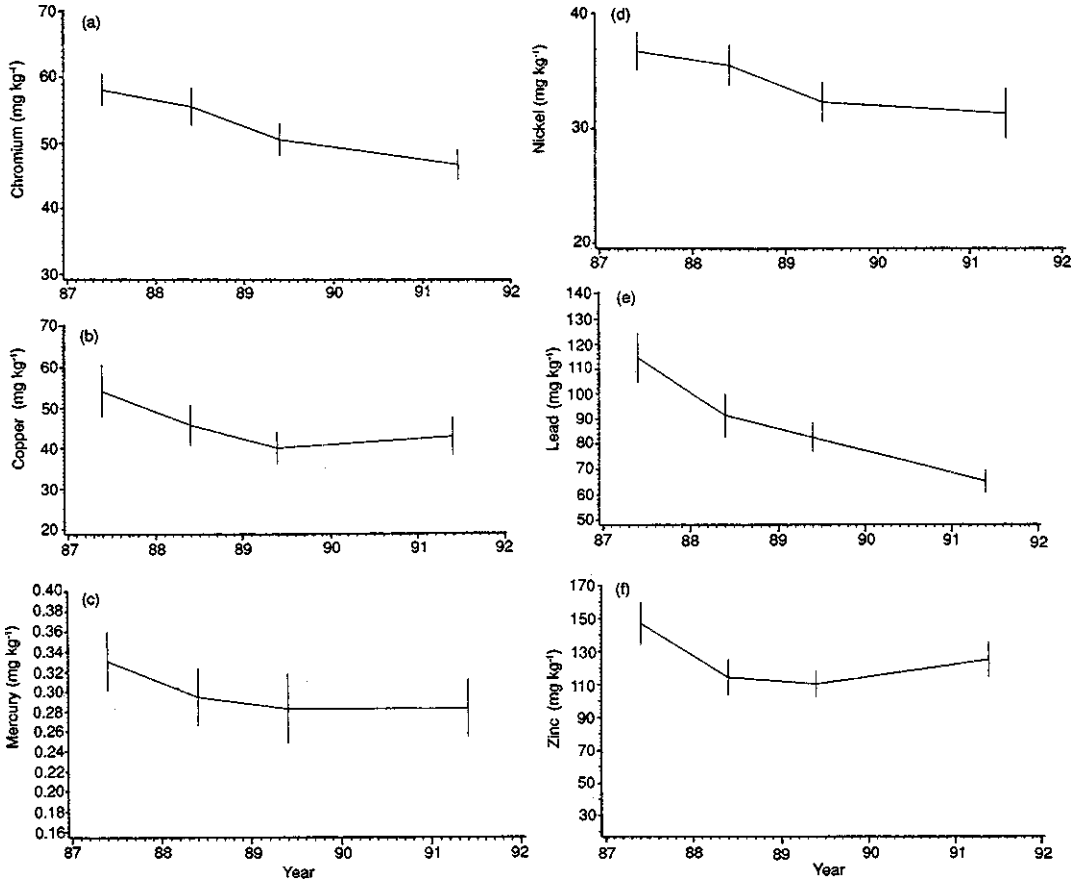

Figure 42. ***The concentrations of various metals in the <63 m fraction of sediment collected in the zone of initial settlement at the southern edge of the Tyne sewage-sludge disposal site; (a) chromium; (b) copper; (c) mercury; (d) nickel; (e) lead and (f) zinc***

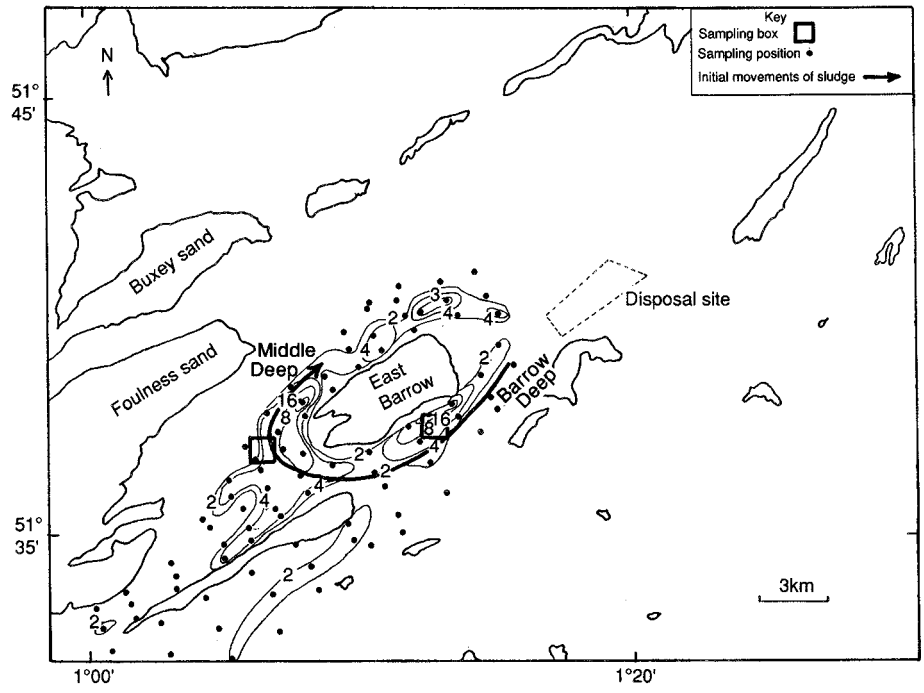

Figure 43. ***Location of the Barrow Deep sewage-sludge disposal site and MAFF sampling zones in the Outer Thames, showing the direction of residual water flow movement and principal areas of initial settlement of sewage sludge as shown by the distribution of radioactivity from labelled sludge (Talbot et al., 1982)***

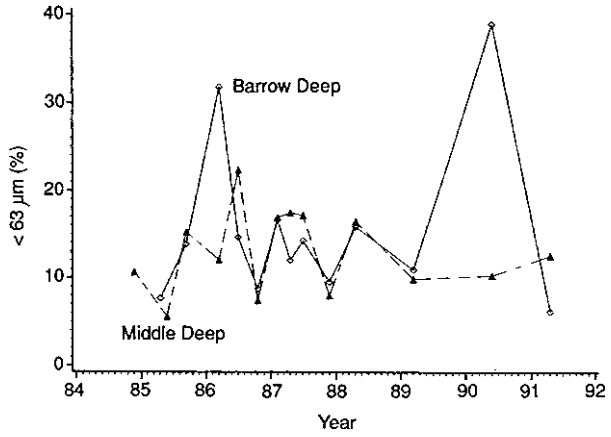

Figure 44. Fines (% <63 m) content of sediment from the Barrow and Middle Deep sampling zones

The strong association of many metals with fines (see for example Figure 45) due to, among other things, adsorption on clay minerals, can cause a strong positive relationship between metal and fines when whole sediments are considered. In the present case, this applies to chromium, copper, mercury, nickel, lead and zinc. It is possible to counter the effects of variations in fines content on concentrations of metals in whole sediment by examining metal/fines ratios; i.e. to normalise the metal data.

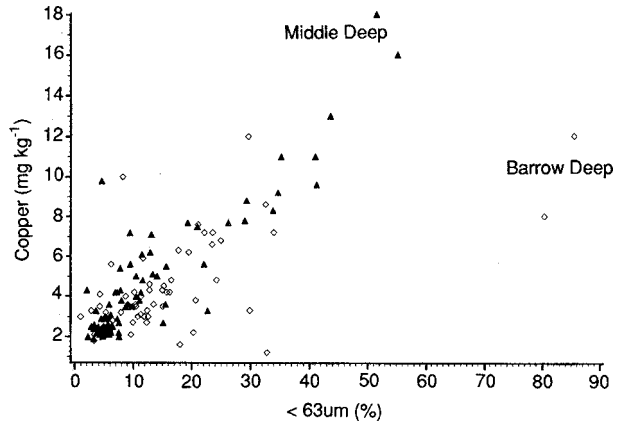

Figure 45. Relationship between copper (mg kg^{-1}) and fine sediment (% < 63 m) in sediments from the Barrow and Middle Deep sampling zones

Figures 46 and 47 show the mean metal/fines ratios and ranges of mercury and lead (as examples of the trace metals analysed) in the Barrow Deep and Middle Deep zones. The most notable feature of these datasets is that while there have been some variations, there is no evidence of a trend, either worsening or improving, over the period studied.

15.3 Nab Tower

Each year approximately 250 000 tonnes of sewage sludge, and between 600 000 and 1 500 000 tonnes of dredged material are deposited at the Nab Tower disposal site.

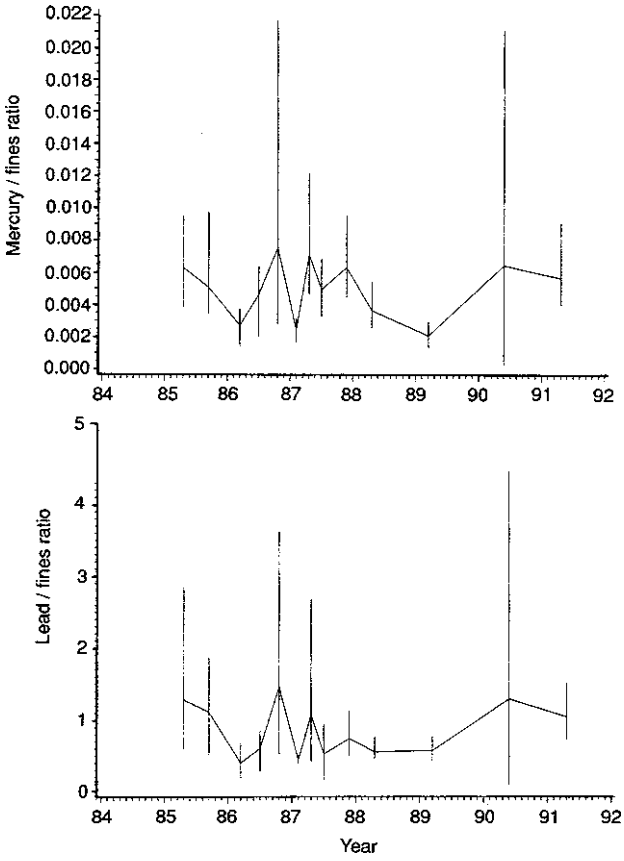

Figure 46. Mean and range of metal/fines ratios of mercury and lead in sediments from the Barrow Deep sampling zone

