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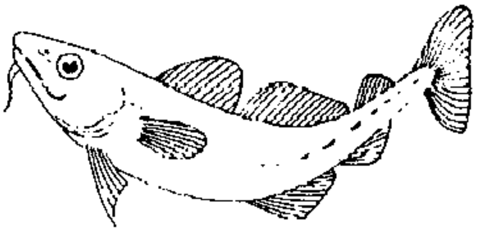
Number 40



Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment and Activities Regulating the Disposal of Wastes at Sea, 1992

MAFF
Directorate of Fisheries Research
Lowestoft, 1994

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD
DIRECTORATE OF FISHERIES RESEARCH

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

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

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CONTENTS Page

FOREWORD

BACKGROUND TO THE WORK

BIOTA

SEA WATER

SEDIMENTS

/continued :

SPECIAL STUDIES

DISPOSAL AT SEA: ENVIRONMENTAL ASSESSMENT STUDIES

DEPOSITS IN THE SEA: LICENSING AND RELATED ACTIVITIES

FOREWORD

This is the fifth report in MAFF's series of publications on 'Monitoring and surveillance of non-radioactive contaminants in the aquatic environment'. Now established as a regular, annual publication, the report describes work carried out chiefly by staff of the Directorate of Fisheries Research's laboratory at Burnham on Crouch in 1992. In addition, the activities on disposal at sea involve the Sea Fisheries Inspectorate and Marine Environment Protection Division of MAFF.

Although dealing largely with information gathered during 1992, the report also includes accounts of certain studies conducted in earlier years, and some data collected more recently. In this way, we hope to provide a picture of marine contamination that is both up-to-date and makes the best use of previous information to identify trends and patterns.

This report comprises a broad range of studies, that indicates the diverse nature of the many types of contaminants and situations that need to be examined. The majority of these are the subject of statutory responsibilities for MAFF, both through national legislation and international agreements. The contexts for the inspection, assessment and monitoring activities are explained in the first few pages of the report.

In support of these direct statutory roles, there is a programme of research and development at DFR, which serves to improve methodology and interpretation, and helps to identify new aspects of concern. The present report includes an account of progress in the development of new sediment bioassays (Section 2) which will allow better assessments of toxic effects in an environmental compartment that is a significant sink for many contaminants. The Rosemaund study (Section 13), now nearing its end, has been instrumental in establishing the importance of migration of pesticides into watercourses, while investigations into the effects of gravel extraction on benthic organisms (Section 15) is producing valuable information that will help to guide future approaches to the licensing of aggregate extraction.

The impact of sewage waste continues to be the subject of strong public concern. This report contains the results of a survey of sewage-derived litter (Section 17), carried out to extend the customary surveillance of chemicals and other indicators. Section 14 reports on studies of oestrogenic effects induced in fish, in response to concern that sewage effluents may introduce oestrogenic chemicals into river systems.

These and other studies are continuing, in a dynamic programme of work by which MAFF seeks to ensure that a careful picture is maintained of the trends and patterns in non-radioactive contamination of UK waters, and that issues of topical concern are investigated promptly and effectively.

Dr P Greig-Smith
Head of Aquatic Environment Protection Division
Ministry of Agriculture, Fisheries and Food
Directorate of Fisheries Research

BACKGROUND TO THE WORK

The Fisheries Laboratory at Burnham-on-Crouch forms a Section of the Aquatic Environment Protection Division of the Directorate of Fisheries Research within the Ministry of Agriculture, Fisheries and Food (MAFF). The Laboratory's primary responsibility is to provide advice to the MAFF policy divisions and other Government departments, on all aspects of non-radioactive contaminants in the aquatic environment. The advice given is used in the formulation of scientifically-based pollution control measures, enforced through Acts of Parliament, many of which define specific monitoring responsibilities for MAFF.

The most direct of these is a duty to license and control the disposal of land-generated wastes from ships at sea. The statutory requirements for this are laid down in Part II of the Food and Environment Protection Act (FEPA) (1985) (Great Britain - Parliament, 1985(a)). Licence applications are scientifically evaluated by inspectors at the Burnham Laboratory and regular visits are made to licence holders to ensure licence conditions are met. The designated disposal sites are monitored by the Laboratory's Environmental Assessment Team, this allows MAFF to confirm predicted impacts of disposal and to take action if unanticipated adverse effects do occur. This also contributes to future licensing decisions and overall licensing Policy. The work also enables the UK to show compliance with the Oslo and Paris Conventions (Great Britain - Parliament, 1972(a,b)) which regulate disposal at sea at an international level. Some examples of monitoring investigations and licensing related data are included in this report.

All discharges to tidal waters require a consent under the Water Resources Act (1991) (Gt. Britain - Parliament, 1991). MAFF is a statutory consultee and fishery implications are assessed at Burnham-on-Crouch. This assessment takes into consideration such factors as resources in the area, toxicity of the effluent, hydrography of the area and any standards contained within National Policy, International agreements or EC Directives.

In order to provide a broad enough base for sound advice under these specific duties and to service other less direct responsibilities under a variety of other Acts and Regulations, the Laboratory conducts a more general programme of environmental quality monitoring. Each year, samples of sea water, sediment and marine organisms are collected in which a wide range of chemical determinands are measured. These include, among others, heavy metals, organochlorine pesticides and polychlorinated biphenyls (PCBs). This programme produces valuable background information on the quality of the marine environment, against which areas such as disposal sites can be assessed. It also aims to ensure that where potentially harmful substances occur, concentrations do not reach levels that present a hazard to either marine organisms or human consumers. Most of this work is carried out in accordance with procedures agreed under the auspices of organisations such as the International Council for the Exploration of the Sea (ICES), the Joint Monitoring Group (JMG) of the Oslo and Paris Commissions and, of late, the North Sea Task Force. As such the information contributes to the database on the status of the marine environment around north-western Europe, much of which was, for example, used in the Quality Status Report on the North Sea in 1993 (NSTF, 1993).

All pesticides in the UK have to be approved by the Ministry's Pesticide Safety Directorate and the Advisory Committee on Pesticides (ACP). The Burnham-on-Crouch Laboratory provides advice on risk assessment to the ACP and has conducted extensive research on pesticide leaching and run-off, the data from which are being used to validate improved computer models of leaching and run-off.

In relation to particular measures of control, the Laboratory undertakes monitoring to demonstrate UK compliance with the European Council Directive on mercury discharges and similar requirements under the Paris Commission. It also undertakes work to assess the effectiveness of control measures taken, under Part III of FEPA, 1985, to regulate the use of antifouling paints containing tributyltin (TBT).

The Laboratory's programme of aquatic environmental studies has been developed over more than 25 years. During this period the Laboratory has achieved a number of environmental protection successes and as a consequence of its work has established a worldwide reputation in the field of aquatic environmental research.

BIOTA

1. MONITORING OF CONTAMINANTS IN MARINE FISH AND SHELLFISH

1.1 Introduction

In 1992, sampling was concentrated in those areas around England and Wales where previous monitoring (MAFF, 1990 and 1991) has indicated that the level of some contaminants could be above the range normally anticipated, as indicated by reference to guidelines issued by the Joint Monitoring Group (JMG) of the Oslo and Paris Commission. This group has developed guidelines for ‘lower’, ‘medium’ and ‘upper’ concentration values for contaminants, based on the results submitted by all of the countries which participate in the Joint Monitoring Programme (JMP) of the two Commissions. These, together with other standards/ guidelines which apply in England and Wales for contaminants in fish and shellfish, are summarised in Appendix 2. The presence of a contaminant in the ‘upper’ category does not necessarily imply any risk either to human health or to the environment, but countries are asked to provide monitoring data from affected areas on a biennial basis to ensure that up-to-date information is always available. In 1992, data were required for this purpose on mercury in fish from Liverpool and Morecambe Bays, on PCBs in fish from Liverpool Bay and on cadmium in mussels from the Humber, the Bristol Channel and off Whitehaven; it is generally these data which are given in the tables which follow. Some additional analysis of fish for mercury was carried out to indicate compliance with the Environmental Quality Standard (EQS) for mercury adopted by the European and Paris Commissions.

Fish samples were also obtained from the Humber and Thames to update information on these major estuaries.

Also reported are the latest results from the continuing study of heavy metal concentrations in oysters (*Ostrea edulis*) from the Fal estuary in Cornwall; these oysters are subject to possible metal contamination as a consequence of discharges from the abandoned Wheal Jane mine.

Sites and areas from which fish and shellfish samples were collected are shown in Figure 1.

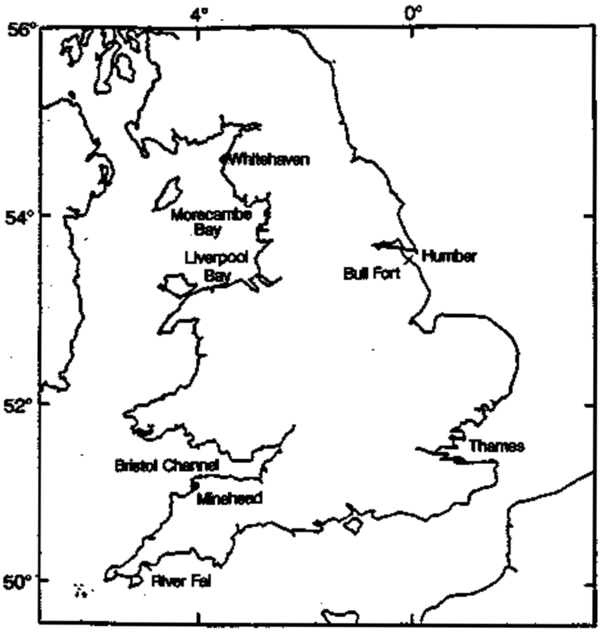

Figure 1. Sites and areas from which fish and shellfish samples were obtained

1.2 Methods

1.2.1 Sampling

With the exception of the special study on oysters in the Fal, samples were obtained from the five internationally agreed ‘core’ monitoring species:- cod (*Gadus morhua*), whiting (*Merlangius merlangus*), dab (*Limanda limanda*), flounder (*Platichthys flesus*) and the blue mussel (*Mytilus edulis*). In addition, samples of plaice (*Pleuronectes platessa*) and sole (*Solea solea*) were also taken and analysed for mercury for the EQS study.

Duplicate analyses were carried out on bulked tissues ¾ fish muscle and liver for metals, fish liver for organochlorine residues and whole soft *Mytilus* body tissue for cadmium.

1.2.2 Analysis

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

Analyses were made of the following organochlorine residues: HCB, a-HCH, g-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. For consistency with the earlier surveys, PCBs have been quantified using the formulation Aroclor 1254 as the standard; a separate report is being prepared to provide data on individual chlorinated biphenyls, commencing with those listed by the International Council for the Exploration of the Seas (ICES) (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 muscle and liver are listed in Table 1, organochlorine residues in fish liver in Table 2, cadmium in mussels in Table 3 and heavy metals in Fal oysters in Table 4. Where duplicate bulked analysis was carried out, generally both results are listed in the tables, in brackets, below the mean. The exception is the time series of results for Fal oysters, where, for the purpose of clarity, only the means have been included.

All concentrations are expressed on a wet tissue weight basis, but 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.

