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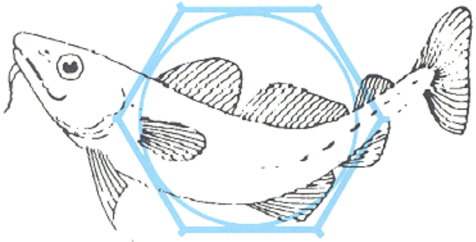

Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment and Activities Regulating the Disposal of Wastes at Sea, 1990

MAFF
Directorate of Fisheries Research
Lowestoft, 1992

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD
DIRECTORATE OF FISHERIES RESEARCH

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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. This, the third report in this series, accordingly covers work carried out in 1990.

As with the second report in the series, in addition to a general coverage of monitoring programmes, it has been decided that these reports should include 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 17 of the present report. Section 18 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 17 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. Previous reports have also included information on fish disease surveys and monitoring related to algal toxins. Such work is undertaken at the Fish Diseases Laboratory, Weymouth and will in future be reported on a regular basis in a separate publication in this series.

C. E. Purdom
Deputy Director
(Aquatic Environment Protection)
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BIOTA

1. MONITORING OF CONTAMINANTS IN MARINE FISH AND SHELLFISH

1.1 Introduction

Extensive monitoring of contaminants in fish and mussels taken from around the coast of England and Wales was carried out in the mid-1980s as part of a baseline survey for the Joint Monitoring Group (JMG) of the Oslo and Paris Commissions (OSPARCOM) (MAFF, 1990). Following the evaluation of the results, the Commissions decided that a supplementary survey should be undertaken in 1990 with the emphasis being applied to sampling more offshore waters. Since the UK had already published some offshore data on the Irish Sea (MAFF, 1987), the UK effort on this occasion was concentrated on the North Sea. The results are presented in this Report. (See Appendix 1 for staff involved in this and other projects throughout the Report).

The locations of all fish and mussel sampling sites are shown in Figure 1. The offshore region to the east and north of the Humber is difficult to fish, owing to the rough nature of the seabed, but efforts will be made in 1991 to improve the coverage of this area.

Some preliminary fish sampling was undertaken in 1989, and these results have also been included where they provide information from areas not covered in 1990. It should be emphasized that the present report is an interim one, since the survey is being continued in 1991 with a view to the utilisation of all of the data in the preparation of the next North Sea Quality Status Report, due out in 1993/94.

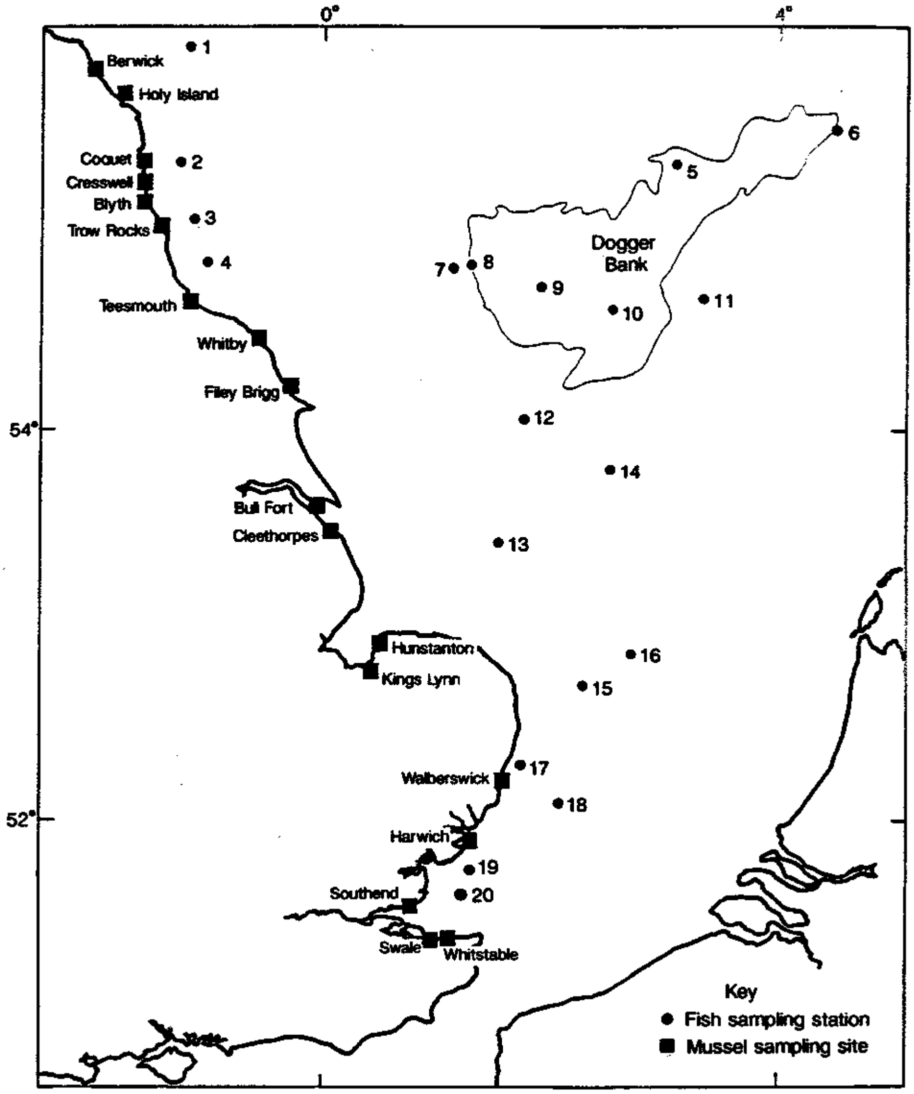

Figure 1. Locations of fish and mussel sampling sites

Staff of the Plymouth Marine Laboratory obtained samples of mussels (*Mytilus edulis*) in 1990 from along the east coast of England for use in studies into biological effects (scope for growth). These samples are being analysed for contaminants at the Burnham-on-Crouch Laboratory. The results for heavy metals are already available and have been included here to provide an up-date on information on this species.

1.2 Methods

1.2.1 Sampling

As agreed by the Commissions, the supplementary JMG survey had focused on a limited number of fish species and, of those recommended, the most widespread were found to be whiting (*Merlangius merlangus*) and dab (*Limanda limanda*). Where possible, samples of 25 of each of these species were obtained, within the agreed length ranges of 20-35 cm and 20-30 cm, respectively. As in earlier studies, samples of mussels consisted of 50 individuals within a 3-5 cm range.

Duplicate analyses were carried out on pooled tissue. As in previous work (MAFF, 1990), bulked fish muscle and *Mytilus* were analysed for heavy metals. However, some (e.g. cadmium and lead) are normally present in such low concentrations in fish muscle that they are below the limits of detection of methods of analysis routinely employed by most laboratories. For this survey, therefore, fish liver was also analysed for heavy metals. Organochlorine pesticides and PCBs tend to accumulate in fatty tissues and analyses for these contaminants were therefore carried out on fish liver.

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 generally determined by an automated cold vapour AAS technique based on the method of Kirkwood (1976); towards the end of 1990, the Burnham-on-Crouch Laboratory changed to 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 chromatography, using electron capture detection after alumina and silica column 'clean-up' and separation. PCBs have been quantified using the formulation Aroclor 1254 as the standard; it is planned to re-analyse some of the samples to provide data on individual chlorinated biphenyls, commencing with the seven on an International Council for the Exploration of the Sea (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 fish tissue are listed in Tables 1 and 2, organochlorines/PCBs in fish liver in Tables 3 and 4 and heavy metals in mussels in Table 5. All concentrations are expressed on a wet tissue weight basis; the percentage of dry matter in muscle and of fat in liver have been included to allow conversion of the data to a dry and lipid weight basis respectively. The mean concentration is given, with both results from the duplicate bulked analysis in brackets underneath.

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/ guidelines which apply in England and Wales for contaminants in fish and shellfish, are summarised in Appendix 2. Because of its recent addition to the JMG's programmes, there are as yet no guidelines for the ranges of metal contaminants in fish liver. The presence of a contaminant in the 'upper' JMG category simply means that it is at a concentration above that normally found in most areas monitored. As such, it does not necessarily imply any risk either to human health or to the environment. Nevertheless, countries are asked to provide monitoring data from affected areas on a biennial basis, to ensure that up-to-date information is always available. It should be noted that samples of mussels were taken for indicator purposes and, as such, do not necessarily represent commercially exploited stocks. Indeed, only those marked with an asterisk in Table 5 were obtained from areas where commercial exploitation is known to take place.

Table 1. Concentrations of metals in whiting

Station no./ area	Date of capture	No. of fish	Mean length (cm)	Tissue	Concentration (mg kg^{-1} wet weight)					Dry matter (%)
					Hg	Cu	Zn	Cd	Pb	
1 (Off Berwick)	Feb. 90	25	27.7	M	0.05 (0.05, 0.05)	0.19 (0.19, 0.18)	3.1 (3.0, 3.1)	-	-	20
				L	0.02 (0.01, 0.02)	1.7 (1.7, 1.7)	13 (13, 13)	0.12 (0.12, 0.12)	<0.7 (<0.6, <0.7)	71
2 (Off Amble)	Feb. 90	25	26.0	M	0.03 (0.03, 0.03)	0.11 (0.11, 0.10)	2.8 (2.8, 2.7)	-	-	20
				L	0.02 (0.02, 0.02)	3.0 (2.9, 3.1)	14 (14, 14)	0.08 (0.07, 0.09)	<0.7 (<0.7, <0.7)	69
3 (Off the Tyne)	May. 90	24	27.2	M	0.07 (0.07, 0.07)	0.20 (0.21, 0.19)	2.7 (2.7, 2.7)	-	-	19
				L	0.04 (0.03, 0.04)	4.8 (4.7, 4.8)	21 (21, 21)	0.18 (0.17, 0.18)	<0.6 (<0.6, <0.6)	49
4 (Off the Tees)	May. 90	25	28.6	M	0.09 (0.09, 0.09)	0.22 (0.22, 0.21)	2.8 (2.7, 2.9)	-	-	18
				L	0.05 (0.05, 0.05)	5.3 (5.2, 5.3)	22 (22, 22)	0.23 (0.22, 0.24)	<0.6 (<0.6, <0.6)	46
5 (N. Dogger)	May. 90	20	28.1	M	0.07 (0.07, 0.07)	0.25 (0.26, 0.24)	3.2 (3.2, 3.1)	-	-	18
				L	0.04 (0.03, 0.04)	4.0 (4.0, 4.0)	25 (24, 25)	0.42 (0.42, 0.41)	<0.6 (<0.6, <0.6)	41
6 (NE tip of Dogger)	Sep. 90	25	27.6	M	0.05 (0.05, 0.05)	0.19 (0.18, 0.19)	3.1 (3.1, 3.1)	-	-	20
				L	0.05 (0.05, 0.05)	3.6 (3.6, 3.6)	15 (15, 15)	0.14 (0.14, 0.13)	<0.6 (<0.6, <0.6)	69
7 (W. of Dogger)	Feb. 90	25	26.0	M	0.05 (0.04, 0.05)	0.18 (0.18, 0.17)	2.9 (2.9, 2.9)	-	-	19
				L	0.03 (0.02, 0.03)	6.3 (6.2, 6.3)	22 (22, 22)	0.37 (0.29, 0.44)	<0.6 (<0.6, <0.6)	60
8 (W. edge of Dogger)	May. 90	23	26.2	M	0.09 (0.08, 0.09)	0.18 (0.18, 0.17)	2.8 (2.8, 2.8)	-	-	18
				L	0.03 (0.03, 0.03)	3.1 (3.1, 3.1)	19 (19, 19)	0.38 (0.35, 0.40)	<0.6 (<0.6, <0.6)	44
11 (E. of Dogger)	Sep. 90	25	23.7	M	0.04 (0.04, 0.04)	0.14 (0.13, 0.15)	3.1 (3.1, 3.0)	-	-	19
				L	0.02 (0.02, 0.02)	3.7 (3.8, 3.6)	15 (15, 15)	0.16 (0.17, 0.15)	<0.6 (<0.6, <0.6)	64
12 (Central Outer Silver Pit)	Sep. 90	25	31.6	M	0.09 (0.09, 0.09)	0.19 (0.19, 0.18)	2.6 (2.6, 2.6)	-	-	20
				L	0.02 (0.02, 0.02)	3.1 (3.0, 3.2)	12 (12, 12)	<0.06 (<0.06, <0.06)	<0.6 (<0.6, <0.6)	76
14 (SE of Outer Silver Pit)	Feb. 90	25	31.5	M	0.09 (0.09, 0.09)	0.37 (0.38, 0.35)	3.2 (3.1, 3.2)	-	-	19
				L	0.02 (0.02, 0.02)	4.0 (3.9, 4.1)	17 (16, 17)	0.13 (0.14, 0.11)	<0.6 (<0.6, <0.6)	69
15 (Smith's Knoll)	Nov. 90	25	25.1	M	0.07 (0.06, 0.07)	0.27 (0.26, 0.27)	3.5 (3.5, 3.5)	-	-	19
				L	0.03 (0.03, 0.03)	4.0 (4.1, 3.9)	16 (15, 16)	0.06 (0.06, 0.06)	<0.6 (<0.6, <0.6)	66
16 (E. of Smith's Knoll)	Sep. 90	25	23.8	M	0.07 (0.06, 0.07)	0.24 (0.23, 0.24)	3.1 (3.1, 3.1)	-	-	18
				L	0.06 (0.06, 0.06)	4.6 (4.6, 4.6)	15 (15, 15)	<0.06 (<0.06, <0.06)	<0.6 (<0.6, <0.6)	64
17 (Off Southwold)	Oct. 90	25	27.9	M	0.09 (0.08, 0.09)	0.15 (0.15, 0.15)	3.0 (3.0, 2.9)	-	-	20
				L	0.05 (0.04, 0.05)	5.1 (5.1, 5.0)	16 (16, 16)	0.08 (0.07, 0.09)	<0.6 (<0.6, <0.6)	70
18 (Gabbard)	Oct. 90	11	29.0	M	0.08 (0.08, 0.08)	0.09 (0.09, 0.09)	3.2 (3.2, 3.2)	-	-	20
				L	0.06 (0.06, 0.05)	4.6 (4.6, 4.5)	16 (15, 16)	0.10 (0.09, 0.11)	<0.6 (<0.6, <0.6)	72
19 (Thames (Swin))	May. 90	25	27.6	M	0.14 (0.14, 0.14)	0.16 (0.17, 0.15)	2.8 (2.8, 2.8)	-	-	18
				L	0.07 (0.07, 0.07)	2.8 (2.8, 2.7)	21 (21, 21)	0.08 (0.08, 0.08)	<0.6 (<0.6, <0.6)	55
19 (Thames (Swin))	Oct. 90	25	28.9	M	0.07 (0.06, 0.07)	0.17 (0.17, 0.17)	3.0 (3.0, 3.0)	-	-	20
				L	0.07 (0.08, 0.06)	4.7 (4.6, 4.7)	15 (15, 14)	0.09 (0.09, 0.08)	<0.6 (<0.6, <0.6)	69
20 (Thames (Barrow Deep))	Jun. 90	25	25.3	M	0.09 (0.09, 0.08)	0.35 (0.36, 0.34)	3.5 (3.3, 3.6)	-	-	19
				L	0.05 (0.05, 0.05)	3.7 (3.7, 3.7)	16 (16, 16)	0.07 (0.06, 0.07)	<0.6 (<0.6, <0.6)	56

M = Muscle; L = Liver

Table 2. Concentrations of metals in dab

Station no./ area	Date of capture	No. of fish	Mean length (cm)	Tissue	Concentration (mg kg^{-1} wet weight)					Dry matter (%)
					Hg	Cu	Zn	Cd	Pb	
1 (Off Berwick)	Feb. 90	25	24.8	M	0.06 (0.06, 0.06)	0.16 (0.15, 0.16)	4.0 (4.1, 3.8)	-	-	21
				L	0.05 (0.05, 0.05)	4.1 (4.3, 3.9)	37 (38, 36)	0.38 (0.32, 0.43)	<0.6 (<0.6, <0.6)	35
2 (Off Amble)	Feb. 90	25	21.6	M	0.05 (0.04, 0.05)	0.19 (0.20, 0.18)	3.8 (3.8, 3.8)	-	-	20
				L	0.06 (0.06, 0.05)	7.7 (7.8, 7.5)	36 (37, 35)	0.18 (0.18, 0.17)	<0.6 (<0.6, <0.6)	37
3 (Off the Tyne)	Apr. 89	25	22.1	M	0.07 (0.07, 0.06)	0.18 (0.17, 0.18)	3.7 (3.8, 3.6)	-	-	18
				L	0.08 (0.08, 0.08)	8.4 (8.3, 8.4)	32 (32, 32)	0.34 (0.33, 0.35)	<0.7 (<0.7, <0.7)	25
5 (N. Dogger)	May. 90	23	22.0	M	0.08 (0.08, 0.08)	0.19 (0.20, 0.17)	4.4 (4.4, 4.3)	-	-	17
				L	0.10 (0.09, 0.10)	5.5 (5.5, 5.4)	30 (29, 30)	0.50 (0.54, 0.45)	<0.6 (<0.6, <0.6)	23
5 (N. Dogger)	Sep. 90	23	21.2	M	0.08 (0.08, 0.07)	0.21 (0.23, 0.19)	4.5 (4.5,4.5)	-	-	19
				L (one sample only)	0.10	4.2	28	0.40	<0.6	31
6 (NE. tip of Dogger)	Sep. 90	25	21.4	M	0.03 (0.03, 0.03)	0.16 (0.15, 0.17)	4.4 (4.5, 4.3)	-	-	21
				L	0.03 (0.03, 0.03)	4.7 (4.6, 4.7)	24 (24, 24)	0.14 (0.14, 0.13)	<0.6 (<0.6, <0.6)	47
7 (W. of Dogger)	Feb. 90	25	22.1	M	0.05 (0.05, 0.05)	0.21 (0.22, 0.20)	4.2 (4.3, 4.1)	-	-	19
				L	0.07 (0.07, 0.07)	5.6 (5.7, 5.5)	42 (41, 42)	0.24 (0.25, 0.22)	<0.7 (<0.6, <0.7)	30
8 (W. edge of Dogger)	May. 90	25	21.6	M	0.08 (0.08, 0.08)	0.18 (0.17, 0.18)	4.4 (4.0, 4.8)	-	-	17
				L	0.12 (0.12, 0.12)	4.3 (4.3, 4.3)	27 (27, 27)	0.55 (0.50, 0.60)	<0.6 (<0.6, <0.6)	22
9 (Central Dogger)	Aug. 89	25	24.4	M	0.11 (0.11, 0.10)	0.18 (0.18, 0.18)	3.8 (3.7, 3.9)	-	-	20
				L	0.08 (0.08, 0.08)	5.3 (5.2, 5.3)	25 (25, 25)	0.43 (0.36, 0.49)	<0.6 (<0.6, <0.6)	35
10 (Central Dogger)	Sep. 90	24	22.3	M	0.05 (0.05, 0.05)	0.12 (0.13, 0.10)	4.2 (4.2, 4.3)	-	-	19
				L	0.05 (0.05, 0.05)	4.7 (4.7, 4.6)	21 (21, 21)	0.29 (0.29, 0.28)	<0.6 (<0.6, <0.6)	36
11 (E. of Dogger)	Sep. 90	25	21.4	M	0.10 (0.09, 0.10)	0.21 (0.23, 0.19)	5.1 (5.1, 5.0)	-	-	20
				L	0.10 (0.10, 0.10)	7.4 (7.3, 7.4)	29 (29, 29)	0.40 (0.39, 0.41)	<0.6 (<0.6, <0.6)	30
12 (Central Outer Silver Pit)	Sep. 90	25	21.2	M	0.06 (0.06, 0.05)	0.14 (0.13, 0.14)	4.1 (4.0, 4.1)	-	-	21
				L	0.06 (0.06, 0.06)	5.8 (5.7, 5.8)	25 (24, 25)	0.17 (0.17, 0.17)	<0.6 (<0.6, <0.6)	37
13 (Sole Pit)	Sep. 90	25	25.8	M	0.11 (0.11, 0.11)	0.22 (0.22, 0.21)	4.3 (4.3, 4.2)	-	-	20
				L	0.12 (0.11, 0.12)	6.8 (6.9, 6.7)	30 (30, 30)	0.57 (0.55, 0.58)	<0.6 (<0.6, <0.6)	40
14 (SE. of Outer Silver Pit)	Feb. 90	25	24.8	M	0.12 (0.11, 0.12)	0.11 (0.12, 0.10)	3.4 (3.3, 3.5)	-	-	19
				L	0.10 (0.10, 0.10)	6.6 (6.7, 6.5)	40 (40, 40)	0.49 (0.43, 0.54)	<0.6 (<0.6, <0.6)	28
15 (Smith's Knoll)	Nov. 90	22	26.5	M	0.10 (0.10, 0.10)	0.22 (0.23, 0.21)	4.0 (4.1, 3.8)	-	-	21
				L	0.10 (0.10, 0.10)	11 (10, 11)	43 (43, 43)	0.30 (0.31, 0.29)	<0.6 (<0.6, <0.6)	44
16 (E. of Smith's Knoll)	Sep. 90	25	22.1	M	0.08 (0.08, 0.08)	0.29 (0.23, 0.35)	4.2 (4.1, 4.2)	-	-	21
				L	0.10 (0.10, 0.10)	5.7 (5.7, 5.7)	24 (24, 24)	0.17 (0.17, 0.16)	<0.6 (<0.6, <0.6)	39
20 (Thames (Barrow Deep))	Jul. 89	24	26.5	M	0.12 (0.12, 0.11)	0.21 (0.21, 0.20)	3.6 (3.6, 3.5)	-	-	22
				L	0.08 (0.08, 0.07)	11 (11, 11)	28 (28, 28)	0.08 (0.07, 0.09)	<0.6 (<0.6, <0.6)	42

M = Muscle; L = Liver

Table 3. Concentrations of PCBs and pesticide residues in whiting liver

Station no./ area	Concentration (mg kg^{-1} wet weight)								Fat (%)
	HCB	α–HCH	γ-HCH	Dieldrin	ppDDE	ppTDE	ppDDT	PCB	
1 (Off Berwick)	0.015	<0.001	0.011	0.031	0.072	0.039	0.030	0.69	59
	(0.014, 0.015)	(<0.001, <0.001)	(0.011, 0.010)	(0.032, 0.029)	(0.079, 0.065)	(0.041, 0.037)	(0.033, 0.027)	(0.68, 0.70)	
2 (Off Amble)	0.012	<0.001	<0.001	0.029	0.053	0.019	0.025	0.76	61
	(0.012, 0.012)	(<0.001, <0.001)	(<0.001, <0.001)	(0.037, 0.021)	(0.050, 0.055)	(0.018, 0.020)	(0.026, 0.023)	(0.78, 0.73)	
3 (Off the Tyne)	0.010	0.003	<0.001	0.057	0.12	0.035	0.033	1.3	31
	(0.012, 0.008)	(0.002, 0.003)	(<0.001, <0.001)	(0.049, 0.064)	(0.12, 0.11)	(0.035, 0.034)	(0.032, 0.034)	(1.4, 1.2)	
4 (Off the Tees)	0.012	0.016	0.007	0.059	0.14	0.044	0.036	1.5	32
	(0.014, 0.010)	(0.017, 0.015)	(0.006, 0.008)	(0.059, 0.059)	(0.16, 0.11)	(0.045, 0.042)	(0.037, 0.035)	(1.5, 1.4)	
5 (N. Dogger)	0.005	<0.001	<0.001	0.017	0.064	0.014	0.022	0.68	20
	(0.005, 0.004)	(<0.001, <0.001)	(<0.001, <0.001)	(0.015, 0.018)	(0.061, 0.066)	(0.013, 0.015)	(0.020, 0.023)	(0.68, 0.68)	
6 (NE tip of Dogger)	0.01	0.008	0.008	0.037	0.067	0.020	0.011	1.1	60
	(0.01, 0.01)	(0.008, 0.008)	(0.008, 0.008)	(0.036, 0.038)	(0.066, 0.067)	(0.019, 0.021)	(0.010, 0.011)	(1.1, 1.1)	
7 (W. of Dogger)	0.008	<0.001	<0.001	0.029	0.074	<0.001	0.017	0.81	36
	(0.008, 0.008)	(<0.001, <0.001)	(<0.001, <0.001)	(0.026, 0.031)	(0.074, 0.073)	(<0.001, <0.001)	(0.017, 0.016)	(0.82, 0.79)	
8 (W. edge of Dogger)	0.005	<0.001	0.003	0.025	0.13	0.020	0.016	1.2	27
	(0.005, 0.005)	(0.001, <0.001)	(0.002, 0.003)	(0.024, 0.025)	(0.13, 0.13)	(0.019, 0.020)	(0.015, 0.016)	(1.2, 1.2)	
11 (E. of Dogger)	0.007	0.004	0.007	0.012	0.071	0.015	0.019	0.96	42
	(0.007, 0.007)	(0.004, 0.003)	(0.005, 0.008)	(0 .010, 0.014)	(0.070, 0.071)	(0.015, 0.014)	(0.019, 0.018)	(0.92, 0.99)	
12 (Central Outer Silver Pit)	0.009	NR	0.012	0.034	0.050	0.040	0.035	2.1	57
	(0.009, 0.009)		(0.012, 0.011)	(0.041, 0.027)	(0.055, 0.044)	(0.044, 0.036)	(0.038, 0.032)	(2.3, 1.8)	
14 (SE of Outer Silver Pit)	0.015	<0.001	0.012	0.047	0.11	0.036	0.043	2.3	55
	(0.015, 0.014)	(<0.001, <0.001)	(0.011, 0.013)	(0.044, 0.049)	(0.11, 0.11)	(0.034, 0.037)	(0.042, 0.043)	(2.3, 2.2)	
15 (Smith's Knoll)	0.023	<0.001	0.019	0.056	0.14	0.029	0.012	3.4	59
	(0.023; 0.023)	(<0.001, <0.001)	(0.019, 0.018)	(0.056, 0.055)	(0.11, 0.17)	(0.030, 0.028)	(0.012, 0.012)	(3.4, 3.4)	
16 (E. of Smith's Knoll)	0.011	<0.001	0.019	0.066	0.11	0.034	0.085	2.5	56
	(0.011, 0.010)	(<0.001, <0.001)	(0.016, 0.022)	(0.067, 0.065)	(0.11, 0.11)	(0.029, 0.038)	(0.075, 0.095)	(2.5, 2.4)	
17 (Off Southwold)	0.013	<0.001	0.017	0.049	0.14	0.055	0.030	2.4	58
	(0.013, 0.013)	(<0.001, <0.001)	(0.015, 0.019)	(0.051, 0.047)	(0.15, 0.13)	(0.050, 0.060)	(0.028, 0.032)	(2.4, 2.4)	
18 (Gabbard)	0.014	<0.001	0.016	0.076	0.17	0.053	0.031	3.6	59
	(0.014, 0.014)	(<0.001, <0.001)	(0.016, 0.015)	(0.077, 0.075)	(0.17, 0.17)	(0.052, 0.053)	(0.030, 0.032)	(3.6, 3.5)	
19 (Thames (Swin))	0.014	0.002	0.012	0.16	0.38	0.14	0.056	4.3	44
	(0.014, 0.013)	(0.002, 0.002)	(0.012, 0.011)	(0.17, 0.15)	(0.39, 0.36)	(0.14, 0.13)	(0.058, 0.053)	(4.4, 4.2)	
19 (Thames (Swin))	0.017	0.015	0.037	0.17	0.23	0.13	0.077	6.2	60
	(0.017, 0.016)	(0.015, 0.015)	(0.039, 0.035)	(0.16, 0.17)	(0.22, 0.23)	(0.11, 0.14)	(0.079, 0.074)	(6.1, 6.3)	
20 (Thames (Barrow Deep))	0.008	<0.001	0.013	0.080	0.13	0.044	0.016	1.9	44
	(0.008, 0.008)	(<0.001, <0.001)	(0.012, 0.013)	(0.086, 0.074)	(0.12, 0.14)	(0.045, 0.043)	(0.016, 0.016)	(1.8, 1.9)	