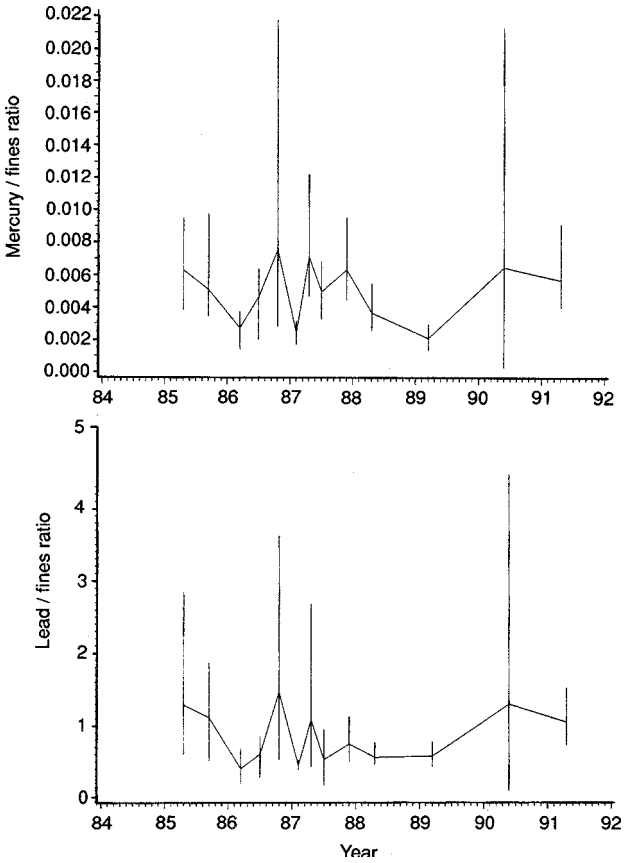

Figure 47. Mean and range of metal/fines ratios of mercury and lead in sediments from the Middle Deep sampling zone

Surveys carried out in 1989 (23 samples), and 1991 (21 samples) were based on a random stratified design and centred on the area of likely impact. This type of survey design was used to monitor changes in sediment quality over time. Samples were wet sieved at 63 µm and the fine fraction digested using *aqua regia*, and analysed by flame atomic absorption spectrophotometry.

Table 27 shows the mean values and standard deviations for the concentrations of metals in the <63 µm sediment fraction in 1989 and 1991. Few firm conclusions can be drawn until further data are collected, although it should be noted that the values are within the ranges found at other UK sewage-sludge disposal sites.

Table 27. Concentrations of metals at the Nab Tower disposal site in 1989 and 1991

Year		Concentration (mg kg^{-1})					
		Hg	Pb	Cu	Zn	Cr	Ni
1989	Mean	0.41	62.6	27.8	106.7	42.3	38.9
	S.D.	0.39	98.8	22.6	46.8	10.2	31
1991	Mean	0.29	72.6	60.9	163	55.9	27.1
	S.D.	0.28	55	48.3	117	11.4	8.4

Note: all cadmium analyses were below the detection limit of the method used (0.2 mg kg^{-1})

16. TRANSECT STUDIES AT THE TYNE SEWAGE-SLUDGE DISPOSAL SITE

In May 1991, sampling was conducted along two parallel transects of stations running through the Tyne sewage-sludge disposal site (Figure 48). The benthic macrofauna (animals retained on an 0.5 mm mesh sieve), along with sediment samples for determinations of trace metal, organic content, and particle size, were taken by Day grab. The meiofauna (animals passing through 0.5 mm mesh sieves) were sampled by Craib core, as part of a collaborative project between MAFF and the Plymouth Marine Laboratory, funded by the Chief Scientist's Group, to assess the utility of this component of the benthos in pollution monitoring. Prior to processing, Eh determinations of intact cores were made at 1 and 6 cm depths, using a 5 mm diameter combination redox electrode (Eh provides an indication of the reduction-oxidation status of sediments, which may be influenced by inputs of organic matter).

The transects took account of previous findings (e.g. Rees *et al.*, 1992(a)) which identified mild enrichment in the immediate vicinity of the disposal site. In the following account, data from adjacent stations across the transects were combined.

Eh values from Craib cores at 1 and 6 cm depths are shown in Figure 49. At 1 cm depth, all values were positive, providing no indication of significant deterioration in sediment quality. Ranges astride each mean indicate appreciable between-core variability at most stations; however, noticeably lower mean values occur in surface sediments at and just beyond the southern part of the disposal site. This trend is not apparent at 6

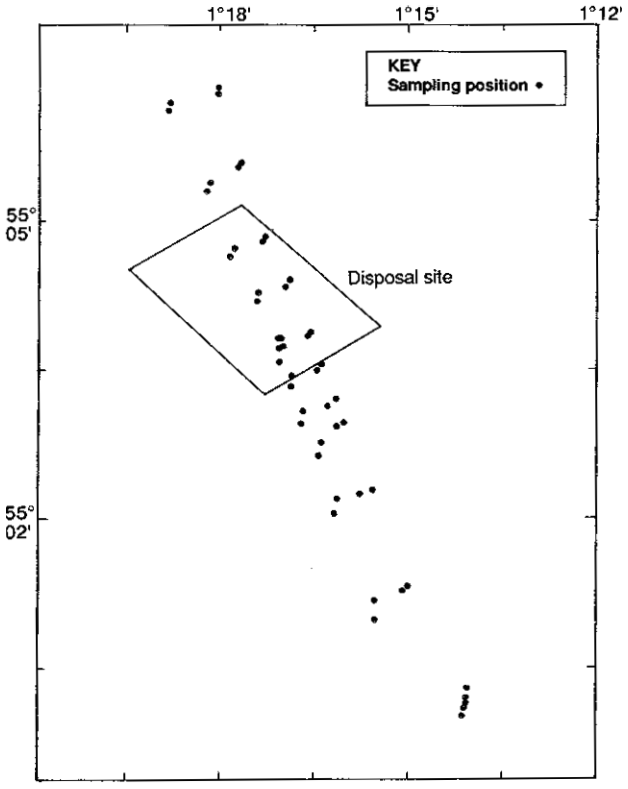

Figure 48. Benthic sampling stations at the Tyne sewage-sludge disposal site, May 1991

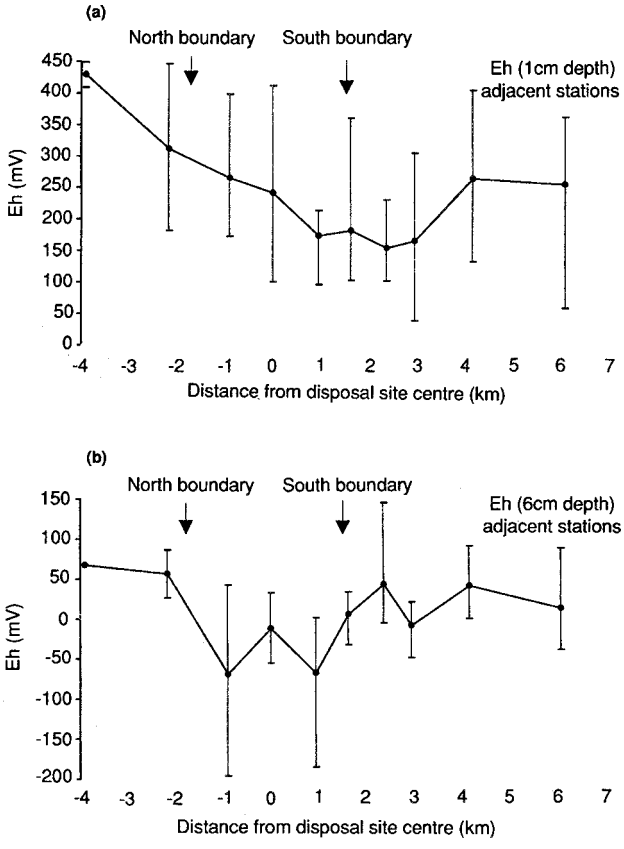

Figure 49. Trends in Eh values along transects through the Tyne sewage-sludge disposal site ($\bar{x}$ + range): (a) 1 cm depth and (b) 6 cm depth

cm, again indicating that the physico-chemical condition of sediments in the general area has not been significantly impaired by sludge disposal. However, the two lowest values were encountered at approximately 1 km either side of the disposal site centre.

Tomato pips are a commonly used indicator of contamination by sewage particulates as they generally survive passage through the human digestive system and sewage treatment works. Trends in counts (Figure 50) provide clear evidence of localised accretion. Peaks at the northern and southern edges of the disposal site can probably be accounted for by the tendency of the disposal vessel to alternate between these locations during discharge. While there is slight attenuation of the curve to the south, indicating some lateral transport, it is clear that — at least for this particulate component — contamination is largely confined to within and immediately beyond the disposal site.

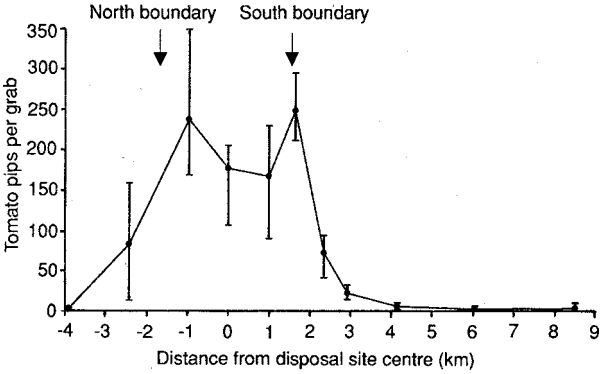

Figure 50. Trends in numbers of tomato pips along transects through the Tyne sewage-sludge disposal site ($\bar{x}$ + range)

In Figure 51, trends in total numbers of macrofauna individuals and taxa, and of tomato pips, are compared with earlier data along a transect running south from the disposal site. (Sampling was conducted in February, 1984 and June, 1986). Peaks in tomato pips correspond with those in abundance and taxa, and add weight to earlier conclusions concerning a localised response of the benthos to mild organic enrichment. A comparison of counts of tomato pips between years suggests that there may be a long-term trend towards accumulation at the disposal site. However, there is no indication of a worsening trend in the fauna - indeed patterns in the 1991 data are very similar to those of 1984 and, for numbers of individuals, are not markedly elevated, compared with trends in 1986. A similar pattern of localised response has been observed in the meiofauna (Somerfield and Gee, in preparation). A full account of macrofauna changes along the length of the transects of Figure 48 awaits completion of sample analyses.

Between-sample variability in macrofauna counts was relatively high in 1991, which may be partly accounted for by adverse weather conditions at the time of sampling. Nevertheless, results were consistent with earlier findings, which indicated mild organic enrichment close to the disposal site. Such changes remain within proposed 'Environmental Quality Standards' for the benthos at this location (MAFF, 1992(b), Rees and Pearson, 1992). Annual sampling at representative stations, as reported in MAFF (1992(a)), is continuing, and will be the subject of future reporting.