1.3.1 Heavy metals in fish tissue

Mercury
In 1992, concentrations of mercury in the 'upper' JMP category (see Appendix 2) were only found in some species taken from Liverpool Bay. Both Liverpool Bay and Morecambe Bay are areas subject to discharges from chloralkali plants and thus to provisions of EC Council Directive 82/176 (European Communities, 1982). Compliance has also to be demonstrated with an agreed Paris Commission EQS (see Appendix 2). Data are submitted biennially to indicate compliance with an EQS of 0.30 mg mercury per kilogramme of representative wet fish flesh. For this purpose, the mercury concentration of each individual fish species is weighted according to the quantity of that particular species landed from the area in question, and an overall representative mean for the area is then calculated. The weighted mean concentration of mercury calculated for 1992 for Liverpool Bay was 0.13 mg kg^{-1} and for Morecambe Bay 0.12 mg kg^{-1}; the standard was therefore met at both sites.

Copper
Concentrations in fish muscle were within the 'expected' range (Appendix 2) except for dab from the Humber (0.89 mg kg^{-1}) and whiting from the Thames (0.71 mg kg^{-1}) which were slightly above this. The maximum concentration in fish liver (22 mg kg^{-1}) was found in flounder taken from Morecambe Bay. This is slightly above the range of values obtained for this species, in samples collected for a recent JMG survey.

Zinc
Concentrations in fish muscle were all within the 'expected' range (Appendix 2). The maximum concentration in fish liver 46 mg kg^{-1} was found in flounder taken from the Thames Estuary. This is equivalent to the top of the range of concentrations found in the JMG Baseline Survey.

Cadmium
Analysis for this contaminant is not carried out on fish muscle as previous work has indicated that concentrations are extremely low and below the limit of detection of the method of analysis used routinely.

Levels in fish liver were low (maximum 0.28 mg kg^{-1}) and considerably less than those reported for dab taken offshore of the Humber in 1991 which were found to contain up to 0.96 mg kg^{-1} (MAFF, 1993(a)).

Lead
As with cadmium, analysis for this contaminant is only undertaken in fish liver. All values were below the limit of detection of the method employed (i.e. <1.6 mg kg^{-1}).

1.3.2 Organochlorine compounds in fish tissue

Concentrations of organochlorine pesticides found in fish liver (Table 2) were generally within the 'expected' levels listed in Appendix 2. Exceptions were the levels of HCB in cod and whiting liver from the Humber Estuary (0.29 and 0.23 mg kg^{-1} wet weight respectively compared with an 'expected' level of up to 0.1 mg kg^{-1} wet weight), HCH in whiting liver from the Thames Estuary (0.064 and 0.056 mg kg^{-1} wet weight compared with an 'expected' level of up to 0.05 mg kg^{-1} wet weight) and ppTDE in whiting liver from Liverpool Bay (1.2 mg kg^{-1} compared with an 'expected' level of up to 0.5 mg kg^{-1}). The levels of PCBs found in fish liver were generally lower than in previous surveys (MAFF 1990, 1991) with only one, whiting from Liverpool Bay (6.6 mg kg^{-1} wet weight), in an 'upper' JMP category (>5 mg kg^{-1} wet weight).

Analysis of the corresponding muscle tissue was undertaken, where the contaminant level in liver was found to be higher than expected. As in previous surveys, concentrations were very much lower — for pesticides all were at or below 0.001 mg kg^{-1} wet weight and for PCBs, all were in the 'medium' or 'lower' JMP categories, i.e. less than 0.05 mg kg^{-1} wet weight.

1.3.3 Cadmium in mussels

Concentrations of cadmium in the 'upper' JMP category i.e. >5 mg kg^{-1} dry weight (Appendix 2) have previously been found in mussels taken from the Humber and Severn estuaries and in the Whitehaven area (MAFF, 1990 and 1991). Results for samples collected in 1992 are listed in Table 3. No commercial landings are recorded for these areas; the samples are simply used as an index of relative contamination.

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

Area of capture/ species	Date of capture	No. of fish	Mean length (cm)	Tissue	Concentration (mg kg^{-1} wet weight)					Dry matter (%)
					Hg	Cu	Zn	Cd	Pb	
Humber Estuary										
Cod	Jun-92	25	23.4	M	0.03	0.44	3.6	-	-	20
					(0.03; 0.03)	(0.38; 0.49)	(3.6; 3.6)			
				L	<0.03	4.3	18	<0.2	<1.6	51
						(4.0; 4.6)	(18; 18)			
Whiting	Jun-92	18	22.3	M	0.04	0.30	2.9	-	-	20
					(0.03; 0.04)	(0.36; 0.24)	(2.9; 2.8)			
				L	<0.02	3.6	16	<0.2	<1.6	54
Dab	Jun-92	18	23.5	M	0.14	0.89	4.3	-	-	23
					(0.14; 0.13)	(0.86; 0.91)	(4.1; 4.5)			
				L	0.12	5.1	21	0.20	<1.6	34
					(0.11; 0.13)	(4.5; 5.7)	(21; 21)	(0.19; 0.21)		
Flounder	Jun-92	25	25.0	M	0.02	0.43	6.4	-	-	21
					(0.02; 0.02)	(0.41; 0.45)	(5.9; 6.8)			
				L	<0.03	10	30	0.19	<1.6	27
						(10; 10)	(29; 30)	(0.21; 0.16)		
Thames Estuary										
Cod	Nov-92	25	38.6	M	0.08	0.21	3.6	-	-	19
					(0.08; 0.08)	-	(2.9; 4.2)			
				L	0.04	6.3	23	<0.05	<1.6	48
					(0.04; 0.03)	(6.2; 6.3)	(23; 22)			
Whiting	Nov-92	25	32.6	M	0.08	0.71	2.5	-	-	18
					(0.08; 0.08)	(0.74; 0.67)	(2.5; 2.5)			
				L	<0.03	7.4	13	0.05	<1.6	63
						(6.6; 8.2)	(13; 12)	(0.05; 0.04)		
Dab	Oct-92	25	25.4	M	0.19	0.22	4.0	-	-	20
					(0.21; 0.17)	(0.21; 0.23)	(4.1; 3.8)			
				L	0.09	16	41	0.24	<1.6	45
					(0.10; 0.08)	(16; 15)	(43; 38)	(0.27;0.20)		
Flounder	Oct-92	25	30.1	M	0.12	0.20	4.6	-	-	19
					(0.11; 0.13)	(0.19; 0.20)	(4.5; 4.6)			
				L	0.10	15	46	0.18	<1.6	27
					(0.11; 0.08)	(15;15)	(45;46)	(0.18; 0.18)		
Liverpool Bay										
Cod	Apr-92	21	25.4	M	0.13	0.30	3.7	-	-	18
					(0.13; 0.12)	(0.30; 0.30)	(3.9; 3.4)			
				L	0.08	5.4	22	<0.16	<1.6	45
					(0.07; 0.08)	(5.6; 5.2)	(20; 23)			
Whiting	Apr-92	25	32.1	M	0.31	0.24	2.7			16
					(0.31; 0.30)	(0.25; 0.23)	(2.7; 2.7)			
				L	0.09	3.3	16	<0.16	<1.6	51
					(0.09; 0.09)	(3.3; 3.2)	(17; 15)			
Dab	Apr-92	25	22.7	M	0.22	<0.1	4.2	-	-	15
					(0.23; 0.21)	-	(3.8;4.5)			
				L	0.27	9.2	23	0.23	<1.6	23
					(0.27; 0.27)	(9.3; 9.1)	(23; 23)			
Flounder	Mar-92	33	28.9	M	0.31	0.24	5.2	-	-	16
					(0.31; 0.31)	(0.27; 0.21)	(5.4;5.0)			
				L	0.29	17	34	<0.16	<1.6	21
Sole *	Oct-92	25	23.0	M	0.08					
					(0.08; 0.08)					
Plaice *	Sep-92	27	28.3	M	0.13					
					(0.05-0.24)$^{+}$					
Morecambe Bay										
Cod	Oct-92	18	39.2	M	0.12	0.20	3.7	-	-	20
					(0.12; 0.12)	(0.21; 0.19)	(3.5; 3.8)			
				L	0.07	5.3	21	<0.16	<1.6	55
					(0.06; 0.07)	(5.4; 5.1)	(21; 20)			
Whiting	Oct-92	22	30.2	M	0.16	ND	3.4	-	-	19
					(0.14; 0.17)		(3.2; 3.6)			
				L	0.05	3.6	14	<0.16	<1.6	60
					(0.05; 0.04)	(3.6; 3.6)	(14; 14)			
Dab	Oct-92	14	28.9	M	0.20	<0.1	4.4	-	-	24
					(0.18; 0.22)		(4.7; 4.0)			
				L	0.08	12	29	<0.16	<1.6	47
					(0.08; 0.08)	(11; 12)	(27; 31)			
Flounder	Oct-92	25	31.5	M	0.19	0.24	4.9	-	-	21
					(0.19; 0.19)	(0.23; 0.24)	(4.5; 5.2)			
				L	0.12	22	39	0.28	<1.6	34
					(0.15; 0.09)	(19; 24)	(37; 41)	(0.26; 0.29)		
Sole *	Oct-92	45	24.4	M	0.09					
					(0.08; 0.09)					
Plaice *	Oct-92	24	26.4	M	0.08					
					(0.08; 0.08)					

Note: * *EQS samples: analysed for mercury only*
$^{+}$*Analysed individually, since also used for trend purposes*
ND= Not determined

Table 2. Concentrations of organochlorine residues in fish liver (expressed as the mean, with the results from duplicate bulked analysis shown in brackets underneath)

Area of capture	Species	Concentration (mg kg^{-1})			
		HCB	a-HCH	g-HCH	Dieldrin
Humber Estuary	Cod	0.29	0.029	0.025	0.30
		(0.28, 0.29)	(0.029, 0.029)	(0.025, 0.025)	(0.29, 0.30)
	Whiting	0.23	0.031	0.026	0.23
		(0.21, 0.25)	(0.030, 0.031)	(0.026, 0.026)	(0.22, 0.23)
	Dab	0.055	0.012	0.010	0.072
		(0.055, 0.055)	(0.012, 0.012)	(0.009, 0.010)	(0.069, 0.075)
	Flounder	0.052	0.006	0.005	0.056
		(0.051, 0.053)	(0.006, 0.006)	(0.005, 0.005)	(0.053, 0.059)
Thames Estuary	Cod	0.006	0.023	0.026	0.079
		(0.005, 0.006)	(0.023, 0.023)	(0.025, 0.026)	(0.074, 0.084)
	Whiting	0.014	0.064	0.056	0.11
		(0.013, 0.014)	(0.064. 0.064)	(0.056, 0.056)	(0.11, 0.11)
	Dab	0.006	0.022	0.019	0.037
		(0.006, 0.006)	(0.021, 0.022)	(0.019, 0.019)	(0.035, 0.038)
	Flounder	0.002	0.006	0.005	0.011
		(0.002, 0.002)	(0.006, 0.006)	(0.005, 0.005)	(0.01, 0.011)
Liverpool Bay	Cod	0.009	0.02	0.017	0.066
		(0.008, 0.009)	(0.02, 0.02)	(0.016, 0.017)	(0.061, 0.070)
	Whiting	0.026	0.039	0.030	0.14
		(0.025, 0.027)	(0.039, 0.039)	(0.029, 0.030)	(0.13, 0.14)
	Dab	0.003	<0.001	0.005	0.016
		(0.003, 0.003)	(<0.001, <0.001)	(0.005, 0.005)	(0.016, 0.016)
	Flounder	0.01	<0.001	0.006	0.025
		(0.009, 0.01)	(<0.001, <0.001)	(0.006, 0.006)	(0.023, 0.026)
Morecambe Bay	Cod	0.002	0.026	0.02	0.045
		(0.002, 0.002)	(0.025, 0.026)	(0.02, 0.02)	(0.043, 0.047)
	Whiting	0.005	0.062	0.049	<0.001
		(0.004, 0.005)	(0.062, 0.062)	(0.048, 0.05)	(<0.001, <0.001)
	Dab	0.008	0.031	0.021	0.018
		(0.008, 0.008)	(0.030, 0.031)	(0.021, 0.021)	(0.018, 0.021)
	Flounder	0.001	<0.001	0.011	<0.001
		(0.001, 0.001)	(<0.001, <0.001)	(0.01, 0.011)	(<0.001, <0.001)