NR = No result

Table 4. Concentrations of PCB's and pesticide residues in dab liver

Station no./ area	Concentration (mg kg^{-1} wet weight)								Fat (%)
	HCB	α–HCH	γ–HCH	Dieldrin	ppDDE	ppTDE	ppDDT	PCB	
1 (Off Berwick)	0.006 (0.006, 0.006)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.061 (0.056, 0.065)	0.045 (0.043, 0.046)	0.008 (0.007, 0.009)	0.028 (0.025, 0.030)	0.33 (0.33, 0.33)	17
2 (Off Amble)	0.004 (0.004, 0.004)	<0.001 (<0.001, <0.001)	<0.001 (0.001, <0.001)	0.029 (0.028, 0.029)	0.018 (0.019, 0.016)	<0.001 (<0.001, <0.001)	0.008 (0.007, 0.008)	0.15 (0.16, 0.14)	20
3 (Off the Tyne)	0.002 (0.002, 0.002)	<0.001 (<0.001, <0.001)	0.001 (0.001, 0.001)	0.009 (0.009, 0.009)	0.011 (0.011, 0.011)	<0.002 (<0.001, 0.002)	0.004 (0.004, 0.004)	0.1 (0.1, 0.1)	5
5 (N. Dogger)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	<0.001 (0.001, <0.001)	0.005 (0.005, 0.004)	0.022 (0.022, 0.022)	<0.001 (<0.001, <0.001)	0.006 (0.006, 0.005)	0.23 (0.23, 0.23)	3
5 (N. Dogger)	0.002 (0.002, 0.002)	0.002 (0.002, 0.002)	0.005 (0.004, 0.005)	0.006 (0.007, 0.005)	0.017 (0.016, 0.017)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.15 (0.14, 0.15)	16
6 (NE. tip of Dogger)	0.003 (0.003, 0.002)	0.003 (0.002, 0.003)	0.005 (0.004, 0.006)	0.012 (0.012, 0.012)	0.017 (0.018, 0.015)	0.012 (0 013, 0.010)	0.025 (0.020, 0.029)	0.19 (0.20, 0.17)	33
7 (W. of Dogger)	0.002 (0.002, 0.002)	0.001 (0.001, 0.001)	<0.001 (<0.001, <0.001)	0.013 (0.013, 0.012)	0.012 (0.012, 0.012)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.13 (0.12, 0.14)	9
8 (W. edge of Dogger)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.003 (0.003, 0.003)	0.027 (0.025, 0.028)	0.001 (0.001, 0.001)	<0.001 (<0.001, <0.001)	0.25 (0.24, 0.25)	3
9 (Central Dogger)	0.003 (0.003, 0.003)	0.003 (0.002, 0.003)	0.005 (0.005, 0.005)	0.002 (0.002, 0.002)	0.012 (0.012, 0.011)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.2 (0.2, 0.2)	18
10 (Central Dogger)	0.004 (0.004, 0.004)	0.003 (0.003, 0.003)	0.003 (0.003, 0.003)	0.007 (0.007, 0.007)	0.015 (0.016, 0.013)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.15 (0.15, 0.14)	20
11 (E. of Dogger)	0.003 (0.003, 0.002)	<0.001 (<0.001, <0.001)	0.003 (0.002, 0.003)	0.008 (0.007, 0.008)	0.027 (0.029, 0.025)	<0.001 (<0.001, <0.001)	<0.001 (<0.001, <0.001)	0.34 (0.34, 0.34)	13
12 (Central Outer Silver Pit)	0.003 (0.003, 0.003)	0.002 (0.002, 0.002)	0.005 (0.005, 0.005)	0.012 (0.015, 0.008)	0.022 (0.021, 0.022)	<0.001 (<0.001, <0.001)	0.006 (0.006, 0.006)	0.27 (0.28, 0.26)	24
13 (Sole Pit)	0.004 (0.004, 0.004)	<0.001 (<0.001, <0.001)	0.005 (0.004, 0.005)	0.024 (0.023, 0.024)	0.035 (0.037, 0.032)	0.007 (0.006, 0.007)	<0.001 (<0.001, <0.001)	0.40 (0.43, 0.37)	25
14 (SE. of Outer Silver Pit)	0.002 (0.002, 0.002)	<0.001 (0.001, <0.001)	0.002 (0.002, 0.002)	0.013 (0.013, 0.013)	0.012 (0.012, 0.011)	<0.001 (<0.001, <0.001)	<0.002 (0.003, <0.001)	0.23 (0.24, 0.22)	9
15 (Smith's Knoll)	0.006 (0.006, 0.006)	0.005 (0.005, 0.005)	0.009 (0.009, 0.009)	0.032 (0.031, 0.033)	0.033 (0.033, 0.033)	0.011 (0.011, 0.010)	<0.001 (<0.001, <0.001)	0.95 (0.95, 0.94)	31
16 (E. of Smith's Knoll)	0.003 (0.003, 0.003)	0.005 (0.005, 0.005)	0.007 (0.007, 0.007)	0.022 (0.024, 0.020)	0.031 (0.031, 0.031)	0.009 (0.009, 0.009)	0.025 (0.024, 0.026)	0.78 (0.77, 0.79)	24
20 (Thames (Barrow Deep))	0.005 (0.005, 0.005)	<0.001 (<0.001, <0.001)	0.013 (0.013, 0.013)	0.050 (0.053, 0.046)	0.069 (0.071, 0.067)	0.017 (0.015, 0.018)	0.018 (0.020, 0.015)	1.3 (1.3, 1.2)	26

Table 5. Concentrations of metals in Mytilus edulis sampled in July 1990

Area of capture	No. of shellfish	Mean length (cm)	Concentration (mg kg^{-1} wet weight)					Dry matter (%)
			Hg	Cu	Zn	Cd	Pb	
Berwick	50	4.3	0.05 (0.04, 0.05)	1.2 (1.1, 1.2)	16 (15, 16)	0.15 (0.14, 0.15)	<0.6 (<0.6, <0.6)	16
Holy Island*	50	4.2	0.04 (0.04, 0.04)	1.6 (1.6, 1.5)	16 (16, 16)	0.16 (0.17, 0.15)	<0.6 (<0.6, <0.6)	18
Coquet Estuary	50	4.3	0.03 (0.03, 0.03)	1.2 (1.2, 1.2)	20 (20, 20)	0.11 (0.10, 0.11)	0.8 (0.8, 0.8)	17
Cresswell	50	3.9	0.06 (0.07, 0.05)	0.9 (0.9, 0.9)	20 (20, 20)	0.24 (0.25, 0.23)	<0.6 (<0.6, <0.6)	16
Blyth	49	4.1	0.04 (0.03, 0.05)	1.3 (1.0, 1.5)	24 (19, 28)	0.14 (0.14, 0.14)	1.9 (1.7, 2.1)	16
Tyne (Trow rocks)	50	4.1	0.06 (0.05, 0.06)	1.4 (1.4, 1.4)	49 (49, 48)	0.11 (0.11, 0.10)	4.6 (4.5, 4.7)	16
Tees mouth	50	4.0	0.01 (0.01, 0.01)	1.6 (1.6, 1.6)	29 (29, 28)	0.22 (0.22, 0.21)	1.9 (2.0, 1.8)	17
Whitby	50	4.0	0.04 (0.04, 0.04)	1.7 (1.7, 1.6)	27 (27, 27)	0.18 (0.17, 0.19)	2.2 (2.2, 2.2)	16
Filey Brigg	50	3.9	0.06 (0.06, 0.06)	1.5 (1.5, 1.5)	19 (19, 19)	0.15 (0.16, 0.14)	2.0 (1.9, 2.0)	20
Bullfort	49	4.2	0.03 (0.03, 0.03)	1.8 (1.8, 1.8)	33 (33, 33)	0.59 (0.57, 0.61)	<0.6 (<0.6, <0.6)	18
Cleethorpes	50	4.0	0.04 (0.04, 0.03)	1.8 (1.7, 1.8)	26 (25, 26)	0.56 (0.56, 0.55)	<0.6 (<0.6, <0.6)	22
Kings Lynn*	50	4.1	0.02 (0.02, 0.02)	1.2 (1.2, 1.2)	17 (17, 17)	0.19 (0.19, 0.18)	0.7 (0.7, 0.7)	18
Hunstanton	50	4.0	0.02 (0.02, 0.02)	1.3 (1.3, 1.3)	14 (14, 13)	<0.06 (<0.06, <0.06)	<0.6 (<0.6, <0.6)	19
Walberswick	50	3.9	0.05 (0.05, 0.05)	1.3 (1.3, 1.3)	18 (18, 18)	0.12 (0.12, 0.12)	1.1 (1.0, 1.1)	14
Harwich	50	3.8	0.05 (0.05, 0.04)	1.2 (1.2, 1.1)	20 (20, 20)	0.21 (0.19, 0.22)	<0.6 (<0.6, <0.6)	18
Creeksea	50	4.0	0.04 (0.05, 0.03)	1.4 (1.3, 1.4)	19 (18, 19)	0.17 (0.18, 0.16)	<0.6 (<0.6, <0.6)	18
Southend	50	4.0	0.02 (0.02, 0.02)	1.6 (1.6, 1.6)	23 (22, 23)	0.29 (0.28, 0.29)	0.7 (0.7, 0.6)	19
Swale	50	4.0	0.02 (0.02, 0.02)	1.5 (1.4, 1.5)	23 (22, 23)	0.23 (0.22, 0.23)	0.6 (0.6, 0.6)	18
Whitstable*	49	3.9	0.04 (0.03, 0.05)	1.4 (1.6, 1.2)	19 (19, 18)	0.39 (0.38, 0.40)	<0.6 (<0.6, <0.6)	17

**Areas where commercial exploitation is known to take place*

1.3.1 Mercury

Concentrations of mercury were generally low, with virtually all values in or near the 'lower' JMP category (see Appendix 2) for both fish and mussels. In an earlier study of dab in the Southern North Sea, Claussen (1988) noted a clear decrease in mercury levels from the coast to the open sea. There was little evidence of such an obvious spatial trend in the present work (see, for example, Figure 2) though some relatively high concentrations were found in fish samples from the Thames (up to a mean of 0.14 mg kg^{-1} wet weight in whiting) and earlier work (MAFF, 1990) indicates that concentrations of mercury in dab from the Humber estuary (0.22 mg kg^{-1} wet weight) are considerably higher than any reported here.

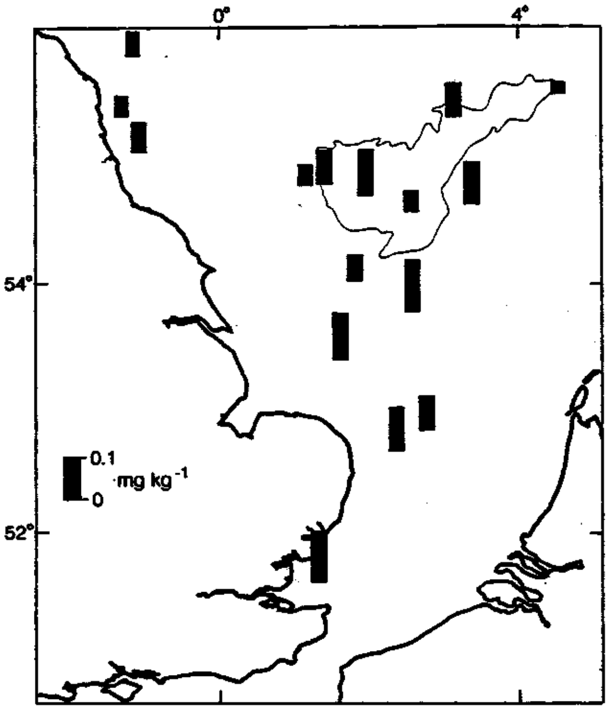

Figure 2. Concentrations of mercury (mg kg⁻¹) in dab muscle

1.3.2 Copper

Levels of copper in fish muscle were generally less than 0.3 mg kg^{-1} wet weight (i.e. in the lower half of the 'expected' ranges) (Appendix 2). Concentrations in fish liver were much higher than those in muscle (by factors of x10 or more), but there appeared to be no clear spatial pattern in the results. Although differences between samples were not great, the highest concentration in mussels (1.8 mg kg^{-1} wet weight) occurred in the Humber area (Bull Fort and Cleethorpes). These stocks are not known to be commercially exploited, but even here the copper values were considerably less than the Food Standards Committee's recommended limit (see Appendix 2).

1.3.3 Zinc

The picture for zinc in fish was similar to that for copper, with concentrations in muscle being within the expected ranges (up to 6.0 mg kg^{-1} wet weight) and values in liver, though generally about ten times higher, showing no obvious spatial pattern. As in earlier work (MAFF, 1990), highest levels in mussels occurred in the Tyne area (49 mg kg^{-1} wet weight). Concentrations in all samples, including those from stocks which are not commercially exploited (such as those in the Tyne) were therefore less than the Food Standards Committee's guideline value of 50 mg kg^{-1} wet weight (see Appendix 2).

1.3.4 Cadmium

Claussen (1988) reported relatively higher cadmium levels in the livers of dab taken from the Dogger Bank when compared with adjacent areas off the coasts of Britain and Germany. In the present study, results were variable but some relatively high values (>0.4 mg kg^{-1} wet weight) were recorded in both whiting and dab livers from stations on the Dogger Bank. However, for dab (Figure 3), there were indications of similar concentrations being present south and west of the Dogger Bank. As mentioned previously, further information on this region should become available following work planned for 1991. Some data were reported in a previous report (MAFF, 1990) for a sample of dab taken from the Humber estuary itself.

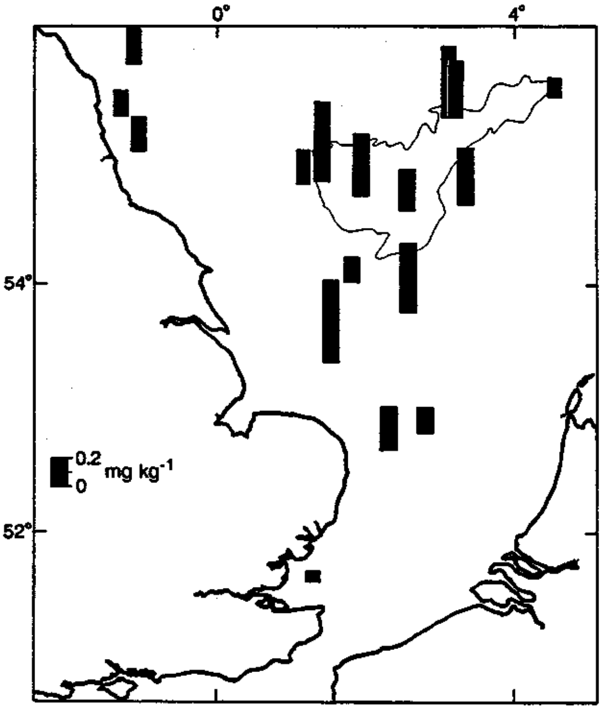

Figure 3. Concentrations of cadmium (mg kg⁻¹) in dab liver

The level of cadmium in the liver obtained from this sample was 0.60 mg kg^{-1} wet weight, and about the same as the highest concentrations found in this study. Cadmium is normally present in very low levels in fish muscle and in the present study, for those fish found to have relatively high concentrations of cadmium in the liver, concentrations in muscle samples were only ~0.005 mg kg^{-1} wet weight.

As in previous years, the maximum concentration of cadmium in mussels taken from the east coast of England occurred in the Humber. Cadmium levels have in the past been in the JMP 'upper' category [>5.0 mg kg^{-1} dry weight (Appendix 2)], but in 1988 and 1989 these had fallen to 4.5 mg kg^{-1} dry weight (MAFF, 1991(a)). The 1990 value reported here (equivalent to 3.3 mg kg^{-1} dry weight) continues the time series of concentrations in the 'medium' JMP category.

Time-trend studies carried out on the horse-mussel (*Modiolus modiolus*) from the outer Humber, reported in Section 2 of this report, indicate that a reduction in cadmium levels for this species has also recently occurred.

1.3.5 Lead

Levels of lead in fish liver were all below the limit of detection for the analytical method used. Concentrations in mussels showed a spatial pattern which was similar to that revealed by earlier work, with the highest values being found along the north-east coast (Blyth to Filey Brigg) but with no levels approaching the limit for shellfish of 10 mg kg^{-1} wet weight (Lead in Food Regulations 1979, see Appendix 2).

1.3.6 Organochlorine pesticides

Concentrations of all organochlorine pesticides in whiting and dab livers were generally low and well within 'expected' values (see Appendix 2). It is difficult to detect any general spatial pattern in the results but, for virtually all of the pesticides, the highest concentrations occurred in samples taken from the Thames estuary area (similar levels to those reported earlier for the area in MAFF, 1990). Relatively high values for dieldrin and DDT were also recorded in dab off Berwick, and there was the occasional comparatively high pesticide concentration in offshore areas (e.g. ppDDT in some samples from Stations 6 and 16).

1.3.7 PCBs

A spatial trend for PCBs in fish liver appeared to be more obvious than that for pesticides, with highest values (some in 'upper' JMG categories - see Appendix 2) being found off the Thames. There was a general decrease northward towards the Dogger Bank area where concentrations were all in the 'lower' JMP categories but, for whiting, 'medium' category values were generally obtained in the area between the Thames and Station 12 (the Outer Silver Pit) (Figure 4). The concentration in the sample of whiting from the Barrow Deep area of the Thames (Station 20) was only in the 'lower' category. In previous years, concentrations have been higher in samples from this area (see, for example, MAFF, 1990).

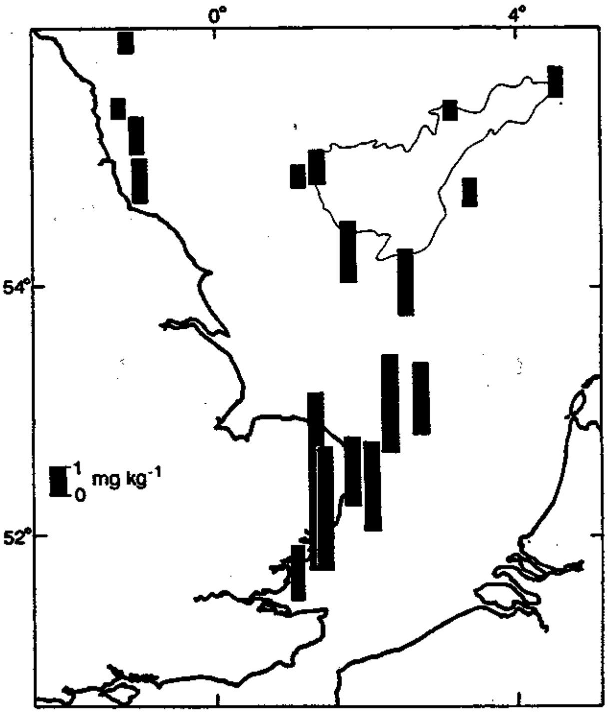

Figure 4. Concentrations of PCB's (mg kg^{-1}) in whiting liver

1.4 Conclusions

Survey work carried out over the 1989/90 period has provided information on contaminant levels in fish taken from areas offshore, off the English North Sea coast. A preliminary evaluation of the data (further work was carried out in 1991 and will be reported later) shows little indication of any general spatial trends for mercury, copper, zinc and organochlorine pesticides. Relatively high concentrations of cadmium were found in the livers of some fish taken from the Dogger Bank, but further data are required from the region off the Humber for comparative purposes. However, there does appear to be a spatial trend in PCBs in fish liver, with highest concentrations off the Thames and generally lower levels in the northern part of the area surveyed and over the Dogger Bank.

Recent information on levels of heavy metals in mussels has confirmed the spatial pattern observed in previous work, with generally low mercury and copper levels, and highest concentrations of zinc, cadmium and lead being found in stocks which are not commercially exploited in the Tyne, Humber and north-east coast areas respectively.

2. BIOLOGICAL AND CHEMICAL STUDIES OF THE HORSE-MUSSEL (*MODIOLUS MODIOLUS*) IN THE HUMBER/WASH REGION

2.1 Introduction

Annual sampling of horse-mussels for biological and chemical analysis has been conducted at up to 5 sites in the Humber/Wash area since 1984. Trends in body burdens of lead, up to 1987, were reported in Rees and Nicholson (1989). The initial objective of the work was to compare populations near to and distant from the Humber sewage-sludge disposal site, but it has become increasingly clear that efflux from the Humber estuary itself is likely to be a major factor determining concentrations of contaminants in biota of the general area. As such, the spatial distribution of sampling stations (Figure 5) provides a useful basis for a regional assessment of the biological consequences, if any, of both diffuse and point-source (sewage-sludge) inputs. The sampling programme has added significance, for the following reasons:

i. the Humber/Wash area is known to be enriched, for example in particulate lead (Turner *et al.*, 1991);

ii. tide-swept superficial gravel deposits are widespread in the area; these are not amenable to quantitative sampling by Day grab, so that traditional assessments of the structure of benthic communities by this means are not feasible;

iii. accurate ageing of horse-mussels from the Humber area has shown that the animals may commonly live for 20 years or more (Anwar *et al.*, 1990). This, along with their sedentary and filter-feeding habit, makes them a potentially ideal indicator of the well-being of subtidal benthic communities;

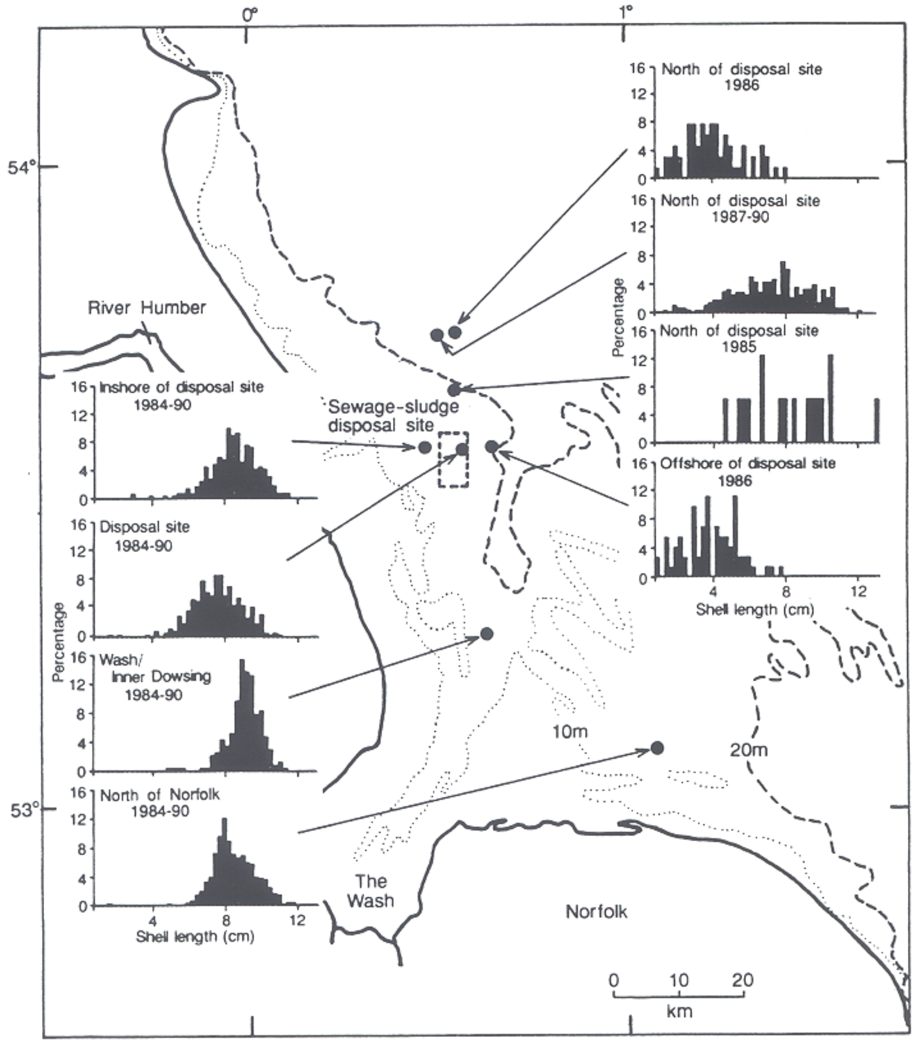

Figure 5. Locations of horse-mussel sampling stations in the Humber/ Wash area. Combined length-frequency distributions at each location are super-imposed

- iv. at least in principle, horse-mussel populations in this region represent a potentially exploitable commercial resource, either as bait for fishing or as human food. They are also subject to natural predation by commercially important species such as edible crabs and whelks;
- v. studies in the Humber/Wash area in relation to anthropogenic influences are important in connection with wider assessments of the quality status of the North Sea.

The purpose of this account is to summarise the chief biological attributes of the sampled populations, and to provide an illustration of their use in the monitoring of trends in concentrations of trace metals. (A full account of this work, along with an assessment of trends in body burdens in relation to biological variables will be published on completion of current research).

2.2 Methods

As far as possible, sampling procedures followed guidelines for temporal trend-monitoring produced by ICES, which involved the separate analyses of whole body tissue from 25 individuals on each sampling occasion, and at each site. Further details are given in Rees and Nicholson (1989). Replicate data are expressed as arithmetic means with 'Least Significant Intervals' (LSI's). For comparisons between years and between sites, intervals which do not overlap may be assumed to be significantly different at the 95% probability level (see Andrews *et al.*, 1980).

2.3 Results and discussion

2.3.1. Size structure

Combined length-frequency histograms at several sites in the Humber/Wash area are shown in Figure 5. There was no clear evidence to suggest systematic growth in length of cohorts through time. The most notable feature was the predominance of older individuals at several sites, with modal sizes typically in the range of 8-9 cm. At these sites, there was evidence only for spasmodic recruitment of juveniles. In contrast, sites to the east and north of the disposal site were characterised by a preponderance of smaller individuals. The reasons for this are presently unclear, but may reflect differences in predator-prey interactions between sites.

2.3.2. Concentrations of cadmium in relation to biological variables and waste inputs

For this comparison, the station at the Humber disposal site and that off Norfolk were selected. Concentrations of contaminants in flesh may be influenced by seasonal and annual variation in condition of animals, as well as by their absolute size (see, for example, Phillips and Segar, 1986). In the following analysis, animal size was standardised by selecting those within the length range of 7-9 cm. (The overall mean length of animals at the five regularly sampled stations was about 8 cm). At the above two sites, the minimum number present on any one sampling occasion within this size category was eight; accordingly, the same number were selected (randomly where more were present) for all sampling occasions. Figure 6(a) confirms that the means and ranges of shell lengths were very similar at both sites between 1984 and 1990.

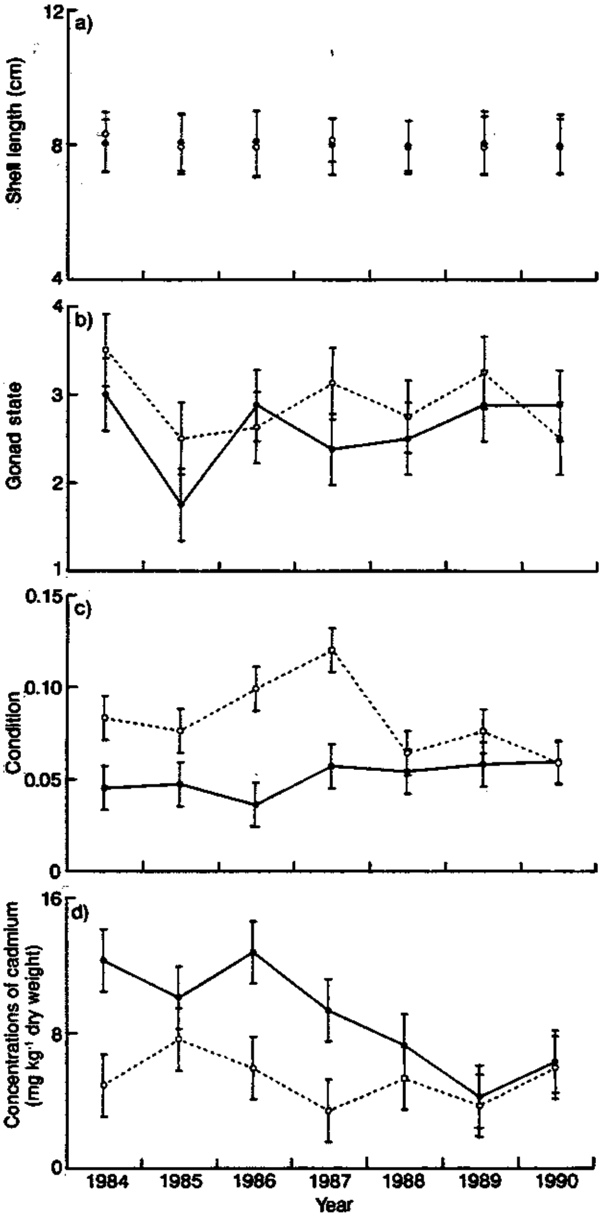

Figure 6. ***Annual trends in biological and chemical variables, derived from analysis of samples of east coast horse-mussels. Results for shell length are expressed as arithmetic means with ranges; others are expressed as arithmetic means with 95% LSI's: (a) shell length; (b) gonad state; (c) condition; and (d) concentrations of cadmium (solid circles: Humber disposal site; open circles: off Norfolk)***

A notable factor influencing condition of animals is that of reproductive state. Unlike the intertidal mussel (*Mytilus*) horse-mussels tend to breed continually (Brown, 1984), and this appears to be the case in the present area. Gonad state was assessed on a subjective four-point scale as described in Rees and Nicholson (1989), with the lowest and highest values representing unripe and ripe individuals, respectively. Condition (of which gonad state is a component) was expressed as dry body weight divided by shell weight.

It will be seen from Figure 6(b) that gonad development was generally somewhat higher at the Norfolk site, though the differences were not usually significant. Significantly higher values for condition were also found off Norfolk in the early years, but this has tended to decrease more recently (Figure 6(c)). Although the trend in condition at the Humber station is marginally upward, the differences between individual years and over the whole sampling period, 1984-1990, are not statistically significant.

Figure 6(d) shows trends in concentrations of cadmium at the two locations. The trend at the Humber location is one of decline with time. Concentrations at both sites were very similar in 1989 and 1990. Correlations of concentrations of cadmium with gonad state at both sites, and with condition at the Norfolk site, yielded no significant relationships. However, there was a significant though weak negative association between concentrations and condition of animals at the Humber site ($r = -0.35$; $df = 54$; $p \approx 0.01$). Further investigations will be required in order to establish whether any causal relationship might exist.

Figure 7 shows trends in the quantities of sewage sludge disposed of to the Humber site from 1984-1990, along with loads of cadmium. In the latter case, the downward trend can be ascribed to improved controls on the quality of effluent entering the sewerage system in recent years. However, such a trend is likely to be matched in the quality of water from the Humber itself, and hence it is possible that this source may also be an important factor accounting for regional-scale trends, especially bearing in mind the highly dispersive nature of the sewage-sludge disposal site itself, and the fact that the quantities discharged are relatively small. Support for this contention is provided by information from the National Rivers Authority (C. Urquhart personal communication). The average annual load of cadmium discharged to the Humber system for the period 1985-1989 amounted to about 6 tonnes, compared with just over 3 tonnes in 1990.

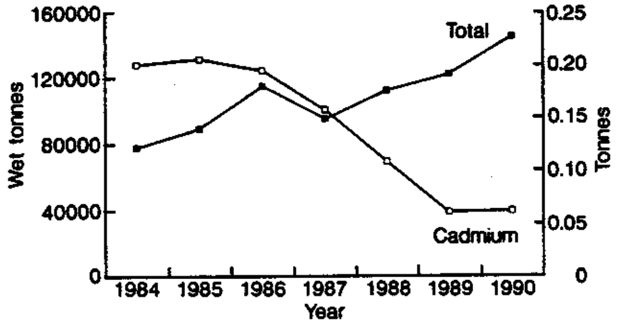

Figure 7. Quantities of sewage sludge (closed squares) and of cadmium (open squares) discharged at the Humber disposal site, 1984-1990

Further work on trends in a range of trace metals in relation to biological and environmental factors is currently underway. This includes a project funded by MAFF on ageing of horse-mussel shells, to be conducted at the University College of North Wales, Marine Sciences Laboratory.