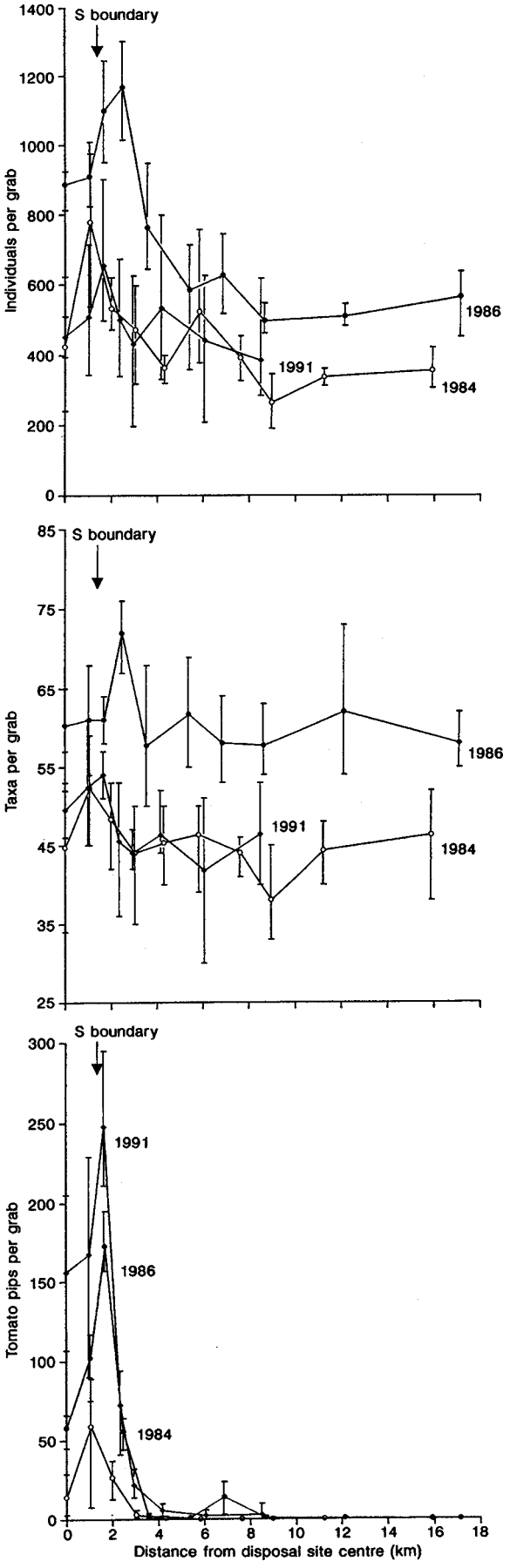

Figure 51. Trends in individuals, taxa and tomato pips along a transect running south from the Tyne sewage-sludge disposal site (x + range)

17. STUDIES AT THE LIVERPOOL BAY SITE Z DREDGED MATERIAL DISPOSAL SITE

17.1 Introduction

Each year approximately 3 million tonnes of dredged material are deposited at Site Z, situated 7 km north west of the entrance to Queens Channel (Figure 52). These dredgings range from clean sands from the approach channels, to muds dredged from Liverpool Docks. Tidal currents flow in an east-west direction, but are likely to be modified by a tongue of mobile sand just inshore of the disposal site.

17.2 Geochemical studies

In 1991, sediments were sampled, using a Day grab, along a transect of stations running from north of the disposal site to the Burbo Bight (Figure 52). The samples were sieved at 63 µm and the fine fraction was digested with *aqua regia* and analysed using flame atomic absorption spectrophotometry.

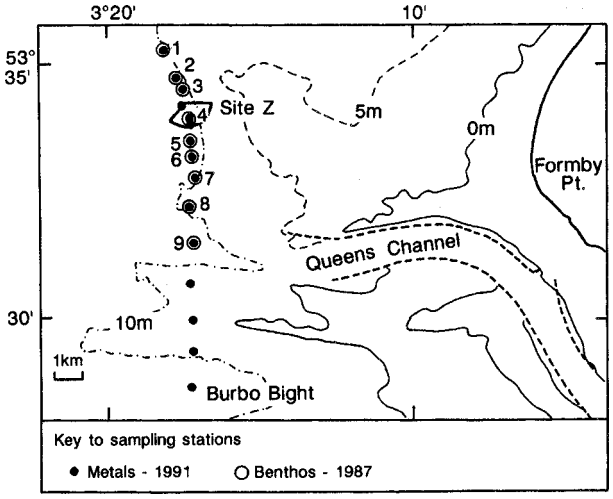

Figure 52. Location of Site Z and sampling stations

All metals show a similar distribution along the transect (Figure 53(a-g)). From relatively high concentrations at the extreme north, concentrations of metals fall until the apex of the disposal site where concentrations are relatively high. Peak concentrations occur at the southern edge of the disposal site and then fall away rapidly, south of the disposal site. Some metals (e.g. Zn, Pb and Cd) exhibit high concentrations in the Queens Channel and Burbo Bight areas.

The highest concentrations of Zn, Hg and Cr occur at the disposal site, while those of Pb, Cu and Cd occur at the mouth of the Mersey. This indicates that dredgings disposal is an important, but not always the most important, source of contamination in these sediments.

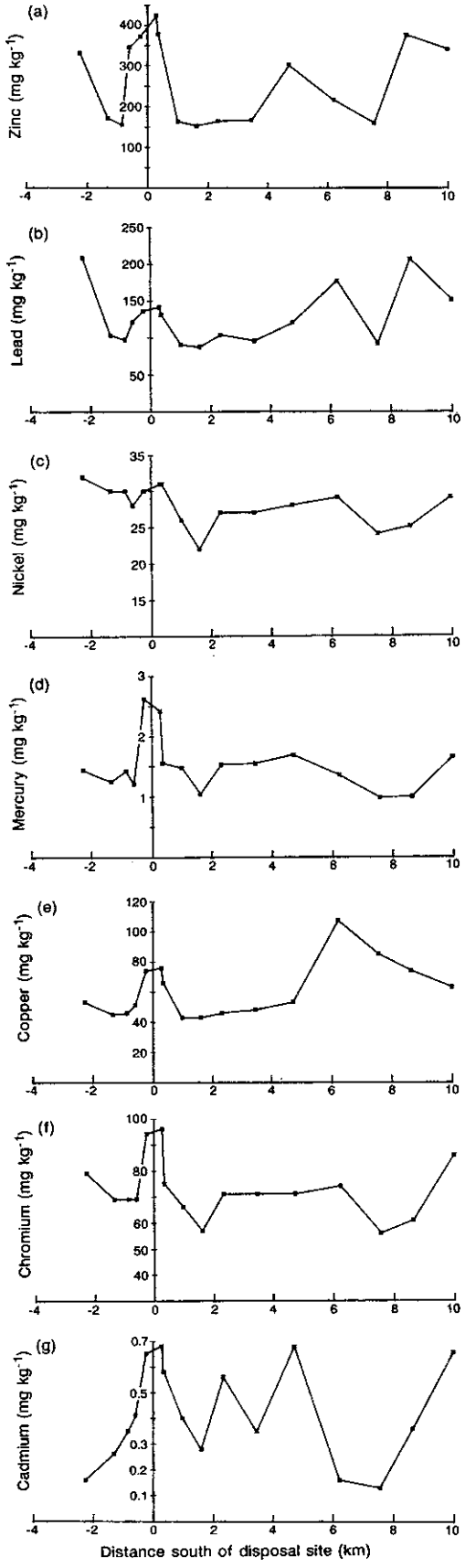

Figure 53. Concentrations of metals in the <63 m fraction of sediment samples collected on a transect through Site Z (see Figure 52 for locations): (a) zinc; (b) lead; (c) nickel; (d) mercury; (e) copper; (f) chromium and (g) cadmium

17.3 Benthic studies

During 1991, analyses were completed of a series of macrofauna samples taken in 1987 by Day grab, along a transect of stations through Site Z. The results complement those of Rowlatt *et al.* (1986) and Rees *et al.* (1992(b)), and will be published separately (Rees *et al.*, in preparation). As the results are of interest in relation to sediment sampling in 1991, for concentrations of trace metals (see above), and also in relation to collaborative MAFF/PML sampling of both the macrofauna and meiofauna at this time, a summary of the main findings is given below.

Station positions are shown in Figure 52. The transect was aligned in a north-south direction so as to minimise gradients associated with depth.

The purpose of the biological sampling was to examine, in some detail, the nature of the response of the benthos to dredgings disposal, rather than to identify the spatial extent of effects. The latter aspect is covered in Rees *et al.* (1992(b)). At each station, three Day grab samples were taken for the macrofauna, which were extracted over a 1 mm mesh sieve. Sub-samples were taken from a fourth sample for analyses of trace metal/organic content and particle size.

Sediments at the disposal site were characteristically muddy in nature, in contrast to the prevailing muddy sand substrate to the north and south. However, muddy samples were also encountered at the two southern-most stations, which impinge upon a 'mud-patch' off the Mersey Mouth, which is known to vary in extent from year to year (Norton *et al.*, 1984). There were also some similarities in the fauna as a consequence. Trace metal and organic content of sediments were largely determined by the 'fines' content (see Rowlatt, 1988 and MAFF, 1991), although there was some evidence of enhanced concentrations of mercury and cadmium at the disposal site itself, which may reflect the disposal of more contaminated muddy material from the River Mersey.

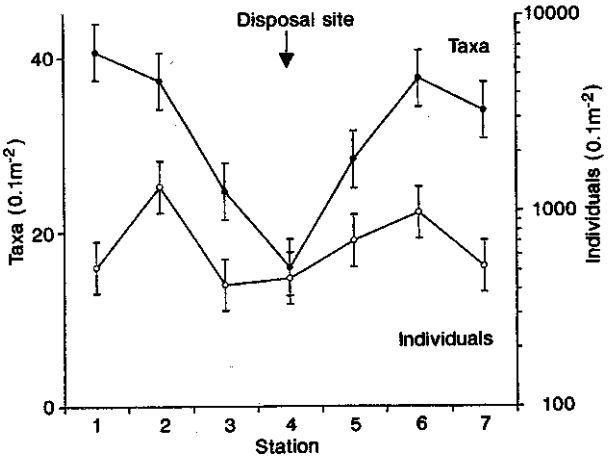

Figure 54. Trends in numbers of taxa and individuals along a transect through Site Z; results are expressed as means with 95% Least Significant Intervals

Trends in numbers of individuals and taxa along the transect are shown in Figure 54. (The two southern-most stations were omitted so as to clarify the effects of dredgings disposal; characteristics of the fauna at these stations are described later). The data are expressed as means with 95% Least Significant Intervals : stations may be assumed to be significantly different in cases where the latter do not overlap (see Andrews *et al.*, 1980). There is a significant reduction in taxa at the disposal site, and a less marked one at immediately adjacent stations. In contrast, there is no marked depression in abundance at the disposal site. Trends in abundance of the numerically dominant taxa are given in Figure 55. Regular disposal of dredgings has the predictable effect of reducing or eliminating taxa common to nearby sediments. However, counts of the tube-dwelling polychaete *Pectinaria* were elevated at the disposal site, accounting for the absence of a marked reduction in total numbers of benthos.