Area of capture	Species	Concentration (mg kg^{-1})				Lipid (%)
		ppDDE	ppTDE	ppDDT	PCB	
Humber Estuary	Cod	0.24	0.44	0.059	1.9	38
		(0.24, 0.24)	(0.43, 0.44)	(0.057, 0.061)	(1.9, 1.9)	(37, 38)
	Whiting	0.16	0.26	0.051	1.3	45
		(0.16, 0.18)	(0.25, 0.27)	(0.050, 0.052)	(1.2, 1.4)	(44, 45)
	Dab	0.70	0.083	0.018	0.67	17
		(0.68, 0.71)	(0.083, 0.083)	(0.017, 0.018)	(0.66, 0.67)	(17, 17)
	Flounder	0.029	0.090	0.011	0.31	7
		(0.029, 0.029)	(0.086, 0.093)	(0.011, 0.011)	(0.30, 0.31)	(7, 7)
Thames Estuary	Cod	0.12	0.04	0.024	1.9	27
		(0.10, 0.13)	(0.04, 0.04)	(0.024, 0.024)	(1.7, 2.0)	(26, 27)
	Whiting	0.21	0.13	0.081	3.1	52
		(0.20, 0.22)	(0.13. 0.13)	(0.079, 0.082)	(2.9, 3.3)	(49, 54)
	Dab	0.050	0.027	0.018	0.60	33
		(0.046, 0.053)	(0.027, 0.027)	(0.018, 0.018)	(0.59, 0.61)	(32, 34)
	Flounder	0.025	0.013	0.007	0.30	8
		(0.025, 0.025)	(0.012, 0.013)	(0.006, 0.007)	(0.30, 0.30)	(8, 8)
Liverpool Bay	Cod	0.17	0.33	0.032	2.7	24
		(0.16, 0.18)	(0.31, 0.35)	(0.030, 0.034)	(2.6, 2.7)	(23, 24)
	Whiting	0.50	1.2	0.14	6.6	43
		(0.48, 0.51)	(1.1, 1.3)	(0.14, 0.14)	(6.3, 6.9)	(42, 43)
	Dab	0.038	0.041	0.01	0.99	6
		(0.037, 0.038)	(0.040, 0.042)	(0.01, 0.01)	(0.97, 1.0)	(5, 6)
	Flounder	0.043	0.17	0.027	0.60	5
		(0.042, 0.044)	(0.15, 0.18)	(0.026, 0.027)	(0.59, 0.60)	(5, 5)
Morecambe Bay	Cod	0.091	0.12	0.028	1.5	32
		(0.089, 0.093)	(0.12, 0.12)	(0.027, 0.028)	(1.5, 1.5)	(32, 32)
	Whiting	0.13	0.27	0.059	2.6	52
		(0.11, 0.15)	(0.27, –)	(0.056, 0.062)	(2.3, 2.8)	(52, 51)
	Dab	0.053	0.051	0.021	0.81	34
		(0.053, 0.053)	(0.051, 0.051)	(0.020, 0.021)	(0.78, 0.84)	(34, 34)
	Flounder	0.022	0.049	0.014	0.40	17
		(0.023, 0.020)	(0.049, 0.048)	(0.014, 0.013)	(0.41, 0.39)	(17, 17)

Table 3. Concentrations of cadmium in the blue mussel (Mytilus edulis) (JMP upper level >5.0 mg kg^{-1} dry weight)

Area	Date of capture	Number of mussels	Mean length (cm)	Concentration in whole bulked tissue (mg kg^{-1} wet weight)	Concentration in whole bulked tissue (mg kg^{-1} dry weight)
Humber Estuary (Bull Fort)	Feb-92	50	4.3	0.66 (0.67; 0.65)	5.8 (5.6; 5.9)
	Jul-92	50	4.5	0.57 (0.56; 0.58)	3.2 (3.1; 3.2)
Bristol Channel (Minehead)	Sep-92	50	4.0	1.4 (1.4; 1.4)	12.9 (11.7; 14.0)
Whitehaven	Aug-92	50	4.8	2.3 (2.4; 2.2)	14.4 (15.0; 13.8)

Table 4. Fal oysters - Time series of contaminants

Date sample taken	Mean length (cm)	Concentration (mg kg^{-1} wet weight)							
		Cd	Cr	Cu	Hg	Ni	Pb	Zn	Fe
Parsons Bank									
Oct-91	6.6	0.25	0.55	95	0.01	<0.5	1.00	540	83
Jan-92*	7.4	0.42	<0.5	149	<0.01	<0.5	<0.6	780	63
Feb-92*	7.6	0.48	<0.5	156	0.02	<0.5	<0.6	612	96
Mar-92*	7.3	0.35	<0.6	158	0.02	<0.6	<0.7	910	96
Apr-92	7.3	0.38	<0.6	133	0.02	<0.6	<0.7	758	51
May-92	7.6	0.47	<0.7	164	<0.01	1.00	<0.6	938	54
Jun-92	6.8	0.42	0.7	129	<0.01	<0.6	<0.7	1006	46
Jul-92	6.9	0.42	0.7	115	<0.01	<0.6	<0.7	948	81
Aug-92	7.3	0.47	<0.6	131	<0.01	<0.6	<0.7	1030	78
Sep-92*	6.8	0.39	<0.6	68	<0.03	<0.6	<0.7	662	44
Oct-92	6.4	0.28	0.9	61	<0.01	<0.6	<0.6	623	66
Nov-92	6.6	0.27	<0.6	97	<0.01	<0.6	<0.6	1063	45
Dec-92	7.7	0.25	0.7	110	<0.01	<0.6	<0.7	651	46
Jan-93	6.3	0.32	0.7	137	<0.01	<0.5	<0.6	812	32
Feb-93	5.8	0.33	1.0	150	<0.01	0.5	<0.6	1010	48
Mar-93	8.1	0.36	0.3	92	0.01	0.18	0.28	662	77
May-93	7.7	0.46	0.9	145	<0.01	0.39	0.44	948	107
Jul-93	7.0	0.49	0.2	97	0.01	<0.3	0.23	1000	61
Sep-93	7.3	0.38	0.5	94	<0.01	<0.7	<0.7	834	51
Nov-93	6.8	0.21	<0.5	62	<0.01	<0.3	<0.6	724	60
Messack Point									
Oct-91	6.1	0.24	0.98	119	0.02	<0.5	<0.6	584	102
Jan-92*	6.7	0.42	<0.6	106	<0.01	<0.5	<0.6	603	36
Feb-92*	7.0	0.46	<0.5	115	0.02	<0.5	<0.6	610	52
Mar-92*	7.1	0.35	<0.6	102	<0.02	<0.7	<0.7	804	67
Apr-92	7.2	0.34	<0.6	107	0.02	<0.6	<0.7	704	43
May-92	6.7	0.50	<0.6	129	<0.01	<0.6	<0.7	897	45
Jun-92	6.7	0.47	<0.6	136	<0.01	<0.6	<0.7	987	33
Jul-92	7.1	0.42	<0.6	126	<0.01	<0.6	<0.7	991	39
Aug-92	7.0	0.47	<0.6	133	<0.01	<0.6	<0.6	1098	59
Sep-92*	7.0	0.41	<0.5	85	<0.03	<0.5	<0.6	787	48
Oct-92	6.5	0.30	<0.5	67	<0.01	<0.5	<0.6	645	44
Nov-92	6.5	0.32	<0.5	74	<0.01	<0.5	<0.6	501	39
Dec-92	6.7	0.24	0.6	100	<0.01	<0.6	<0.7	536	24
Jan-93	6.4	0.30	0.7	114	<0.01	<0.6	<0.7	692	28
Feb-93	6.4	0.31	0.8	136	<0.01	<0.5	<0.6	958	62
Mar-93	6.7	0.36	0.42	86	0.01	0.2	0.23	524	55
May-93	7.0	0.42	0.75	146	<0.01	0.24	0.32	755	90
Jul-93	6.8	0.41	0.21	102	<0.01	<0.3	0.20	787	63
Sep-93	6.9	0.31	0.38	104	<0.01	<0.7	<0.7	636	39
Nov-93	6.8	0.28	<0.5	66	<0.01	<0.3	<0.6	633	80

*Note: * = mean of 2 or more samples*

Concentrations in samples from the Severn and Whitehaven remained in the 'upper' JMP category, though the value for Whitehaven is the lowest recorded by MAFF. Over the last few years, the cadmium concentrations in Humber mussel samples collected in the summer (June/July) have fallen to the JMP 'medium' category i.e. 2-5 mg kg^{-1} dry weight (MAFF, 1991 and 1992). Although the concentration in a sample of mussels collected in February 1992 was just in the 'upper' category, the concentration in the July sample was however, again in the 'medium' category.

1.3.4 Heavy metals in oysters from the River Fal

In November 1991, the Restronguet Creek area of the River Fal was subject to a large discharge of water contaminated with heavy metals, as a result of the closure and abandonment of the nearby Wheal Jane tin mine. Since this time, water from the mine has continued to overflow into the area, although treatment is applied in the form of lime dosing and settlement, so as to minimise the quantity of metal reaching the estuary. MAFF carried out a pre-discharge survey in expectation of the overflow and, since its occurrence, has conducted a regular sampling programme to monitor heavy metal levels in oysters (*Ostrea edulis*), the main commercially produced shellfish in the area.

Samples of oysters were taken from two main stock areas, Parsons Bank and Messack Point. Sampling frequency varied, being greatest (every two weeks) between January-March 1992 when the discharges were at a peak; at the time of writing (October, 1993), samples are collected every two months.

Results for the first year of monitoring were included in an earlier report in this series (MAFF, 1993(a)). Further results are now available and the time series of results for two years of monitoring are shown in Table 4.

Concentrations of Cr, Ni, Hg and Pb were generally all below the limits of detection of the method of analysis used (see Section 1.2.2). Concentrations of Cu and Fe showed some fluctuations but there was no significant increase compared with levels prior to the event. Concentrations of Cd increased at the beginning of 1992 to 0.4-0.5 mg kg^{-1} wet weight and remained around this level throughout the summer period (Figure 2). In September 1992, concentrations began to decrease falling back to pre-discharge levels (0.25 mg kg^{-1}) by December 1992. In January 1993, concentrations began to rise again reaching 0.4-0.5 mg kg^{-1} by July; these subsequently fell again to 0.2-0.3 mg kg^{-1} in November 1993.

Concentrations of Zn also began to increase at the beginning of 1992 reaching levels of ~1000 mg kg^{-1} wet weight by August. As with Cd, concentrations began to decrease in September falling to ~600 mg kg^{-1} by December 1992. Concentrations began to increase again in January 1993 reaching the previous level of ~1000 mg kg^{-1} by February. However, by November 1993 they had again fallen back to ~600-725 mg kg^{-1}.