3. USE OF BIOASSAYS TO ASSESS MARINE SEDIMENT AND WATER QUALITY

3.1 General introduction

Earlier work on this subject was described in a previous report (MAFF, 1991(a)). In July 1990, an extensive water and sediment sampling survey was carried out on MAFF's vessel *RV CIROLANA* (details of the stations worked are given in Section 5-Table 7 and Section 11-Table 12). A major part of the sampling carried out on this cruise, and the deployment of an oyster embryo bioassay, was undertaken in fulfilment of MAFF's responsibilities under the North Sea Task Force Monitoring Master Plan (NSTFMMP) (North Sea Task Force, 1990) the results of which are to be utilised in the production of a North Sea Quality Status Report in 1993/94. During the cruise, an algal bioassay was also deployed. The results from the two bioassays described here focus on the north-east coast of England (the algal bioassay was employed only in this area in 1990) and provide a snapshot of the usefulness of these bioassays in measuring sediment and water quality. The full results of this work will be published, alongside those from similar work planned for 1991, in the next annual report on monitoring of non-radioactive contaminants. By that time, greater experience will be available and a more comprehensive interpretation should be possible.

3.2 Algal growth bioassay

3.2.1 Introduction

This bioassay is primarily used to assess the algal growth potential of sea-water samples. Chemical analysis alone may reveal the nutrient status of sea water, but in conjunction with a biological measure, a more powerful and meaningful conclusion can be made about its potential to support growth of phytoplankton, as this will take account of any phytotoxic substances present that may inhibit algal growth. Furthermore, it is not only the quantity of N, P and Si present in sea water that drives the eutrophication process, but frequently the relative proportions of these elements. Clearly, one way to measure this is with algae which will integrate the quantity and ratio of nutrients present.

3.2.2 Methods

The bioassay is now routinely deployed on-board MAFF's research vessels. Surface sea-water samples are taken at 1 m depth in glass bottles. Replicate 125 ml volumes of this sample in 250 ml flasks are inoculated with a stock algal culture of *Thalassiosira pseudonana* at a concentration of 5 x 10^4 cells ml^{-1}. After 5 days incubation in constant light conditions, the algal cell densities are measured directly by counting under a microscope or spectrophotometrically. Nutrients are determined in a separate sub-sample of each sea-water sample by the autoanalyser technique described by Kirkwood (1989).

3.2.3 Results and discussion

Figure 8 shows algal cell growth for *T. pseudonana* at sites along the Tweed, Tyne, Wear, Tees and Humber estuaries of the north-east coast of England and for open sea locations out to the Dogger Bank. In all

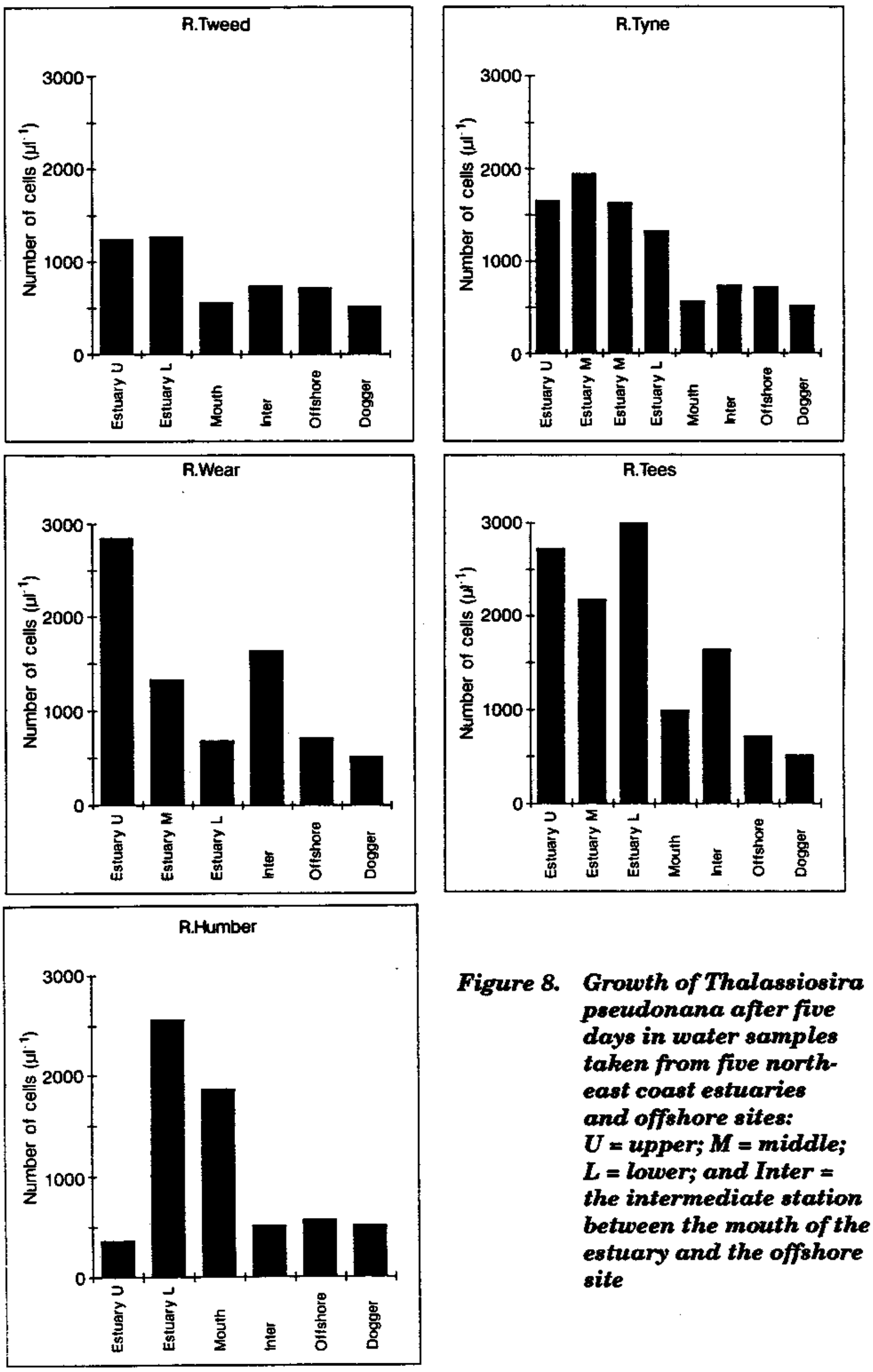

Figure 8. Growth of Thalassiosira pseudonana after five days in water samples taken from five north-east coast estuaries and offshore sites: U = upper; M = middle; L = lower; and Inter = the intermediate station between the mouth of the estuary and the offshore site

cases, algal growth was greatest at the estuarine sites and in general there was a trend for algal growth to increase with progression up the estuary. In the Humber estuary, a comparatively reduced growth was observed at the most landward site. An explanation for this may be found in Figure 9 which shows that concentrations of PO_4 were much lower than those in the other estuaries. In general terms, algal growth was most enhanced in the Tees and, in descending order, in the Tyne, Humber, Wear and Tweed. This may reflect the pattern of nutrients found in the estuaries (Figure 9), which itself derives from the degree of urbanisation at each estuary.

Nutrient levels vary considerably in the open sea throughout the year and are higher in winter than in summer. The results shown here are from a survey carried out in July; a winter survey would probably show algal growth to be higher at open sea locations and even to be reduced in estuaries where winter flows dilute the input of nutrients.

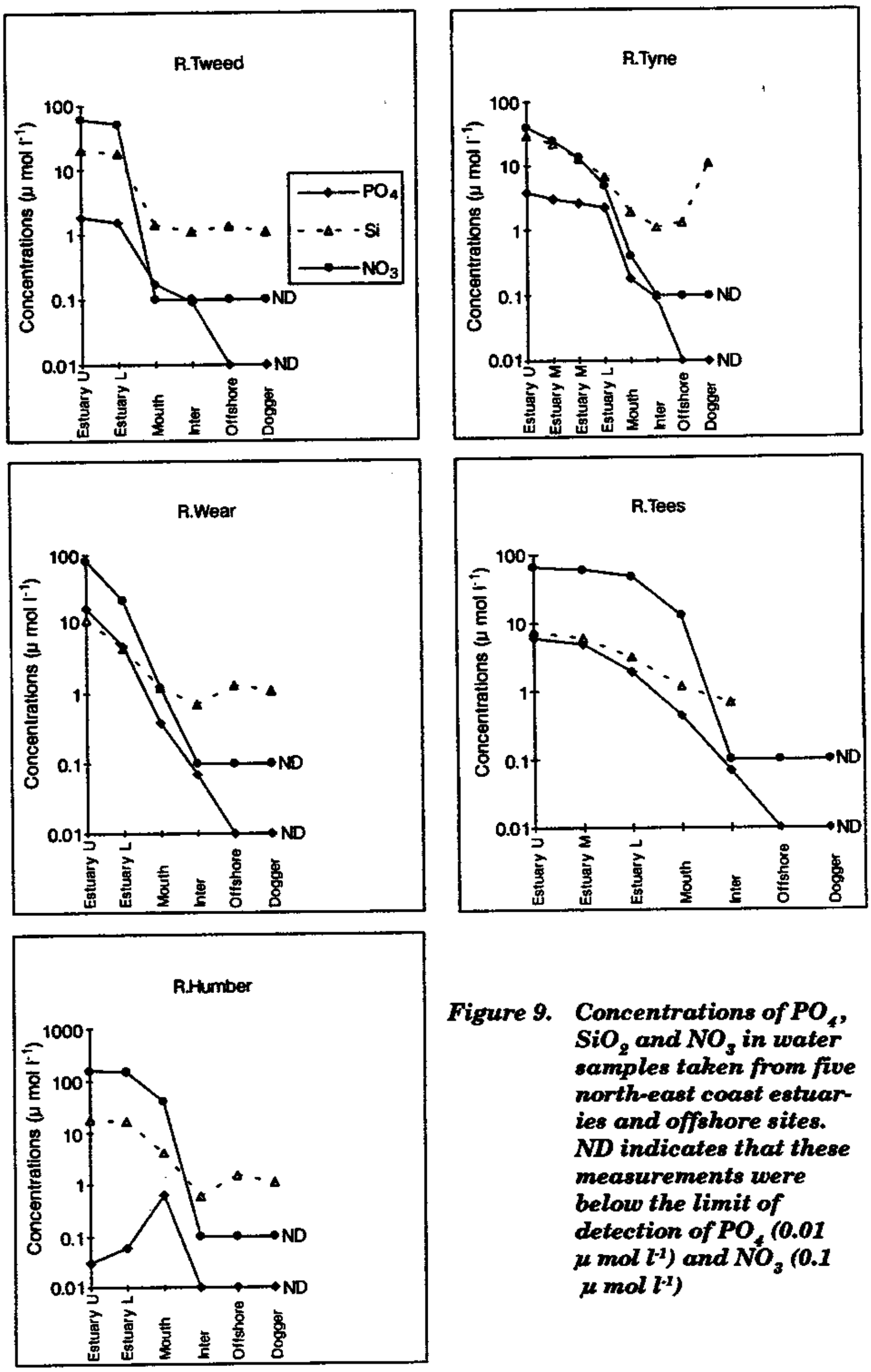

Figure 9. ***Concentrations of PO_4, SiO_2 and NO_3 in water samples taken from five north-east coast estuaries and offshore sites. ND indicates that these measurements were below the limit of detection of PO_4 (0.01 µ mol l^{-1}) and NO_3 (0.1 µ mol l^{-1})***

3.3 Oyster embryo bioassay

3.3.1 *Introduction*

The NSTF (1990) has adopted four techniques for monitoring biological effects, one of which is the oyster embryo bioassay. On the north-east coast survey, samples were taken at two to four sites within each major river, and one at the estuary mouth, at a nearshore and offshore site and a station well offshore, located on the Dogger Bank.

3.3.2 *Methods*

The NSTFMMP requires the bioassay to be used for water column and sediment samples. For the water samples, 2.5 l is taken from 1m depth and assayed within 6 h. To enable all samples to be compared on a similar salinity basis (below a salinity of 25, growth would be affected purely by this factor), upper estuarine samples are adjusted to that of full sea water by the addition of artificial sea salts. For each assay, oysters pre-conditioned for spawning are opened, the gametes stripped and the eggs artificially fertilized. The embryos are placed in the water samples and their development after 24 h is assessed by microscopic examination (see Thain, 1991). The embryos' development in the test water is compared to the development in a control water obtained from the Atlantic (the South-Western Approaches). The result is expressed as Percent Net Response (PNR), i.e.:

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

For sediment samples, 200 ml of sediment are placed with 500 ml of 'control' reference sea water in a 1 litre screw-capped polycarbonate container and shaken on an orbital shaker at a speed of 100 revolutions min^{-1}. After 3 h, the contents are filtered through Whatman GFC filter paper and the resulting elutriate assayed as described above for water samples.

3.3.3 *Results and discussion*

No deleterious water quality was measured with the bioassay in any of the water samples taken from the estuaries of the Rivers Tweed, Tyne, Wear or Humber, nor was any response recorded at any open sea locations. The River Tees samples did however show a response with no embryos developing in the upper estuarine sample and very few in the middle estuarine sample (PNR values of 100 and 71 respectively). It should be emphasized that these results are only a snap-shot in time, and furthermore, that the sampling locations do not extend to the tidal limit in each river.

The results for the sediment elutriate bioassays for each estuary are shown in Figure 10. PNR values

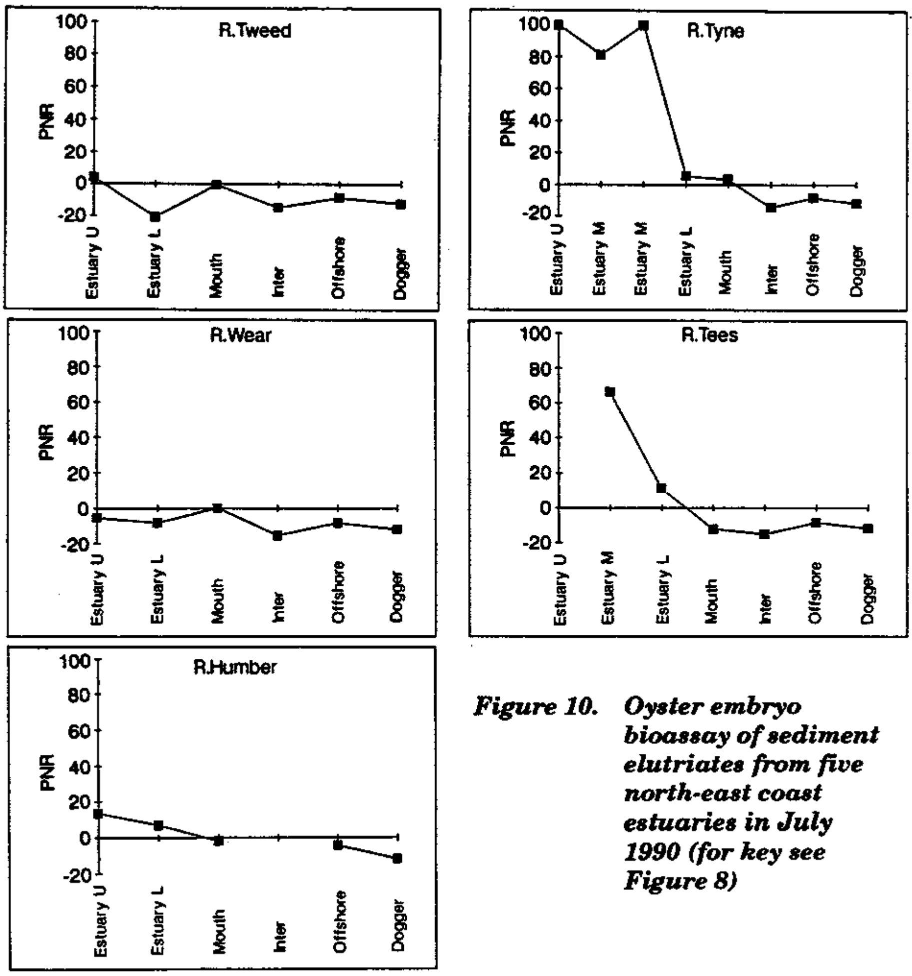

Figure 10. Oyster embryo bioassay of sediment elutriates from five north-east coast estuaries in July 1990 (for key see Figure 8)

of 0, or negative, indicate that embryonic development was similar to that in the 'control' reference sea water. No observable effects were measured in sediment elutriates from the Tweed, Wear and Humber estuaries. On the River Tees, a suitable sample could not be obtained from the upper estuarine site, but the sediment elutriate sample from the middle estuary exhibited some degree of toxicity. The most toxic sediments were found at the upper and middle estuarine sites on the River Tyne, where two of the three sediment elutriates completely inhibited embryonic development. The results of this study show that a water column bioassay can usefully be employed to assess sediment quality. However, it must be emphasised that, although it has a use in measuring relative site differences, it does not impart information on absolute sediment toxicity to benthic species under field conditions. Research is currently underway to develop a whole-sediment bioassay which can give a clearer picture of the toxic effects experienced by sediment-dwelling fauna. The above results should therefore be regarded as preliminary results.

The oyster embryo bioassay was also deployed at all stations sampled on the south and west coasts of England (see Table 7). No indications of poor water quality or sediment elutriate toxicity were observed, with the exception of Station 55 on the River Mersey where the sediment elutriate induced embryonic abnormality (a PNR of 69 was recorded).

The commitment to the NSTF programme is continuing and, as mentioned previously, both bioassays are again being utilised on MAFF research cruises in 1991. The full results from both the 1990 and 1991 work will be published later in this series.

4. SURVEYS OF CONTAMINANTS IN MARINE MAMMALS

Since the 1988 seal epizootic, the Burnham-on-Crouch Laboratory has been actively involved in the analysis of marine mammals for persistent organic contaminants, with particular emphasis on chlorobiphenyls. The majority of these data have already been published elsewhere (Law *et al.*, 1989, 1991; Morris *et al.*, 1989) and a summary of the data was included in a previous report in this series (MAFF, 1991(a)).

Current work is focused on samples provided by the DOE-funded initiative on marine mammals and will be published in detail elsewhere. This initiative is a direct response to the need for further information on the status of marine mammal stocks in the UK and the DOE coordinates the work of several institutes, including the Institute of Zoology, the Natural History Museum, the Sea Mammal Research Unit of the Natural Environment Research Council (NERC), as well as the Scottish Office Agriculture and Fisheries Department (SOAFD) and MAFF. The role of the Burnham-on-Crouch Laboratory is one of providing analytical support for the determination of trace metals and organics in samples of blubber (for organics) and liver (for metals).

The determination of trace organics in marine mammal tissues still presents considerable difficulties and inter-laboratory agreement has in the past been poor. Under the auspices of ICES/the Intergovernmental Oceanographic Commission (IOC)/OSPARCOM, the Burnham-on-Crouch Laboratory has been an active participant in a series of international intercomparison exercises aimed at improving the quality of analytical data on chlorobiphenyls in marine media. The stepwise progression of these exercises has led to the isolation of particular problem areas of the analysis, and recommendations regarding methodology have been made (De Boer *et al.*, in press).

The rationale behind the expression of residue data for chlorobiphenyls on an individual compound basis, rather than as a technical formulation, is based on the knowledge that transformation of chlorobiphenyls by degradative and metabolic processes occurs following their release to the environment and it is now generally accepted that the individual congener approach is preferred. However, the decision to switch from formulation expressed data to the individual congener approach is not one that can be taken lightly without full appreciation of the analytical problems. The cost of the analysis is also a major consideration. Pure individual chlorobiphenyl congener standards are now available as solid materials, but are expensive and are likely to continue to be expensive. The analytical equipment needs to be of the highest specification, well maintained and operated in accordance with good laboratory practice and by experienced personnel. Unfortunately, these constraints may restrict the rate at which laboratories can progress from formulation to congener specific analysis, but it is important that these problems are appreciated. The switch from formulation data to individual congeners does not automatically result in an improvement of the quality of data produced. Indeed, it is quite likely, without an understanding of the pitfalls, that the quality will deteriorate further rather than improve.

The data summarised in this account (Table 6) relate to a number of samples of marine mammals submitted to the Burnham-on-Crouch Laboratory for analysis during 1990. The samples submitted during this period were analysed using the standard protocols developed in recent years by the Laboratory (Allchin *et al.*, 1989). As well as the 'standard suite' of persistent organochlorines, a total of thirteen individual chlorobiphenyls including the 'ICES seven' (ICES, 1986) were determined. Subsequent samples have been analysed for a total of twenty-five chlorobiphenyls. This still excludes the so-called 'coplanar' compounds (BZ#77,126 and 169) which require the application of highly specialised fractionation techniques to allow their detection at the ultra-trace concentrations expected. These techniques are being actively developed at the Laboratory and will be applied retrospectively to a number of these samples.

Table 6. Concentrations of organic contaminants (mg kg^{1} wet weight) in marine mammals

Location	Date found	Species	Sex	Age	HEL (%)	CB28	CB31	CB52	CB101	CB105	CB118	CB128
Ballykinlar, Co. Down	Oct. 88	C	M	P	11	< 0.005	< 0.005	0.026	0.084	< 0.005	0.032	0.18
Northern Ireland	Oct. 88	C	M	J	41	< 0.005	< 0.005	0.13	0.27	< 0.005	0.081	0.57
Whiteabbey, Co. Antrim	Oct. 88	C	F	J	37	< 0.005	< 0.005	0.13	0.21	< 0.005	0.097	0.23
River Tees	Dec. 88	G	F	P	61	0.048	NA	0.077	0.25	< 0.005	0.16	NA
River Tees	Dec. 88	C	M	P	74	0.088	NA	0.068	0.16	< 0.005	0.091	NA
Portaferry, Co. Down	Jan. 89	G	F	P	38	< 0.005	< 0.005	0.026	0.055	< 0.005	0.028	0.048
Portrush, Co. Antrim	Feb. 89	G	F	P	31	< 0.005	< 0.005	0.29	0.3	< 0.005	< 0.005	0.7
County Sligo, I. Rep.	Mar. 89	WSD	$	J	33	< 0.005	< 0.005	< 0.005	0.011	< 0.005	0.017	0.007
Newport, W.Wales	Nov. 89	P	F	A	64	< 0.005	< 0.005	0.047	0.17	< 0.005	0.19	0.079
Aberystwyth, N.Wales	Feb. 90	SD	F	J	74	< 0.005	< 0.005	0.26	0.43	< 0.005	0.44	0.14
Penzance, Cornwall	Feb. 90	SW	$	$	56	< 0.005	< 0.005	< 0.005	0.22	< 0.005	0.22	0.06
Carmarthen Bay, S.Wales	Mar. 90	CD	F	A	76	< 0.005	< 0.005	0.73	0.81	< 0.005	1.1	0.8

Location	Date found	Species	Sex	Age	HEL (%)	CB138	CB149	CB153	CB170	CB180	CB183	CB187
Ballykinlar, Co. Down	Oct. 88	C	M	P	11	1.3	0.094	1.7	0.32	0.69	0.23	0.63
Northern Ireland	Oct. 88	C	M	J	41	3.3	0.49	4.1	0.93	2	0.77	1.5
Whiteabbey, Co. Antrim	Oct. 88	C	F	J	37	1.5	0.38	2.3	0.83	2	0.71	1.3
River Tees	Dec. 88	G	F	P	61	2.9	NA	4.7	NA	1.8	NA	NA
River Tees	Dec. 88	C	M	P	74	1.1	NA	1.7	NA	0.35	NA	NA
Portaferry, Co.Down	Jan. 89	G	F	P	38	0.33	0.076	0.45	0.067	0.16	0.056	0.15
Portrush, Co.Antrim	Feb. 89	G	F	P	31	4.1	1.3	5	1.1	2.2	0.93	2.2
County Sligo, I. Rep.	Mar. 89	WSD	$	J	33	0.045	0.03	0.055	0.014	0.04	0.016	0.036
Newport, W.Wales	Nov. 89	P	F	A	64	0.63	0.33	0.75	0.29	0.68	0.2	0.62
Aberystwyth, N.Wales	Feb. 90	SD	F	J	74	0.95	0.64	1.1	0.23	0.54	0.17	0.6
Penzance, Cornwall	Feb. 90	SW	$	$	56	0.45	0.27	0.52	0.11	0.25	0.08	0.24
Carmarthen Bay, S.Wales	Mar. 90	CD	F	A	76	4.2	2.4	4.7	1.1	2.2	0.81	2.1

Location	Date found	Species	Sex	Age	HEL (%)	HCB	α–HCH	β–HCH	γ–HCH	Dieldrin	pp'-DDE	pp'-TDE	pp'DDT
Ballykinlar, Co. Down	Oct. 88	C	M	P	11	<0.005	0.006	<0.005	<0.005	0.007	1.4	0.33	0.79
Northern Ireland	Oct. 88	C	M	J	41	<0.005	0.007	<0.005	<0.005	0.024	3.5	0.081	1.5
Whiteabbey, Co. Antrim	Oct. 88	C	F	J	37	<0.005	0.008	<0.005	<0.005	0.03	0.64	0.052	0.54
River Tees	Dec. 88	G	F	P	61	<0.005	0.043	<0.005	0.013	0.31	3	0.14	1.4
River Tees	Dec. 88	C	M	P	74	<0.005	<0.005	<0.005	<0.005	0.11	1.6	0.038	0.34
Portaferry, Co. Down	Jan. 89	G	F	P	38	<0.005	0.007	<0.005	<0.005	0.024	0.29	0.022	0.092
Portrush, Co. Antrim	Feb. 89	G	F	P	31	0.16	0.014	<0.005	0.007	0.23	6.5	0.3	3.2
County Sligo, I. Rep.	Mar. 89	WSD	$	J	33	0.005	0.007	<0.005	<0.005	0.033	0.054	0.093	0.034
Newport, W. Wales	Nov. 89	P	F	A	64	0.02	0.01	<0.005	< 0.005	0.1	0.31	0.4	0.17
Aberystwyth, N. Wales	Feb. 90	SD	F	J	74	0.23	0.03	<0.005	< 0.005	0.5	6.4	2.5	1.6
Penzance, Cornwall	Feb. 90	SW	$	$	56	0.09	< 0.005	<0.005	< 0.005	0.09	3.2	0.94	1.2
Carmarthen Bay, S. Wales	Mar. 90	CD	F	A	76	0.19	0.02	<0.005	0.06	0.84	8	3.1	1.4

G=grey seal	*Halichoerus grypus*	*SD=striped dolphin*	*Stenella coeruleoalba*	*P = pup (i.e. <1 yr)*
C=common seal	*Phoca vitulina*	*WSD=white-sided dolphin*	*Lagenorhynchus acutus*	*J = juvenile*
P=harbour porpoise	*Phocoena phocoena*	*SW=sperm whale*	*Physeter catodon*	*A = adult*
CD=common dolphin	*Delphinus delphis*	*NA = not analysed*		*$ = no information*
		HEL(%) = percentage of hexane extractable lipid		

Interpretation of organic contaminant data in marine mammals is difficult, particularly in individually stranded animals, when little or nothing is known about their life history and on whom full post-mortem analysis has not been performed. The feeding territory of some species is very large and the stranding of a carcass at one particular location is not necessarily (particularly with cetaceans) an indication that the animal was resident in, or even died in, that area. There are also considerable differences between sexes, with males generally having burdens that increase with age, whereas breeding females will depurate considerable percentages of their total body burden to their offspring, partly via placental transfer but mainly during lactation. The condition of the animal at the time of sampling is also important as demonstrated by Borrell and Aquilar (1989) who showed the loss of organochlorine compounds from the tissues of a decomposing stranded dolphin.

Given these problems and the fact that the DOE-funded study is still in its early stages, with few of the post-mortems fully reported, it is difficult to interpret the organochlorine residue data reported here. Few of the residues were present in particularly high concentrations and it is noteworthy that most of the higher concentrations were found in samples from animals where the extractable lipid content of the blubber was low. Whether this simply reflected the state of decomposition of the carcass or was related to the cause of death will only become apparent as more data are collected and coupled to the post-mortem findings of the cause of death.

It is of interest to note that the data set includes residue levels found in the blubber of a sperm whale stranded on the Cornish coast, giving a rare opportunity to study this off-shore, deep-water species.

SEAWATER

5. INTRODUCTION: GENERAL SAMPLING PROCEDURES

Most of the work reported in this section was carried out on *RV CIROLANA*, during cruise 6, 5-27 June 1990. The cruise track is illustrated in Figure 11. In order to avoid duplication, the basic information about this cruise will be given here and referred to again in later sections. Table 7 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. As mentioned in Section 3 of this report, some of the monitoring carried out in 1990 was undertaken to provide information for the North Sea Task Force Monitoring Master Plan. Where designated NSTFMMP stations were visited, their allotted number is also indicated in Table 7. Table 8 lists the water column data (depth, salinity, concentrations of nutrients, surface water temperature and sediment type) for each of the stations, where available.

The strategy adopted for estuarine sampling was to anchor the ship close to the mouth of the estuary, and to collect samples both from the ship, and from a small inflatable boat which could operate in shallow water 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 5.1 (Station 3, Table 7).