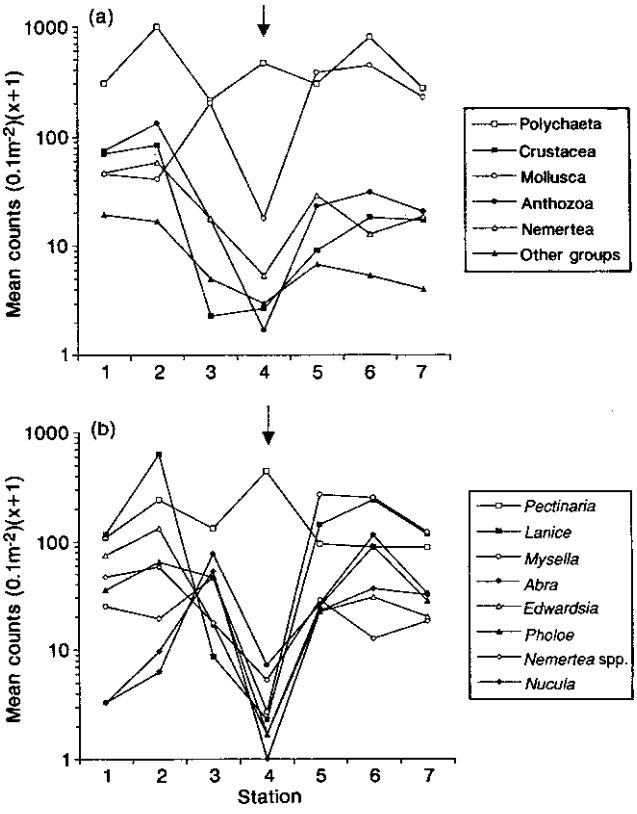

Figure 55. Trends in abundance of: (a) the major faunal groups and (b) the numerically dominant taxa along a transect through Site Z. Error bars are omitted for clarity. (Arrows indicate location of disposal site)

Another feature of the fauna at the disposal site was the localised presence in low numbers of the oligochaete *Tubificoides*, the polychaete *Capitomastus* and the bivalve mollusc *Macoma*. These taxa may have survived the process of dredging of estuarine sediments, and deposition at the site (cf. Wildish and Thomas, 1985).

 A comparison of the fauna at the disposal site with

the two muddy stations at the southern end revealed a number of similarities, e.g. high densities of the polychaete *Pectinaria* were present at all three stations, and an examination of their size structure showed that all were characterised by a preponderance of small-sized individuals: The numbers of taxa present were also similar (on average, 19 and 17 taxa per grab, compared with 16 at the disposal site). However, the polychaete *Pholoe* and the bivalve mollusc *Abra* were present in appreciable numbers at the southern-most stations, giving rise to a more equitable distribution of counts among the taxa and hence a somewhat higher diversity.

There was some evidence to suggest enhanced survival and/or growth of *Pectinaria* and other taxa at stations immediately to the south of the disposal site; this may be a consequence of the stabilising or nutritional properties of dispersing dredged material, or efflux of organic matter from the River Mersey itself.

Changes in the benthos within and immediately adjacent to the disposal site were consistent with effects of dredged material disposal; however, these effects were not widespread along the transect. The combination of a high frequency of physical disturbance, high ambient suspended solids load associated with dispersing fine material, elevations in certain contaminants and the development of anoxia beneath the surface of muddy deposits is likely to account for modifications to the fauna at the disposal site itself.

Pectinaria appears to fulfil a role as an opportunistic coloniser of newly-deposited sediment within the disposal site; its capacity to thrive in relatively exposed shallow coastal deposits is reflected in its widespread occurrence in muddy substrates of the inner Bay (see, e.g. Eagle, 1975). However, dense populations of classical indicators of disturbance such as the polychaete *Capitella* were not found.

The responses of the macrofauna to dredgings disposal were less negative than might at first have been anticipated, e.g. there was no evidence of a widespread area of azoic sediments. The proliferation of *Pectinaria* within the disposal site, and possible enhancement in biomass on the periphery, might suggest some beneficial consequences, e.g. for fish-feeding. However, there may be an attendant risk of enhanced bioaccumulation of contaminants associated with muddy dredgings, and then transfer through the food chain. Work is currently in progress to address this issue.

18. A PRELIMINARY ASSESSMENT OF THE SEDIMENTS AND BENTHIC FAUNA AT THE BARROW-IN-FURNESS DREDGED MATERIAL DISPOSAL SITE

Deepening of the approach channels to Barrow-in-Furness during 1991, produced approximately 8 million tonnes of dredged material, most of which was deposited at a site to the south of the southern tip of Walney Island. In December 1991, MAFF carried out a survey of the area to examine the sediment types and assess the impact of the operation on the benthic fauna (Figure 56).

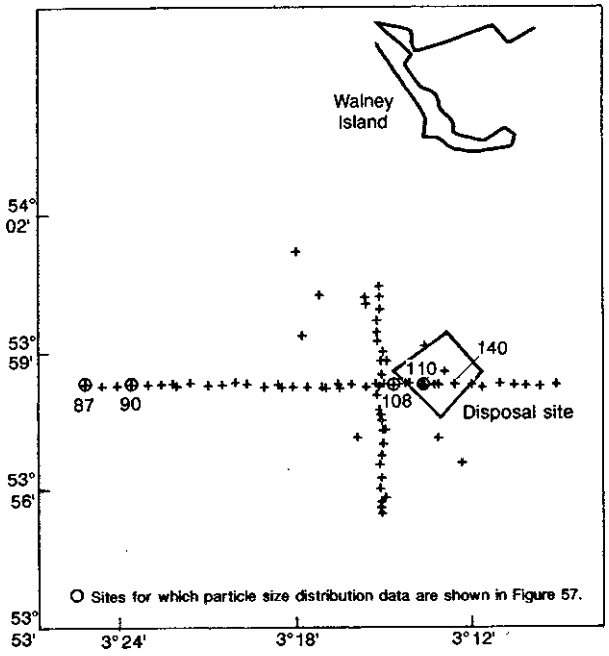

Figure 56. The Barrow-in-Furness dredged material disposal site and sampling grid used in the MAFF Survey, December 1991

Figure 57 shows the particle size distribution of the sediments at representative sites along the east-west transect shown in Figure 56. These distributions were generated by a combination of sedigraph analysis of material passing a 63 μm sieve and dry sieving of the material retained. It can be seen that inshore the sediments are coarser than offshore and they are better sorted.

The larger benthic organisms retained on a 5 mm mesh sieve were identified on the ship at the sampling sites described above (Figure 56). The purpose was to

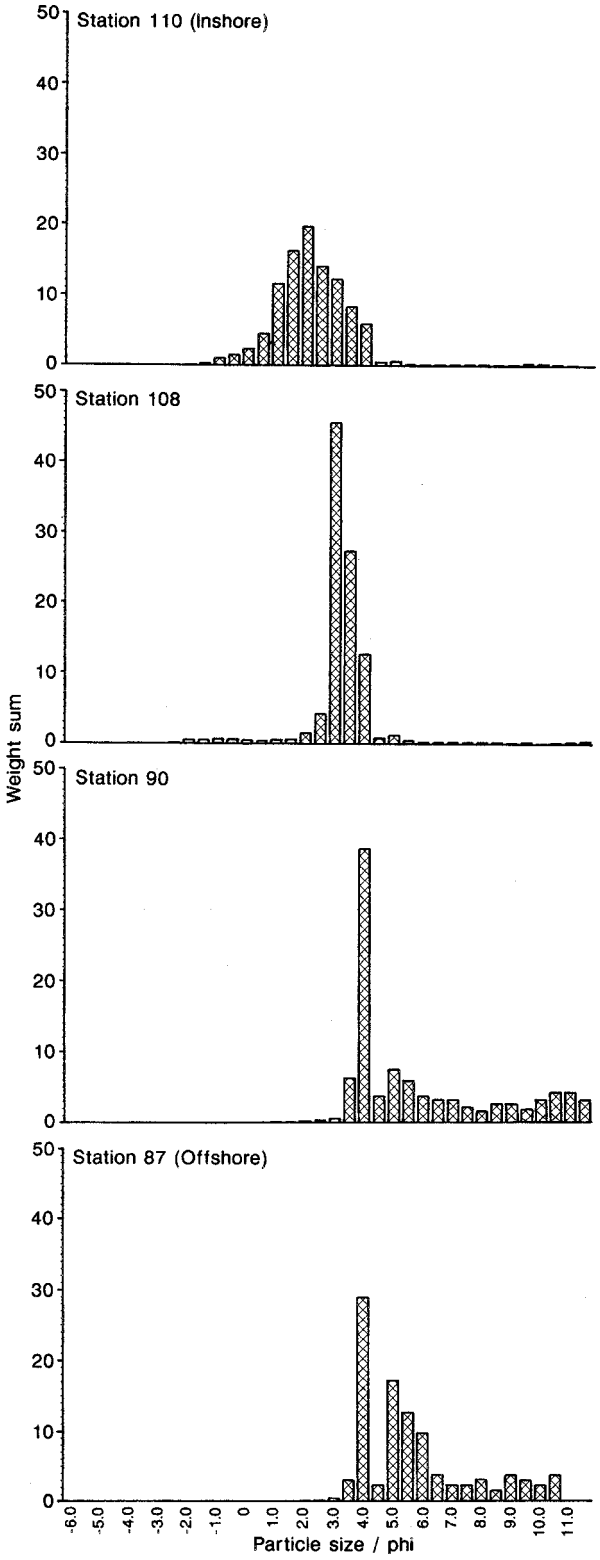

Figure 57. Particle size distribution of various sediments collected in December 1991. Location of sampling sites are shown in Figure 56

identify any gross changes in the nature of the fauna as an aid to the selection of representative stations for follow-up sampling (and later laboratory analysis), employing a 1 mm mesh sieve for extraction of organisms (see Figure 56).

The outcome of cluster analyses by station and by taxon is shown in Figure 58, for the east-west transect. The Bray-Curtis index (Bray and Curtis, 1957) was employed as a measure of affinities between stations or taxa, and the resulting matrices were grouped using the Average-Linkage sorting procedure of Lance and Williams (1967). (Colonial organisms such as hydroids and bryozoans, which were recorded on a presence/absence basis, were excluded from these analyses). The major distinction is between an offshore assemblage dominated by the tube-dwelling polychaete *Pectinaria* and the brittle-star *Amphiura*, and an inshore assemblage characterised by the presence of the bivalve *Nucula* and the polychaete *Nephtys*.

Explanations for this difference can be found in a shift from mud to muddy sands (with an appreciable shell content), coinciding with a higher rate of shallowing at about this transition point in an inshore direction. Muddy sands at the inshore end of the transect in the vicinity of the disposal site were characterised by the presence of clay lumps and a reduced complement of larger taxa. (Figures 58 and 59: note that no quantifiable animals were recorded at Stations 109 and 110, and hence these were excluded from the cluster analyses).