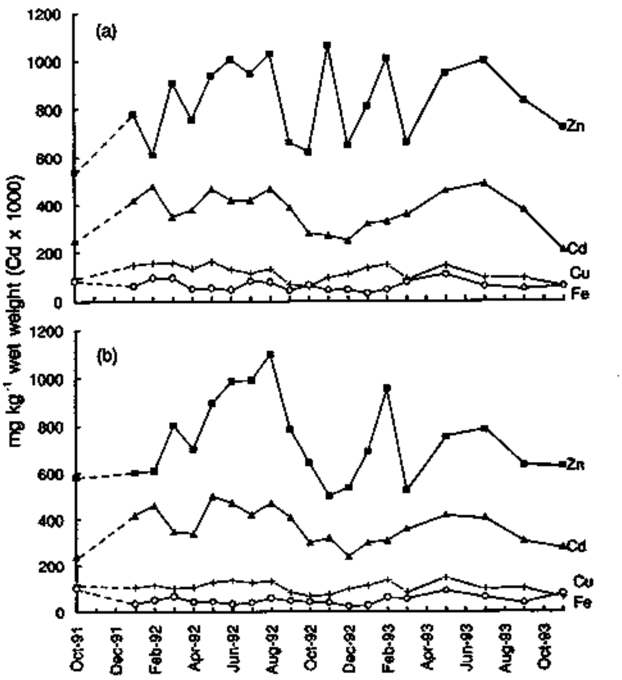

Figure 2. Time series of concentrations of metals in oysters from the Fal Estuary: (a) Parsons Bank; (b) Messack Point

1.4 Conclusions

The level of mercury in some fish species taken from Liverpool Bay and cadmium in mussels taken from non-commercial populations in the Humber, Bristol Channel and Whitehaven remain relatively high and regular monitoring will be continued. However, results indicate that the EQS for mercury in fish flesh was again met in Liverpool and Morecambe Bays.

The levels of organochlorine compounds found give little indication of serious contamination in the major estuaries of England and Wales though, for specific contaminants, concentrations above the 'expected' range continue to occur in some areas and regular monitoring will continue in these areas.

In no case do the levels of contaminants found in commercially exploited fish or shellfish give cause for concern relative to human health.

Monitoring, over the past two years, of heavy metals in oysters near the Wheal Jane overflow has not indicated any significant overall increase in concentrations. It seems quite probable that the results reflect an annual cycle related to changes in body weight, with a fairly constant total body burden, i.e. the variations seen are probably not related to short-term fluctuations in inputs, but to the long-term historical contamination of the area. Throughout 1992 and 1993, MAFF was able to advise that there was no cause for concern with regard to the consumption of oysters. Monitoring will continue, though it is anticipated that the frequency of sampling will be further reduced, unless the situation alters significantly.

2. THE USE OF BIOASSAYS TO ASSESS THE QUALITY OF MARINE WATER AND SEDIMENT

2.1 General introduction

In 1990 and 1991, the oyster embryo and algal growth bioassays were used on MAFF research cruises to monitor the biological water and sediment quality in the North Sea and English Channel (MAFF, 1993(a)). The aim of this work was to provide data conducted in accordance with the North Sea Task Force Monitoring Master Plan (NSTF, 1990). In 1992, these bioassays were again deployed on a MAFF research cruise CIROLANA 7, 26 June-19 July 1992 (see Section 4), together with whole sediment bioassays, with the aim of providing data for the UK National Monitoring Programme.

2.2 Algal growth bioassay

2.2.1 Introduction

The algal growth bioassay is a five day test to assess the potential for algal growth in natural seawater samples.

Chemical analysis is often used to describe the nutrient status of marine waters. The use of this bioassay complements such nutrient analysis and gives an indication of the biological significance of the nutrients present and whether growth is being stimulated or suppressed by the presence of contaminants.

2.2.2 Method

The test methodology has been described in two previous reports in this series (MAFF, 1992 and 1993). Briefly, two species of algae, the brown flagellate *Isochrysis galbana* and the green flagellate *Tetraselmis suecica* were individually inoculated into seawater samples, taken from 0.5-1 m below the surface, at a concentration of 5 x 10^4 cells ml^{-1}. After 5 days incubation on a shaking table in constant light conditions, the algal cell densities were estimated spectrophotometrically.

2.2.3 Results and discussion

Figure 3 shows the cell growth for *T. suecica*, together with the concentrations of nutrients present, in samples from the estuaries of the Tweed, Tyne, Wear, Tees and Humber. Samples were taken from the upper estuary to open sea locations along a transect out to the Dogger Bank.

In all five estuaries growth was highest at the uppermost sampling site followed by a gradual decline in cell growth towards the mouth of each estuary. Algal growth was similar in samples from intermediate offshore and Dogger Bank locations. Comparing estuaries, growth was highest in the Tees, similar in the Tyne, Wear and Humber and lowest in the Tweed; cell density in the upper estuary sample in the Tweed was 50% of the cell density of the upper Tees sample. As the Tweed is considered to be a relatively natural estuary the higher growth rates observed in the Tees presumably reflect growth promotion due to nutrients.

A similar pattern of growth was measured for *I. galbana* (Figure 4). However, in the upper Tweed the algal cell concentration was only 25% of that measured in the Tees.

The poor growth exhibited by *T. suecica* in one of the middle estuary sites on the Tyne is not what would be expected from the concentration of key nutrients present in the water. Water chemistry data available for the Tyne (see Sections 6, 7, 8 and 11) show that neither the concentrations of dissolved metals nor the four groups of organic chemicals measured are likely to have produced the observed growth inhibition. This result suggests that another group (or groups) of chemicals must be present which is not consistently analysed for. This confirms the need to conduct bioassays alongside chemical analysis of samples, since a particular assay will respond to all the chemicals present in the sample acting jointly upon the biological system. The bioassay can therefore be used in this way to focus upon the samples which require more detailed analysis. Such analysis could in turn lead to the identification of the hitherto unidentified pollutants and eventual controls on their release.

2.3 Oyster embryo bioassay

2.3.1 Introduction

This bioassay was used in 1990 in accordance with the requirements of the North Sea Task Force Monitoring Master Plan (NSTF, 1990) and in 1991 and 1992 as part of the UK National Monitoring Programme. The results reported here were obtained during a research cruise in 1992.

2.3.2 Method

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

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

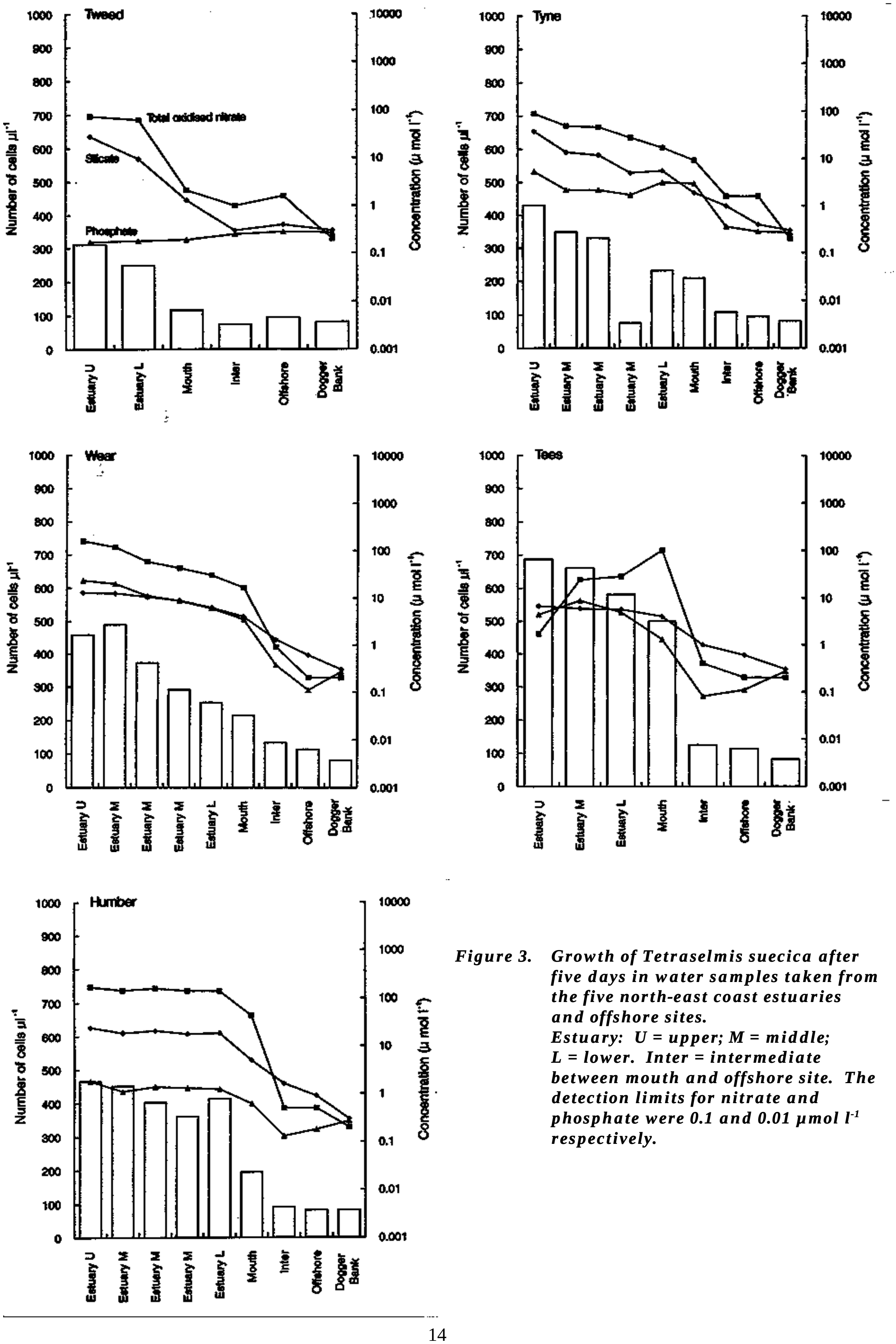

Figure 3. Growth of Tetraselmis suecica after five days in water samples taken from the five north-east coast estuaries and offshore sites. Estuary: U = upper; M = middle; L = lower. Inter = intermediate between mouth and offshore site. The detection limits for nitrate and phosphate were 0.1 and 0.01 µmol l^{-1} respectively.