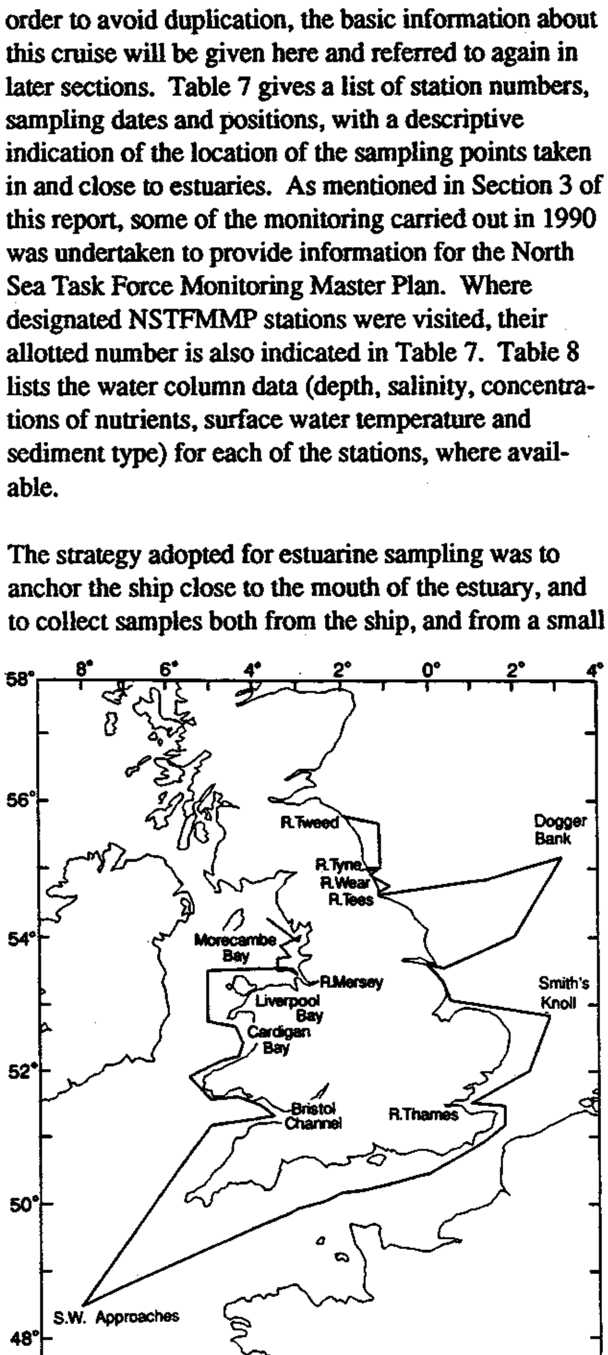

Figure 11. Cruise track of CIROLANA 6, 5-27 June 1990

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

6. NUTRIENTS

Concentrations of nutrients (see Table 8) were at a maximum in the estuarine samples, the highest values observed being : phosphate, 17 µmol l^{-1} (R. Wear - Station 13); silicate, 30 µmol l^{-1} (River Tyne - Station 9); nitrate, 160 µmol l^{-1} (River Humber - Station 25); and nitrite, 14 µmol l^{-1} (River Tees - Station 18). In offshore and coastal surface waters, concentrations of nitrate and phosphate were generally low or undetectable, as would be expected during the summer months. At the MAFF reference station in the Western Approaches (Station 45) nutrients were completely depleted in surface water, but detectable at depth. At 169 m (5 m above the bottom) concentrations of phosphate, silicate, nitrate and nitrite were 0.3, 2.7, 6.4 and 0.25 µmol l^{-1} respectively.

Much concern has been focused on the potentially deleterious consequences for marine ecosystems of nutrient run-off from land, occurring as a result of intensive agricultural practices and sewage discharges. A collaborative study (the Joint Nutrient Study: JONUS) has been established between MAFF, the Plymouth Marine Laboratory of NERC, and the Universities of East Anglia and Essex to study the fate of nutrients discharged to east-coast estuaries, and to estimate their net fluxes to the sea. The MAFF staff involved in this programme are based at Lowestoft, and the full details of their findings will be published elsewhere. Only data on the basic nutrient concentrations are published here as a broad indicator of the likely origin of the bulk of the water sampled.

Table 7. Stations sampled during CIROLANA 6, 5-27 June 1990

Station	Date (June)	Position		Location	NSTFMMP no.
1	6	55° 45.7' N	1° 57.85' W	Tweed/mouth	
2		55° 45.83' N	1° 59.58' W	Tweed/lower estuary	
3		55° 46.12' N	2° 0.35' W	Tweed/upper estuary	
4		55° 0.5' N	1° 7.6' W	Tyne/intermediate	14
5	7	55° 0.41' N	1° 23.76' W	Tyne/mouth	
6		55° 0.63' N	1° 24.98' W	Tyne/lower estuary	
7		54° 59.17' N	1° 27.86' W	Tyne/middle estuary	
8		54° 58.92' N	1° 31.82' W	Tyne/middle estuary	
9		54° 58.09' N	1° 36.25' W	Tyne/upper estuary	
10		55° 0.46' N	1° 7.76' W	Tyne/intermediate	14
11	8	54° 54.88' N	1° 20.3' W	Wear/mouth	
12		54° 55.04' N	1° 21.62' W	Wear/lower estuary	
13		54° 54.58' N	1° 22.87' W	Wear/upper estuary	
14		54° 44.04' N	0° 52.84' W	Tees/intermediate	15
15	9	54° 38.93' N	1° 7.35' W	Tees/mouth	
16		54° 38.52' N	1° 8.66' W	Tees/lower estuary	
17		54° 37.12' N	1° 9.32' W	Tees/middle estuary	
18		54° 36.22' N	1° 9.9' W	Tees/upper estuary	
19		54° 50.1' N	1° 20.24' E	Offshore Tyne/Tees	43
20	10	55° 10.1' N	3° 6.3' E	Dogger Bank	47
21		53° 59.94' N	2° 0.21' E	Silver Pit	53
22		53° 32.41' N	0° 20.12' E	Humber/intermediate	16
23	11	53° 33.33' N	0° 6.37' E	Humber/mouth	
24		53° 37.1' N	0° 2.82' W	Humber/lower estuary	
25		53° 36.0' N	0° 3.0' W	Humber/upper estuary	
26		53° 3.65' N	0° 28.88' E	Wash	17
27	12	52° 50.4' N	2° 50.1' E	E.Smith's Knoll	18
28		51° 59.97' N	2° 19.95' E	Gabbard	25
29		51° 30.34' N	0° 57.88' E	Thames Barrow	19
30		50° 56.1' N	1° 17.05' E	E.English Channel	69
31	13	50° 27.5' N	0° 0.2' W	E.English Channel	
32		50° 24.3' N	0° 17.6' W	E.English Channel	
33		50° 20.6' N	0° 36.5' W	E.English Channel	
34		50° 17.2' N	0° 56.8' W	E.English Channel	
35		50° 14.25' N	1° 16.69' W	Central English Channel	
36		50° 11.37' N	1° 33.49' W	Central English Channel	
37		50° 10.62' N	1° 48.77' W	Central English Channel	
38		50° 10.09' N	2° 1.02' W	Central English Channel	
39		50° 6.88' N	2° 12.16' W	Central English Channel	
40		50° 2.42' N	2° 26.29' W	Central English Channel	
41		49° 59.42' N	2° 43.29' W	Central English Channel	
42		49° 56.11' N	3° 1.81' W	W.English Channel	
43		49° 50.5' N	3° 19.1' W	W.English Channel	
44		49° 45.7' N	3° 35.6' W	W.English Channel	
45	14	48° 29.2' N	8° 1.29' W	S.W. Approaches	
46	15	51° 18.87' N	3° 33.14' W	Bristol Channel	
47		51° 29.72' N	3° 54.89' W	Bristol Channel	
48		51° 36.56' N	4° 25.77' W	Bristol Channel	
49		51° 38.7' N	5° 7.6' W	Bristol Channel	
50	16	52° 10.13' N	4° 44.7' W	Cardigan Bay	
51		52° 13.92' N	4° 21.48' W	Cardigan Bay	
52		52° 25.8' N	4° 16.56' W	Cardigan Bay	
53		52° 40.9' N	4° 25.9' W	Cardigan Bay	
54	17	53° 32.7' N	3° 15.3' W	Mersey	
55		53° 26.4' N	3° 0.7' W	Mersey	
56		53° 28.0' N	3° 2.9' W	Mersey	
57		53° 30.5' N	3° 5.6' W	Mersey	
58		53° 31.9' N	3° 9.3' W	Mersey	
59		53° 30.2' N	3° 21.04' W	Mersey	
60		53° 30.09' N	3° 29.04' W	Liverpool Bay	
61		53° 41.78' N	3° 29.96' W	Ribble	
62	18	53° 41.99' N	3° 10.43' W	Ribble	
63		53° 51.9' N	3° 25.16' W	Morecambe Bay	
64		53° 55.6' N	3° 12.4' W	Morecambe Bay	
65	19	53° 59.37' N	3° 2.19' W	Morecambe Bay	
66		54° 1.1' N	2° 56.26' W	Morecambe Bay	
67		53° 58.6' N	3° 0.0' W	Morecambe Bay	
68		53° 55.68' N	3° 0.2' W	Morecambe Bay	
69		54° 16.34' N	3° 42.97' W	E. Isle of Man	

Table 8. Water column data for stations sampled during CIROLANA 6, 5-27 June 1990

Station	Depth (m)	Salinity (psu)	Concentrations (μ mol l^{-1})				Temperature (°C)
			Phosphate	Silicate	Nitrate	Nitrite	
1	15	33.7 ¶	0.17	1.4	< 0.1	-	11.3
2	-	13.9	1.5	18	52	0.8	-
3	-	5.1	1.8	20	60	1.1	-
4	74	-	0.09	1.1	< 0.1	-	-
5	12	34.2 ¶	0.19	1.5	0.4		10.4
6	-	32.2	2.2	6.8	4.8	0.7	-
7	-	29.5	2.6	13	14	2.0	-
8	-	25.6	3.0	22	25	3.7	-
9	-	18.8	3.8	30	40	4.7	-
10	74	-	-	-	-	-	-
11	13	33.7 ¶	0.82	1.5	3.4	-	10.7
12	-	31.4	4.8	4.4	22	-	-
13	-	22.8	17	12	83	-	-
14	56	34.13 (2 m)	0.07	0.7	< 0.1	-	11.3
"	"	34.15 (40 m)	0.37	2.2	1.2	-	8.1
15	12	33.9 ¶	0.47	1.3	13	1.6	11.8
16	-	33.1	1.9	3.1	48	7.5	-
17	-	31	4.8	5.9	61	11	-
18	-	29.7	5.9	7.4	65	14	-
19	29	34.7	< 0.01	1.3	< 0.1	-	12.3
20	29	35.0	< 0.01	1.1	< 0.1	-	12.6
21	80	34.8	< 0.01	1.5	< 0.1	-	11.7
22	15	34.3	< 0.01	0.6	< 0.1	-	10.9
23	18	30.3 ¶	0.95	7.5	76	0.3	12.8
24	-	26.8	0.06	17	150	4.5	-
25	-	26.3	0.03	18	160	3.3	-
26	37	34.0	< 0.01	0.6	0.2	-	13.2
27	35	34.6	0.08	1.0	0.4	-	12.2
28	49	34.7	0.25	0.6	3.2	0.2	-
29	16	34.2	0.3	1.0	6.8	0.2	14.5
30	30	33.7	< 0.01	0.7	0.4	-	13.1
31	57	-	-	-	-	-	-
32	45	-	-	-	-	-	-
33	49	-	-	-	-	-	-
34	62	-	-	-	-	-	-
35	52	-	-	-	-	-	-
36	57	-	-	-	-	-	-
37	59	-	-	-	-	-	-
38	60	-	-	-	-	-	12.8
39	60	-	-	-	-	-	-
40	65	-	-	-	-	-	-
41	66	-	-	-	-	-	-
42	65	35.1	< 0.01	1.2	1.2	-	13.3
43	69	-	-	-	-	-	-
44	73	35.3	< 0.01	0.4	< 0.1	-	14.1
45	174	35.56 (2 m)	< 0.01	< 0.1	< 0.1	-	15.1
"	"	35.58 (34 m)	0.01	0.2	< 0.01	-	13.1
"	"	35.59 (94 m)	0.02	0.2	0.5	-	11.6
"	"	35.58 (169 m)	0.3	2.7	6.4	0.25	11.5
46	58	31.3	1.3	5.9	40	0.5	14.3
47	24	32.7	0.1	0.4	6.2	-	-
48	24	33.7	0.3	0.6	1.3	-	-
49	36	34.5	< 0.01	0.8	0.7	-	12.6
50	38	34.4	< 0.01	1.9	< 0.1	-	13.4
51	20	34.3	0.09	2.6	< 0.1	-	14.8
52	20	34.2	0.04	2.3	0.1	-	15.5
53	18	34.3	0.1	1.2	0.3	-	14.1
54	14	31.7 ¶	1.6	2.4	68	1.7	15.6
55	-	29.0	5.5	11	120	12	15.9
56	-	29.2	5.2	11	110	12	-
57	-	30.1	4.3	8.0	86	8.7	-
58	-	30.7	3.6	6.5	69	1.1	-
59	17	31.8	1.4	3.4	40	1.1	-
60	27	32.7	5.5	1.6	6.8	0.2	-
61	40	32.7	1.7	0.8	2.0	-	-
62	16	31.9 ¶	1.3	1.7	13	-	15.1
63	19	32.6	0.5	0.6	< 0.1	-	15.9
64	22	32.7	0.2	0.8	< 0.1	-	14.5
65	20	32.2 ¶	1.0	0.8	< 0.1	-	14.7
66	41	32.1	1.1	1.6	0.3	-	-
67	-	32.1	1.0	1.4	0.2	-	-
68	-	30.5	1.6	0.7	1.3	-	-
69	41	33.1	0.6	0.8	0.1	-	13.7

¶ *Results for salinity and nutrients are for samples taken at or near low water*

7. ALPHA- and GAMMA-HEXACHLOROCYCLOHEXANE

The insecticide lindane (γ-HCH) is included in the UK DOE's 'Red List' of dangerous substances where input to water is to be controlled (Anon., 1990). Alpha-HCH is a product of photolytic (and possibly microbial) degradation of γ-HCH, and is also a constituent of technical hexachlorocyclohexane (t-HCH); t-HCH has not been used in the UK for several years, but lindane is still commonly used in agriculture in England and Wales. An estimated 5.5 tonnes were used on grassland and fodder crops in 1989 (Davis *et al.*, 1991), 5.3 tonnes were used as seed treatments in 1988 and, also in 1988, a further 16.5 tonnes were used on arable farm crops (Davis *et al.*, 1990). Timber preservation is a further, important use for γ-HCH. Although precise data are not currently available, it is thought that this latter use peaked in 1987-88 and is declining rapidly as professional users of timber treatment products switch to alternative biocides (Health and Safety Executive, personal communication).

In December 1989, a survey was conducted of the concentrations of HCH in surface waters around England and Wales, following the loss of *MV PERINTIS* in the English Channel with a cargo which included 5.8 tonnes of lindane (MAFF, 1991(a)).

The concentrations of α- and γ-HCH found in the 1990 survey are given in Table 9. Alpha-HCH was not detected in samples from the North Sea (detection limit 0.1 ng l^{-1}), and was generally found at low concentrations (< 0.1 to 0.38 ng l^{-1}) elsewhere. The highest concentration observed was 1.3 ng l^{-1} at Station 58 in the River Mersey. Gamma-HCH was detected at the majority of sites sampled (39 of 56) at concentrations up to 3.7 ng l^{-1}, the highest of which was again found at Station 58. Concentrations of 1 ng l^{-1} or greater were measured in samples from Stations 8 (River Tyne), 58 and 59 (River Mersey), and 67 and 68 (Morecambe Bay). The range of concentrations seen in the earlier (1989) survey was from < 0.1 to 1.3 ng l^{-1}, the highest value occurring in one of the samples from the Humber estuary (MAFF, 1991(a)). All of the concentrations found were below the safe level of 10 ng l^{-1} set in the UK and, if further samples taken during a *CIROLANA* cruise in June 1991 yield similar concentrations, it is unlikely that further monitoring for these compounds will be conducted on any basis other than a local basis close to estuarine inputs.

Table 9. Hexachlorocyclohexanes (HCHs) and total hydrocarbon concentrations (THCs) in sea water. (Station details are given in Table 7)

Station	Alpha-HCH (ng l^{-1})	Gamma-HCH (ng l^{-1})	THC in water # (μg l^{-1} Ekofisk crude oil equivalents)
1	<0.1	<0.1	0.5
2	<0.1	0.19	1.8
3	<0.1	0.25	1.9
4	-	-	0.3
5	<0.1	<0.1	0.7
6	<0.1	0.34	16
7	<0.1	0.83	17
8	<0.1	1	18
9	<0.1	0.41	22
11	<0.1	<0.1	1.1
12	<0.1	0.1	7.2
13	<0.1	0.22	16
14	-	-	2
15	<0.1	<0.1	5.4
16	<0.1	<0.1	55
17	<0.1	<0.1	35
18	<0.1	<0.1	51
19	-	-	1.9
20	-	-	0.7
21	-	-	0.4
22	-	-	0.6
23a	<0.1	<0.1	4.9
23b	<0.1	<0.1	-
24	<0.1	<0.1	18
25	<0.1	<0.1	10
26	<0.1	<0.1	1.2
27	<0.1	<0.1	0.4
28	-	-	0.3
29	<0.1	<0.1	1.2
30	0.14	0.55	0.4
31	<0.1	0.56	-
32	0.1	0.49	-
33	0.1	0.53	-
34	<0.1	0.25	-
35	0.12	0.49	-
36	0.13	0.53	-
37	0.14	0.43	-
38	0.13	0.42	0.3
39	0.12	0.33	-
40	0.14	0.38	-
42	-	-	<0.3
44	<0.1	0.08	<0.3
45	-	-	<0.3
46	0.11	0.54	2.8
47	0.18	0.49	1.2
48	0.12	0.36	0.3
49	0.13	0.3	0.3
50	<0.1	<0.1	< 0.3
51	<0.1	<0.1	0.3
52	<0.1	<0.1	<0.3
53	-	-	<0.3
54a	<0.1	0.11	2.1
54b	<0.1	0.13	-
55	0.25	0.54	12
56	0.18	0.48	10
57	0.14	0.32	8.4
58	1.3	3.7	7.3
59	<0.1	1.7	2.1
60	0.24	0.69	1.1
61	-	-	0.7
62	0.23	0.77	1.6
63	0.25	0.62	0.8
64	0.22	0.67	1.1
65	-	-	1
66	0.26	0.86	2.9
67	0.38	1.1	7.1
68	0.15	1.1	6.6
69	0.23	0.47	0.4

Samples marked a and b are replicates
\# *Mean blank value = 0.13*
3 x sd of blank = 0.22
Limit of detection = 0.3

8. HYDROCARBONS

A simple measurement of the concentration of total hydrocarbons present in the water samples taken in the 1990 survey was made using fluorescence spectrometry (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 (see Table 9).

Low total hydrocarbon concentrations (THC's) were recorded offshore and in coastal areas away from estuaries. They were at or below the limit of detection (0.3 µg l^{-1} Ekofisk crude oil equivalents) in the western English Channel and Western Approaches, Carmarthen Bay and Cardigan Bay, and at two offshore stations in the North Sea. In the River Tweed, the highest THC was only 1.9 µg l^{-1} (Station 3) although the sample was taken at a salinity of 5.1. In other estuaries, the maximum THC's were higher: 22 µg l^{-1} in the Tyne (Station 9), 16 µg l^{-1} in the Wear (Station 13), 55 µg l^{-1} in the Tees (Station 16), 18 µg l^{-1} in the Humber (Station 24), and 12 µg l^{-1} in the Mersey (Station 55). These concentrations are of the same order as those reported previously (Law, 1981) and in no case would visible oil contamination be apparent.

9. PRELIMINARY RESULTS OF MEASUREMENT OF SURFACTANTS IN UK WATERS

Although vast amounts of anionic, cationic and non-ionic surfactants are released into estuarine and marine systems as a result of domestic and industrial use, there is little information available concerning concentrations of these compounds in nearshore waters. In order to augment the few data available for UK waters, the Burnham-on-Crouch Laboratory has carried out determinations of concentrations of nonylphenols in areas perceived to have the highest inputs (i.e. in estuaries and at sewage-sludge disposal grounds). The studies form part of a wider collaborative programme undertaken with scientists employed by companies producing anionic surfactants. The joint study is not yet complete and this report provides only preliminary data based purely on the surveys carried out by MAFF.

One clearly defined route for surfactants to enter the marine environment is via sewage. Giger *et al.* (1984) have shown that nonlyphenols arise from the breakdown of alkylphenol polyethoxylates in anaerobically digested sewage sludges. Ten sludge samples and seven effluents from sewage treatment works in the UK were analysed in 1984-85 for the presence of nonylphenols. Concentrations of nonylphenols in effluents were generally low (<2 to 21 ng l^{-1}), but were present at higher levels in sludges derived from primary treatment of sewage (0.3 mg kg^{-1} dry weight) and anaerobically digested sewage (up to 4 mg kg^{-1} dry weight). Dispersion of the nonylphenols was then determined following sludge disposal and compared with the dispersion of coprostanol, a faecal sterol commonly used as a marker for the presence of sewage in environmental samples. The results of the survey were submitted to ICES (Waldock and Thain, 1986).

Dispersion rates, determined by the experiment, showed that concentrations of surfactants would decrease rapidly below acute toxic thresholds. It therefore seems unlikely that nonylphenols will pose a threat to marine species. In the example of the most heavily contaminated sewage (4 mg kg^{-1} nonylphenol) deposited into the outer Thames estuary, calculated maximum environmental concentrations would be less than 10 µg l^{-1} within one hour of discharge.

In addition to following disposal events, concentrations of nonylphenols have been measured on two occasions in the estuary of the River Mersey and in Liverpool Bay. The Mersey has been regarded historically as an impacted riverine system and receives inputs from a variety of industrial processes as well as raw and primarily treated sewage effluents. Samples of water were collected during two cruises on MAFF research vessels in 1989 and 1990, and concentrations of nonylphenols, coprostanol, and a variety of other potential organic contaminants were measured. Figure 12 shows an example of one transect from the outer Mersey to a sewage-sludge disposal area in Liverpool Bay (1990 cruise). The concentrations of nonylphenols and coprostanol measured in 1989 and 1990 are shown in Table 10; Figure 13 presents the 1990 cruise data in pictorial form.

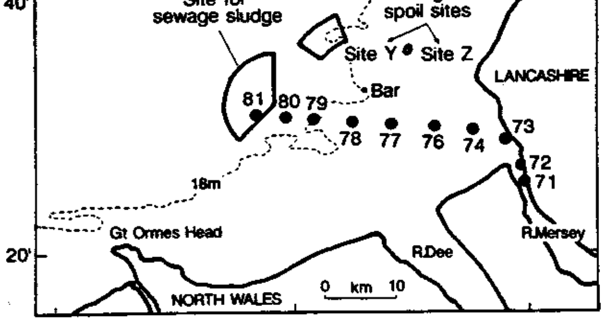

Figure 12. Stations sampled in Liverpool Bay in 1990

Table 10. Concentrations of nonylphenols and coprostanol (ng l^{-1}) in water samples taken from the River Mersey and Liverpool Bay

	Station no.	81	80	79	78	77	76	74	73	72	71
1989	Nonylphenol	-	-	-	-	-	<25	<25	-	<25	314
	Coprostanol	-	-	-	-	-	2	5	-	13	100
1990	Nonylphenol	<50	<50	<50	<50	<50	<50	<50	67	97	160
	Coprostanol	<5	<5	<5	<5	<5	<5	10	81	105	438

It is clear from the data obtained, that the levels of nonylphenols in the water samples were closely related to the concentration of coprostanol, confirming that the source of inputs was sewage discharges to the River Mersey. Further from sources of sewage input, the concentrations attenuated to values below the detection limit. Despite semi-continuous inputs of sewage sludge to the Liverpool Bay disposal ground, no increase in concentrations of either coprostanol or nonylphenols was measured close to the point of disposal activity. Data for concentrations of nitrate and phosphate are also depicted in Figure 13, and further emphasise the extent of the influence of contaminated Mersey water to the waters of Liverpool Bay.

Since the study area represents a worst case scenario, it appears unlikely that measurable concentrations of nonylphenols would be encountered in the open sea.

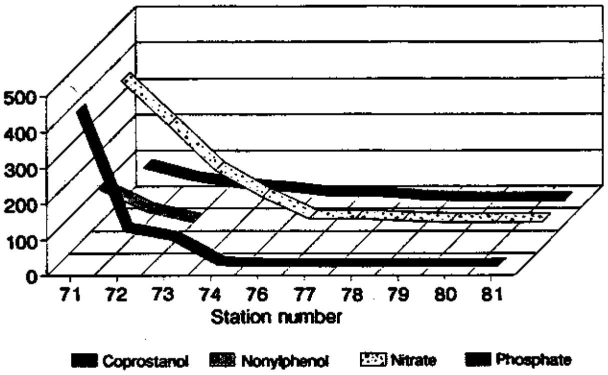

Figure 13. Concentrations of coprostanol (ng l^{-1}), nonylphenols (ng l^{-1}), nitrate (µ mol l^{-1} x 5), and phosphate (µ mol l^{-1} x 5) found in samples from Liverpool Bay in 1990

SEDIMENTS

10. BASELINE SEDIMENT SURVEY

As a contribution to an OSPARCOM Joint Monitoring Group's (JMG) sediment baseline programme and to the 1993/94 North Sea Quality Status Report, the Burnham-on-Crouch Laboratory is conducting a joint survey with Cambridge University of sediment quality around England and Wales. The samples were collected from MAFF vessels and charter vessels using Day or Shipek grabs depending on the substrate type and sampling was undertaken at the station positions shown in Figure 14. Sample collection is now complete and, in the first phase of sediment analysis, concentrations of metals have been measured in a subset of the samples collected from the North Sea and English Channel.

The full set of data is not yet available, as analysis is still incomplete. However, as an example of the results which are becoming available, Figure 15 shows the distribution of the concentrations of total zinc in the <2 mm sediment fraction. Higher concentrations found near the coast can be attributed to the generally higher clay mineral content of sediments in this area, as demonstrated by Figure 16 which shows that the concentration of zinc is correlated with that of lithium, a surrogate for clay mineral content in this type of sediment. The lithium/zinc correlation not only points to the association of zinc and clays but also provides a method for normalising the concentration of metal to eliminate the effects of sediment bulk composition and thereby reveal any anthropogenic influences.

Table 11 shows the strength of correlation of each pair of determinands measured so far, together with the

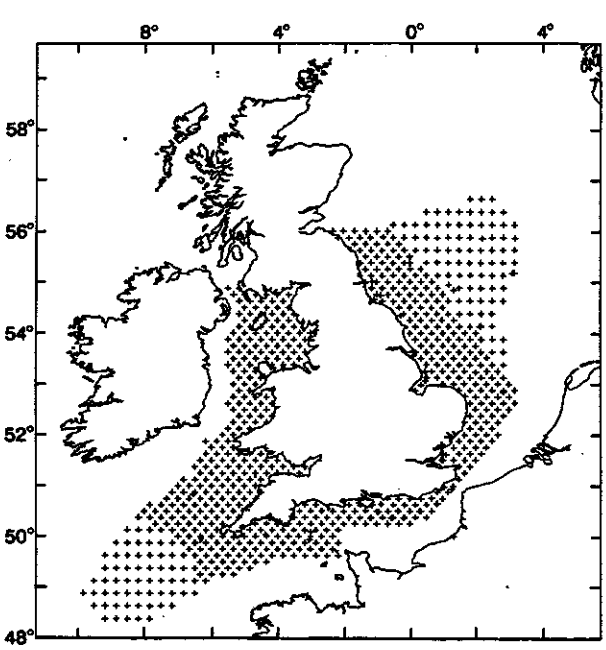

Figure 14. Sediment baseline sampling stations

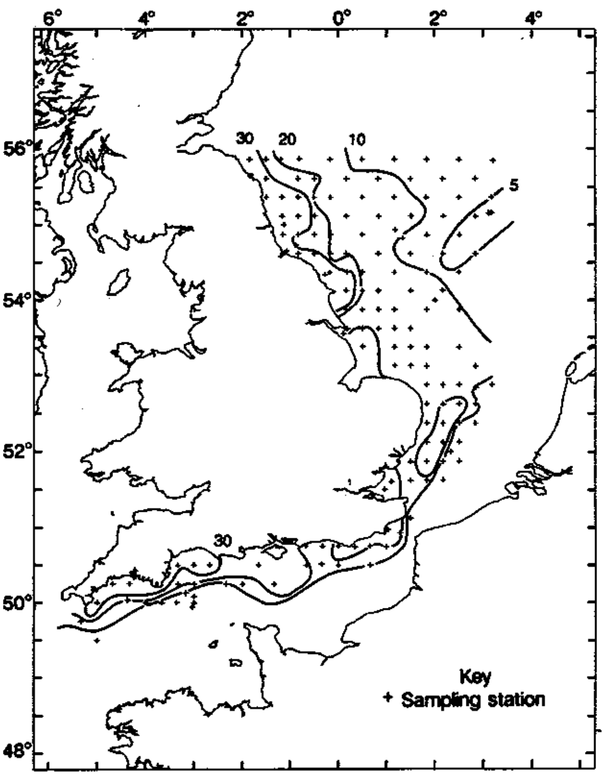

Figure 15. Distribution of total zinc (mg kg^{-1}) in the <2mm fraction of sediment throughout the North Sea and English Channel

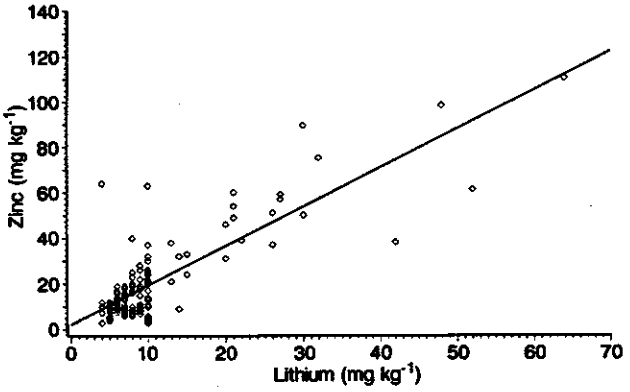

Figure 16. Correlation of zinc and lithium in the <2mm fraction of sediment collected from the North Sea and English Channel

Table 11. Correlation matrix of determinands in the <2 mm fraction of sediments collected from the North Sea and English Channel

Al	Cd	Cr	Cu	Fe	Hg	Li	Pb	Zn	
1.00000 0.0 166	-0.03955 0.6816 110	0.76070 0.0001 143	0.40384 0.0005 70	0.44818 0.0001 165	0.57902 0.0001 146	0.61098 0.0001 164	0.42667 0.0001 166	0.66490 0.0001 166	Al
	1.00000 0.0 110	0.11356 0.2681 97	0.36128 0.0042 61	-0.06572 0.4972 109	-0.08682 0.3929 99	0.04307 0.6566 109	-0.01800 0.8519 110	0.01248 0.8971 110	Cd
		1.00000 0.0 143	0.39263 0.0017 61	0.53156 0.0001 142	0.52705 0.0001 127	0.51321 0.0001 141	0.41825 0.0001 143	0.76704 0.0001 143	Cr
			1.00000 0.0 70	0.29774 0.0130 69	0.23911 0.00551 65	0.41647 0.0003 70	0.23211 0.0532 70	0.49274 0.0001 70	Cu
				1.00000 0.0 165	0.26671 0.0012 145	0.45588 0.0001 163	0.58177 0.0001 165	0.76754 0.0001 165	Fe
					1.00000 0.0 146	0.3424 0.0001 145	0.27459 0.0008 146	0.50009 0.0001 146	Hg
						1.00000 0.0 164	0.39686 0.0001 164	0.62721 0.0001 164	Li
							1.00000 0.0 166	0.55806 0.0001 166	Pb
								1.00000 0.0 166	Zn

Pearson product moment correlation coefficient
Probability >/R/under Ho:Rho = O (low values indicate strong positive correlation)
Number of observations

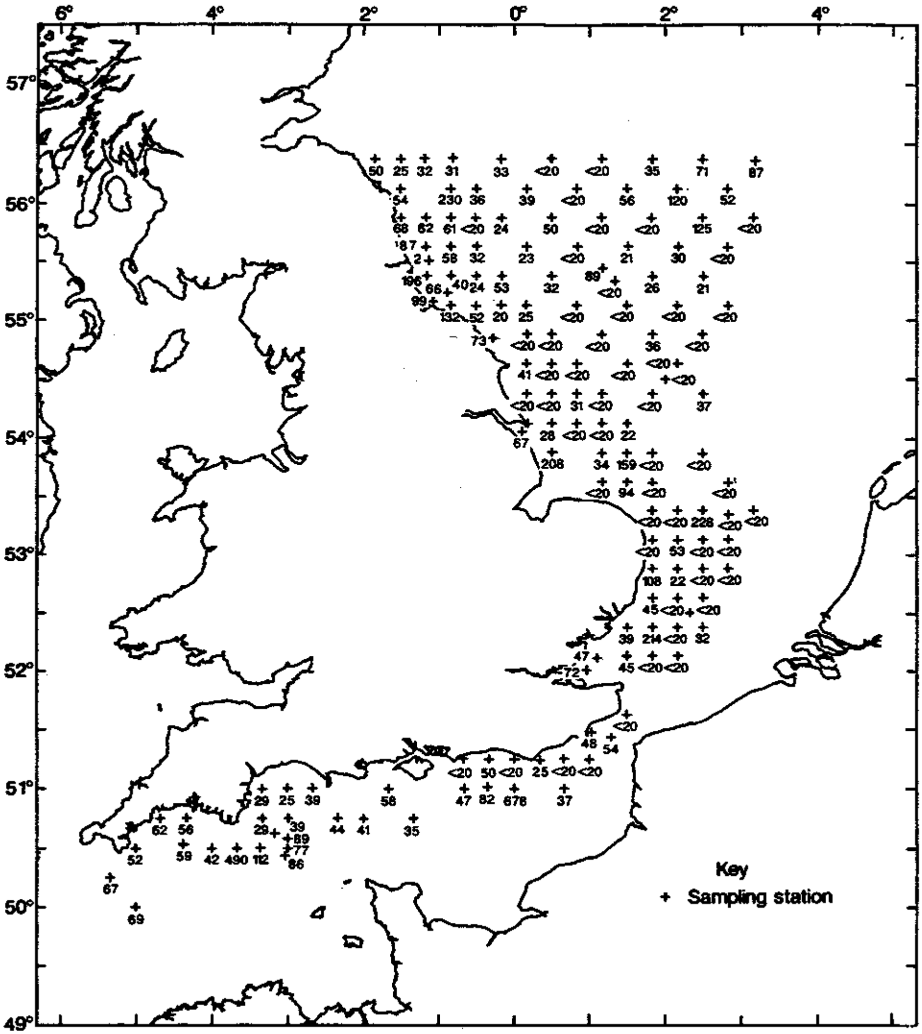

Figure 17. Concentrations of total cadmium (µg kg^{-1}) in the <2mm fraction of sediment throughout the North Sea and English Channel

probability of that correlation, given the null hypothesis of the correlation being 0, and the number of samples used to calculate the value. There is a strong positive correlation between most trace metals and lithium, although cadmium is a clear exception. The distribution of this metal is shown in Figure 17 and, while no firm conclusions can be drawn at this stage (occasional relatively high values were found in many of the areas surveyed), there appear to be regions of relatively high concentrations off the north-east coast of England and in the Western Channel.