Sediments inshore of Station 140 appeared to be characterised by the presence of stones with associated epifauna, though it was not possible to obtain satisfactory samples with the Day grab from these locations.

Conclusions

1. Gross characterisation of the fauna retained on a 5 mm mesh sieve provided evidence of a dichotomy mid-way along the east-west transect which could be linked with a shift from muddy sands to muds and which is considered to be an entirely natural feature of the area.

2. Sediments in the immediate vicinity of the disposal site could be characterised by the presence of dredged material, and a reduced range - or absence - of larger benthic species. Stony substrates inshore of the disposal site were unsuitable for quantitative sampling; this feature is again considered to reflect a natural gradient of change in the locality.

3. Stations representative of the major faunal and substrate types have been established with a view to follow-up monitoring of any changes relating to dredged material disposal which might occur on localised or wider scales. The results will be reported at a later date.

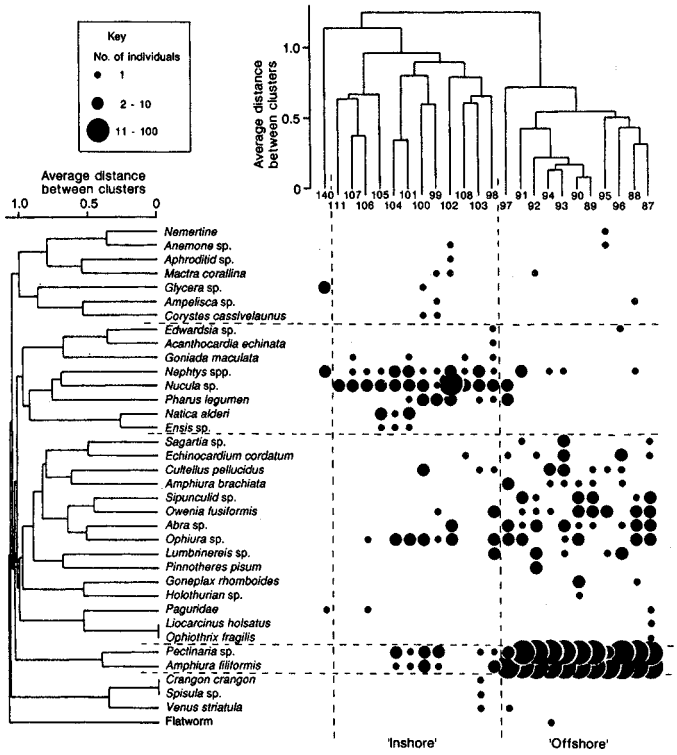

Figure 58. Combined output of cluster analysis of quantitative data by station and by taxon

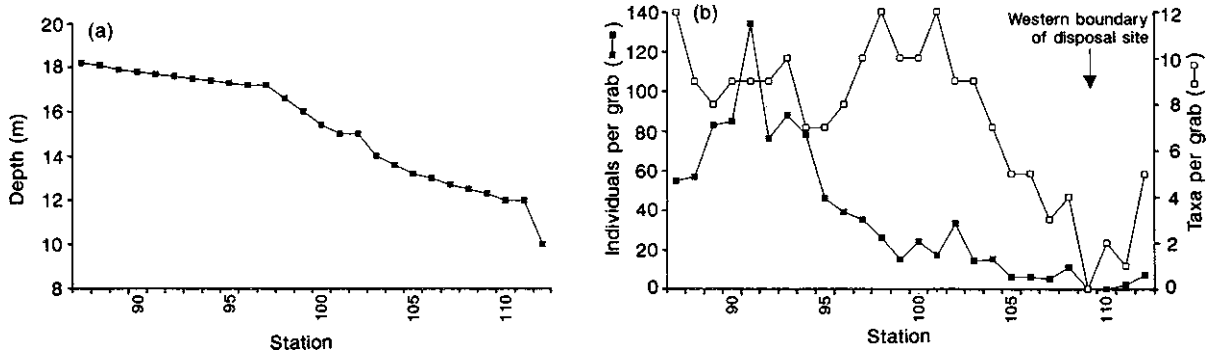

Figure 59. Approximate depths at Lowest Astronomical Tide for stations along the east-west transect off Barrow-in-Furness: (a) trends in total numbers of taxa (including colonial organisms) and (b) individuals along the same transect

DISPOSAL AT SEA: LICENSING AND RELATED ACTIVITIES

19. LICENSING OF DISPOSAL OF WASTES AT SEA

19.1 Introduction

This section gives information about the licensing of disposal of wastes at sea in the UK during 1991 under Part II of FEPA (1985) (Great Britain - Parliament, 1985(a)). It fulfils an undertaking by the Government to report on the licensing, enforcement and monitoring of activities related to the disposal of wastes at sea.

In accordance with that undertaking and for convenience, licensing statistics for Scotland and Northern Ireland are included in this Section to provide statistics for the UK as a whole.

An initial report, describing the licensing system, and giving information on wastes licensed in 1986 and 1987, was published in 1989 (MAFF/DAFS, 1989) and the information on licensing and enforcement for 1988 and 1989 was included, together with a report on the related monitoring activity in England and Wales, in a report in this series (MAFF, 1991). A further report was prepared for 1990 (MAFF, 1992(a)).

19.2 Legislation and licensing authorities

The disposal of waste at sea, as opposed to discharge into the sea via pipelines, is controlled by a system of licences issued under Part II of FEPA. Certain operations (e.g. deposit of scientific instruments, navigation aids), are exempt from licensing under the Deposits in the Sea (Exemptions) Order 1985 (Great Britain - Parliament, 1985(b)). In England and Wales, the licensing function rests with MAFF and in Scotland with the Scottish Office Agriculture and Fisheries Department (SOAFD). In Northern Ireland, the issuing of licences for disposal at sea is the responsibility of the Department of the Environment for Northern Ireland (DOE(NI)).

Section 147 of the Environmental Protection Act (1990) (Great Britain-Parliament, 1990) provides for further additional material about sea disposal activities to be added to the public registers, established under FEPA. In future the registers will also contain information about applications, breaches of the legislation and enforcement action.

19.3 Enforcement

Scientists from the Burnham-on-Crouch Laboratory have powers to enforce licence provisions by visits to production units, storage sites and disposal vessels. They may take samples, and check records including logbooks. They carried out 65 inspections in 1991. The Sea Fisheries Inspectorate, with staff based on the coast, detects unlicensed disposal operations and enforces licence conditions relating to the disposal of the wastes in the designated disposal area. They made 162 inspections in 1991.

In Scotland, similar enforcement powers are held by staff of the SOAFD Marine Laboratory and by the Scottish Sea Fisheries Inspectorate (SSFI). The Laboratory made nine enforcement visits in 1991 and a further twelve visits were made by the SSFI. In Northern Ireland, enforcement duties are carried out by officers of the Department of the Environment's Environmental Protection Division. Two inspections were considered necessary in 1991.

MAFF investigated seven reports of unlicensed operations or of breach of licence conditions. In three cases warning letters were sent and in one further case it was established that no licence was needed since the disposal was above high water. It was not possible to establish who was responsible for another case of unlicensed disposal and no action was taken when MAFF were informed by a responsible body that they had been unable to retrieve chains and anchors accidentally lost off the North Cornwall Coast.

In 1991, MAFF successfully prosecuted a marina and the skipper of a disposal vessel for breach of conditions set in a licence. The marina was fined £1 000 on each of three charges and £500 costs. The skipper was fined £300 on each of the three charges and £100 costs.

In Scotland, SOAFD investigated six reports of unlicensed work or cases where it was alleged that there had been a breach of licence conditions. One further case was investigated in Northern Ireland.

19.4 Report on licensing activities

Tables 28 to 33 give details, over the period 1987-1991, of the number of sea disposal licences issued, the quantity of waste licensed, and the quantity actually

Table 28(a). Liquid industrial wastes licensed for disposal at sea in 1991 (1)

Licensed quantity (t)	Company	Description of waste	Disposal sites	Quantity deposited (t)
165,000	Imperial Chemical Industries Ltd	Ammoniacal liquor	Tees	160,619
40,000	Sterling Organics Ltd.	Phenolic liquor	Tyne/ Spurn Head	31,326

Notes: (1) No liquid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report
Licensed quantities : licences were issued throughout the calendar year 1991 and were generally valid for twelve months
Tonnages deposited : relate to quantities deposited in the calendar year 1991, which may be covered by two licences, including one issued in 1990

Table 28(b). Summary of liquid industrial wastes licensed and disposed of at sea(1)

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1987	21	338,871	264,131	0.02	0.35	0.66	0.01	0.57	0.76	0.83
	1988	19	311,411	249,744	0.03	0.29	0.88	0.01	0.41	0.63	0.76
	1989	16	292,968	248,454	0.02	0.19	0.78	0.00	0.27	0.81	0.60
	1990	5	228,000	209,961	0.02	0.10	0.46	0.00	0.19	0.56	0.27
	1991	2	205,000	191,945	0.03	0.32	0.10	0.00	0.45	0.49	0.07

Notes: (1) No liquid industrial wastes were licensed or disposed of in Scotland or Northern Ireland in the period covered by this report
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

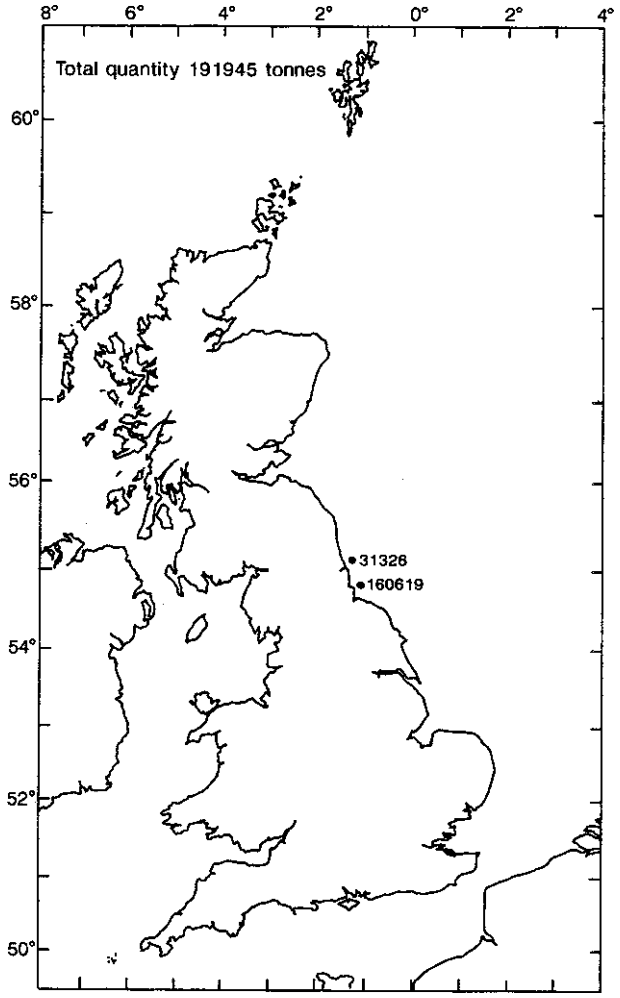

Figure 60. Liquid industrial wastes disposed of at sea in 1991

deposited, together with information on those contaminants in the wastes which the UK is required to report internationally to meet obligations under the Oslo and London Conventions (Great Britain - Parliament 1972 (a-b)).