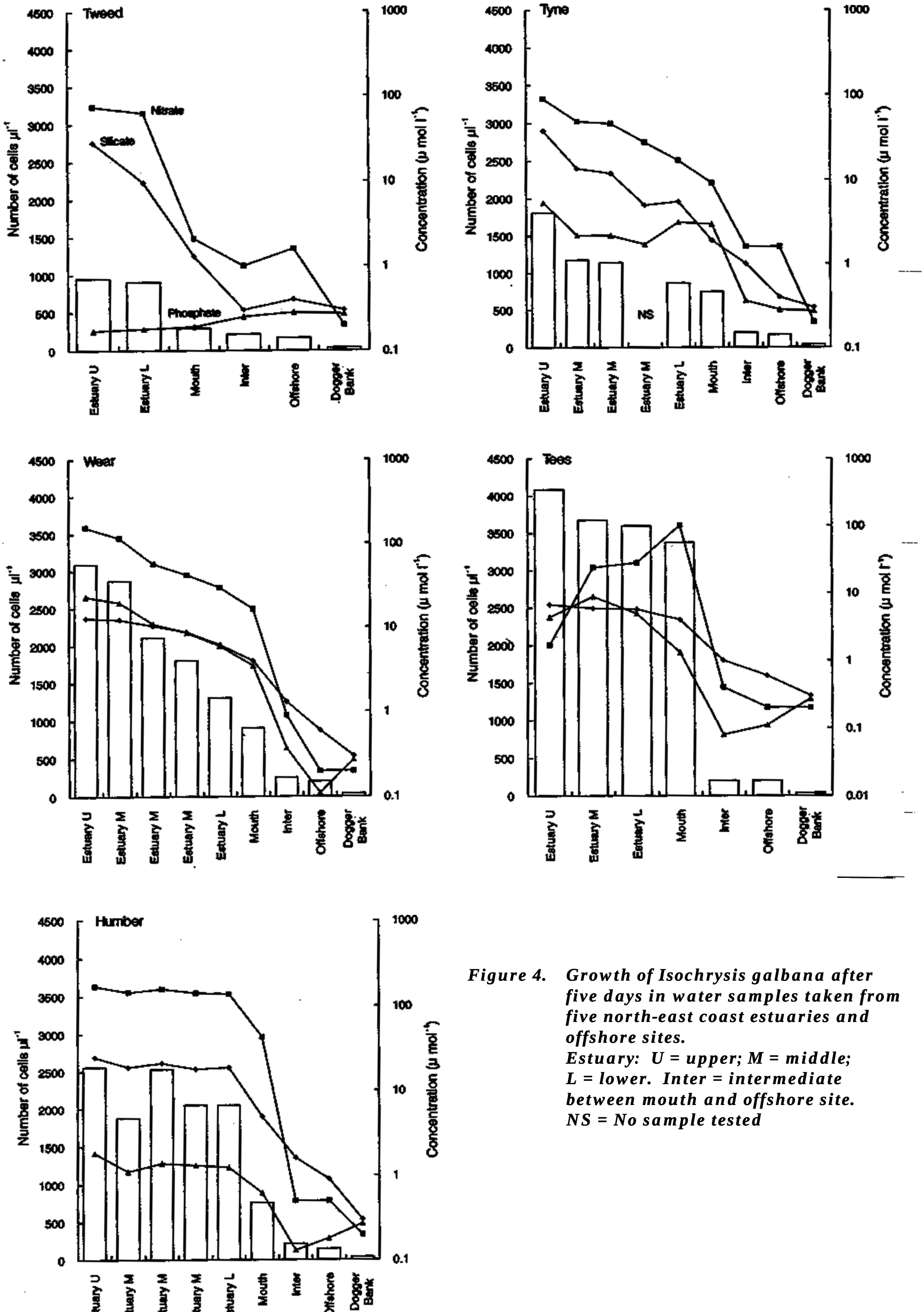

Figure 4. Growth of Isochrysis galbana after five days in water samples taken from five north-east coast estuaries and offshore sites. Estuary: U = upper; M = middle; L = lower. Inter = intermediate between mouth and offshore site. NS = No sample tested

2.3.3 Results and discussion

Sampling stations are given in Table 5: Latitude and longitude positions (see Section 4, Table 8) show the precise location of the sampling stations. Each result is expressed as a Percent Net Response (PNR)

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

A value of 0 or close to 0 indicates the measured response of the sample is similar to that of the controls. A negative value indicates the water quality is better than the control. Poor water quality is defined when PNR values (usually >10) are statistically different from the reference sea water. A PNR value of 100 occurs when the sample is highly toxic and all the embryos have either died or shown abnormal development over the 24 h exposure period.

Poor water quality was not detected at any of the stations. Sediment elutriates were found to be toxic at three locations; on the River Tees at Redcar jetty and No. 25 buoy (PNR values of 100 and 85.2 respectively) and on the River Wear at Wearmouth Bridge (PNR value of 27). Similar results for the sediment at Redcar jetty (River Tees) were obtained in both 1990 and 1991.

2.4 Whole sediment bioassays

2.4.1 Introduction

In previous surveys, sediment toxicity has been measured by exposing oyster embryos to sediment elutriates. For many reasons it is not the preferred technique and serious consideration has been given to developing and deploying whole sediment bioassays i.e. with animals that live in and re-work sediments. In 1992, one such bioassay was deployed for the first time on-board a MAFF research vessel and used alongside the oyster embryo bioassay.

2.4.2 Arenicola marina bioassay

This technique uses the annelid polychaete *Arenicola marina* ¾ the common lugworm. Animals of approximately 1 g in size are collected from the foreshore and maintained in tanks containing a layer of sand and running sea water until ready for use. For each assay replicate sediment samples of 2 kg in weight are placed in small tanks (polythene sandwich boxes) with 4 cm of static overlying water and aeration. Five worms are placed into each tank and the worms left to burrow into the sediment. After 10 days the contents of each tank are sieved and the number of surviving worms recorded. During the exposure period observations on feeding behaviour are recorded by making daily counts of the casts on the surface of the sediment.

2.4.3 Results and discussion

The results are shown in Table 5. Sediments from Redcar jetty and No. 25 buoy on the River Tees were toxic to *A. marina*, 100% and 30% mortality respectively. Twenty seven casts were recorded in the control sediment and statistically significant (P<0.05) disruption in feeding activity was recorded for 10 of the sediments. This obviously occurred when the lugworms were dying, but also at the River Tyne Jarrow Slake site where mortality was 0 and the number of casts was 0. Although feeding behaviour of *A. marina* may be used to measure sub-lethal toxicity it is important that further work is carried out on this end point. Substrate type and the degree of anoxia do not appear to affect casting, but like many behavioural responses the casting activity may be variable. Thus a small change in feeding activity may be statistically different from a control, but in practice may not signify an adverse sub-lethal effect. Nevertheless, this end point shows promise, as was demonstrated at Jarrow Slake.

Table 5. Bioassay results for water and sediment samples collected in 1992, during CIROLANA 7, 26 June-19 July 1992

Station	Location	Oyster Bioassay		*Arenicola marina* % mortality	Total casts in 10 days
		Water PNR	Sediment PNR		
1	River Humber : Bull anchorage	-3.6	-7.1	0	23
2	River Humber : Pyewipe out fall	5.5	10.5	0	23
3	River Humber : Diffuser	-7.7	3.3	0	18*
4	River Humber : No. 6B buoy	8.7	5.9	0	17*
5	River Humber : No. 10A buoy	-3.0	-0.1	0	12*
6	River Humber : No. 11A buoy	3.3	-1.1	0	11*
16	River Tees : Redcar jetty	-0.4	100.0	100	0*
18	River Tees : No. 25 buoy	-5.5	85.2	30	12*
19	Off River Tees	7.7	5.5	0	27
21	River Wear : Between South Piers	4.9	2.7	0	21
23	River Wear : Wearmouth Bridge	-0.1	27.0	0	34
24	River Wear : Hetton Staiths	5.9	7.1	0	19*
25	River Wear : Deptford Quay	-4.2	6.8	0	25
26	River Wear : Queen Alexandra Bridge	5.5	-0.1	0	21
27	River Tyne (mouth)	-5.5	7.4	0	13*
28	River Tyne : Lloyd's Hailing Station	0.0	11.2	0	25
29	River Tyne : buoy off Jarrow Slake	-4.2	13.4	0	0*
31	River Tyne : Ouse Burn	3.7	11.5	10	16*
33	River Tyne : Team Confluence	-1.1	15.0	0	25
34	Off River Tyne	5.2	-6.7	0	21
39	Dogger Bank	-4.8	-3.2	0	27

PNR = Percent Net Response
** Disruption in feeding behaviour: significantly different from the control (P<0.05)*

Initial results indicate the usefulness of the bioassay: It is inexpensive to deploy; test animals are available throughout the year; and it is sensitive with acute and sub-lethal endpoints. Further evaluation will be carried out during the coming year on the suitability of the test for monitoring, assessment of dredge spoil toxicity and for toxicity testing of pure compounds spiked into sediments.

2.5 Conclusions

The algal and oyster embryo bioassays continue to provide easily applicable techniques for shipboard assessment of biological water quality. The oyster bioassay still appears to be of value in assessing sediment quality even though there are difficulties in interpreting results produced by elutriation techniques. The *A. marina* bioassay showed a response to three sediments, two of which also gave a positive response with the oyster embryo bioassay. This shows the value of using a battery of test species when conducting bioassays as not all species are sensitive to the same contaminants. The whole sediment bioassay was suitable for ship-board deployment but requires further evaluation in terms of sensitivity and ease of application for monitoring purposes.

3. SURVEYS OF CONTAMINANTS IN MARINE MAMMALS

Early in 1992, there was a mass mortality of dolphins in the south-west of England, with over one hundred animals stranding on the coasts of Devon and Cornwall. Detailed *postmortem* studies were conducted on the carcases of thirty-eight common dolphins (*Delphinus delphis*) to determine the cause of death. The results of these examinations and subsequent analyses were compared with those of twenty common dolphins stranded in the same area over the previous fifteen months. There was no evidence that death was caused by any infectious or parasitic disease, or as a result of acute intoxication with algal toxins, trace metals or organochlorine compounds measured. (The metals and organochlorines data are presented in Tables 6 and 7 respectively). Table 7 gives information on chlorinated organics in eleven dolphins from the mass stranding incident; metals data for these animals are included in Table 6, but this also shows data for other animals which were stranded in 1992. None of the concentrations of contaminants, either individually or collectively, were unusually high and the study concluded that the dolphins died from entanglement in nets used in fisheries for mackerel or pilchards, on which the dolphins were also feeding (Kuiken *et al.*, submitted). Better estimates of mortality of marine mammals in fisheries conducted around the UK are needed in order to assess the current status of local populations (MAFF, 1993(a)), in particular of small cetaceans (dolphins and porpoises) whose populations may be declining (Evans and Scanlan, 1988).

Of the results for trace metals given in Table 6, those for arsenic are of particular interest, as no comparable data have previously been presented for the UK. Fish and marine invertebrates (especially crustaceans) can contain quite high arsenic concentrations, often as much as 100 mg kg^{-1}. Almost all of this arsenic is in the form of organoarsenical compounds, which are relatively non-toxic. The concentrations of arsenic found in the twelve common dolphin livers analysed ranged from 0.51 to 2.6 mg kg^{-1} wet weight, similar to the concentrations reported for pilot whales (*Globicephala melas*) from the Færoe Islands (0.28 to 1.8 mg kg^{-1}; Julshamn *et al.*, 1987) and Newfoundland (Canada) (0.01 to 2.9 mg kg^{-1}; Muir *et al.*, 1988).