Work is proceeding on the measurement of other determinands (e.g. organic carbon and grain size) which will further assist in explaining the distribution of metals.

11. TOTAL HYDROCARBONS IN SEDIMENTS

Samples of sediment were taken during *CIROLANA* 6, 5-27 June 1990 and *CORYSTES* 11, 12-22 October 1990. Station positions are listed in Tables 12 and 13 respectively. The sediment descriptions are based on inspection of the samples at the time of collection. Samples were analysed for hydrocarbons using the fluorescence spectrometry technique described previously (Law *et al.*, 1988). Total hydrocarbon concentrations (THCs) were determined at excitation and emission wavelengths of 310 and 360 nm respectively, and all concentrations are expressed as µg g^{-1} Ekofisk crude oil equivalents. Synchronous excitation/emission spectra ($\Delta\lambda = 25$ nm) were run for each sample. The extracts were also analysed by capillary gas chromatography following clean-up on alumina.

Table 12. Stations sampled during CIROLANA 6, 5-27 June 1990 and total hydrocarbon concentrations in surface sediments ($\mu g\ g^{-1}$ dry mass Ekofisk crude oil equivalents)

Station	Date (June)	Position		Location	NSTFMMP no.	Sediment type	THC in sediment
1	6	55° 45.7' N	1° 57.85' W	Tweed/mouth		Sand	5.9
2		55° 45.83' N	1° 59.58' W	Tweed/lower estuary		Sand	1.7
3		55° 46.12' N	2° 0.35' W	Tweed/upper estuary		Sand	14
4		55° 0.5' N	1° 7.6' W	Tyne/intermediate	14		No sample
5	7	55° 0.41' N	1° 23.76' W	Tyne/mouth		Fine sand	53
6		55° 0.63' N	1° 24.98' W	Tyne/lower estuary		Sand and gravel	95
7		54° 59.17' N	1° 27.86' W	Tyne/middle estuary		Mud	750
8		54° 58.92' N	1° 31.82' W	Tyne/middle estuary		Mud	640
9		54° 58.09' N	1° 36.25' W	Tyne/upper estuary		Mud	560
10		55° 0.46' N	1° 7.76' W	Tyne/intermediate	14	Mud	190
11	8	54° 54.88' N	1° 20.3' W	Wear/mouth		Stones and sand	490
12		54° 55.04' N	1° 21.62' W	Wear/lower estuary		Sand	61
13		54° 54.58' N	1° 22.87' W	Wear/upper estuary		Sand and stones	310
14		54° 44.04' N	0° 52.84' W	Tees/intermediate	15	Sand and mud	96
15	9	54° 38.93' N	1° 7.35' W	Tees/mouth		Sand	69
16		54° 38.52' N	1° 8.66' W	Tees/lower estuary		Sand	13
17		54° 37.12' N	1° 9.32' W	Tees/middle estuary		Mud	570
18		54° 36.22' N	1° 9.9' W	Tees/upper estuary			No sample
19		54° 50.1' N	1° 20.24' E	Offshore Tyne/Tees	43	Fine sand	5.7
20	10	55° 10.1' N	3° 6.3' E	Dogger Bank	47	Fine sand and shell	1.3
21		53° 59.94' N	2° 0.21' E	Silver Pit	53	Fine sand	16
22		53° 32.41' N	0° 20.12' E	Humber/intermediate	16		No sample
23	11	53° 33.33' N	0° 6.37' E	Humber/mouth		Fine sand	110
24		53° 37.1' N	0° 2.82' W	Humber/lower estuary		Mud and sand	160
25		53° 36.0' N	0° 3.0' W	Humber/upper estuary		Mud	340
26		53° 3.65' N	0° 28.88' E	Wash	17	Stones, sand and mud	58
27	12	52° 50.4' N	2° 50.1' E	E.Smith's Knoll	18	Sand and shell	1.3
28		51° 59.97' N	2° 19.95' E	Gabbard	25	Medium sand	0.4
29		51° 30.34' N	0° 57.88' E	Thames Barrow	19	Mud	62
30		50° 56.1' N	1° 17.05' E	E.English Channel	69	Shell and sand	5
31	13	50° 27.5' N	0° 0.2' W	E.English Channel			No sample
32		50° 24.3' N	0° 17.6' W	E.English Channel			No sample
33		50° 20.6' N	0° 36.5' W	E.English Channel			No sample
34		50° 17.2' N	0° 56.8' W	E.English Channel			No sample
35		50° 14.25' N	1° 16.69' W	Central English Channel			No sample
36		50° 11.37' N	1° 33.49' W	Central English Channel			No sample
37		50° 10.62' N	1° 48.77' W	Central English Channel			No sample
38		50° 10.09' N	2° 1.02' W	Central English Channel			No sample
39		50° 6.88' N	2° 12.16' W	Central English Channel			No sample
40		50° 2.42' N	2° 26.29' W	Central English Channel			No sample
41		49° 59.42' N	2° 43.29' W	Central English Channel			No sample
42		49° 56.11' N	3° 1.81' W	W.English Channel		Coarse sand, stones and shell	0.9
43		49° 50.5' N	3° 19.1' W	W.English Channel			No sample
44		49° 45.7' N	3° 35.6' W	W.English Channel		Sand, stones and shell	4.9
45	14	48° 29.2' N	8° 1.29' W	S.W. Approaches		Sand and shell	0.7
46	15	51° 18.87' N	3° 33.14' W	Bristol Channel			No sample
47		51° 29.72' N	3° 54.89' W	Bristol Channel			No sample
48		51° 36.56' N	4° 25.77' W	Bristol Channel		Sand and mud	7.3
49		51° 38.7' N	5° 7.6' W	Bristol Channel		Sand and shell	2.6
50	16	52° 10.13' N	4° 44.7' W	Cardigan Bay		Sand and shell	8.6
51		52° 13.92' N	4° 21.48' W	Cardigan Bay		Mud and stones	20
52		52° 25.8' N	4° 16.56' W	Cardigan Bay			No sample
53		52° 40.9' N	4° 25.9' W	Cardigan Bay		Shingle	3.3
54	17	53° 32.7' N	3° 15.3' W	Mersey		Fine sand	37
55		53° 26.4' N	3° 0.7' W	Mersey		Mud	240
56		53° 28.0' N	3° 2.9' W	Mersey		Mud	140
57		53° 30.5' N	3° 5.6' W	Mersey		Sand	1.1
58		53° 31.9' N	3° 9.3' W	Mersey			No sample
59		53° 30.2' N	3° 21.04' W	Mersey		Sand and shell	35
60		53° 30.09' N	3° 29.04' W	Liverpool bay		Sand	92
61		53° 41.78' N	3° 29.96' W	Ribble		Mud and sand	61
62	18	53° 41.99' N	3° 10.43' W	Ribble		Sand and mud ¶	60
63		53° 51.9' N	3° 25.16' W	Morecambe Bay		Mud and sand	50
64		53° 55.6' N	3° 12.4' W	Morecambe Bay			No sample
65	19	53° 59.37' N	3° 2.19' W	Morecambe Bay		Fine sand	4.7
66		54° 1.1' N	2° 56.26' W	Morecambe Bay		Sand	4
67		53° 58.6' N	3° 0.0' W	Morecambe Bay		Sand	3.5
68		53° 55.68' N	3° 0.2' W	Morecambe Bay		Mud	82
69		54° 16.34' N	3° 42.97' W	E. Isle of Man		Mud	82

¶ Contained Spisula elliptica (ca. 2500 m^{-2})

Table 13. Total hydrocarbon concentrations in sub-surface sea water and surface sediments (µg l^{-1} and µg g^{-1} dry mass (respectively) Ekofisk crude oil equivalents). Stations sampled during CORYSTES 11,12-22 October 1990

Station	Date	Position	Location	NSTF no.	THC in water	Sediment type	THC in sediment
5	13	50° 55.69' N 1° 16.08' E	Dover Strait	69	0.5	Sand	1.7
24	14	50° 38.25' N 0° 52.01' W	Isle of Wight	70	1	Shell and sand	1.9
36	16	50° 4.98' N 2° 59.8' W	Central Channel	72	0.3	No sample	No sample
46	-	50° 1.96' N 4° 22.03' W	MBA station E1	73	2.6	Sand	4.5
49	-	50° 19.52' N 4° 16.98' W	Off Rame Head	-	No sample	Sand and mud	88
50	-	50° 18.8' N 4° 15.8' W	Off Rame Head	-	No sample	Mud and sand	220
52	-	50° 18.8' N 4° 15.3' W	Off Rame Head	-	No sample	Sand and mud	370
53	-	50° 18.07' N 4° 14.45' W	Off Rame Head	-	No sample	Sand and mud	98

THCs ranged from 0.4 to 750 µg g^{-1}, with the higher concentrations generally being found within industrialised estuaries (Tables 12 and 13). In the River Tweed, a non-industrial river draining a mainly agricultural area, THCs were low, at 1.7 to 14 µg g^{-1} (*CIROLANA* 6, 1990; Stations 1 to 3). In the industrialised estuaries the ranges were: Tyne, 53 to 750 µg g^{-1} (Stations 5 to 9); Wear, 61 to 490 µg g^{-1} (Stations 11 to 13); Tees, 13 to 570 µg g^{-1} (Stations 15 to 18); Humber, 110 to 340 µg g^{-1} (Stations 23 to 25); Mersey, 1.1 to 240 µg g^{-1} (Stations 54 to 59). The one sample collected in the Thames (Station 29) had a THC of 62 µg g^{-1}, and in Morecambe Bay THCs ranged from 3.5 to 82 µg g^{-1} (Stations 65 to 68). In almost all cases, the GC traces indicate the presence of heavily weathered and/or biodegraded oil, but at Station 11 (just outside the River Wear) fresher material was observed, with an *n*-alkane range similar to that of diesel oil, and a prominent pristane peak. As has been found previously, (Law, 1981; Law and Fileman, 1985; Fileman and Law, 1988) away from estuarine areas, THCs were generally much lower, but one offshore mud sample (10 miles off the River Tyne; Station 10) yielded a THC of 190 µg g^{-1}. High THCs, from 88 to 370 µg g^{-1}, were also found in four samples taken from the vicinity of the Rame Head disposal site, which receives dredged material from Devonport dockyard and other areas adjacent to Plymouth Sound and the River Tamar (*CORYSTES* 11, 1990, Stations 49 to 53; Table 13).

These concentrations should be seen against the recently adopted OSPARCOM 'No Observed Effect Concentration' (NOEC) for oil in sediment around offshore oil and gas installations of 10 µg g^{-1} dry mass or 2-3 times background levels. It is hoped to carry out analysis of some of these samples for specific polycyclic aromatic hydrocarbons (PAH) in the future as the composition of the PAH found can give information on the source of the hydrocarbon (e.g. pyrolysis products versus offshore oil inputs). There is also interest in the environmental concentrations of some high molecular PAH's which are known to be carcinogenic.

The preliminary account of the use of the oyster embryo bioassay (Section 3) indicated that some toxicological effects were occurring at a number of the estuarine stations found to have high concentrations of THCs. However, it should be borne in mind that estuarine sediments are contaminated with a wide range of compounds, both organic and inorganic, and further work would be required to determine which of these were responsible for the effects seen by the oyster embryo bioassay.

GENERAL STUDIES

12. TBT AND THE MARINE ENVIRONMENT

12.1 Small estuaries and marinas

Results from the joint MAFF/DOE monitoring programme for the toxin tributyltin (TBT) used in antifouling paints have been summarised in previous reports in this series (MAFF, 1990(a), 1991(a)). The main purpose of early surveys was to follow changes in the concentrations of TBT in water, bivalves and sediment after legislation was passed in 1986 and 1987 to control the use of TBT on small boats (i.e. those less than 25 m). By 1989, concentrations of TBT in small estuaries had decreased markedly with a concomitant increase in bivalve growth performance. There were further reductions in concentrations of TBT in most of the water, sediment and bivalve samples collected in 1990 from the Crouch, Blackwater, Dart, Kingsbridge and Teign estuaries (Tables 14-16). There was also further improvement in oyster growth, both in terms of meat production and shell-shape (Tables 17-18). Concentrations of TBT in the water of two marinas in Plymouth and Dartmouth decreased considerably in 1990, although this was not the case for the other marinas/harbours sampled during that year (Table 14). The concentration of TBT in water from Teignmouth Harbour has remained in the 20 to 25 ng l^{-1} range since 1986. This harbour is used mainly by ships as opposed to small boats.

Table 14. Mean summer (May to September) concentrations (ng l^{-1}) of tributyltin in UK estuaries, marinas and harbours, 1986-1990

Estuary/location	Shellfish site	1986	1987	1988	1989	1990
Crouch	Fambridge	15±8	33±27	21±8	13	7±5
	Bridgemarsh	22±12	17±12	13	8	4
	Creeksea	35±17	17±9	22±14	8±2	4±0.4
	Burnham	45±17	31±18	23±18	11±4	7±6
	Bush Shore	26±9	22±15	13±5	8	5
	Roach Mouth	26±12	18±13	15±12	8±2	3
	Holliwell Buoy	11	26±23	10±4	3±2	3±1
	Holliwell Point	16	6±5	6±5	2	1±1
Blackwater	West Mersea	38±21	36±29	76±43	25	24±17
Dart	Blackness Point	38±33	13±4	13±5	8±3	9±10
Kingsbridge	Frogmore	15±6	11±6	51±95[b]	5±2	5±6
Teign	Arch Brook	12±10	7±6	6±2	6±4	5±3
	Marinas/harbours					
Plymouth	Sutton marina	1156±84	882±323	274±79	266±134	133±79
Dartmouth	Dart marina	95±66	85±32	21±4	16±6	6±8
Kingsbridge	Salcombe	117±84	62±71	30±18	21±15	16±11
Teign	Teignmouth	22±24	23±25	19±12	25±15	24±16
Beaulieu River	Bucklers Hard*	93±45	1087±1845[a]	82±9	25±7	25±17
Southampton Water	Hythe marina			1956±2472[c]	93±62	103±135

** 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

Table 15. Summer concentrations (µg g^{-1} dry weight) of tributyltin in sediments, 1986-1990

Estuary	Site	1986	1987	1988	1989	1990
Crouch	Fambridge	0.08	0.03	0.02	0.10	0.03
	Bridgemarsh	0.15	0.04	0.08	0.05	0.01
	Creeksea	0.04	0.11[a]	0.07	0.04	0.07
	Burnham	0.36	0.15[b]	0.31	0.10	0.04
	Bush Shore	0.07	0.05	0.27	0.07	0.02
	Roach Mouth	0.05	0.02	0.02	0.04	0.01
	Holliwell Buoy	<0.01	0.01	0.02	0.02	0.01
	Holliwell Point	0.01	0.01	0.05	<0.01	0.01
Blackwater	West Mersea	0.66	0.26	0.15	0.53[a]	0.07
Dart	Blackness Point	0.08	0.19	0.17	0.07	0.05
Kingsbridge	Frogmore	0.06[b]	0.03	0.07	0.04	0.02
Teign	Arch Brook	0.05[b]	0.02	0.03	0.02	0.01
Beaulieu	Bucklers Hard	4.56	10.83	1.11	0.23	2.70

All values are for August samples except: [a] July; [b] September

Table 16. Mean summer (June to August) concentrations (µg g^{-1} wet weight) of tributyltin in Crassostrea gigas, 1986-1990

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

NS = not sampled

Table 17. Meat weight values (g wet weight) for Crassostrea gigas, August 1986-1990

Estuary	Site	1986	1987	1988	1989	1990
Crouch	Fambridge	1.29	0.70	2.59	5.66	6.53
	Bridgemarsh	1.26	0.67	2.62	2.84	3.85
	Creeksea	0.97	0.84	2.23	2.90	3.63
	Burnham	0.77	0.49	2.39	2.77	3.88
	Bush Shore	1.24	1.24	2.44	NS	4.47
	Roach Mouth	1.18	1.14	2.50	5.92	NS
	Holliwell Buoy	1.39	1.44	NS	5.96	4.04[a]
	Holliwell Point	2.72	2.06	4.11	9.28	5.55
Blackwater	West Mersea	1.00	0.97	2.53	3.65	6.10
Dart	Blackness Point	1.56	0.89	3.19	7.09	12.4
Kingsbridge	Frogmore	1.77	1.32	4.53	8.07	10.4
Teign	Arch Brook	1.56	1.24	5.22	6.23	8.79
Beaulieu	Bucklers Hard	0.37	0.12	0.95	1.24	2.40

NS - not sampled

[a] *July sample*

Table 18. Shell thickness index values* for Crassostrea gigas, August 1986-1990

Estuary	Site	1986	1987	1988	1989	1990
Crouch	Fambridge	4.4	4.9	6.9	9.8	18.4
	Bridgemarsh	4.8	4.7	8.1	10.2	24.6
	Creeksea	5.1	5.3	8.1	9.8	21.9
	Burnham	5.0	5.0	7.9	10.2	15.7
	Bush Shore	5.1	5.2	9.8	NS	32.0
	Roach Mouth	5.4	5.3	9.0	8.6	NS
	Holliwell Buoy	9.6	6.5	NS	19.8	36.4[a]
	Holliwell Point	12.4	9.6	23.3	21.0	44.9
Blackwater	West Mersea	5.0	4.0	6.9	14.9	17.7
Dart	Blackness Point	10.6	6.0	12.5	14.7	29.4
Kingsbridge	Frogmore	7.6	5.4	9.3	13.1	36.3
Teign	Arch Brook	10.6	12.4	25.7	18.9	44.9
Beaulieu	Bucklers Hard	3.2	8.1	4.3	6.7	17.7

NS - not sampled

[a] *July sample*

* *Shell thickness index =* $\dfrac{\textit{Length of upper shell valve}}{\textit{Thickness of upper shell valve}}$

12.2 Inputs of TBT from shipping activity

In 1990, the emphasis of the TBT monitoring programme was changed to permit a more complete assessment of the impact of continued use of TBT on large ships. An area of particular concern was the discharge of TBT from dry docks where ships are cleaned-off and re-coated with antifouling paint. Since September 1990, water samples have been taken from Falmouth docks on a weekly basis (see Figure 18 for location of sampling site). For completeness, the available data for 1991 are also included in this report. Concentrations of TBT in sub-surface waters (Figure 19) have varied considerably over the sampling period, ranging from 11 ng l^{-1} to 558 ng l^{-1}. In May 1991, a detailed survey of Falmouth docks and the open water of the Fal estuary was carried out (Figure 18). The concentration of TBT within the dockyard area was 150 ng l^{-1}. In the Penryn River only approximately 100 m from the docks, the levels of TBT were much lower,

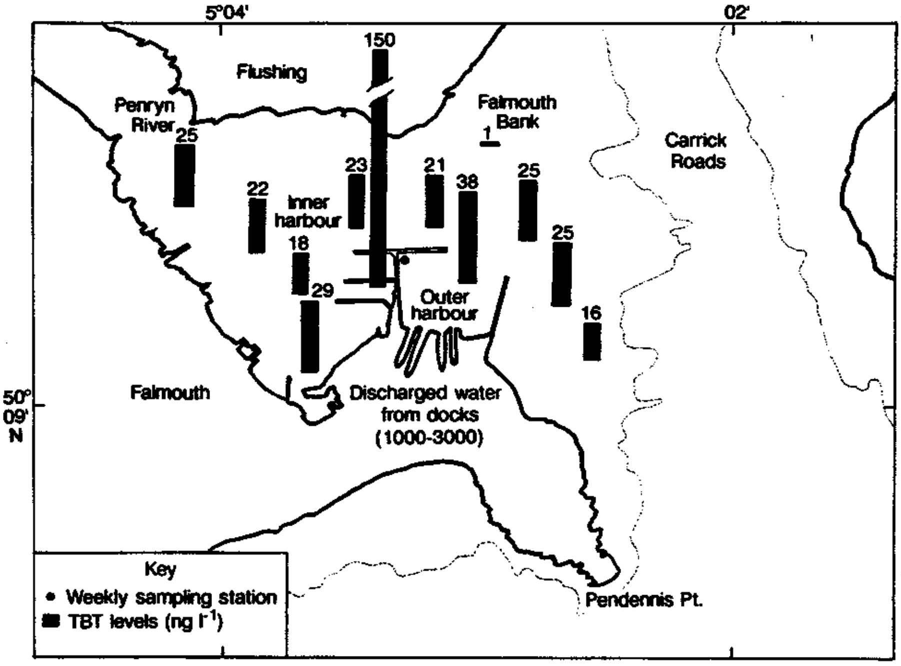

Figure 18. Concentrations of TBT in surface water samples from the Falmouth dock area

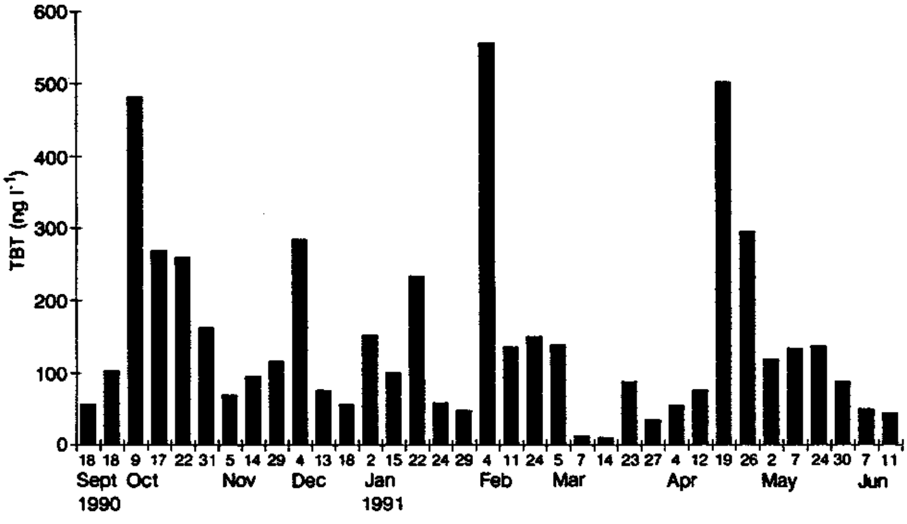

Figure 19. Concentrations of TBT in surface water samples from the outer harbour of Falmouth docks (see Figure 18)

being in the range 20 to 30 ng l^{-1}. Surprisingly, however, even at the mouth of the estuary, concentrations were still greater than 15 ng l^{-1} although, near the northern bank, where the Penryn River meets the Carrick Roads, the concentration was 1 ng l^{-1}. This suggests that there was a discrete plume of contaminated water from the Penryn River extending into the cleaner water of the Carrick Roads.

TBT-contaminated wash water from dry-docking operations at Falmouth docks is collected in a holding tank which periodically discharges into the estuary. Data supplied by the National Rivers Authority (NRA) Southwest Region indicate that the concentration of TBT in the discharge is generally in the range 1000 to 3000 ng l^{-1}. The survey of concentrations of TBT in the receiving water, close to the point of discharge, indicates that the discharge is diluted by a factor of approximately 10 in sub-surface waters once it enters the estuary.

Sediment samples taken from the Eastern Wharf at Falmouth, near to the docks, had extremely high concentrations of TBT (up to 27 $\mu g\ g^{-1}$ dry weight), indicating the presence of paint chippings in the sediment (Table 19). Similarly high concentrations have also been recorded in sediment samples from Lowestoft and Portsmouth docks.

Table 19. Concentrations of organotin compounds (ug g^{-1} dry weight) in dredged sediment taken from Falmouth eastern wharf, on 2 August 1991

Monobutyltin	Dibutyltin	Tributyltin
0.01	0.04	0.44
0.07	1.68	27.0
0.02	0.02	0.46
0.05	0.56	9.48
0.09	2.03	22.4

In 1987, the DOE published a set of guidelines for boatyards and others involved in antifouling of large vessels (Department of the Environment, 1987(a)). These give advice on the storage, application and removal of antifoulants (with particular emphasis on TBT-based paints), and waste disposal. Waste, (e.g. cleaned-off marine growths, paint tins, used shot-blasting grit) must be cleared up and taken to a site licensed for disposal. The Ministry of Defence operates its own voluntary code of practice for the application and removal of TBT-based paints. This has proved to be very effective in minimising the discharge of TBT from its dry docking facilities. Discussions are presently taking place with the DOE and the NRA regarding the control of discharges of TBT-contaminated effluents from dry docks.

13. THE ROSEMAUND PESTICIDE 'RUN-OFF' STUDY

13.1 Introduction

One of the functions of the Burnham-on-Crouch Fisheries Laboratory under the Food and Environment Protection Act, 1985 (Great Britain - Parliament, 1985(a)) is to provide advice on the aquatic environmental aspects of pesticides to MAFF Pesticides Safety Division, to the Health and Safety Executive and to the Advisory Committee on Pesticides. This applies both to new notifications and to older pesticides which are subject to review. The two most important aspects of this advice concern assessment of toxicity and evaluation of exposure, leading to predictions of hazard. With exposure, the process of evaluating likely concentrations necessitates prediction of the concentrations of pesticides in surface fresh waters and sediment. This is a particularly difficult procedure and is one for which the fundamental science is still developing. Current procedures involve calculation of a reasonable 'worst-case' concentration based on an accidental overspray of a shallow, static waterbody at the maximum recommended field application rate, but it is clear that this approach gives no information about the long-term exposures which might result, for example, from leaching and 'run-off'.

The most promising new method for predicting leaching is to construct computer models of the processes which are thought to control the transport of pesticides, and a large number of such models have been developed as research tools. One of them, the Mackay fugacity model, looks particularly attractive because it only requires the relatively simple physico-chemical data that are available before a pesticide is released into the environment. Unfortunately, however, neither the Mackay model, nor other more complex models, have received proper validation with reliable field data, particularly for agro-ecosystems in the United Kingdom. At present, therefore, they cannot be used operationally, and aquatic assessments of pesticides continue to have to err substantially on the safe side by basing hazard predictions on calculation of overspray, as mentioned above.

A research project to remedy this situation was therefore set up in 1987, in collaboration with the Building Research Establishment (BRE) and the Agricultural Development and Advisory Service (ADAS). Subsequently, other organisations have become involved, notably the Institute of Hydrology (IoH), the Welsh Region of the National Rivers Authority (NRA), Essex and Birmingham Universities, and the Soil Survey and Land Resource Centre (SSLRC). The primary aim of the project (from the point of view of the MAFF Directorate of Fisheries Research) was to generate reliable field data on the leaching and 'run-off' of pesticides from a controlled catchment, in order that validated models of aquatic exposure can be brought into operational use (Bird *et al.*, 1991). The data from the field experiments are presented in this report for the first time.