19.5 Licensing of liquid industrial wastes

Table 28(a) shows details of the two remaining liquid industrial wastes licensed for disposal at sea in the period covered by this report. Table 28(b) summarises the quantity of wastes licensed and the amounts deposited, including metallic contaminants, for the period 1987-1991. The locations of the disposal sites and the quantities deposited at each site are given in Figure 60. Disposal of liquid industrial wastes ceased at the end of 1992.

19.6 Licensing of solid industrial wastes

Table 29(a) gives details of licences issued for the disposal of solid industrial wastes, and Table 29(b) details the quantities of metal contaminants in the material deposited. Figure 61 shows the locations of the relevant disposal sites together with the quantities deposited at each site. The bulk of the material licensed was stone extracted during coal mining. The material is accepted internationally as inert in the marine environment and as such the presence of the metal contaminants is of no significance in terms of potential ecological impact.

One licence to National Power for sea disposal of 50 000 tonnes of pulverised fuel ash was terminated in May 1991 after generation ceased at the two Stella power stations on the Tyne. The other licence was terminated at the end of 1992.

Table 29(a). Solid industrial wastes licensed for disposal at sea in 1991[(1)]

Licensed quantity (t)	Company and source of waste	Description of waste	Disposal sites	Quantity deposited (t)
	British Coal Collieries			
700,000	Dawdon/Seaham	Minestone	Bankside, Seaham	759,414
850,000	Easington	Minestone	Foreshore at Easington	709,917
650,000	Ellington	Minestone	Foreshore at Ellington	494,741
100,000	Point of Ayr	Minestone	Foreshore at Point of Ayr	85,336
875,000	Wearmouth	Minestone	Sunderland	531,899
975,000	Westoe	Minestone	Souter Point/North Tyne/Sunderland	775,044
1,100,000	Wearmouth/Westoe	Mine tailings	Souter Point/North Tyne/Sunderland	873,455
	National Power			
500,000	Blyth Power Station	Pulverised fuel ash	Blyth A	301,054
	Other Waste			
300	Fish farm	Dead salmon	Atlantic	460

Notes: [(1)] No solid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report. Also, the table excludes one licence valid into 1991, but on which no waste was disposed of in 1991
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

Table 29(b). Summary of solid industrial waste licensed and disposed of at sea [1]

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1987	7	4,290,100	4,275,382*	0.23	20	197	0.23	63	251	486
	1988	10	5,756,200	4,211,615	0.26	21	193	0.25	63	244	470
	1989	10	5,928,917	4,835,508	0.28	24	200	0.24	68	245	481
	1990	9	6,575,000	4,919,654	0.29	24	182	0.21	64	223	441
	1991	9	5,750,000	4,530,860	0.28	23	172	0.21	60	212	414
Scotland	1987	1	250	117	0.00	0	0	0.00	0	0	0
	1988	3	570	30	0.00	0	0	0.00	0	0	0
	1989	0	0	102	0.00	0	0	0.00	0	0	0
	1990	0	0	0	0.00	0	0	0.00	0	0	0
	1991	1	300	460	0.00	0	0	0.00	0	0	0
UK total	1987	8	4,290,350	4,275,499	0.23	20	197	0.23	63	251	486
	1988	13	5,756,770	4,211,645	0.26	21	193	0.25	63	244	470
	1989	10	5,928,917	4,835,610	0.28	24	200	0.24	68	245	481
	1990	9	6,575,000	4,919,654	0.29	24	182	0.21	64	223	441
	1991	9	5,750,300	4,531,320	0.28	23	172	0.21	60	212	414

Notes: [1] *no solid industrial wastes were licensed or disposed of in Northern Ireland in the period covered by this report*
* *tonnage deposited was covered by 5-year licences issued in 1983*
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

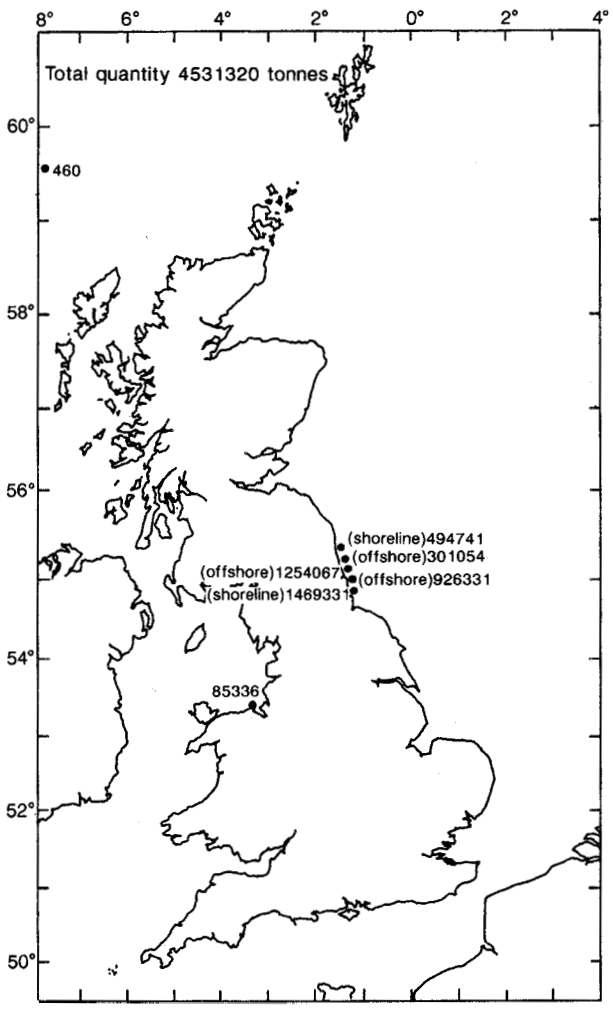

Figure 61. Solid industrial wastes disposed of at sea in 1991

19.7 Licensing of sewage sludge

Table 30(a) gives the details of licences issued for disposal of sewage sludge at sea. Total quantities of key metallic contaminants in sewage sludge licensed for disposal at sea are shown in Table 30(b). Figure 62 shows the location of the disposal sites for sewage sludge and the quantities deposited at each site.

At the 1987 Second International Conference on the Protection of the North Sea, (Department of the Environment, 1987), the Government indicated it was taking urgent action to reduce the contamination by persistent, toxic or bioaccumulatable materials present in sewage sludge deposited in the North Sea and to ensure that the quantities of such contaminants disposed of in the North Sea did not increase above 1987 levels. Earlier reports explained that to apply this control, limits were set for a series of key contaminants deposited under each licence. Table 31 compares the aggregated figures (in tonnes) authorised for disposal in the North Sea in 1991, with the estimated quantity (in tonnes) at 1987 licensed levels.

Although the bulk of these contaminants comes from general domestic sources rather than industry, the licence levels for 1991 were below both those for 1990 and the 1987 baseline figures.

On 5 March 1990, the Minister of Agriculture, Fisheries and Food announced that practical alternatives could now be found and, accordingly, that disposal at sea of sewage sludge was to be terminated. He added that a major capital investment programme would be involved to bring these alternative disposal outlets into being and it was noted that planning permission would be required in some cases. Nevertheless, the final licences for disposal at sea would be terminated by the end of 1998.

Table 30(a). Sewage sludge licensed for disposal at sea in 1991

Country	Licensed quantity (t) [1]	Company and source of waste	Disposal sites	Quantity deposited (t) [1]
England	80,000	Anglian Water (Cliff Quay STW, Ipswich)	Roughs Tower	70,672
and Wales	81,569	Anglian Water (Colchester STW)	Roughs Tower	35,061
	150,000	Anglian Water (Tilbury STW)	Roughs Tower	128,084
	554,000	Northumbrian Water (Howdon, Chester-le-Street, Cramlington, Washington STWs)	Tyne/Spurn Head	540,000
	105,000	Northumbrian Water (Portrack, Billingham, Guisborough, Ayton STWs)	Tyne/Spurn Head	64,035
	1,965,000	North West Water (Davyhulme, Liverpool, Warrington STWs)	Liverpool Bay	1,876,091
	5,000	North West Water (Walney Island)	Liverpool Bay	3,409
	300,000	Southern Water (Woolston, Portswood, Millbrook, Slowhill Copse STWs)	Nab Tower	262,194
	58,000	South West Water (Countess Wear STW)	Lyme Bay	38,456
	80,000	South West Water (Plympton, Radford, Camel's Head, Ernesettle, Ivybridge, Saltash, Newton Ferrers STWs)	Plymouth	57,322
	4,500,000	Thames Water (Beckton, Crossness, Riverside, Deephams STWs)	Barrow Deep	3,983,592
	55,000	Welsh Water (Ponthir, Nash, Llanfoist, Magor STWs)	Bristol Channel	45,412
	300,000	Wessex Water (Avonmouth, Keynsham, Bath STWs)	Bristol Channel	233,018
	140,000	Yorkshire Water (Knostrop STW)	Spurn Head	111,738
Scotland	500,000	Lothian Regional Council	St Abb's Head/Bell Rock	285,535
	2,500,000	Strathclyde Regional Council	Garroch Head	1,698,500
Northern Ireland	80,000	Dept. Environment (Northern Ireland)	Belfast Sludge	302,370[2]

Notes: [1] *all figures are for tonnage in wet weight*
[2] *includes 200,000 t yr^{-1} disposed of by DOE (NI) Water Services under an administrative authorisation*
STW = Sewage Treatment Works
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