Table 6. Concentrations of metals (mg kg^{-1} wet weight) in the livers of marine mammals stranded in 1992

Location	Date	Species§	Sex	Age	Dry %	Cr	Ni	Cu	Zn	As	Se	Cd	Hg	Pb
Llanon, West Wales	25.3.92	G	F		26.1	0.67	< 0.53	3.6	33	NA	37	0.15	65	0.05
Borth, West Wales	17.1.92	P	F	J	28.4	< 0.51	< 0.51	9.6	35	NA	3.7	< 0.06	2.5	0.08
Tywyn, West Wales	15.5.92	P	M		20.6	< 0.5	< 0.5	2.9	17	NA	2.8	0.15	0.83	0.04
Tywyn, West Wales	29.7.92	P	M		22.2	< 0.53	< 0.53	5.0	23	NA	17	0.84	1.2	0.09
Pembrey, South Wales	4.8.92	P	M		30.7	1.9	< 0.58	10	28	NA	17	< 0.07	31	0.12
Looe Beach, Cornwall	2.1.92	CD	F	J	22.1	< 0.56	< 0.56	2.7	21	0.86	NA	2.7	3.1	0.12
Penzance, Cornwall	7.1.92	CD	M	J	26.2	0.74	< 0.67	4.9	32	0.7	NA	0.16	11	0.05
Porthleven Harbour, Cornwall	8.1.92	CD	F	J	25.1	< 0.68	< 0.68	5.3	37	0.91	NA	0.12	7.4	0.05
Praa Sands, Cornwall	14.1.92	CD	M	J	30.5	< 0.57	< 0.57	4.2	24	0.73	NA	0.14	6.1	0.04
Penzance, Cornwall	2.2.92	CD	F	J	27.9	< 0.65	< 0.65	5.6	32	0.89	3.8	0.72	6.4	0.12
Porthtowan, Cornwall	2.2.92	CD	F	J	27.6	< 0.57	< 0.5	4.1	64	2.6	2.2	0.48	2.8	0.02
St. Keverne, Cornwall	2.2.92	CD	M	J	26.7	< 0.52	< 0.52	5.1	30	0.51	8.5	0.94	24	0.05
Falmouth, Cornwall	7.2.92	CD	F	J	29.2	< 0.63	< 0.63	4.6	34	1.6	2.5	0.6	2.7	0.14
Falmouth, Cornwall	7.2.92	CD	F	J	24.9	< 0.61	< 0.61	6.0	36	1.5	4.1	0.65	8.1	0.03
Bigbury, Devon	13.2.92	CD	F	J	26.6	0.64	< 0.62	5.1	34	1.4	NA	0.47	0.3	0.06
Bigbury, Devon	13.2.92	CD	F	J	25.3	< 0.62	< 0.62	5.7	27	0.85	NA	0.47	5.6	0.09
Kingsbridge, Devon	13.2.92	CD	F	J	26.7	< 0.54	< 0.54	4.7	31	0.77	NA	0.72	11	0.15
Newport, Fishguard, West Wales	3.8.92	CD	F	A	30.7	0.81	< 0.62	6.5	24	NA	18	0.66	50	0.11
Newport, Fishguard, West Wales	3.8.92	CD	F	C	34.5	1.0	< 0.57	5.3	37	NA	1.7	< 0.07	1.5	0.1
Harlech, North Wales	19.2.92	BD	M		33.4	< 0.52	< 0.52	8.1	37	NA	81	0.09	191	0.61

Note: §: G grey seal (Halichoerus grypus)
CD common dolphin (Delphinus delphis)
A Adult
C Calf
P porpoise (Phocoena phocoena)
BD bottlenose dolphin (Tursiops truncatus)
J Juvenile
NA not analysed

Table 7. Concentrations of organochlorine contaminants (mg kg^{-1}) in the blubber of sub-adult dolphins stranded in Devon and Cornwall, Jan-Feb 1992

Location	Date found	Sex	%HEL	CB#18	CB#31	CB#28	CB#52	CB#49	CB#47	CB#44	CB#66	CB#101	CB#110
Looe Beach, Cornwall	2.1.92	F	82	0.056	nd §	nd	1.8	0.32	0.56	0.049	2.0	1.4	0.49
Penzance, Cornwall	7.1.92	M	80	nd	nd	nd	0.91	0.21	0.27	nd	1.1	0.87	0.24
Porthleven Harbour, Cornwall	8.1.92	F	84	nd	nd	nd	1.3	0.22	0.38	0.031	1.5	1.3	0.32
Praa Sands, Cornwall	14.1.92	M	84	nd	nd	nd	0.91	0.17	0.26	nd	1.0	0.75	0.26
Penzance, Cornwall	2.2.92	F	92	nd	nd	nd	0.75	0.25	0.22	nd	1.0	0.89	0.25
Porthtowan, Cornwall	2.2.92	F	88	nd	nd	nd	0.39	0.43	0.11	0.03	0.58	0.83	0.14
St. Keverne, Cornwall	2.2.92	M	64	nd	nd	nd	0.48	0.15	0.14	nd	0.62	0.39	0.2
Falmouth, Cornwall	7.2.92	F	90	nd	nd	nd	0.32	0.27	0.09	nd	0.5	0.38	0.13
Falmouth, Cornwall	7.2.92	F	92	nd	nd	nd	0.28	0.22	0.07	nd	0.41	0.49	0.09
Bigbury, Devon	13.2.92	F	84	nd	nd	nd	0.33	0.34	0.1	nd	0.49	0.61	0.11
Bigbury, Devon	13.2.92	F	86	nd	nd	nd	0.47	0.14	0.12	nd	0.6	0.51	0.14

Location	Date found	CB#151	CB#149	CB#118	CB#153	CB#105	CB#141	CB#138	CB#158	CB#187	CB#183
Looe Beach, Cornwall	2.1.92	3.3	7.5	1.9	17	1.2	0.11	14	0.72	5.7	1.9
Penzance, Cornwall	7.1.92	1.4	3.7	0.97	7.9	0.68	0.077	6.6	0.33	2.7	0.87
Porthleven Harbour, Cornwall	8.1.92	2.0	4.9	1.5	11	0.88	0.11	9.1	0.49	3.9	1.3
Praa Sands, Cornwall	14.1.92	1.4	3.6	1.1	8.5	0.68	0.05	6.9	0.34	3.0	0.98
Penzance, Cornwall	2.2.92	1.5	4.0	1.1	9.2	0.69	0.093	7.2	0.34	3.3	1.0
Porthtowan, Cornwall	2.2.92	0.68	2.0	1.2	4.2	0.28	0.12	3.1	0.15	1.4	0.41
St. Keverne, Cornwall	2.2.92	1.2	2.4	0.59	8.1	0.44	0.025	5.8	0.27	3.1	1.0
Falmouth, Cornwall	7.2.92	0.79	1.9	0.71	4.7	0.35	0.029	3.5	0.17	1.6	0.52
Falmouth, Cornwall	7.2.92	0.51	1.4	0.58	3.3	0.27	0.06	2.4	0.12	1.2	0.39
Bigbury, Devon	13.2.92	0.58	1.6	0.94	3.3	0.21	0.07	2.6	0.13	1.1	0.38
Bigbury, Devon	13.2.92	0.83	2.0	0.76	4.8	0.4	0.035	3.8	0.19	1.5	0.53

Location	Date found	CB#128	CB#156	CB#180	CB#170	CB#194	Sum 7CB ¶	Sum 25CB
Looe Beach, Cornwall	2.1.92	2.2	0.72	5.5	2.6	0.38	41.6	36.4
Penzance, Cornwall	7.1.92	1.0	0.34	2.5	1.2	0.18	19.8	17.6
Porthleven Harbour, Cornwall	8.1.92	1.5	0.49	3.6	1.7	0.27	27.8	24.5
Praa Sands, Cornwall	14.1.92	1.0	0.38	2.9	1.3	0.26	21.1	18.0
Penzance, Cornwall	2.2.92	1.0	0.4	3.3	1.4	0.23	22.4	19.2
Porthtowan, Cornwall	2.2.92	0.49	0.18	1.4	0.64	0.09	11.1	10.6
St. Keverne, Cornwall	2.2.92	0.77	0.35	3.3	1.4	0.31	18.7	14.3
Falmouth, Cornwall	7.2.92	0.51	0.21	1.7	0.78	0.12	11.3	9.8
Falmouth, Cornwall	7.2.92	0.36	0.15	1.3	0.52	0.09	8.4	7.4
Bigbury, Devon	13.2.92	0.4	0.16	1.1	0.54	0.076	8.9	8.4
Bigbury, Devon	13.2.92	0.57	0.2	1.6	0.72	0.097	11.9	10.4

Location	Date found	HCB	a-HCH	g-HCH	Dieldrin	pp-DDE	pp-DDT	pp-TDE	Sum DDT
Looe Beach, Cornwall	2.1.92	0.068	0.26	0.19	1.8	2.4	11	2.2	15.6
Penzance, Cornwall	7.1.92	0.087	0.24	0.19	0.78	2.5	5.3	1.4	9.2
Porthleven Harbour, Cornwall	8.1.92	0.073	0.18	0.19	1.0	2.6	7.5	1.7	11.8
Praa Sands, Cornwall	14.1.92	0.08	0.21	0.2	0.77	1.9	5	1.1	8.0
Penzance, Cornwall	2.2.92	0.058	0.16	0.16	1.0	1.4	5.4	1.1	7.9
Porthtowan, Cornwall	2.2.92	0.06	0.22	0.21	0.54	1.2	2.1	0.6	3.9
St. Keverne, Cornwall	2.2.92	nd	0.064	0.11	0.44	0.71	8.3	0.57	9.6
Falmouth, Cornwall	7.2.92	0.048	0.092	0.13	0.45	0.72	2.8	0.49	4.0
Falmouth, Cornwall	7.2.92	0.06	0.11	0.15	0.6	0.87	2.7	0.57	4.1
Bigbury, Devon	13.2.92	0.079	0.25	0.27	0.5	1.3	1.7	0.6	3.6
Bigbury, Devon	13.2.92	0.06	0.1	0.12	1.0	1.1	5.9	1.0	8.0

Note: § : nd :not detected (limit of detection 0.001 for a standard sample of 5g tissue)
¶ ICES 7 CBs (# 28, 52, 101, 118, 138, 153, 180)
HEL = Hexane extractable lipid

A review of the literature concerning metals in marine mammals has recently been published (Law, in press), and summarises available information for mercury, copper, zinc, cadmium, lead, arsenic, chromium and nickel; additional data for trace metals in UK marine mammals can also be found in earlier papers (Law *et al.*, 1991(a) and 1992). A report summarising data for both metals and organochlorines in marine mammals analysed during the period 1988-92 is also in preparation (MAFF, in press). Although some of the very highest residue concentrations may affect disease or other stress resistance in marine mammals, most of the concentrations found are at the lower end of the overall range reported for marine mammals generally and as such reflect what must now be considered ‘normal’ or ‘background’ levels of contamination.

SEA WATER

4. GENERAL SAMPLING PROCEDURES

Most of the work reported in this section was carried out on *RV CIROLANA* cruise 7, 26 June-19 July 1992. In order to avoid duplication, the basic information about this cruise will be given here and referred to again in later sections. The cruise track is illustrated in Figure 5, and Table 8 gives a list of station numbers, sampling dates and positions, with a descriptive indication of the location of sampling points within and close to estuaries. Some of the monitoring carried out in 1992 was undertaken to provide data for both the Monitoring Master Plan of the North Sea Task Force, and the UK National Monitoring Programme (NMP). Where stations designated under these programmes were visited, the relevant sampling point numbers are also indicated in Table 8. In the case of the NMP it should be noted that the final numbering scheme (January 1994) has been used in this report. Table 9 lists the water column data (depth, salinity, concentrations of nutrients, surface water temperature and sediment type) for each of the stations, where available.

As outlined previously (MAFF, 1992), the strategy adopted for inshore sampling was to anchor the ship close to the mouth of an estuary and to collect samples both from the ship and from a small inflatable boat which could operate in shallow water around and within the estuary. Where the small boat was used, sampling took place at pre-arranged sites adjacent to named or numbered buoys, quays, jetties, etc., which could be readily identified in order that the sampling position could be determined accurately. Whenever possible, the sampling from the small boat was carried out around low water, whilst samples were being collected over longer periods from the ship. Water samples taken within estuaries generally had a minimum salinity within the range 10 - 30; in the River Tweed the minimum salinity observed was 3.98 at the upstream site.