13.2 Methods

It was clear from the beginning, that a study of this type had to be conducted on a surface-water dominated catchment (i.e. having little loss to groundwater) in order to maximise the concentrations appearing in surface streams and, furthermore, that inputs of pesticides would have to be strictly controlled and recorded. An almost ideal site in both respects was found at the ADAS Rosemaund Experimental Husbandry Farm (EHF) near Hereford. This encloses an entire water catchment of about 180 ha which slopes gradually (1:40) to a small stream that eventually drains into the River Lugg. The shallow underlying geology consists of essentially impermeable Devonian siltstones and mudstones at 1-3 m depth, which are overlain by silty clay loams and some alluvial gley soils. These soils are generally low in organic carbon content (≤ 1.7%) and, although the subsoil is impermeable, it is subject to cracking and is underlain by field drains at 1 m depth (mean spacing = 20 m). All of these factors tend to maximise the transport of pesticides to the stream, thus providing an almost ideal setting for generation of reasonable worst-case leaching and 'run-off' data.

The main crops grown at Rosemaund consist of winter cereals, forage grass and hops, with some oilseed rape, beans/peas, roots and maize. All monitoring experiments to date have involved operational pesticide applications at rates consistent with good agricultural practice. Full details of these experiments are given by Bird *et al.* (1991). In essence, the experiments have monitored concentrations of pesticides in soil (0-1 m depth), soil water (0.5, 1.0 and 1.5 m depth), field drain water and stream water. Some samples have also been taken of crops, soil fauna, stream fauna and stream sediment, but full data are not yet available from these analyses. The water sampling in drains and stream has mainly relied on the use of automatic samplers triggered by the increased water flow-rates associated with rainstorms. Soil has been obtained by coring, and soil water has been obtained by *in situ* suction samplers. In addition to the storm-triggered samples, water and soil samples have been taken manually at intervals between storms.

The experiments have monitored both autumn and spring application of pesticides, and to date have focused largely on those products whose physico-chemical properties, and/or history of appearance in UK surface waters, suggest that leaching is likely. The majority of pesticides monitored have therefore been the water soluble herbicides (triazines, ureas and phenoxy acids), but organophosphorus and organochlorine insecticides have also been included. The long-term aim is to study a range of chemical structures which is as broad as possible, including those not expected to leach, but an over-riding constraint is the availability of suitably sensitive analytical methods. To date, these methods have been confined to gas-liquid chromatography (Bird *et al.*, 1991).

13.3 Results

The detailed results of the experiments so far, have been reported already by Bird *et al.* (1991), Brooke and Matthiessen (1991), and Williams *et al.* (1991). A further paper is in preparation by Matthiessen *et al.* A summary of the maximum concentrations found in drain and stream water is shown in Table 20, together with information on application rate, crop, type, and the delays between spraying and the first 'run-off' event and between the onset of rainfall and the start of the event. With the exception of triclopyr, it can be seen that all pesticides were detected intermittently in drains or stream at concentrations in excess of 0.5 µg l^{-1}, the maximum peak value being 46.8 µg l^{-1} for MCPA in a field drain. Without exception, these peak values all followed rainfall events in excess of 10 mm rain in 24 h, and the highest values usually occurred if significant rain fell within a week of spraying. The apparent discrepancy in dates and peak concentrations between the stream and the drains is partly explained by the fact that not all of the drains were continuously monitored; it may also indicate a certain amount of overland flow during the heaviest rain storms. In a typical rainfall event of ≥ 10 mm, stream flow increased sharply within 4-8 hours of the onset of rainfall, and peak concentrations of pesticide were generally associated with peak water flows. An important point to note, however, is that peak concentrations were never maintained for more than 6 h, and concentrations generally returned to background levels within 12-24 h. The transience of these events, combined with the relatively low concentrations involved, mean that the total quantities of pesticide mobilised from the experimental fields into the stream were always small (< 10 g for any individual pesticide in any season). The overwhelming majority of pesticide applied to the fields (> 99%) remained in the soil and was lost by degradation.

Table 20. Pesticide application rates in the Rosemaund experiments and peak concentrations found in drains and stream water

Pesticide	Application rate (kg ha^{-1}) and date	Area sprayed (ha) and crop	Peak concentrations (μg l^{-1}) and date	Delay from spraying to first pesticide event (days)	Minimum delay from rainfall to start of event (hours)
Mecoprop	2.0 17 Nov. 87	10.8 Barley	(D)> 1.8 20 Nov. 87 (S) 11.7 19 Nov. 87	2	<8
Dicamba	0.4 20 Nov. 87	5.2 Grass	(D)<0.1 (S)0.7 18 Dec. 87	28	<48
2,4-D	1.0 20 Nov. 87	5.2 Grass	(D)>1.3 18 Dec. 87 (S)1.0 18 Dec. 87	28	<48
Simazine	1.1 8 Dec. 88	5.4 Beans	(D)1.4 27 Feb. 89 (S)11.0 1 Feb. 89	27	<15
Triclopyr	0.2 8 Dec. 88	5.2 Grass	(D)<0.01 - (S)<0.01 -	-	-
Isoproturon	1.0 1 Nov. 89 0.4 17 Nov. 89	5.2 + 10.8 Wheat	(D)13.7 10 Nov. 89 (S)5.4 13 Dec. 89	2	<9
Lindane	0.5 1 Nov. 89	5.2 Wheat	(D)4.4 8 Nov. 89 (S)0.3 14 Dec. 89	8	<5
Mecoprop	0.6 20-22 Mar. 90	10.8 Wheat	(D)0.2 3 Apr. 90 (S)1.4 16 May 90	14	<4
Dichlorprop	2.6 20-22 Mar. 90	10.8 Wheat	(D)0.2 3 Apr. 90 (S)2.2 3 Apr. 90	12	<4
Isoproturon	2.1 23 Nov. 90	6.0 Wheat	(D)26.2 29 Nov. 90 (S)17.2 25 Dec. 90	6	<6
Dimethoate	0.3 28 Nov. 90	16.0 Wheat	(D)2.8 14 Dec. 90 (S)3.0 25 Dec. 90	1	<5
Oxydemeton-methyl	0.1 28 Feb. 91	15.0 Wheat	(D) - (S) 0.76 4 Mar. 91	4	10
MCPA	1.7 28 Feb. 91	15.0 Wheat	(D) 46.8 19 Mar. 91 (S) 12.7 16 Mar. 91	4	2

(D) = drain water
(S) = stream water

Concentrations found in whole soil are of little relevance in an aquatic environment hazard assessment context and will not be reported here, but concentrations in soil water are of interest. In the case of the isoproturon application in November 1989 (Table 21), residues at the level of the field drains (1 m), not surprisingly, reached concentrations similar to those found in drain outflows (up to 19.8 µg l^{-1}). However, concentrations below the drains at 1.5 m peaked at 54.2 µg l^{-1}, indicating that not all pesticide was directly intercepted by the drains. This sub-drain reservoir may subsequently be available for horizontal transport by seepage into the stream, but most probably degrades *in situ*. More importantly, the results in Table 21 indicate rapid transport of pesticide to drain level at speeds suggestive of so-called by-pass or 'crack flow', rather than percolation through micro-pores. In other words, some (perhaps most) of the pesticide reaching the stream does not have the chance to contact the majority of the soil particles and to reach equilibrium. Surface application of aqueous-borne dye tends to confirm that 'crack flow' transport can be highly significant. This has profound implications for modelling.

Table 21. Concentrations of isoproturon (µg l^{-1}) in soil water in one of the Rosemaund experiments, by date

Depth (m)	15 Nov. 89	29 Nov. 89	18 Dec. 89	20 Mar. 90
0.5	3.8-17.2	2.0-9.0	0.4-16.4	0.8*
1.0	0.1-17.8	0.3-19.8	0.3-9.3	0.1-2.7
1.5	0.8-53.8	1.1-54.2	1.0-49.0	0.4-5.5

* *Only one sample*

Modelling to date has concentrated on a development of the Mackay Level 2 fugacity model, a fundamental assumption of which is that the chemical is allowed to reach equilibrium with the soil, water, sediment and air phases (Brooke and Matthiessen, 1991; Williams *et al.*, 1991). Detailed modelling has been conducted on the mecoprop experiment of November 1987, the simazine experiment of December 1988, the isoproturon and lindane experiments of November 1989, and the mecoprop experiment of March 1990. Predicted concentrations in the stream all overestimate the measured concentrations by factors of between 5 and 500, if it is assumed that all of the water falling as rain reaches the stream immediately. However, for individual storms, the data show that only about 20% of the incident rain volume transfers rapidly to the stream. Re-running the model with only 20% of the chemical which leaves the soil block being allowed to enter the stream, produces predictions which are at worst no more than 100 times greater than the observations and at best match the observations closely.

13.4 Summary and conclusions

Concentrations of individual pesticides in the Rosemaund stream reached transient peaks of up to 17 µg l^{-1} immediately after rainstorms. For the pesticides in question, these aquatic exposure levels were almost certainly of no significance for aquatic life, and limited bioassay data from Rosemaund are available to support this view (Bird *et al.*, 1991). Although peak concentrations generally exceeded the European Communities' Drinking Water Directive (European Communities, 1975) maximum acceptable concentration (MAC) of 0.1 µg l^{-1} for individual pesticides, the nearest abstraction points for drinking water are situated over 35 km downstream. The Rosemaund discharge would have received dilution of over 3000 times by this point and human water consumers are very unlikely to be at risk, even though the dilution water would, itself, be contaminated to some extent. Given suitable rainfall, the pesticides tested were able to reach the stream very quickly after spraying (presumably by 'crack flow') and were therefore unlikely to be completely in equilibrium with the 3 soil compartments (particles, water, air) before transfer to the stream. This may mean that the Mackay fugacity model will have to be abandoned, although reasonable results (less than a factor of 10 between observations and predictions) have been obtained with it, in 3 out of the 5 cases examined in detail. Further field data are currently being obtained from Rosemaund and additional modelling, using the MacKay and other models, will be conducted when these data are available.

DISPOSAL AT SEA (DAS) GROUP: FIELD ASSESSMENT STUDIES

14. MONITORING OF SEWAGE-SLUDGE DISPOSAL AREAS

14.1 The integrated monitoring programme

Monitoring at UK sewage-sludge disposal sites is overseen by the Co-ordinating Group on the Monitoring of Sewage-Sludge Disposal Sites (CGMSD), a sub-group of the Marine Pollution Monitoring Management Group (MPMMG). In England and Wales, MAFF is the licensing authority for sludge disposal at sea. In addition to a core monitoring programme undertaken by MAFF, licensees conduct a complementary, 'self-monitoring' programme (Rowlatt *et al.*, 1991) designed and coordinated by the Burnham-on-Crouch Laboratory, which is also responsible for production of the annual CGMSD report on all aspects of UK sewage-sludge disposal site monitoring (see, for example, MAFF, 1991(b)). Not all sites are surveyed every year and the following sub-sections discuss only the work carried out by MAFF in 1990.

14.2 Tyne

14.2.1 General introduction

Sewage sludge has been deposited at the Tyne site (Figure 20) since 1978. The quantities were initially small, but rose to the present level of about 500 000 tonnes by 1984. Previous monitoring has shown little impact of the disposal operation on the chemical quality of local sediment in terms of contamination by heavy metals and a range of organochlorine compounds, although there has been a minor effect on benthos and some accumulation at the site of sewage-derived solids (e.g. litter and tomato pips) (Rowlatt *et al.*, 1991).

14.2.2 Annual survey for change in the benthos

The benthic macrofauna at the Tyne sewage-sludge disposal site was sampled along a transect of stations in 1984 and 1986, and subsequently has been sampled at annual intervals at two stations, (Stations 2 and 8, Figure 20) one of which was known to fall within the zone of settlement of sewage-sludge particulates. The rationale for this sampling design, along with sampling methodology, has been described in detail by Rees *et al.* (1985 and 1992). The present account extends the time-series to include data from sampling in 1990.

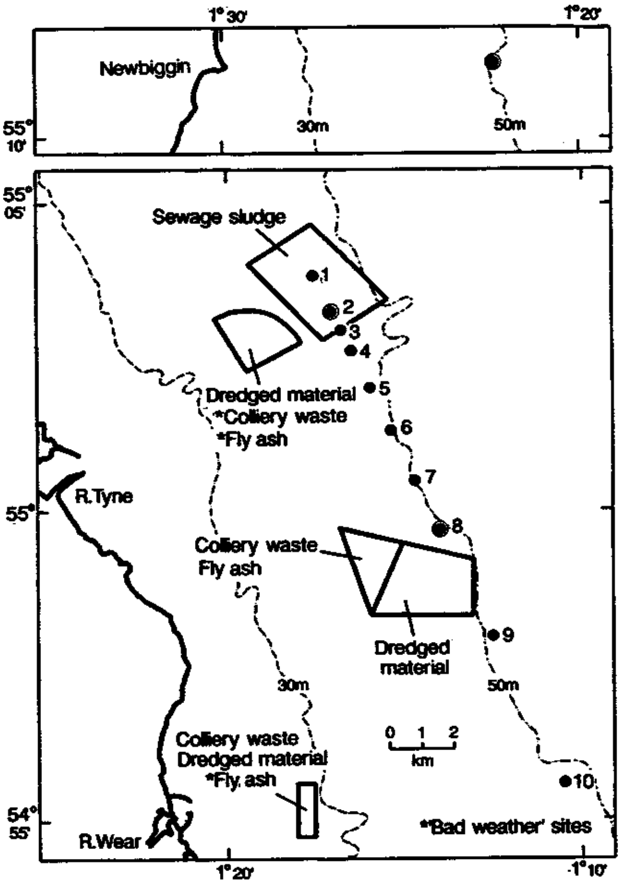

Figure 20. Benthic sampling stations, June 1986. Sites currently in use for sea disposal are specified. Double circles indicate that stations are sampled annually

It should be noted that, from 1986 onwards, sampling was conducted in May/June. Prior to 1986, sampling had been conducted in February/March.

14.2.3 Results

Carbon and nitrogen: Figure 21(a-b) shows concentrations of organic carbon and total nitrogen in sediments respectively at the two sampling stations, expressed as arithmetic means with 'Least Significant Intervals' (LSI's). In comparisons between years and between sites, intervals which do not overlap may be assumed to be significantly different at the 95% probability level (see Andrews *et al.*, 1980). Carbon and nitrogen values are variable between years, with highest values occurring in 1987 and 1988. Except in 1989, concentrations were significantly higher at the disposal site (Station 2). However, this was not necessarily due to the disposal of sewage sludge, as the concentrations of both C and N are affected by the presence of coal particles, which are a characteristic feature of sediments in this area.

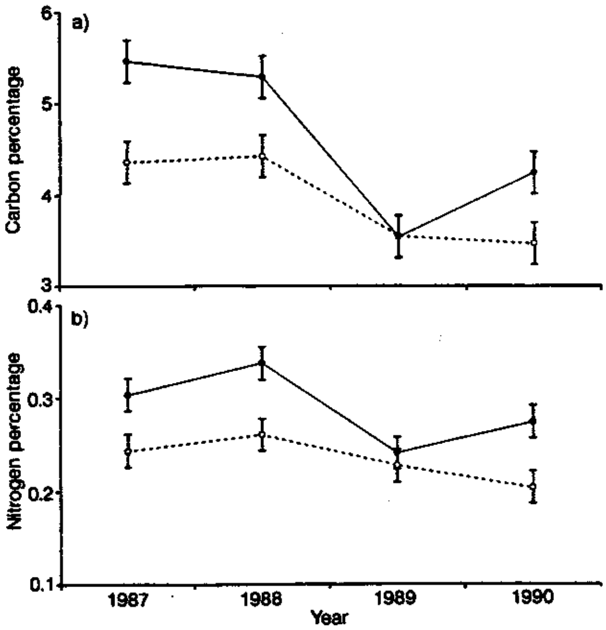

Figure 21. Trends in (a) carbon (%) and (b) nitrogen (%) off the Tyne: arithmetic means with 95% LSI's of three sub-samples from the <63 µm fraction of sediment at the disposal site (solid symbols) and the southern reference site (open symbols)

Particle size: In 1986, sediments at the disposal site (Station 2) were finer than those at the reference site (Station 8), with an average of 40% of particles in the silt/clay fraction at the former, compared with 18% at the latter.

However, in other years, sediments at the disposal site were marginally coarser (median diameter typically about 180 µm) and better sorted (typically about 0.5 ø) than at the reference site (median diameter: 160 µm; sorting: 0.7 ø).

Artefacts: Inspection of the residual sediment during the analysis of macrofauna has consistently revealed evidence of contamination by a wide range of sewage-derived artefacts at the disposal site, especially tissue paper, vegetable peel and plastic fragments. (Such items are occasionally recorded at the reference site, indicating that dispersal processes are operative in the area).

Tomato pips are a useful indicator of sewage contamination, and Figure 22 shows accumulations at the disposal site. Ranges astride each mean indicate appreciable variability in numbers between samples. Lower counts in the winter months (1984 and 1985) may be a natural consequence of a seasonal shift in human diet. However, the general trend since 1986 has been upward, with high counts being recorded in 1987 and 1988.

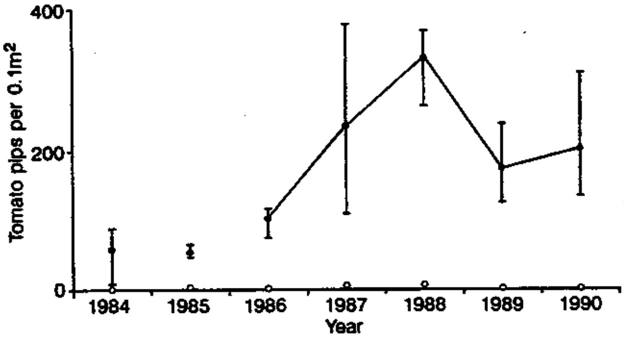

Figure 22. Trends in tomato pips off the Tyne: arithmetic means and ranges of three samples at the disposal site (solid symbols) and southern reference site (open symbols)

Benthic macrofauna: Annual trends in four univariate measures of benthic community structure are presented in Figure 23(a-d). Figure 23(a) is notable both for the occurrence of significantly higher numbers of individuals at the disposal site, and for an element of synchrony in the progress of changes at the two sites. The former is suggestive of mild enrichment at the disposal site (Rees *et al.*, 1992). The latter is considered to reflect equality in the influence of natural environmental factors. There is no evidence of any divergence in animal counts which might indicate a trend towards accumulation of organic matter at the disposal site.

Such synchrony is not evident in numbers of taxa (Figure 23(b)); significant divergency in 1989 would be consistent with mild enrichment, but this effect was not evident in 1990. Figure 23(c) and (d) shows that values of diversity and evenness are similar between stations, and that changes tend to be synchronous. Mean values for evenness are consistently lower at the disposal site, indicating a marginally higher proportional numerical dominance by one or two taxa; however, the effect is not statistically significant for most years.

To date, 135 taxa have been recorded from the disposal site location, and 116 taxa from the reference site location. Trends in numbers of individuals of the major taxa, along with biomass relationships, will be the subject of future reporting.

14.2.4 Conclusions

The sediments near the River Tyne contain relatively high concentrations of carbon (Rowlatt *et al.*, 1991) due to the presence of coal from both natural and industrial sources. Thus, although organic carbon values are generally higher at the disposal site than at the reference site, it is not possible to deduce a causal link with sewage-sludge deposition. Sediments at the

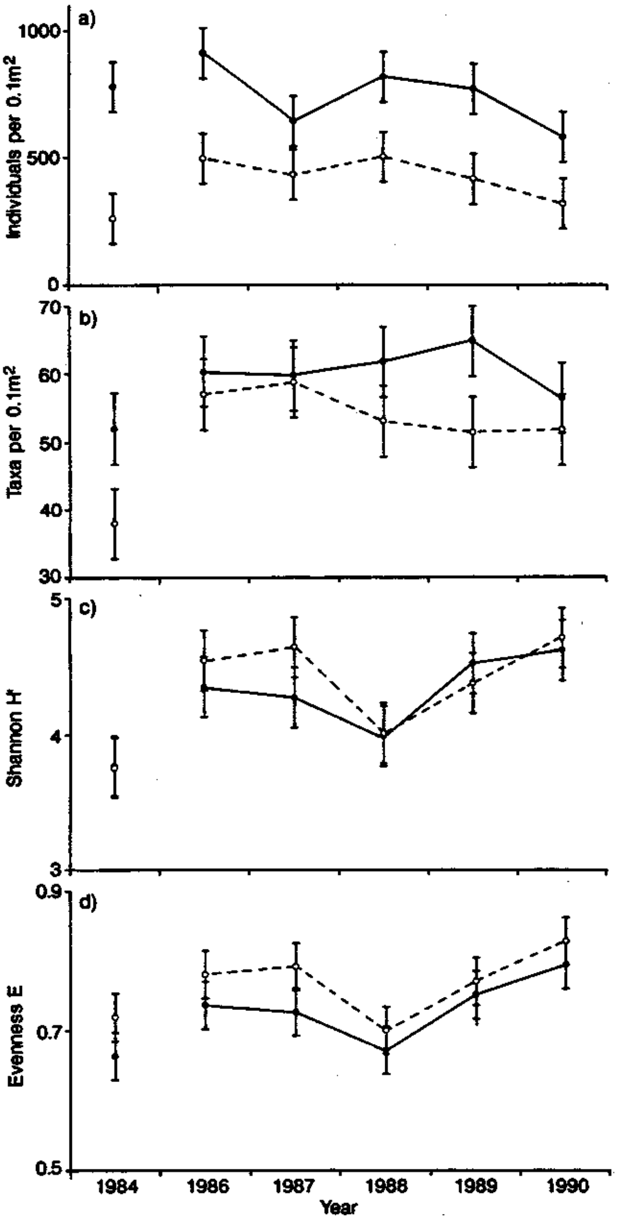

Figure 23. Trends in univariate measures of benthic community structure off the Tyne: arithmetic means with 95% LSI's for three replicates at the disposal site (solid circles) and southern reference site (open circles): (a) individuals; (b) taxa; (c) Shannon diversity; and (d) evenness

disposal site may, however, be clearly distinguished by the presence in quantity of a range of sewage-derived artefacts, with tomato pips providing a particularly useful indicator.

An underlying synchronicity in temporal trends for certain measures of benthic community structure at the two sites is considered to reflect the influence of similar, widely-operating, natural environmental factors.

Persistently elevated animal counts at the disposal site, along with marginal reductions in values of evenness, are consistent with earlier spatial studies and provide good circumstantial evidence of mild organic enrichment in the immediate vicinity of the disposal site (Rees *et al.*, 1992). These effects are minor, when compared, for example, with those arising at the Garroch Head sewage-sludge disposal site in Scotland, where quiescent conditions promote the build-up of organic matter, which in turn induces dramatic effects on the benthos (see, e.g. Pearson, 1987).

Furthermore, there is little evidence to indicate a worsening trend with time, suggesting that natural processes of dispersion are presently effective. However, since substantial quantities of sludge have only been disposed of at this location over a relatively short period (<10 years), monitoring at this site will continue, in order to ensure that the scale of the biological response, along with physical contamination, remains within acceptable bounds.

14.2.5 Transect study of sediment chemistry

In order to assess the impact of sludge disposal on sediment chemistry, samples were collected in May 1990 along a line extending from the mouth of the River Tyne, through the sewage-sludge disposal site to about 17 km offshore (Figure 24).

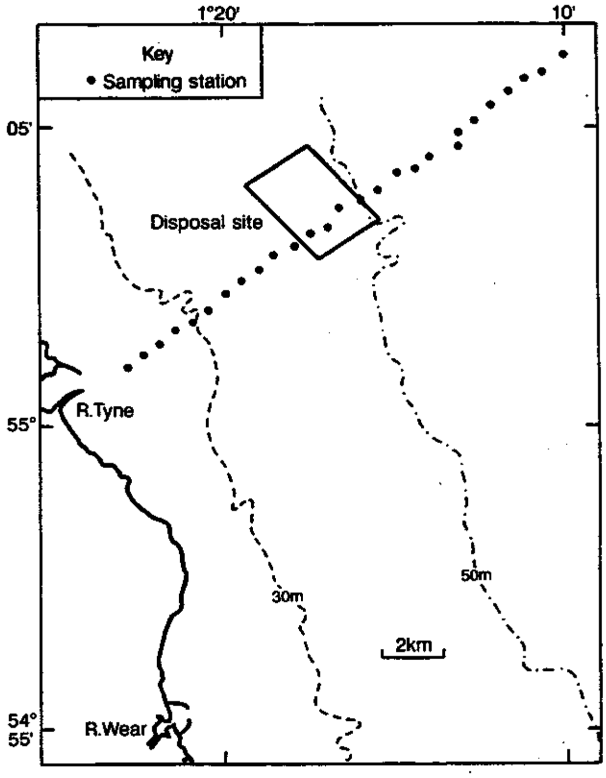

Figure 24. Sediment sampling stations off the River Tyne in May 1990

Faecal bacteria (*E. coli* and faecal streptococci) were determined in surface scrapes of sediment to detect any influence of sewage sludge. Concentrations of carbon and nitrogen were measured in the <2 mm and <63 µm fractions of the sediment by CHN analyser, after pre-treatment with SO_2 solution to remove carbonates. Metals (Al, Cd, Cu, Cr, Hg, Ni, Pb and Zn) were measured in hydrofluoric acid digests of the <2 mm fraction of the sediments and (all except Al) in *aqua regia* digests of the <2 mm and <63 µm fractions. Concentrations of trace metals in sediment extracts were determined by atomic absorption spectrophotometry (AAS).

Figure 25 shows the concentrations of metals in the fine (< 63 µm) sediment fraction along the transect, using copper as an example, together with the numbers of *E. coli* in the sediment. There is no evidence of elevated concentrations of copper (or other metals) in the region of the disposal site where the bacteria indicate that sludge settles. Thus, it may be concluded that there is no detectable accumulation of trace metals from sewage sludge at the sea bed.

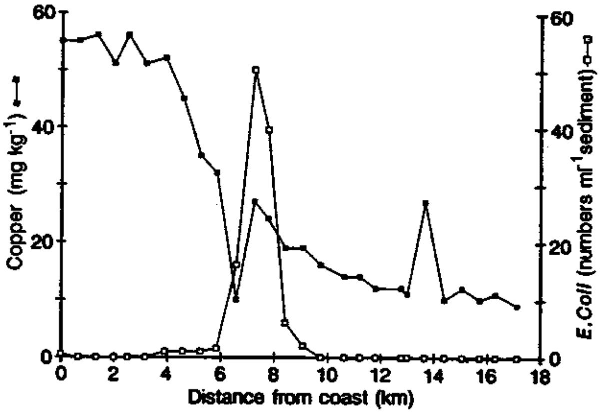

Figure 25. Concentrations of copper in the <63 µm fraction of sediment and numbers of E. coli in surface sediment along a transect off the River Tyne in May 1990

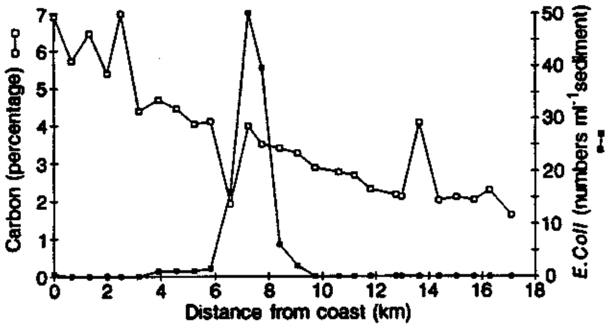

Figure 26. Concentrations of carbon in the <63 µm fraction of sediment and numbers of E. coli in surface sediment along a transect off the River Tyne in May 1990

A similar approach can be applied to interpretation of the data on concentrations of carbon in the sediment and, as Figure 26 shows, there is no evidence of any accumulation of organic carbon at the disposal site. It must be borne in mind when considering organic carbon data, that coal is generally present in these sediments due largely to erosion of coal measures and past industrial practices and that these may mask any subtle influence of sewage sludge on carbon content. One possible method to counteract this problem is to consider C/N ratios which may give a more sensitive indication of the presence of sludge (which has a higher concentration of nitrogen relative to carbon than coal). Figure 27 shows both the C/N ratio and *E. coli* counts and provides no indication of sewage-sludge disposal effects disrupting the general decreasing offshore trend in the C/N ratio.

It may be concluded from the above that sewage-sludge disposal off the Tyne has had no significant effects on sediment chemical quality.

The transect data also provide an insight into the chemical composition of the sediments. Figure 28 shows the concentration of copper in sediments separated into the following components:

(a) the total concentration measured after complete dissolution of the <2 mm sediment using hydrofluoric acid;

(b) the *aqua regia* soluble fraction of the <2 mm sediment which includes all copper other than that contained in lattice positions in the most resistant minerals; and

(c) the component of the copper in the *aqua regia* soluble portion of the fine (<63 µm) sediment fraction.