Table 30(b). Summary of sewage sludge licensed and disposed of at sea

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England	1987	17	8,682,510	6,692,654	3.24	104	136	1.08	21	156	439
and Wales	1988	16	7,503,580	7,267,935	2.45	92	130	0.95	18	144	355
	1989	15	8,321,305	7,373,212	2.17	85	128	0.94	18	157	297
	1990	14	8,553,579	7,098,944	1.76	68	113	0.88	18	109	238
	1991	14	8,373,569	7,449,084	1.88	67	121	0.92	17	103	269
Scotland	1987	2	2,300,000	1,978,041	0.41	34	29	0.19	2	28	54
	1988	2	2,300,000	2,004,963	0.42	45	33	0.19	3	26	56
	1989	2	2,300,000	1,940,575	0.46	53	32	0.17	3	24	58
	1990	2	3,000,000	1,946,430	0.27	34	30	0.16	3	18	39
	1991	2	3,000,000	1,984,035	0.24	35	41	0.10	3	20	44
Northern	1987	1	100,000	309,489*	0.09	3	3	0.08	1	4	15
Ireland	1988	1	90,000	291,904*	0.06	3	3	0.03	1	2	18
	1989	1	80,000	329,060*	0.05	3	4	0.03	1	3	12
	1990	1	80,000	290,030*	0.03	2	3	0.03	0	2	11
	1991	1	80,000	302,370*	0.04	2	3	0.04	0	3	11
UK total	1987	20	11,082,510	8,980,184*	3.73	140	167	1.35	24	188	508
	1988	19	9,893,580	9,564,802*	2.92	140	166	1.17	22	172	429
	1989	18	10,701,305	9,642,847*	2.68	141	164	1.14	22	184	367
	1990	17	11,633,579	9,335,404*	2.06	104	147	1.06	21	129	288
	1991	17	11,453,569	9,735,489*	2.15	104	165	1.06	20	125	325

Notes: * *includes 200,000 t yr^{-1} disposed of by DOE(NI) Water Services under an administrative authorisation*
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

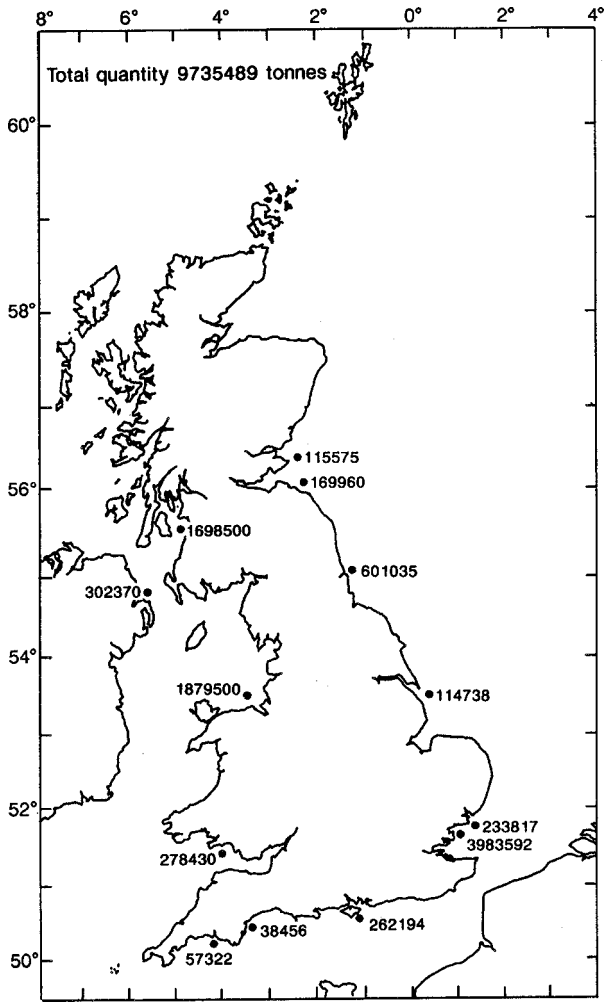

Figure 62. Sewage sludge disposed of at sea in 1991

Table 31. Contaminants in sewage sludge authorised for disposal in the North Sea in 1991 compared against estimated quantities in 1987

Year	Hg	Cd	Cr	Ni	Cu	Zn	Pb
1987	1.2	3.7	56.2	19.4	133.6	468.2	146.4
1991	1.1	2.4	43.5	15.9	128.4	340.0	126.2

On 5 March 1991, the Minister answered a parliamentary question about the steps that the water companies were taking to find alternative outlets for sludge currently disposed to sea. It was clear that most of the companies were considering incineration as providing the most secure and reliable method of dealing with the sludge.

19.8 Licensing of dredged material

The bulk of the dredged material licensed for disposal at sea is silt and sand, but coarse sand and shingle can occur in 'maintenance' dredging, and shingle, cobbles, small rocks and heavy clay can be present in 'capital' material arising from development. Table 32 shows the numbers of licences issued, the quantity licensed, and the quantity deposited, together with figures for the quantity of a range of trace metals which enter the sea in the dredged materials. A proportion of the trace metals associated with dredged material occurs within the mineral structure or is tightly bound, such that generally it will not be available to marine organisms. Figure 63 shows the main disposal sites used in 1991 and the quantities deposited at each site.

Table 32. Summary of dredged material licensed and disposed of at sea

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1987	107	28,689,146	38,692,856*	35.0	1410	1438	14.6	997	2876	6661
	1988	131	61,645,223	34,691,093	23.2	1165	1091	11.0	753	2199	5191
	1989	138	66,408,100	40,810,718	18.6	1234	1037	9.3	638	1877	4938
	1990	135	63,983,920	33,728,978	12.2	1023	834	6.8	484	1426	3724
	1991	108	57,782,520	39,886,812	7.4	1189	773	7.0	518	1263	3394
Scotland	1987	35	8,813,850	3,927,264	1.3	120	67	1.6	43	101	206
	1988	25	4,148,690	3,506,685	1.2	114	89	1.5	43	123	259
	1989	27	4,252,950	3,154,756	1.1	106	106	1.3	40	141	313
	1990	21	3,031,960	2,109,114	0.8	61	59	0.8	36	116	210
	1991	26	5,147,245	2,788,611	0.6	70	53	0.6	22	79	167
Northern Ireland	1987	5	338,400	547,052	1.2	6	4	0.0	6	4	15
	1988	9	1,534,200	1,077,023	0.1	10	7	0.1	8	11	26
	1989	6	383,300	338,521	0.1	2	2	0.0	2	2	7
	1990	6	261,700	317,082	0.1	3	3	0.0	3	5	8
	1991	10	807,400	519,049	0.1	2	2	0.1	3	2	6
UK total	1987	147	37,841,396	43,167,172	37.5	1537	1509	16.2	1046	2981	6881
	1988	165	67,328,113	39,274,801	24.6	1289	1187	12.5	805	2333	5476
	1989	171	71,044,350	44,303,995	19.8	1343	1145	10.6	679	2021	5258
	1990	162	67,277,580	36,155,174	13.5	1086	895	7.6	523	1546	3942
	1991	144	63,737,165	43,194,472	8.1	1260	828	7.8	543	1344	3566

Notes: * *some deposits were covered by 5-year licences*
For information on licensed quantities and tonnages deposited see footnote to Table 28(a)

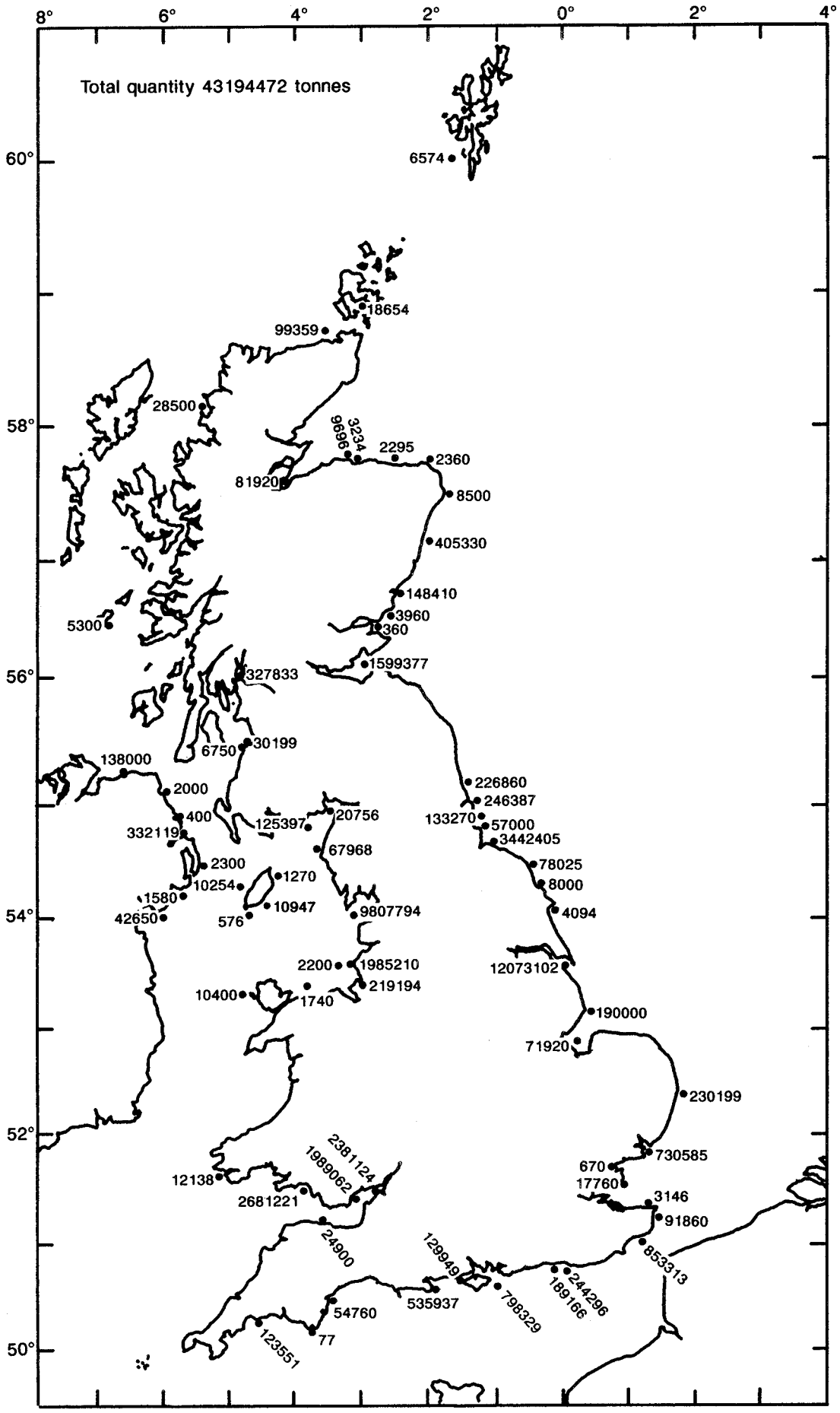

Figure 63. Dredged material disposed of at sea in 1991

19.9 Licensing of marine incineration

No wastes were incinerated at sea in 1991; this was in accordance with international policy, which as explained in last year's report decreed that no further waste would be licensed for marine incineration after 1990.

19.10 Other materials deposited at sea

Under Part II of FEPA, licences are also required for certain activities which do not involve the disposal of wastes, but nevertheless involve the deliberate and permanent deposition of substances in the sea. Each request for such a licence is carefully considered, but generally the anticipated impact is minimal and no monitoring is required. Specifically, such activities involved construction work below mean high water, the use of tracers, the application of biocides and burials at sea.