An additional series of stations along the Danish coast and outward from the German Bight was visited in 1992. This latter transect was the same as that utilised during the joint ICES/Intergovernmental Oceanographic Commission (IOC) Bremerhaven Biological Effects Workshop in March 1990 (Stebbing and Dethlefsen, 1992; Stebbing *et al.,* 1992), and was worked primarily to provide bioassay samples along a previously established and acknowledged gradient of contamination.

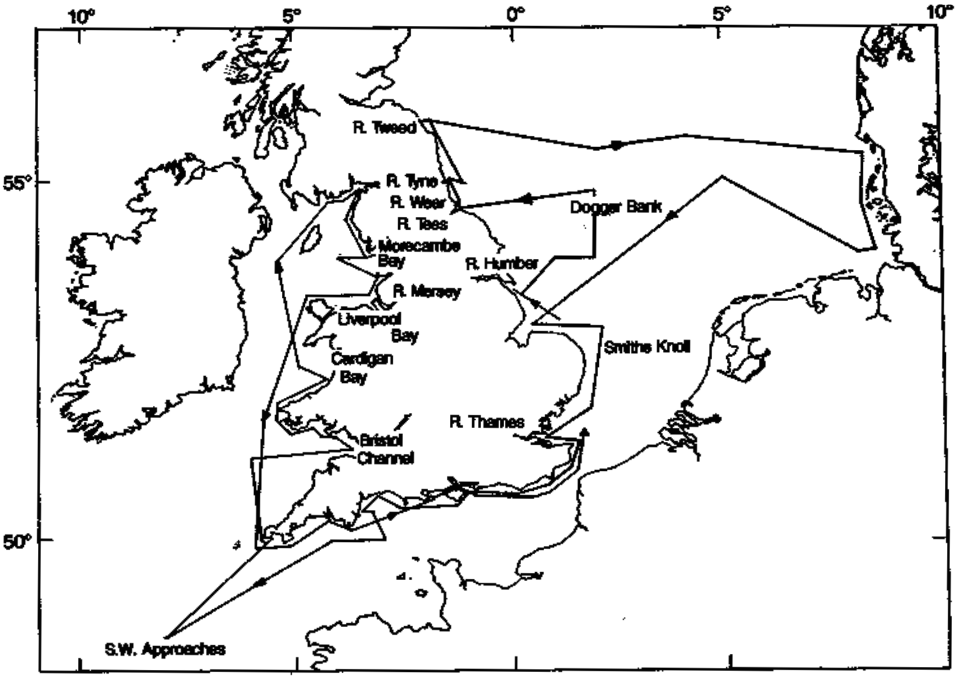

Figure 5. Cruise track of CIROLANA 7, 26 June-19 July 1992

Table 8. Stations sampled during CIROLANA 7, 26 June-19 July, 1992

Station	Date	Position	Location	NSTF	NMP
1	26-Jun	53° 32.96' N 0° 6.01' E	River Humber : Bull anchorage		
2		53° 36.0' N 0° 3.6' W	River Humber : Pyewipe outfall		
3		53° 36.1' N 0° 5.9' W	River Humber : Diffuser		
4		53° 36.6' N 0° 4.6' W	River Humber : No. 6B buoy		
5		53° 37.4' N 0° 8.7' W	River Humber : No. 10A buoy		
6		53° 38.6' N 0° 11.0' W	River Humber : No. 11A buoy		
7		53° 31.97' N 0° 19.85' E	Off River Humber	16	375
8		54° 0.11' N 1° 0.15' E			
9		54° 0.0' N 1° 59.48' E	Off Humber/Wash (Outer Silver Pit)	53	345
10	27-Jun	54° 30.03' N 1° 59.91' E	Dogger Bank : SW Patch		
11		55° 0.07' N 2° 0.17' E	Dogger Bank		
12		54° 49.9' N 1° 19.76' E	Offshore Tyne/Tees	43	285
13		54° 44.95' N 0° 00.12' W			
14	28-Jun	54° 39.1' N 1° 7.25' W	By mouth of River Tees		
15		54° 38.52' N 1° 8.66' W	River Tees : No. 8 buoy		
16		54° 37.4' N 1° 9.34' W	River Tees : Redcar jetty		325
17		54° 36.22' N 1° 9.9' W	River Tees : ICI No. 4 buoy		
18		54° 35.3' N 1° 11.4' W	River Tees : No. 25 buoy		
19		54° 44.12' N 0° 52.87' W	Off River Tees	15	295
20		54° 54.92' N 1° 20.23' W	By mouth of River Wear		
21		54° 55.05' N 1° 21.43' W	River Wear : Between South Piers		275
22		54° 55.01' N 1° 21.78' W	River Wear : North Dock Basin		
23		54° 54.58' N 1° 22.87' W	River Wear : Wearmouth Bridge		
24		54° 54.68' N 1° 23.43' W	River Wear : Hetton Staiths		
25		54° 54.99' N 1° 23.55' W	River Wear : Deptford Quay		
26		54° 54.8' N 1° 24.24' W	River Wear : Queen Alexandra Bridge		265
27	29-Jun	55° 00.47' N 1° 23.40' W	By mouth of River Tyne		
28		55° 0.47' N 1° 25.84' W	River Tyne : Lloyd’s Hailing Station		235
29		54° 59.17' N 1° 27.86' W	River Tyne : buoy off Jarrow Slake		
30		54° 59.09' N 1° 31.49' W	River Tyne : Hebburn		225
31		54° 58.25' N 1° 35.2' W	River Tyne : Ouse Burn		
32		54° 58.09' N 1° 36.25' W	River Tyne : Tyne Bridge		
33		54° 57.45' N 1° 38.1' W	River Tyne : Team Confluence		
34		55° 0.53' N 1° 7.80' W	Off River Tyne	14	245
35	30-Jun	55° 46.08' N 1° 57.79' W	By mouth of River Tweed		
36		55° 46.12' N 2° 0.35' W	River Tweed : Berwick/Tweed Bridges		
37		55° 45.83' N 1° 59.58' W	River Tweed : buoy off Spittal Point		
38		55° 29.95' N 2° 0.09' E			
39	1-Jul	55° 10.41' N 3° 9.37' E	Dogger Bank	47	
40		55° 29.96' N 4° 10.0' E	Bremerhaven Workshop transect : station 9		
41		55° 25.04' N 8° 09.11' E	Off Esbjerg : No. 4 buoy		
42		55° 00.07' N 7° 57.69' E	Off Sylt : by ODAS-S buoy		
43	2-Jul	54° 37.97' N 7° 54.12' E	Amrumbank-W buoy		
44		54° 14.95' N 8° 06.04' E	North of Steingrund-O buoy		
45		54° 03.98' N 8° 07.51' E	Bremerhaven Workshop transect : station 1		
46		54° 01.98' N 8° 02.85' E	Bremerhaven Workshop transect : station 2		
47		54° 00.08' N 8° 00.06' E	Bremerhaven Workshop transect : station 3		
48		54° 00.86' N 7° 48.85' E	Bremerhaven Workshop transect : station 4		
49		54° 06.53' N 7° 24.02' E	Bremerhaven Workshop transect : station 5		
50		54° 25.01' N 6° 14.90' E	Bremerhaven Workshop transect : station 6		
51		54° 50.11' N 5° 34.96' E	Bremerhaven Workshop transect : station 7		
52	3-Jul	55° 06.02' N 5° 00.07' E	Bremerhaven Workshop transect : station 8		
53		53° 03.96' N 0° 29.13' E	Off the Wash	17	385
54	4-Jul	52° 49.95' N 2° 50.06' E	Smiths Knoll	18	395
55		51° 59.91' N 2° 20.01' E	Off Thames	25	475
56		51° 30.85' N 0° 58.22' E	Thames (Warp)	19	465
57		50° 56.02' N 1° 16.81' E	South Varne	69	485
58	5-Jul	50° 44.96' N 0° 09.98' W	Off Brighton / Perintis 1		
59		50° 40.02' N 0° 30.20' W	Perintis 2		
60		50° 39.05' N 0° 50.20' W	Selsey Bill / Perintis 3	70	495
61		50° 29.90' N 1° 30.11' W	Perintis 4		
62		50° 30.05' N 1° 59.75' W	Perintis 5		
63		50° 29.96' N 2° 39.95' W	Perintis 6 / Carole 1		
64		50° 40.03' N 3° 10.10' W	Perintis 7 / Carole 2		
65		50° 24.79' N 3° 30.22' W	Perintis 8 / Carole 3		
66		50° 25.17' N 3° 16.07' W	Perintis 9 / Carole 4		
67		50° 23.16' N 3° 05.96' W	Perintis 10 / Carole 5		
68		50° 05.23' N 2° 59.62' W	Perintis 11 / Central Channel	72	535
69	6-Jul	50° 02.01' N 4° 22.03' W	Perintis 12 / MBA Stn. E1	73	585
70		49° 35.09' N 5° 14.81' W	Perintis 13		
71		49° 10' N 6° 09.94' W	Perintis 14		
72		48° 49.98' N 7° 05.07' W	Perintis 15		
73		48° 30.01' N 7° 59.94' W	Western Approaches reference station		595
74	7-Jul	51° 15.00' N 6° 00.00' W	Celtic Deep		605
75	8-Jul	51° 18.00' N 3° 36.18' W	Severn Estuary		615
76		51° 29.7' N 3° 54.80' W	Swansea Bay		
77		51° 36.79' N 4° 25.70' W	Carmarthen Bay		
78		52° 13.94' N 4° 21.24' W	New Quay		
79		52° 21.50' N 4° 10.47' W	Cardigan Bay		655