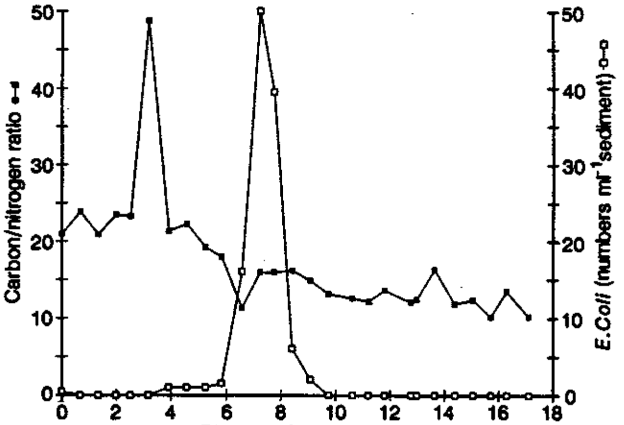

Figure 27. Carbon/nitrogen ratio in the <63 µm fraction of sediment and numbers of E. coli in surface sediment along a transect off the River Tyne in May 1990

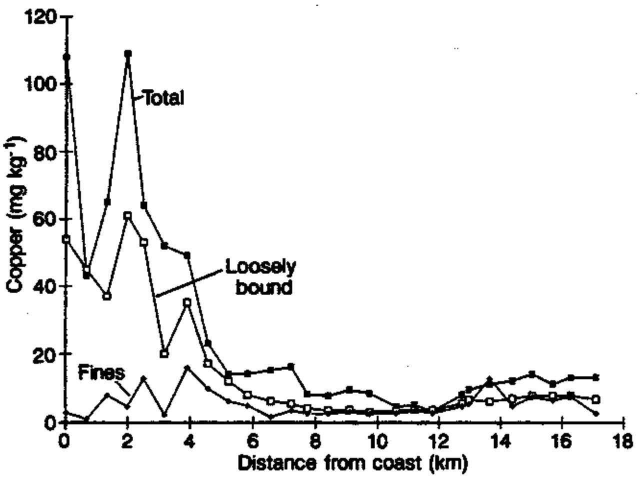

Figure 28. Concentrations of copper in various fractions of sediment along a transect off the River Tyne in May 1990: total = hydrofluoric acid soluble copper in whole sediment; loosely bound = aqua regia soluble copper in whole sediment; and fines = aqua regia soluble copper in the <63 µm fraction of sediment

There is a marked difference between nearshore and offshore sediments. The former contain higher concentrations of copper, most of which is in the coarse fraction with a significant proportion in the lattice structure of minerals. The latter contain most copper in the fine sediment fraction. Part of this difference may be explained by the generally finer nature of the offshore sediments. This, however, is not the full explanation and it is clear that the provenance of the sediments also has a marked influence, the nearshore material being recent erosion products which still contain a wide range of minerals, while the offshore sediments are composed of sands which, due to extended weathering, tend to have only low concentrations of metals. The offshore sediments are also more remote from anthropogenic influences than those near the Tyne.

Research is in progress to investigate the differences between the nearshore and offshore sediments; preliminary findings suggest that both natural erosion products and colliery waste have pronounced effects on sediment composition near the mouth of the Tyne.

14.3 Liverpool Bay

Each year, about 1.5 million wet tonnes of sewage sludge are deposited at a designated site in Liverpool Bay (Figure 29) by North West Water plc (formerly the North West Water Authority). The Burnham-on-Crouch Laboratory has studied the area since the 1970s to determine any effects on water, sediment or fish quality.

This work has included the analysis of sediment samples collected near the disposal site (Figure 29), where effects on sediment chemistry have been observed (Norton *et al.*, 1984 and unpublished data).

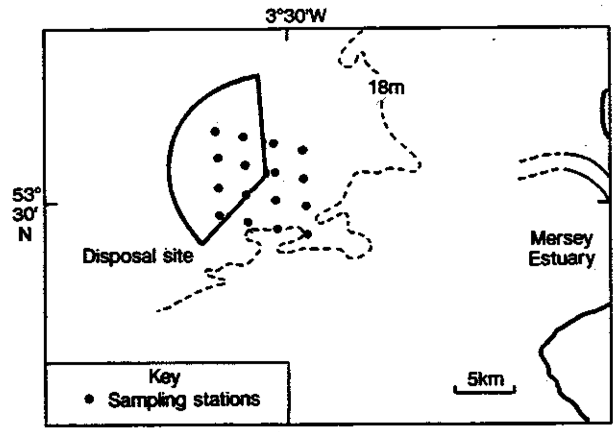

Figure 29. The sewage-sludge disposal site and sediment sampling grid in Liverpool Bay

The sediment samples were collected and analysed using the methods described by the CGMSD (MAFF, 1989) except that, for reasons of consistency with earlier work, the <90 µm sediment fraction was used rather than the <63 µm fraction.

Concentrations of trace metals in the sediments exhibit highly skewed distributions and are therefore described using non-parametric measures. Figure 30(a-c) shows the median and range of concentrations of mercury,

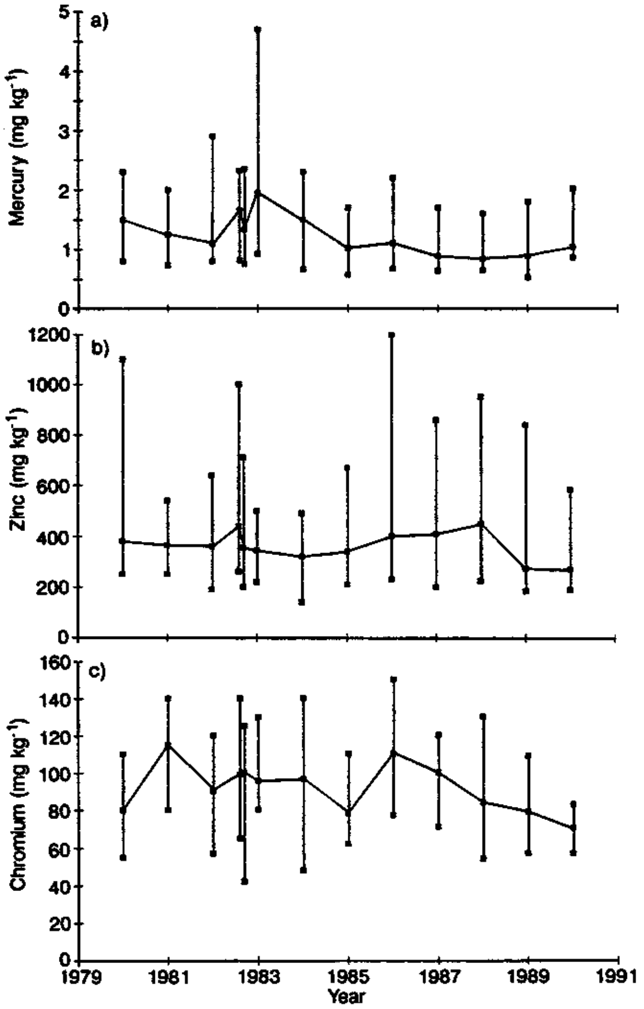

Figure 30. Time series of concentrations of metals in the <90 µm fraction of sediment collected near the Liverpool Bay sewage-sludge disposal site: (a) mercury; (b) zinc; and (c) chromium. Bars show median value and range

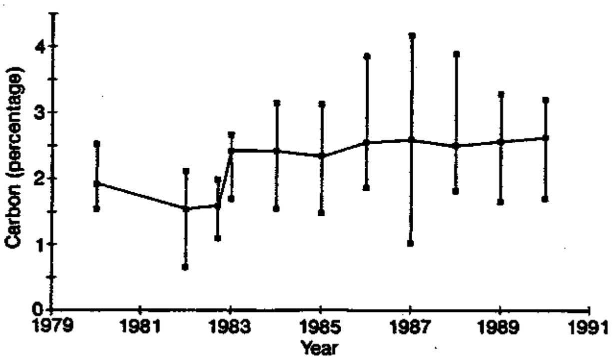

Figure 31. Time series of concentrations of carbon in the <90 µm fraction of sediment collected near the Liverpool Bay sewage-sludge disposal site. Bars show median value and range

zinc and chromium from 1980 to 1990. The most notable feature of the time series is the absence of any marked trend over the decade of study. Figure 31 shows carbon data presented in the same manner as that of metals. This shows a consistent median value during the period 1983 to 1990, although it appears that the concentrations may have been slightly lower in the early years of the decade. The concentration ranges show considerable overlap between all years suggesting that this small difference is of little significance.

When considering data from this sandy area, it must be recognised that the <90 µm fraction in general represents less than 5% of the total sediment and that, when considered on a whole sediment basis, the concentrations of metals are very low.

It is known that sediments in the vicinity of the disposal site contain higher concentrations of several metals (e.g. Cu, Hg, Zn and Pb) than sediments further away (Norton *et al.*, 1984) and it seems reasonable to conclude that there has been some accumulation of metal contaminants in the area as a direct consequence of the sewage-sludge disposal operation. However, the fact that there has been no increase during the past 10 years indicates that the system is in a steady state and has been so for at least the last decade.

15. MONITORING OF ACTIVITIES RELATED TO AGGREGATE EXTRACTION

15.1 Physical effects - Hastings Shingle Bank side-scan sonar survey

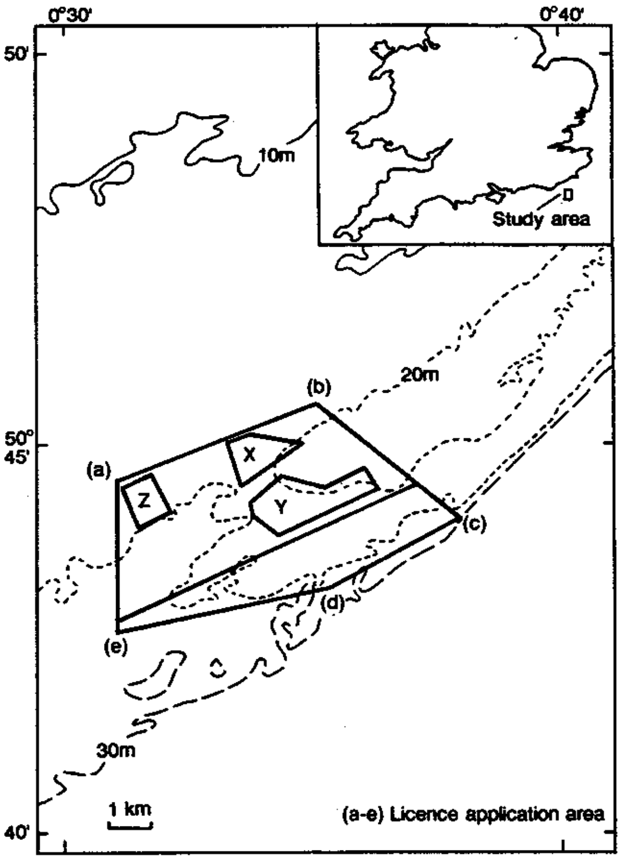

Figure 32. Location map of the Hastings Shingle Bank, showing positions of licensed extraction zones X, Y and Z

In August 1988, zones X, Y and Z in the northern sector of the Hastings Shingle Bank were licensed for aggregate extraction (Figure 32). Side-scan sonar surveys of this area had been conducted previously in 1986 and 1987, during which the general sediment distribution upon the Bank was established (MAFF, 1990). The benthic fauna inhabiting the Bank was also surveyed in 1986 to establish a baseline against which future surveys of the benthos might be compared. Together, these surveys provide an accurate description of the condition on the Bank prior to dredging.

In 1990, a further, detailed side-scan sonar survey was made of zones X and Y (the areas where extraction activities were thought to have been concentrated since licences were issued). The aim of this survey was to locate the areas which had been dredged most intensively since the licence was granted, so allowing subsequent benthic surveys to be targeted appropriately.

The track followed during the survey is shown in Figure 33 and an interpretation of the resultant side-scan sonar record is presented in Figure 34. Due to errors inherent in the Decca navigation system employed during the conduct of this survey, the apparent position of the ship as shown by the side-scan track is known to be approximately 250 m SE of its actual position. This error is taken into account in the interpretation presented in Figure 34.

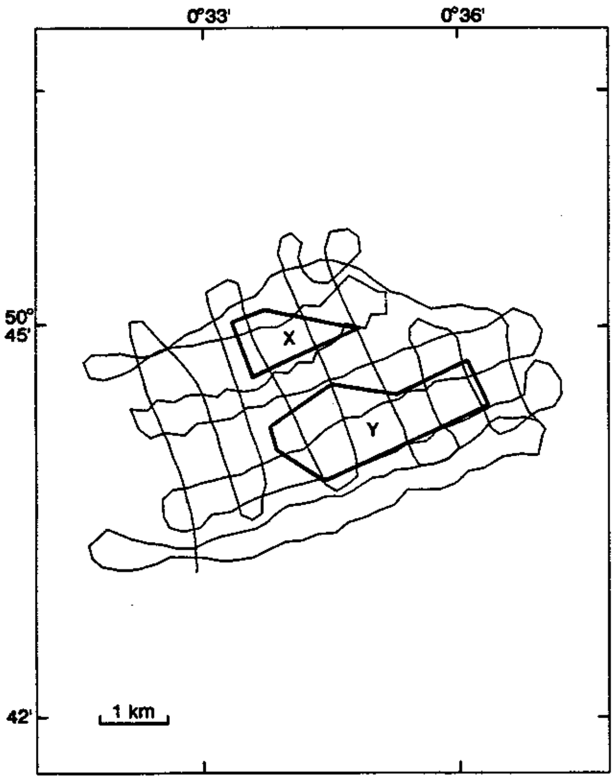

Figure 33. Side-scan track followed during the survey of aggregate extraction zones X and Y on the Hastings Shingle Bank in 1990

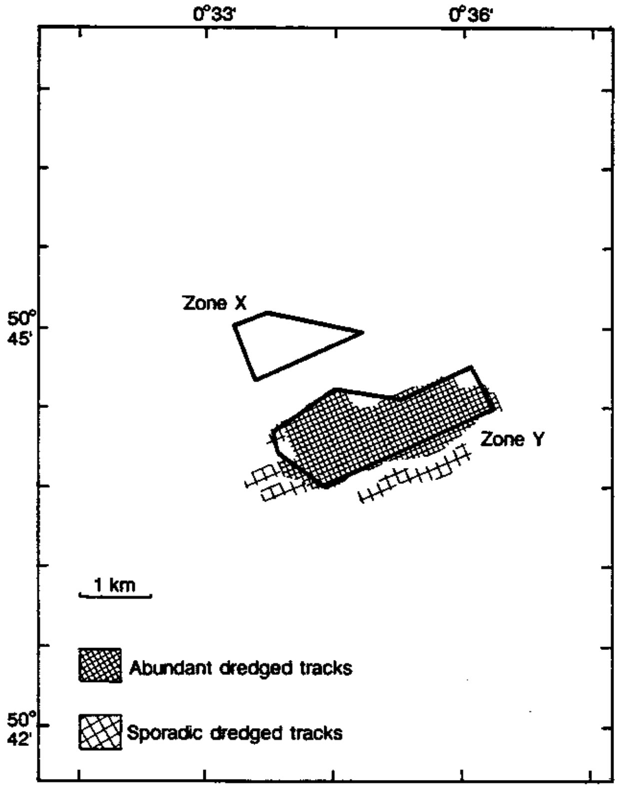

Figure 34. The distribution of dredged tracks, as determined by the side-scan sonar survey of aggregate extraction zones X and Y on the Hastings Shingle Bank in 1990

In terms of its sedimentology, the sea bed does not appear to have changed significantly since the 1986 and 1987 surveys, consisting principally of sand (built up into sand waves) in the west and north-east of the licensed area and gravel elsewhere. Evidence of dredging activity is completely lacking within zone X, but is abundant in zone Y; virtually the whole of this zone is completely covered by dredged tracks. There is no evidence yet of a localised lowering of the sea bed level, as was ascertained in licence area 213 off the Isle of Wight (MAFF, 1991(a)). Only a few dredged tracks exist outside of the licensed zones. These occur to the south and west of zone Y and generally cover only 1-2% of the sea bed. It thus appears that the vast majority of dredging activity has been confined to the licensed zone. No evidence was seen of a 4m deep experimental pit dredged on the Bank in 1971 (Dickson and Lee, 1973).

15.2 Effects on benthos - preliminary results of an inter-regional comparison of gravel assemblages off the eastern and southern coasts of England

Whenever new proposals for sand and gravel extraction are made, concerns are expressed about the impact on fisheries and benthos. With a view to quantifying the validity of these concerns, a three-year study was initiated in 1990 to examine the impact on the benthos arising from the commercial exploitation of marine aggregate deposits off the English coast, including processes of recolonisation following disturbance. The first part of this study has involved an assessment of regional differences in the nature of sand and gravel assemblages (Kenny *et al.*, 1991). Stations were located close to areas licensed for aggregate extraction, but were not subject to the direct influence of dredging. The areas surveyed included the Isle of Wight and Hastings off the southern coast, and Lowestoft and North Norfolk off the eastern coast of England (Figure 35).

Preliminary results indicate that, although there is significant regional variability in the structure of faunal assemblages, it is nevertheless possible to identify similarities between those associated with the gravel deposits of Hastings, the Isle of Wight and North Norfolk. In particular, the bryozoans *(Flustra foliacea* and *Alcyonidium* sp.), the slipper limpet *(Crepidula fornicata)* and the ascidian *(Dendrodoa grossularia)*, are present at all of these locations, as are a number of species of hydroids such as *Abietinaria abietina*. Figure 36 shows the output from cluster analysis of presence/absence data, which expresses the 'distance' (or dissimilarity) between stations or groups of stations. It is evident that stations have initially been

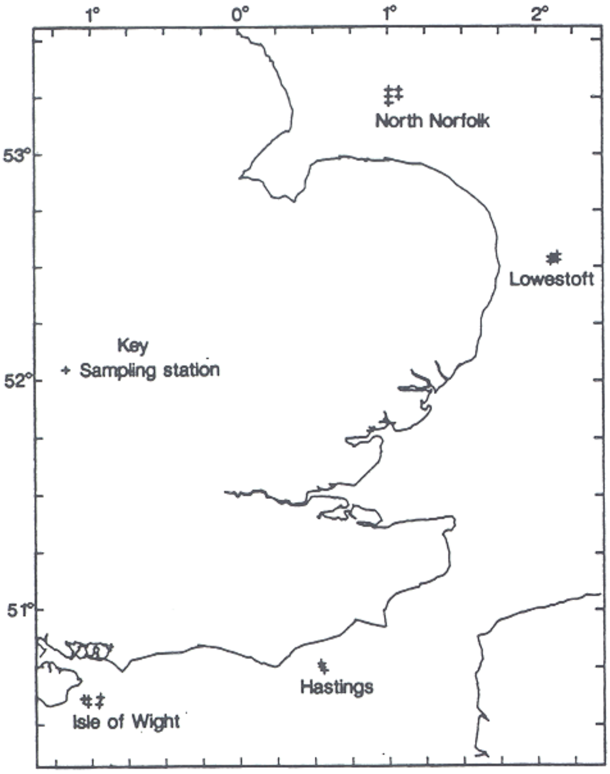

Figure 35. The location of sampling stations around aggregate extraction areas

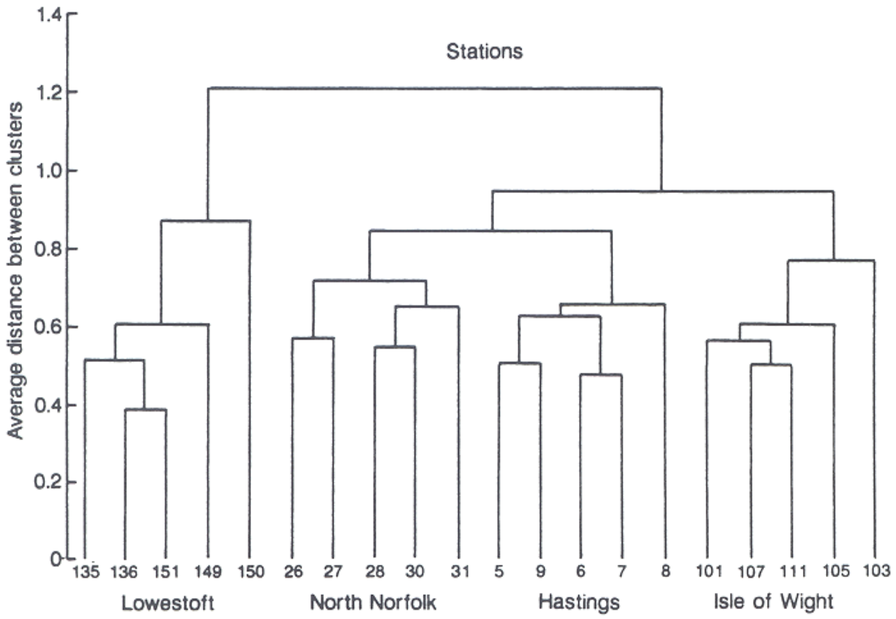

Figure 36. Average linkage cluster analysis between stations for the benthos retained on a 5 mm mesh. (Polychaetes were excluded from the analysis)

clustered according to their respective regions. Below this, the greatest 'distance' occurs between the stations off Lowestoft and those from the other three regions.

The combined output of cluster analysis by stations and by species (Figure 37) again highlights the contrast between Lowestoft and the other three regions; there is clearly a much reduced range of species at the former location. This reduction may be attributed to the abrasive effects of sand shifting under strong tidal currents and the effects of disturbance caused by storms. A side-scan survey of a licensed extraction site in the Lowestoft area, conducted in 1990 (unpublished data), showed very few dredge tracks. Yet, it is known that this site has been, and continues to be, extensively dredged. It therefore appears that the tracks are quickly obscured by shifting sand, and this view is supported by the notable presence of sand waves and ripples. Such conditions are of course not conducive to the development of either a stable benthic fauna or populations of the longer-lived benthic species.

It may be concluded from the present data, that gravel deposits off the UK continental shelf have the ability to support different faunal assemblages, depending on the local conditions. The most important of these conditions are probably tides and wave action.

Future work on this project will concentrate on a major field study of recolonisation by benthic organisms following experimental dredging at a selected site. The detailed appraisal of regional variability in the nature of gravel assemblages will continue.

Stations sampled

135	136	151	149	150	26	27	28	30	31	5	9	6	7	8	101	107	111	105	103	
●	●	●	●	●	●	●				●	●	●	●							*Sertularia cupressina*
●	●	●	●						●	●	●	●	●	●		●				*Sagartia* sp
	●	●	●	●	●	●						●	●	●						*Ophiura albida*
			●	●						●	●	●	●							*Psammechinus miliaris*
												●	●				●			*Urticina felina*
												●	●	●	●	●	●		●	*Nucula nucleus*
			●			●	●			●	●	●		●						*Campanularia* sp
							●		●	●	●	●								*Anomia* sp
							●	●		●	●	●	●	●						*Sarsia exima*
					●		●			●	●	●	●	●	●					*Polyplacophora*
					●		●	●		●	●	●	●					●		*Galathea intermedia*
					●	●	●	●		●				●						*Nermertesia antennina*
					●	●	●	●	●	●	●			●	●	●				*Balanus* sp
					●	●	●	●	●	●	●				●					*Pisidia longicornis*
●	●				●	●	●	●	●	●	●	●	●	●						*Electra* sp
					●	●	●	●	●	●	●		●	●						*Scrupocellaria scruposa*
		●					●	●	●	●	●	●	●		●	●	●	●	●	*Abietinaria abietina*
								●	●	●		●	●		●	●	●	●		*Ascophoran*
		●					●	●		●	●	●	●	●		●	●	●	●	*Halecium* sp
					●	●	●			●		●	●	●	●	●	●	●	●	*Alcyonidium* sp
					●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	*Flustra foliacea*
					●	●		●	●	●	●	●	●			●	●	●	●	*Dendrodoa* sp
										●	●		●			●	●			*Upogebia deltaura*
										●	●	●			●	●	●			*Paguridae*
										●		●	●							*Tubulipora* sp
							●	●			●				●	●	●	●		*Crepidula fornicata*
										●				●				●	●	*Sertularella polyzonias*
										●				●						*Cyclostomata*
							●	●		●										*Liocarcinus* sp
							●			●		●								*Bicellariella* sp
								●		●					●			●		*Ocenebra erinacea*
					●			●		●		●		●				●		*Polycarpa* sp
										●	●	●	●	●						*Calycella syringa*
										●	●	●	●	●						*Echinocyamus pusillus*
										●	●	●	●	●						*Cellaria fistulosa*
										●	●	●	●	●						*Bugula fulva*
											●	●	●	●						*Alcyonium digitatum*
										●	●									*Hyas coarctacus*
										●	●									*Venerupis rhomboides*
										●	●									*Crisia aculeata*
										●	●			●						*Spisula elliptica*
●						●	●	●	●											*Diphasia attenuata*
							●	●	●											*Modiolus modiolus*
					●		●	●	●		●									*Plumularia* sp
					●		●	●	●				●							*Porifera*
					●	●	●					●								*Ophiothrix fragilis*
●	●	●	●			●	●		●	●										*Tubularia indivisa*
							●													*Pinnotheres pisum*
							●													*Chlamys* sp
			●													●				*Holothuroidea*
																●				*Gibbula cineraria*
																●	●	●	●	*Bowerbankia pustulosa*
					●															*Corophium* sp
																	●			*Disporella hispida*
																			●	*Macropodia* sp
								●												*Astarte montagui*
						●														*Ensis* sp
						●														*Nymphon* sp
					●															*Crassoster papposus*
					●															*Bugula plumosa*
				●							●									*Buccinum undatum*
														●						*Acanthodoris pilosa*
												●								*Aserias rubens*
									●											*Phascolion strombi*
									●											*Cancer pagurus*
									●											*Mysidacea*
									●											*Golfingia vulgaris*
									●				●							*Ebalia* sp
													●							*Lyonsia norwegica*
											●		●							*Ampelisca* sp
											●									*Scalpellum scalpellum*
											●									*Gibbula tumida*
											●									*Lunatia poliana*
											●									*Timoclea ovata*
											●									*Hinia incrassata*
									●											*Calliostoma zizyphinum*
										●										*Obelia* sp
										●										*Photis longicaudata*
										●										*Mya arenaria*
										●										*Membranipora* sp
										●										*Rhodophyceae*
Lowestoft					North Norfolk					Hastings					Isle of Wight					

Figure 37. The combined output of cluster analysis, by station and by species for each region

16. PHYSICAL ASPECTS OF SEDIMENTS OFF THE NORTH-EAST COAST OF ENGLAND

16.1 Introduction

In the past 50 years, several types of waste have been deposited in the sea off the north-east coast of England: sewage-sludge, dredged material, colliery waste, flyash and liquid industrial waste (Eagle *et al.*, 1979). Since the late 1960s, MAFF has monitored these disposal operations and, since 1974, has undertaken this work as part of its responsibilities initially under the Dumping at Sea Act 1974 (Great Britain - Parliament, 1974) and more recently the Food and Environment Protection Act 1985 (Great Britain - Parliament, 1985(a)). A large amount of basic sediment data has been collected during various surveys using grabbing, coring, dredging, side-scan sonar, remote underwater video and divers. As part of the DOE/MAFF Cohesive Sediment Dynamics Study (COSEDS), these data have been amalgamated and, where necessary, additional information collected to allow the production of various sediment charts of the area.

16.2 General distribution of sediment types

Figure 38 shows a chart of sediment distribution produced primarily from side-scan sonar surveys of the sea bed, collected between 1985 and 1990, along the survey tracks shown in Figure 39. The side-scan data were supplemented by grab samples collected throughout the survey area (see, for example, Figure 40) and also by underwater video images where these proved necessary for ease and certainty of interpretation. Sediments up to about 5 km from the coast are highly variable, being dominated by mixtures of mud, sand and boulders, here termed 'rough ground'. Some rocky outcrops also occur, although these are usually close to the coast. The sediments further offshore are primarily sand/mud mixtures and show only limited local variability.

This chart (Figure 38), together with particle size data, will be used as basic information for a computer modelling exercise for the prediction of cohesive sediment and contaminant transport off this part of England.