20. ADVICE ON FISHERY IMPLICATIONS OF PIPELINE DISCHARGES

This section gives a brief summary of activities carried out during 1991 in connection with provision of advice on fishery implications of pipeline discharges. The background to this work in relation to MAFF's responsibilities as a statutory consultee under the Water Act 1989 (Great Britain-Parliament, 1989) was described in a previous report in this series (MAFF, 1991).

On 1 December 1991, the Water Resources Act, 1991 (Great Britain - Parliament, 1991) replaced the Water Act, 1989 as the legislation for controlling discharges from land-based sources. Under Schedule 10 of this Act, the National Rivers Authority (NRA) is obliged to send MAFF copies of all new applications for consent to discharge effluent from land, in England and Wales, to coastal and territorial waters (from the limit of the highest tide or the fresh water limit). The NRA has a duty to consider any comments which are made to them within six weeks of the application being advertised. Although not obliged to do so under the Water Resources Act, the NRA has also agreed to seek MAFF's views on all reviews of conditions for existing consents, even those where effluent quality is being improved.

Table 33 shows the number of applications for each type of effluent sent to MAFF for comment during 1991.

Table 33. Discharge consent applications assessed by MAFF in 1991

Sewage (including storm and emergency overflows)	Trade effluent	Surface water	Total
128	55	15	198

About half of the applications received during 1991 were for discharges which had no potential for adverse fishery implications and therefore required no comment from MAFF. Approval of the remainder was subject to certain conditions such as freedom from hazardous materials, more stringent treatment or tighter consent limits.

Although the number of applications received was about 35% higher than the 1990 total, there was no indication that the process of positive determination of the 'deemed' consents (see MAFF, 1991) had started. Thus, there remain about 3000 unconditional 'deemed' consents for existing discharges still to be translated into full consents which properly reflect the needs of the receiving water, by the October 1992 deadline. These include many environmentally significant industrial effluents as well as numerous sewage discharges which are in need of improvement.

As in previous years, the majority of the applications received during 1991 were for sewage discharges. In commenting on these, emphasis was placed on the need to reduce inputs of persistent plastics or other material which could foul fishing gear and to improve the microbial quality of shellfish harvesting areas. The latter assumed greater importance with the publication on 15 July 1991 of EC Directive 91/492 laying down the health conditions for the production and the placing on the market of live bivalve molluscs (European Communities, 1991(a)). This Directive, which will come into effect on 1 January 1993, will require all waters which are used for commercial harvesting of live bivalve molluscs to be classified according to the level of bacterial contamination present in the mollusc flesh. The classification will determine the degree of treatment which the shellfish will need to undergo before being sold for human consumption; in very heavily contaminated areas harvesting will be prohibited altogether.

Where sewage disposal arrangements need upgrading in order to achieve improvements in shellfish quality under Directive 91/492, relocation of existing outfalls or higher levels of treatment may suffice. However, in some areas, disinfection of the sewage effluent may also be necessary. During 1991, several water companies carried out trials to determine the safety and efficacy of various sewage disinfection techniques, many of which included MAFF involvement. To ensure protection of fisheries, MAFF has formulated an internal policy on sewage disinfection. The use of chemical disinfectants for

improving the microbiological quality of sewage and sewage effluent is seen as an interim measure and before agreeing to its use there should be an assurance that plans for improvements by other means are in place. In applications where there is no shellfishery interest MAFF would not object to the interim use of chemical disinfection providing the consent stipulates a requirement to monitor trihalomethane (THM) concentrations pre- and post-disinfection, such that THM concentrations do not rise by an agreed factor as a result of disinfection. Where there are shellfish interests, there is an additional requirement to monitor THM levels outside the plume mixing zone as well as a request that the dischargers measure concentrations of bacteria and viruses before and after disinfection. A more cautious approach is taken where long-term application is proposed including field monitoring investigations and regular review of the discharge consent.

MAFF also provides an input to working groups aimed at the development of a National policy.

Another development which is likely to have future implications for fisheries is EC Directive 91/271 concerning urban waste water treatment (European Communities, 1991(b)) which was published on 21 May 1991. This will require all sewage and related discharges to be treated to a certain standard by a given date (between 1998 and 2005) depending on the size of the discharge, the area (coastal, estuary or fresh water) and the 'sensitivity' of the receiving water. During the year, MAFF provided advice in connection with determination of UK policy on implementation of this Directive, particularly with regard to whether areas should be designated as 'sensitive', 'normal', or 'less sensitive'.

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APPENDIX 1. Areas of monitoring mentioned in the text and staff responsible for the projects

Report section	Project	Staff
1.	Contaminants in marine fish and shellfish	A Franklin* J Jones*
2.	Bioassay studies	J Thain* D Sheahan*
3.	Contaminants in marine mammals	C Allchin* C Kelly
4.	General sea water sampling procedures	R Law*
5.	Nutrients in sea water	R Law*
6.	Trace metals in sea water	R Laslett*
7.	Hexachlorocyclohexanes in sea water	C Allchin* R Law*
8.	Atrazine and simazine in sea water	C Allchin* R Hamer
9.	Hydrocarbons in sea water	R Law* J Osborne
10.	Baseline sediment survey	S Rowlatt* D Lovell
11.	Chlorobiphenyls in sediments	C Kelly*
12.	TBT Studies	M Waldock M Waite*
13.	Rosemaund pesticide study	P Matthiessen* R Rycroft
14.	Aggregate extraction activities	R G Lees* A Kenny* H Rees*
15.	Sewage sludge disposal areas - metals	S Rowlatt* D Limpenny P Manning
16.	Sewage sludge disposal areas - Tyne	H Rees* S Rowlatt* D Limpenny M Lambert P Manning
17.	Dredged material disposal areas - Liverpool Bay Site Z	S Rowlatt* H Rees* D Limpenny M Lambert P Manning
18.	Dredged material disposal areas - Barrow-in-Furness	S Rowlatt* H Rees* D Limpenny M Lambert P Manning
19.	Licensing and Enforcement	G Boyes* C M G Vivian
20.	Pipeline discharges	F Franklin*

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APPENDIX 2. Standards/guidelines for contaminants in fish and shellfish

A2.1 Metals

(a) Mercury

The European and Paris Commissions have adopted an Environmental Quality Standard (EQS) for mercury, which requires that the mean concentration of mercury in the flesh of a representative sample of fish, locally caught from areas receiving significant inputs of mercury, shall not exceed 0.3 mg kg^{-1} on a wet weight basis (EC Directive Nos. 82/176 and 84/156 - European Communities, 1982 and 1984).

For the purposes of the Joint Monitoring Programme (JMP) of the Oslo and Paris Commissions, the following arbitrary, purely descriptive, guidelines have been adopted.

Level	Fish flesh and crustaceans	Molluscs
Lower	<0.1 mg kg^{-1} wet weight	<0.6 mg kg^{-1} dry weight
Medium	0.1-0.3 mg kg^{-1} wet weight	0.6-1.0 mg kg^{-1} dry weight
Upper	>0.3 mg kg^{-1} wet weight	>1.0 mg kg^{-1} dry weight

(b) Cadmium

There are no standards or guidelines in England and Wales for fish flesh. The expected values are <0.2 mg kg^{-1} wet weight.

The JMP guidelines for cadmium in mussels are as follows:

Level	Mussel tissue	Approximate equivalent
Lower	<2 mg kg^{-1} dry weight	(<0.4 mg kg^{-1} wet weight)
Medium	2-5 mg kg^{-1} dry weight	(0.4-1.0 mg kg^{-1} wet weight)
Upper	>5 mg kg^{-1} dry weight	(>1.0 mg kg^{-1} wet weight)

From past DFR work, 'expected' values (i.e. using data from areas not known to be severely contaminated) would be up to 0.3 mg kg^{-1} wet weight for crustaceans but up to 10 mg kg^{-1} wet weight for crab 'brown' meat.

(c) Lead

From the Lead in Food Regulations 1979 (Great Britain - Parliament, 1979): lead in fish should not exceed 2.0 mg kg^{-1} wet weight, and lead in shellfish 10.0 mg kg^{-1} wet weight.

From past work, 'expected' values are 0.2-0.3 mg kg^{-1} wet weight in fish, up to 1.0 mg kg^{-1} wet weight in crustaceans, and up to 4.0 mg kg^{-1} wet weight in some molluscs.

(d) Copper

From the Food Standards Committee's Report on Copper (MAFF, 1956), revised recommendations for limits for copper content of food are as follows:

> 'levels of copper in food should not exceed 20 mg kg^{-1} wet weight (but higher levels in shellfish are permitted if copper is of natural occurrence)'.

From past DFR work, 'expected' levels in fish are up to 0.6 mg kg^{-1} wet weight (in excess of 1.0 mg kg^{-1} wet weight in fatty fish such as herring) up to 5.0 mg kg^{-1} wet weight for molluscs (with very much higher values for some gastropods) and 20-30 mg kg^{-1} wet weight for crustaceans.

(e) Zinc

From the Food Standards Committee's Report on Zinc (Ministry of Food, 1953), as a guideline:

> 'levels of zinc in food should not exceed 50 mg kg^{-1} wet weight (but higher levels are permitted in food which naturally contain more than 50 mg kg^{-1}, such as herring and shellfish).'

'Expected' values commonly found are up to 6.0 mg kg^{-1} wet weight in most fish flesh, (though up to 10 mg kg^{-1} in flounder and considerably more in fatty fish), up to 100 mg kg^{-1} wet weight in crustaceans and well in excess of 100 mg kg^{-1} wet weight for some molluscs.

A2.2 Pesticides/PCBs

There are no standards in fish and shellfish from England and Wales.

(a) HCB

The 'expected' value is up to 0.10 mg kg^{-1} wet weight in fish liver.

(b) HCH

Codex Alimentarius Commission's maximum residue limit (MRL) (FAO/WHO, 1987) is 2 mg kg^{-1} in meat fat for γ-HCH. The 'expected' values are up to 0.05 mg kg^{-1} wet weight for each of α- and γ-HCH in fish liver.

(c) Dieldrin

Codex Alimentarius Commission's MRL is 0.2 mg kg^{-1} in meat fat. The 'expected' values are 0.2-0.3 mg kg^{-1} wet weight in fish liver.

(d) Total DDT

Codex Alimentarius Commission's MRL is 5 mg kg^{-1} in meat fat. The 'expected' values are up to 0.5 mg kg^{-1} wet weight for each of DDE, TDE and pp DDT in fish liver.

(e) PCBs

JMP guidelines are as follows (all mg kg^{-1} wet weight):

Level	Fish muscle	Cod[1] liver	Flounder[2] liver	Molluscs	Crustaceans
Lower	<0.01	<2.0	<0.50	<0.02	<0.01
Medium	0.01-0.05	2.0-5.0	0.50-1.0	0.02-0.10	0.01-0.05
Upper	>0.05	>5.0	>1.0	>0.10	>0.05

[1] *Values used for all roundfish in this report*

[2] *Values used for all flatfish in this report*

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