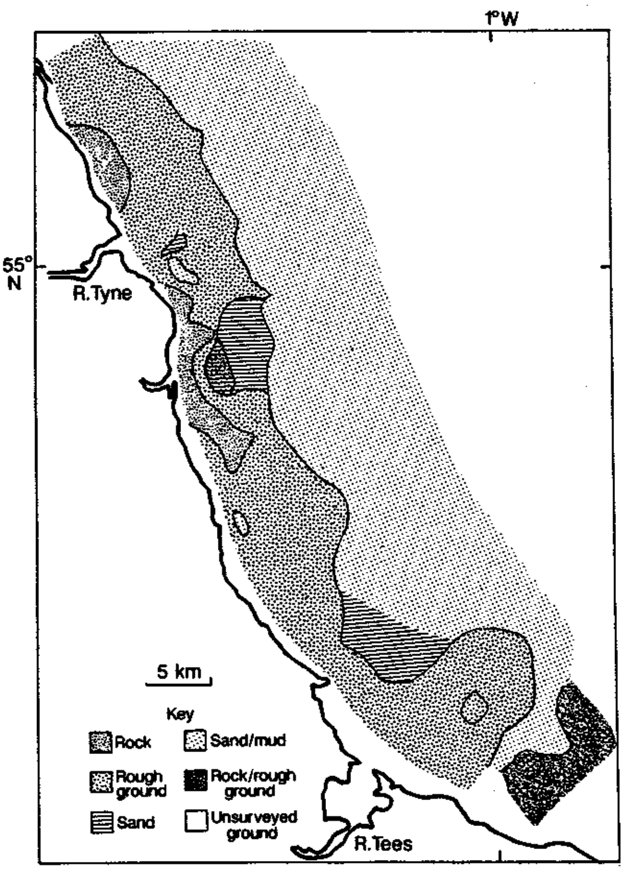

Figure 38. Sediment distribution off the north-east coast of England

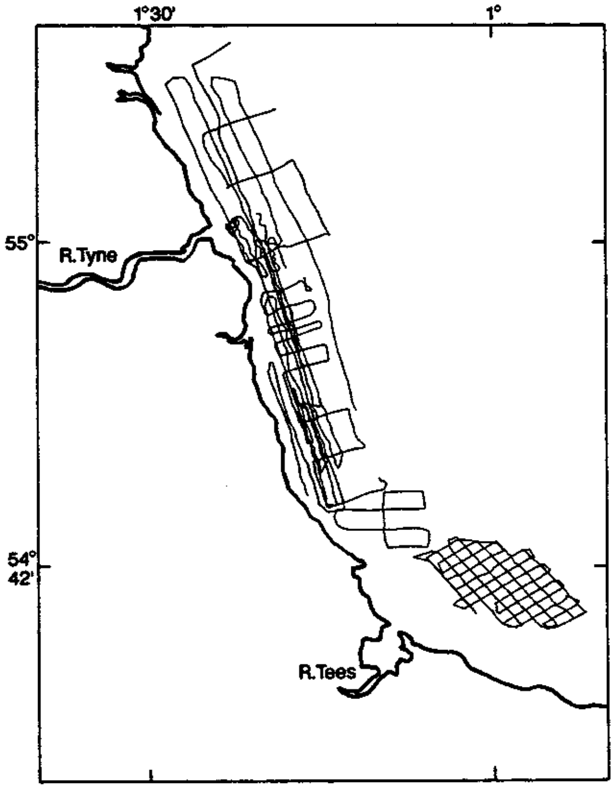

Figure 39. Tracks of side-scan surveys used to produce the sediment map shown in Figure 38

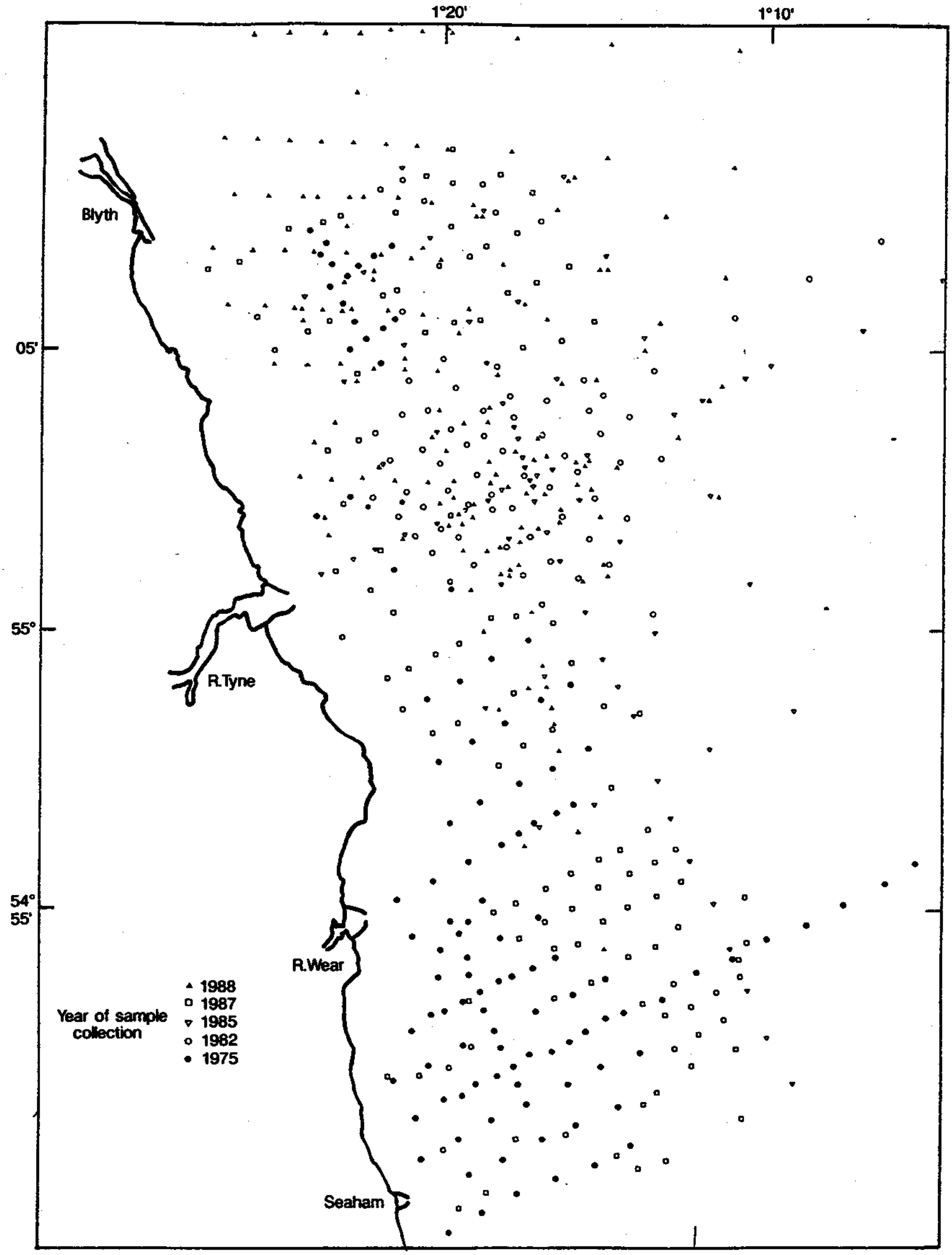

Figure 40. Grab sampling stations used to produce the sediment map shown in Figure 38

DISPOSAL AT SEA: LICENSING AND RELATED ACTIVITIES

17. LICENSING OF DISPOSAL OF WASTES AT SEA

17.1 Introduction

This section gives information about the licensing of disposal of wastes at sea in the UK during 1990 under Part II of the Food and Environment Protection Act 1985 (FEPA), (Great Britain - Parliament, 1985(a)). This section also 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). Information on licensing and enforcement for 1988 and 1989, together with a report on the related monitoring activity in England and Wales, was included in a previous report in this series (MAFF, 1991(a)).

17.2 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)).

The Environmental Protection Act 1990 (Great Britain - Parliament, 1990) made certain changes to FEPA. Section 146 makes it an offence for any vessel (including foreign vessels loaded in foreign ports) to deposit waste within UK Continental Shelf Limits without a licence. In addition, it also provides for an increase in the powers of enforcement officers to enable them to bring into port any vessel suspected of illegal disposal activities and increases the maximum fine on summary conviction from £2 000 to £50 000. Neither powers were used in 1990.

Section 147 provides for additional material about sea disposal activities to be added to the public registers, in addition to the information on the licences issued, which is already available. In future, the public register will also contain information about applications, breaches of the legislation and enforcement action.

17.3 Enforcement activities

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 62 inspections in 1990. The Sea Fisheries Inspectorate, with staff based at the ports, detects unlicensed disposal operations and enforces licence conditions relating to the depositing of the wastes in the designated disposal area. They made 124 inspections in 1990.

In Scotland, similar enforcement powers are held by staff of the SOAFD Marine Laboratory and by the Scottish Sea Fisheries Inspectorate (SSFI). SOAFD made three enforcement visits in 1990 and a further three 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. No inspections were considered to be necessary in 1990.

MAFF investigated eight reports of unlicensed disposal in 1990, including one scuttling of a vessel. Four warning letters were issued and one operation was found to be outside the scope of the Act. In Scotland, SOAFD investigated thirteen cases where operations had been either unlicensed or where licence conditions had been contravened. In Northern Ireland, one case was investigated.

1990 saw completion of the installation of marine position recorders on all vessels engaged in the disposal at sea of sewage sludge and industrial waste. These tamper-proof recorders keep track of the location and disposal activities of the vessels involved in these operations.

17.4 Report on licensing activities

Tables 22 to 26 give details, over the period 1986-1990, of the number of sea disposal licences issued, the quantity of waste licensed, and the quantity deposited,

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* *Now known as The Scottish Office Agriculture and Fisheries Department (SOAFD)*

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)). During the period of this report, a number of enquiries were received by the licensing authorities about the possibility of sea disposal for a wide range of wastes. In many cases, these enquiries were not followed by submission of applications. Three applications were refused in England and Wales in 1990. No applications were refused in Scotland or Northern Ireland. No licences were granted for new arisings of liquid or solid industrial waste or sewage sludge.

17.5 Licensing of liquid industrial wastes

Table 22(a) lists the liquid industrial wastes licensed for disposal at sea in the period covered by this report, together with the name of the producer, the disposal sites used, the quantity licensed, and the quantity deposited. Table 22(b) summarises the quantity of wastes licensed and the amounts deposited, including metallic contaminants, for the period 1986-1990. Locations of the disposal sites and the quantities of waste deposited at each site are given in Figure 41.

Table 22(a). Liquid industrial wastes licensed for disposal at sea in 1990 (1)

Licensed quantity (t)	Company	Description of waste	Disposal sites	Quantity deposited (t)
-	Allied Colloids Ltd	Ammoniacal liquor	Tees	501(2)
2,850	Fine Organics Ltd	Oxygen demanding waste	Tees/Tyne/Spurn Head	878
165,000	Imperial Chemical Industries Ltd	Ammoniacal liquor	Tees	151,087
900	Orsynetics Ltd	Caustic alkali	Tyne/Spurn Head	431
40,000	Sterling Organics Ltd	Phenolic liquor	Tyne/Spurn Head	35,778
19,250	Tate and Lyle Refineries Ltd	Industrial slurry	South Falls/Roughs Tower	21,005
-	Woolcombers Ltd	Oxygen demanding waste	Tyne/Spurn Head	281(2)

Notes

(1) No liquid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report

Also, the table excludes three 1989 licences valid into 1990, but on which no waste was disposed of in 1990

(2) Disposed of under licence issued in 1989

Licensed quantities : licences were issued throughout the calendar year 1990 and were generally valid for twelve months

Tonnages deposited : relate to quantities deposited in the calendar year 1990, which may be covered by two licences, including one issued in 1989

Table 22(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	1986	28	365,995	283,066	0.03	0.55	1.84	0.01	0.80	1.78	2.06
	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
Northern Ireland	1986	0	0	650	0.00	0.00	0.59	0.00	0.00	0.01	0.03
UK total	1986	28	365,995	283,716	0.03	0.55	2.43	0.01	0.80	1.79	2.09
	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

Notes

(1) No liquid industrial wastes were licensed or disposed of in Scotland in the period covered by this report, nor in Northern Ireland after 1986

For information on licensed quantities and tonnages deposited see footnote to Table 22(a)

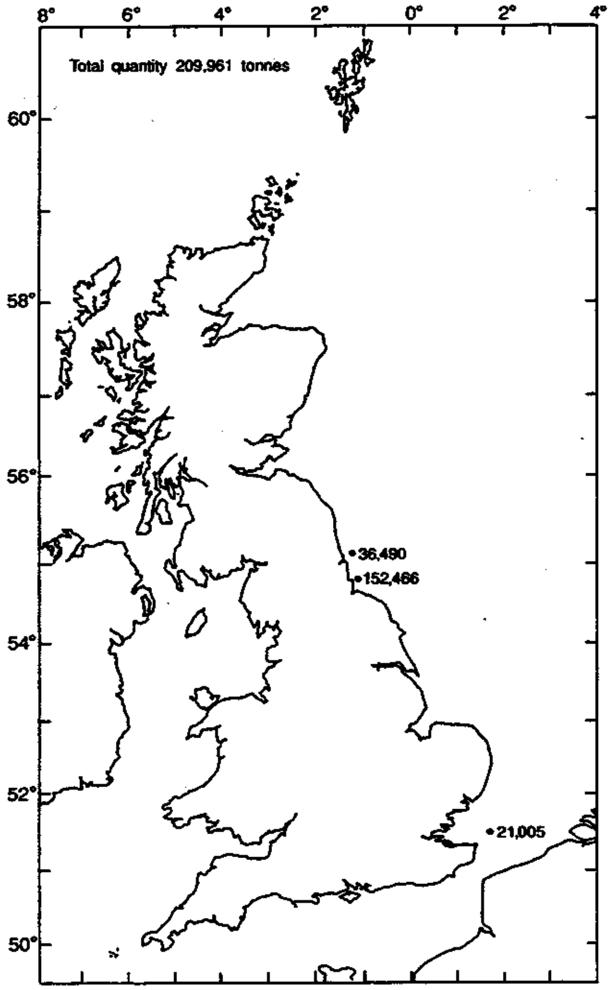

Figure 41. Liquid industrial wastes disposed of at sea in 1990

On 22 February 1990, the Minister of Agriculture, Fisheries and Food announced that the UK Government did not intend that liquid industrial waste or flyash would be disposed of at sea after the end of 1992. He noted, however, that in two cases, those of ICI and Sterling Organics, it might not be feasible to meet this deadline, and stated that their licences would be extended into 1993 only if absolutely necessary on technical grounds, and for the shortest possible part of that year.

All licences for liquid industrial waste, other than those issued to these two companies, were terminated by the end of 1990.

17.6 Licensing of solid industrial wastes

Table 23(a) gives details of licences issued for the disposal of solid industrial wastes, and Table 23(b) details the quantities of metal contaminants in the material deposited. Figure 42 shows the locations of the relevant disposal sites and the quantities of waste deposited at each. The bulk of the material licensed was stone extracted during coal mining. The material is accepted internationally as being inert in the marine environment and, as such, the presence of the metal contaminants is of no significance in terms of potential ecological impact.

Table 23(a). Solid industrial wastes licensed for disposal at sea in 1990(1)

Licensed quantity (t)	Company and source of waste	Description of waste	Disposal sites	Quantity deposited (t)
	British Coal Collieries			
1,150,000	Dawdon/Seaham	Minestone	Bankside, Seaham	1,176,334
875,000	Easington	Minestone	Foreshore at Easington	617,979
525,000	Ellington	Minestone	Foreshore at Ellington	387,135
100,000	Point of Ayr	Minestone	Foreshore at Point of Ayr	117,296
1,000,000	Wearmouth	Minestone	Sunderland	756,145
725,000	Westoe	Minestone	Souter Point/ North Tyne/ Sunderland	631,721
1,650,000	Wearmouth and Westoe	Mine tailings	Souter Point/ North Tyne/ Sunderland	964,655
	National Power			
500,000	Blyth Power Station	Pulverised fuel ash	Blyth A	242,034
50,000	Stella Power Stations	Pulverised fuel ash	Souter Point/ North Tyne/ Sunderland	25,675
-	**Jacksons Fuels**	Washed sand	Foreshore at Hartlepool	680(2)

Notes

(1) *No solid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report*

(2) *Disposed of under licence issued in 1989*

For information on licensed quantities and tonnages deposited see footnote to Table 22(a)

Table 23(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	1986	5	302,800	4,209,783*	0.22	19	195	0.22	63	249	490
	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
Scotland	1986	2	410	19	0.00	0	0	0.00	0	0	0
	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
UK total	1986	7	303,210	4,209,802	0.22	19	195	0.22	63	249	490
	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

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 22(a)

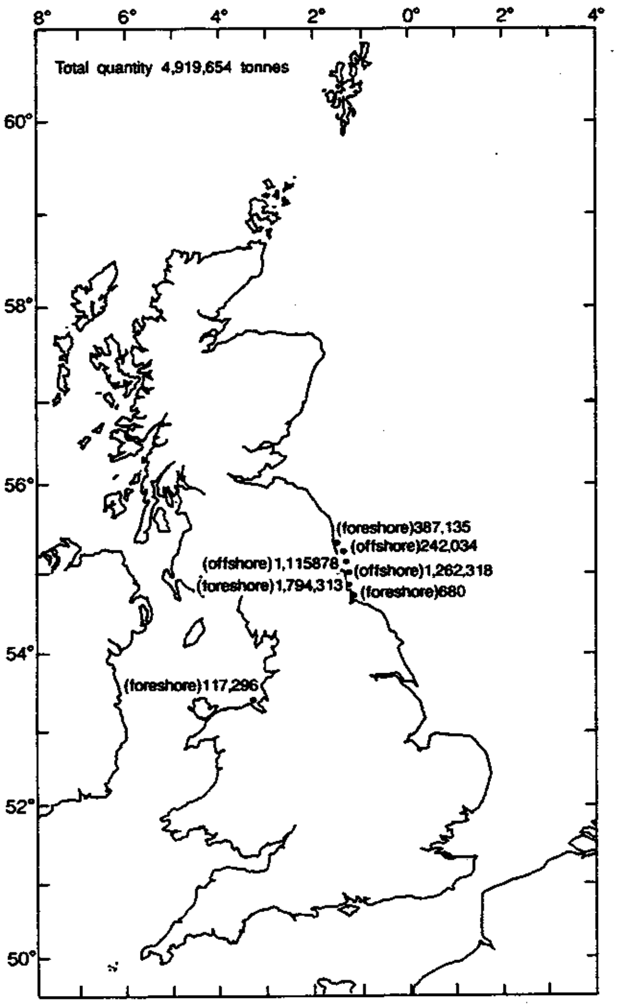

Figure 42. Solid industrial wastes disposed of at sea in 1990

Two licences were also renewed for power station flyash. Both of these licences are expected to finish before the end of 1992 in accordance with the Government's decision to terminate the disposal of flyash at sea (see sub-section 17.5).

On 14 June 1990, the Parliamentary Secretary to the Ministry of Agriculture, Fisheries and Food announced that the Government was determined that the disposal of colliery material on the beaches in the north-east of England should be ended as soon as practicable. MAFF stated that licences would be ended in five years unless, as a result of a planning application, it was demonstrated that no practical means were available to British Coal for disposal of the minestone on land.

British Coal were also required to seek permission to dispose of, on land, the minestone currently disposed of at sea from the Westoe and Wearmouth collieries. A deadline of the end of 1997 was set for terminating this sea disposal operation, unless application of the planning system demonstrated that no practical means were available for land disposal.

17.7 Licensing of sewage sludge

Table 24(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 24(b). Figure 43 shows the location of the disposal sites for sewage sludge and the quantities of sludge deposited at each site.

Table 24(a). Sewage sludge licensed for disposal at sea in 1990

Country	Licensed quantity (t) (1)	Company and source of waste	Disposal sites	Quantity deposited (t)(1)
England/ Wales	80,000	Anglian Water (Cliff Quay STW, Ipswich)	Roughs Tower	67,780
	81,569	Anglian Water (Colchester STW)	Roughs Tower	38,755
	150,000	Anglian Water (Tilbury STW)	Roughs Tower	157,303
	554,000	Northumbrian Water (Howdon, Chester-le-Street, Cramlington, Washington STWs)	Tyne/Spurn Head	481,000
	105,010	Northumbrian Water (Portrack, Billingham, Guisborough, Ayton STWs)	Tyne/Spurn Head	70,737
	1,965,000	North West Water (Davyhulme, Liverpool, Warrington STWs)	Liverpool Bay	1,732,589
	5,000	North West Water (Walney Island)	Liverpool Bay	3,216
	300,000	Southern Water (Woolston, Portswood, Millbrook, Slowhill Copse STWs)	Nab Tower	252,860
	58,000	South West Water (Countess Wear STW)	Lyme Bay	37,130
	80,000	South West Water (Plympton, Radford, Camel's Head, Ernesettle, Ivybridge, Saltash, Newton Ferrers STWs)	Plymouth	67,969
	4,500,000	Thames Water (Beckton, Crossness, Riverside, Deephams STWs)	Barrow Deep	3,803,118
	55,000	Welsh Water (Ponthir, Nash, Llanfoist, Magor STWs)	Bristol Channel	53,338
	400,000	Wessex Water (Avonmouth, Keynsham, Bath STWs)	Bristol Channel	207,574
	140,000	Yorkshire Water (Knostrop STW)	Spurn Head	125,567
Scotland	500,000	Lothian Regional Council	St Abb's Head/ Bell Rock	274,930
	2,500,000	Strathclyde Regional Council	Garroch Head	1,671,500
Northern Ireland	80,000	Dept. Environment (Northern Ireland)	Belfast Sludge	290,030(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 22(a)

Table 24(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 andWales	1986	17	8,702,925	7,532,480	3.26	91	140	1.08	23	161	473
	1987	17	8,682,510	6,692,654	3.24	104	136	1.08	21	156	439
	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
Scotland	1986	2	2,300,000	2,002,817	0.49	30	28	0.20	2	30	57
	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
Northern Ireland	1986	0	0	285,436*	0.09	3	3	0.08	1	5	12
	1987	1	100,000	309,489*	0.09	3	3	0.08	1	4	15
	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	0.04	0	2	11
UK total	1986	19	11,002,925	9,820,733*	3.84	124	170	1.37	26	196	541
	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	1.07	21	129	288

Notes

* *Includes 200,000 t yr^{-1} disposed of by DOE(NI) Water Services under an administrative authorisation*

§ *These figures are corrected versions of those previously published in AEMR 26 (MAFF 1991(a))*

For information on licensed quantities and tonnages deposited see footnote to Table 22(a)

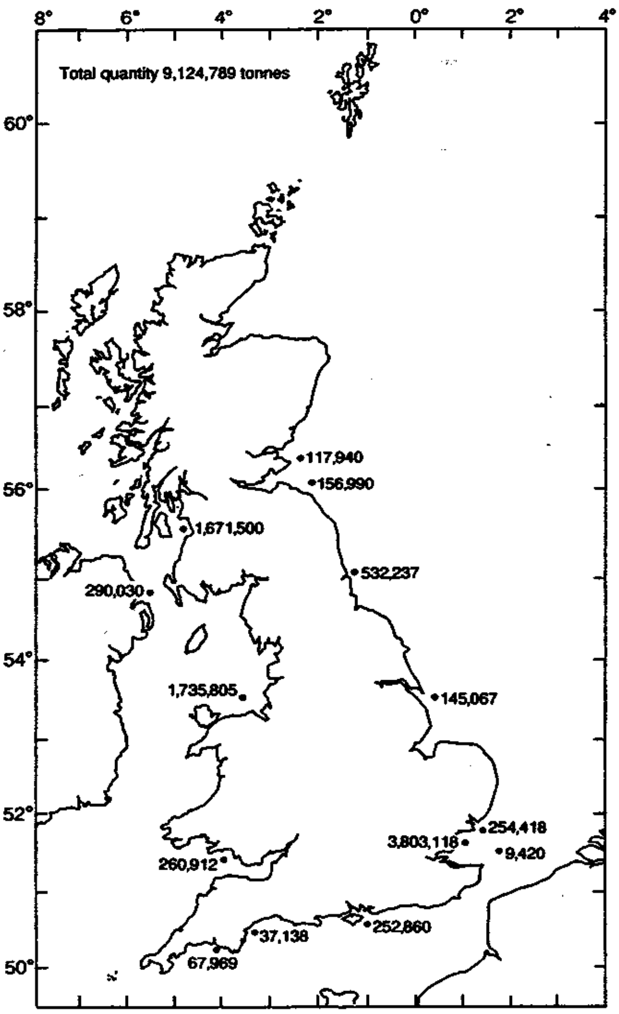

Figure 43. Sewage sludge disposed of at sea in 1990

At the Second International Conference on the Protection of the North Sea in 1987, (Department of the Environment, 1987(b)), the Government agreed to take urgent action to reduce the contamination by persistent, toxic or bioaccumulable 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. The previous report (MAFF, 1991(a)) explained that to apply this control, limits were set for a series of key contaminants deposited under each licence. Table 24(c) compares the aggregated figures (in tonnes) authorised for disposal in the North Sea in 1990, with the estimated quantities (in tonnes) at 1987 licensed levels.

Table 24(c). Contaminants in sewage sludge in 1990 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
1990	1.2	3.4	49.7	18.6	132.5	435.6	136.0

Although the bulk of these contaminants comes from general domestic sources rather than from industry, the licensed levels for 1990 were below both those for 1989 and the 1987 baseline figures. Moreover, the figures for metals actually disposed of in 1990 (Table 24(b)) show reductions in all cases.

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.

17.8 Licensing of dredged materials

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 25 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 material. A proportion of the trace metals associated with dredged material occurs within the mineral structure or is tightly bound, such that generally it would not be available to marine organisms. Figure 44 shows the main disposal sites used in 1990 and the quantities deposited.

Table 25. 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	1986	96	27,917,115	31,698,959*	29.7	1211	1230	13.5	848	2405	5338
	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
Scotland	1986	23	3,460,850	3,516,224	1.7	132	86	1.9	43	113	251
	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	67	64	0.8	30	106	207
Northern Ireland	1986	2	1,544,000	1,927,575	3.3	41	48	0.4	40	68	498
	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
UK total	1986	121	32,921,965	37,142,758	34.8	1384	1363	15.8	931	2585	6086
	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	13.2	1093	900	7.6	518	1537	3939

*Notes: * Some deposits were covered by 5 year licences*
For information on licensed quantities and tonnages deposited see footnote to Table 22(a)

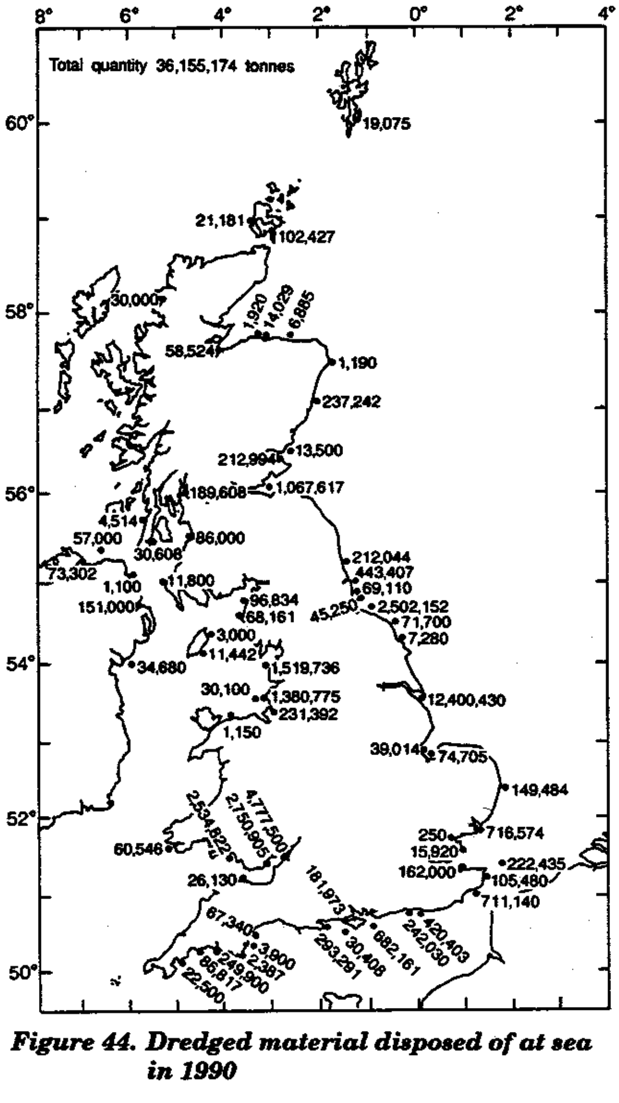

Figure 44. Dredged material disposed of at sea in 1990

17.9 Licensing of marine incineration

On 5 March 1990, the Minister of Agriculture, Fisheries and Food announced that, because land-based disposal facilities were available, no UK waste would be licensed for marine incineration after the end of 1990.

Table 26 gives details of licences issued and material incinerated in 1990. The waste itself does not enter the marine environment except for a very small percentage which is not destroyed by combustion (right-hand column, Table 26). It is assumed that metallic contaminants in this waste are emitted with exhaust gases. These data are also shown in Table 26.

The internationally approved site was an area of 15 miles radius centred on a position 54°17'N, 3°45'E. This site is 150 miles east of Flamborough Head and in the Dutch sector of the North Sea.

Table 26. Licensing of marine incineration, quantities of waste burned and contaminants emitted

Year	Licences issued	Licensed quantity (t)	Amount burned (t)	Estimated amount of contaminants released (kg)							
				Cd	Cr	Cu	Hg	Ni	Pb	Zn	Organohalogens[$]
1986	20	7530	3754	<0.30	<1.9	<3.2	<0.20	<1.9	<2.3	<1.9	38.0
1987	11	10,681	4551	0.90	14.1	65.0	0.40	92.0	15.2	13.8	45.0
1988	13	7497	5407	0.54	12.3	59.2	0.31	84.2	8.2	10.3	30.1
1989	12	6900	4152	0.65	19.6	8.0	0.13	8.7	4.4	8.3	8.3
1990	8	4431	4231	2.2	106.4	26.6	0.21	28.0	7.9	30.9	10.8

Notes

[$] *Based on an average destruction efficiency of 99.995%*

Licensed quantities : licences were issued throughout the calendar year 1990 and were only valid up until 31 December 1990

Tonnages burned : relate to quantities burned in the calendar year 1990, which may be covered by two licences, including one issued in 1989

17.10 Other materials deposited at sea

Under Part II of the Food and Environment Protection Act, 1985 (Great Britain - Parliament, 1985(a)), 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 involve construction work below mean high water, the use of tracers, the application of biocides and burials at sea.

18. ADVICE ON FISHERY IMPLICATIONS OF PIPELINE DISCHARGES

This section gives a brief summary of activities carried out during 1990 in connection with provision of advice on fishery implications of pipeline discharges. The background to this work, which relates mainly to MAFF's responsibilities as a statutory consultee under the Water Act, 1989 (Great Britain - Parliament, 1989), has been described previously (MAFF, 1991(a)).

Table 27 shows the number of applications for each type of effluent sent to MAFF for comment during 1990.

Table 27. Applications for consent to discharge assessed by MAFF in 1990

Sewage (including storm and emergency overflows)	Trade effluent	Surface water	Total
88	28	30	146

Tighter consent limits were recommended for 17% of the applications received in 1990 - an increase of 7% over the 1988/1989 figure. The majority of these recommendations were adopted by the consenting authority. There was also a significant increase (from 15 to 45%) in the number of applications for which approval was conditional on action being taken to prevent discharge of sewage solids, persistent plastics or other hazardous materials.

The number of formal applications received was less than in the previous two years. This is likely to have been at least partly related to Water Service Companies having to reconsider their strategy for sewage disposal, following the announcement by the Secretary of State for the Environment, on 5 March 1990, that all significant (>1500 m^3 day^{-1}) discharges of sewage to coastal waters should receive some treatment before discharge. Positive determination of 'deemed' consents for existing discharges has also been delayed while the recently-formed National Rivers Authority (NRA) has been developing procedures for charging for consents and defining statutory water quality objectives for estuaries. The Burnham-on-Crouch Laboratory has provided comments on both the proposals for water quality objectives and a new classification scheme for estuarine and coastal waters. It has also commented on draft reports for eight estuaries in which objectives and standards have been proposed. Another change during the year was the submission of Environmental Impact Statements (e.g. for proposed new power stations) to MAFF for comment, with a total of seven reports being received during the year. These are welcome developments, in that they give MAFF an opportunity to comment on fishery implications of major schemes at an early stage in the planning process.

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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.	Biological and chemical studies of the horse-mussel *(Modiolus modiolus)*	H Rees
3.	Bioassay studies	J Thain D Sheahan
4.	Contaminants in marine mammals	C Allchin C Kelly
5.	General sea-water sampling procedures	R Law
6.	Nutrients in sea water	R Law
7.	Hexachlorocyclohexanes in sea water	C Allchin
8.	Hydrocarbons in sea water	R Law J Osborne
9.	Surfactants	M Waldock
10.	Metals in sediments	S Rowlatt D Lovell
11.	Hydrocarbons in sediments	R Law J Osborne
12.	TBT studies	M Waldock M Waite
13.	Rosemaund pesticide study	P Matthiessen R Rycroft
14.	DAS*: Sewage-sludge disposal areas	S Rowlatt H Rees D Limpenny M Lambert P Manning
15.	DAS: Aggregate extraction activities	R G Lees A Kenny H Rees
16.	DAS: Sediments off the north-east coast	S Rowlatt
17.	DAS: Licensing and enforcement	C M G Vivian G Boyes
18.	DAS: Pipeline discharges	F Franklin

* *DAS = Disposal at Sea*

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 estuaries 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 foods 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.*

A2.3 References

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