Table 8. *Continued*

Station	Date	Position	Location	NSTF	NMP
80		52° 29.95' N 4° 59.95' W	Offshore Cardigan Bay		665
81	9-Jul	53° 52.66' N 5° 30.00' W	JMG530		
82		54° 00.05' N 5° 19.98' W	JMG544		
83		54° 07.10' N 5° 25.09' W			
84		54° 07.47' N 5° 09.96' W	JMG559		
85		54° 15.12' N 4° 59.99' W	JMG569		
86		54° 20.04' N 4° 50.07' W			
87		54° 22.74' N 4° 50.29' W	JMG583		
88		54° 30.08' N 4° 40.00' W	JMG593		
89		54° 37.54' N 4° 30.16' W	JMG610		
90		54° 52.50' N 3° 25.00' W	Solway Firth		15
91		54° 45.01' N 4° 00.04' W	Offshore Solway		25
92	10-Jul	53° 57.84' N 3° 20.15' W	Off Morecambe Bay		795
93		54° 00.03' N 3° 50.07' W	SE Isle of Man		805
94		54° 40.34' N 3° 10.01' W	Off River Ribble		
95		53° 43.4' N 3° 02.6' W	River Ribble : Salter’s Bank		
96		53° 44.1' N 2° 59.6' W	River Ribble : Lytham St. Anne’s		
97		53° 44.7' N 2° 56.2' W	River Ribble : Lytham Creek		
98		53° 44.5' N 2° 54.7' W	River Ribble : Freckleton Saltings		
99		53° 43.4' N 2° 53.7' W	River Ribble : Bank’s Marsh		
100		53° 44.0' N 2° 51.7' W	River Ribble : Asland confluence		
101		53° 58.51' N 3° 03.41' W	Morecambe Bay		
102		54° 1.1' N 2° 56.3' W	No. 7 buoy		
103		53° 58.6' N 3° 0' W	River Lune		
104		53° 55.65' N 3° 0.1' W	River Wyre		
105		53° 57.7' N 3° 2.5' W	Off Lune / Wyre		785
106		53° 37.51' N 4° 30.02' W	Irish Sea		775
107		53° 29.93' N 3° 41.46' W	Outer Liverpool Bay		715
108	11-Jul	53° 28.65' N 3° 15.94' W	Burbo Bight / inner Liverpool Bay		705
109		53° 31.83' N 3° 8.8' W	River Mersey : C1 buoy		765
110		53° 30.5' N 3° 5.6' W	River Mersey : C12 buoy		
111		53° 28.0' N 3° 2.9' W	River Mersey : C20 buoy		
112		53° 26.4' N 3° 0.7' W	River Mersey : Canada buoy		
113		53° 24.56' N 3° 0.48' W	River Mersey : Seacombe Ferry		755
114		53° 21.4' N 2° 58.2' W	River Mersey : E1 buoy		
115	12-Jul	53° 28.72' N 3° 15.89' W	River Mersey : Anchor		
116	13-Jul	53° 26.91' N 3° 00.83' W	River Mersey : West Alexandra dock		
117	14-Jul	53° 28.92' N 3° 17.11' W	River Mersey : Anchor		
118		53° 31.83' N 3° 8.8' W	River Mersey : C1 buoy		765
119		53° 28.0' N 3° 2.9' W	River Mersey : C20 buoy		
120		53° 26.4' N 3° 0.7' W	River Mersey : Canada buoy		
121		53° 24.56' N 3° 0.48' W	River Mersey : Seacombe Ferry		755
122		53° 25.0' N 3° 12.1' W	River Dee : Hilbre Swash		
123		53° 26.2' N 3° 04.6' W	New Brighton : north cardinal beacon		
124	15-Jul	52° 50.0' N 5° 16.4' W	off Lleyn Peninsula		
125		49° 55.45' N 5° 49.92' W	off Wolf Rock		
126	16-Jul	50° 20.01' N 4° 17.72' W	Rame Head : DSDG station 1		
127		50° 19.17' N 4° 16.24' W	Rame Head : DSDG station 2		
128		50° 19.03' N 4° 15.81' W	Rame Head : DSDG station 3		
129		50° 18.84' N 4° 15.32' W	Rame Head : DSDG station 4		
130		50° 18.68' N 4° 15.17' W	Rame Head : DSDG station 5		
131		50° 18.61' N 4° 14.84' W	Rame Head : DSDG station 6		
132		50° 17.91' N 4° 13.60' W	Rame Head : DSDG station 7		
133		50° 19.29' N 4° 14.44' W	Rame Head : DSDG station 8		
134		50° 19.00' N 4° 14.91' W	Rame Head : DSDG station 9		
135		50° 18.90' N 4° 15.11' W	Rame Head : DSDG station 10		
136		50° 18.66' N 4° 16.10' W	Rame Head : DSDG station 11		
137		50° 18.44' N 4° 16.71' W	Rame Head : DSDG station 12		
138		50° 18.09' N 4° 18.25' W	Rame Head : DSDG station 13		
139		50° 18.20' N 4° 15.63' W	Rame Head : DSDG station 14		
140		50° 20.84' N 4° 09.07' W	Plymouth Sound : near Drake’s Island		
141		50° 24.37' N 4° 12.05' W	River Tamar : Royal Albert Bridge		
142		50° 21.85' N 4° 11.03' W	River Tamar : South Rubble buoy		
143		50° 21.45' N 4° 9.9' W	River Tamar : West Vanguard buoy		
144		50° 21.55' N 4° 8.28' W	Plymouth Sound : West Mallard buoy		
145	17-Jul	50° 39.08' N 1° 54.09' W	off Poole Harbour		
+		50° 37.16' N 1° 55.80' W	off Swanage		
146		50° 42.7' N 2° 1.3' W	Poole Harbour : No. 75 (Lake) buoy		
147		50° 42.18' N 1° 58.2' W	Poole Harbour : No. 55 (Stakes) buoy		
148		50° 41.15' N 1° 57.35' W	Poole Harbour : No. 42 (Brownsea) buoy		
149	18-Jul	50° 42.70' N 1° 02.92' W	Solent : Mother Bank anchorage		
150		50° 52.28' N 1° 22.68' W	Southampton Water : NW Netley buoy		
151		50° 51.05' N 1° 20.33' W	Southampton Water : Cadland buoy		
152		50° 50.12' N 1° 18.62' W	Southampton Water : Hamble Point buoy		
153		50° 48.38' N 1° 17.0' W	Southampton Water : Calshot buoy		
154	19-Jul	51° 25.8' N 1° 44.4' E	Drill Stone		

5. NUTRIENTS

Nutrient analyses were not conducted primarily with the aim of assessing trends in concentration, either spatially or temporally, but essentially to provide information to aid interpretation of the bioassay results (particularly those involving algal growth). In nutrient studies trend measurements are usually made in mid-winter, when primary production is low or zero and concentrations of nutrients are therefore at a maximum, whereas samples from the study described here were taken in mid-summer. As expected in summer, concentrations of nutrients were generally low away from estuaries. In the offshore waters of the North Sea, English Channel and Irish Sea, concentrations of nitrate were either close to or below the detection limit of 0.2 µmol l^{-1} (Table 9). Concentrations were at a maximum in the estuarine samples, the highest values being: ammonia, 500 µmol l^{-1} (River Tees, Redcar jetty; Station 16); nitrate + nitrite, 171 µmol l^{-1} (River Humber, off Immingham; Station 6); phosphate, 23 µmol l^{-1} (River Wear, Queen Alexandra Bridge; Station 26); and silicate, 37.9 µmol l^{-1} (River Tyne, Team Confluence; Station 33). The locations at which the highest values for particular nutrient species were observed, and the maximum values found, are very similar to those seen in surveys conducted in mid-summer 1990-91. Table 10 shows the nutrient data for stations sampled in both 1991 and 1992. Although both sets of results represent single snapshots from a fuller picture, the reported concentrations tended to be highest and similar, in the same estuaries in both years. However, in some cases, the lower concentrations varied by an order of magnitude. More detailed studies of the fate of nutrients discharged to estuaries and the processes involved, are undertaken by staff of the MAFF Lowestoft laboratory and will be reported elsewhere.

6. DISSOLVED TRACE METALS

6.1 Introduction

Coastal waters around England and Wales have been monitored for dissolved copper, cadmium and lead for a number of years (MAFF, 1990). Analytical methods were changed in 1991 to include the determination of zinc, nickel and manganese in sea water. The same analytical methods were used in 1992 as in 1991 and these results are therefore directly comparable with those reported for 1991 (MAFF, 1993(a)).

6.2 Methods

Samples were collected from the estuaries of the Rivers Tweed, Tyne, Wear, Tees, Humber and Mersey, and the Bristol Channel (Figure 5). Offshore samples were collected from the English Channel, the North Sea and the Irish Sea. They were collected from the upper 5 m of the water column using a buoyed device deployed away from the side of the vessel (Harper, 1987). The sea water was filtered (0.45 µm), and acidified (0.1% HNO_3) under clean conditions. Dissolved trace metals were determined by chelation/solvent extraction/ furnace atomic absorption spectrometry, using a modification of the method described by Statham (1985).

6.3 Results and discussion

6.3.1 Offshore regions

The ranges of the metal concentrations in each region are compared with those obtained during 1991, in Table 11. In both years, concentrations of Cd, Cu, Ni, Pb and Zn are similar to those reported in other literature, although those for Mn are a little higher. As suggested previously (MAFF, 1993(a)), this may be a seasonal effect; in some areas concentrations of dissolved Mn are higher in spring and summer than in winter, possibly due to enhanced benthic inputs (e.g. Balzer *et al.*, 1987). Since the data were not normally distributed (Shapiro-Wilk statistic, SAS Institute Inc., 1985), a non-parametric test (Kruskal-Wallis Chi-squared approximation) was used to analyse the data statistically. All the data were used to test for significant differences between the North Sea, Irish Sea and English Channel. Only stations that had been sampled in both years were included for comparisons between 1991 and 1992. Results are summarised in Table 12. Concentrations of dissolved metals did not vary between 1991 and 1992 at the 90% confidence level. However, significant differences were observed between the three geographical areas. For all metals, the highest concentrations were observed in the Irish Sea. Some of these high concentrations were associated with water of low salinity and may be attributable to anthropogenic inputs. The Mersey has previously been identified as a significant source of copper and nickel to Liverpool Bay (Nimmo *et al.*, 1989) and both the Mersey and the phosphate plant near St. Bees Head are major sources of cadmium to the Irish Sea (MAFF, 1990). For all metals, average dissolved concentrations in the North Sea are higher than those in the English Channel (Table 12). This would be anticipated since a large proportion of the water passing through the Channel is from the Atlantic Ocean and is relatively uncontaminated. In all cases concentrations of dissolved metals are much lower than the EQSs for saline waters.

Table 9. Water column data for stations sampled during CIROLANA 7, 26 June-19 July 1992

Station	Depth (m)	Salinity	Temperature °C	Concentrations of nutrients (μmol l^{-1})					Sediment type
				Phosphate	Silicate	Nitrate	Nitrite	Ammonia	
1	13	31.227		0.62	4.9	43	0.52	1.9	sand
2		26.73		1.24	18.7	138	0.84	4.8	mud
3		25.985		1.38	21.1	158	0.81	4.3	mud
4		26.847		1.31	17.9	141	0.82	4	mud
5		26.629		1.11	18.9	144	0.85	4.8	mud
6		24.859		1.82	24.7	170	0.96	6.1	mud
7	16	34.374		0.13	1.6	0.3	0.2	1.3	stones and sand
8	43	34.737		0.1	0.6	< 0.2	0.25	2.1	sand
9	71	37.72		0.18	0.9	< 0.2	0.31	2.4	mud and sand
10	19	34.988		0.19	0.6	0.27	0.23	1.5	sand
11	27	34.98		0.18	0.4	0.31	0.09	1.0	sand
12	30	34.891		0.13	0.5	< 0.2	0.22	0.8	sand and shell
13	73	34.809		0.07	0.5	< 0.2	0.07	1.0	sand
14	12	34.405		0.08	1.0	< 0.2	0.32	1.0	sand
15		33.424		1.31	4.0	91.5	9.47	193	stones and sand
16		32.953		4.9	5.7	24.9	3.22	500	mud
17		32.978		8.76	5.9	22.9	3.22	360	NS
18		32.095		4.38	6.7	0.53	1.17	450	mud
19	55	34.496		0.11	0.6	< 0.2	0.35	1.5	mud and sand
20	17	34.19		0.37	1.3	0.73	0.17	2.2	stones and sand
21		32.029		3.49	4.0	14.4	2.03	7.4	sand
22		30.613		5.96	6.2	26.3	3.38	16.7	mud
23		28.632		8.6	8.8	37	5.03	23.3	sand and mud
24		25.815		11	10.4	50	7.09	24.2	sand and mud
25		19.68		19.6	12.3	102	14	32.1	sand and mud
26		15.905		23	12.8	140	14.7	31.5	sand and mud
27	16	34.236		0.36	1.0	< 0.2	3.9	1.9	sand
28		31.813		2.98	1.9	7.7	1.44	20	sand
29		29.755		3.13	5.5	14.6	2.38	22.7	mud
30		26.787		1.71	5.0	24.3	3.39	13.5	NS
31		20.839		2.18	11.9	40.6	5.41	19.6	mud
32		19.795		2.19	13.6	43	5.92	20.8	NS
33		13.065		5.36	37.9	82.3	7.65	29.2	mud
34	73	34.205		0.28	0.4	< 0.2	6.13	1.1	mud
35	20	33.341		0.19	1.3	1	1.08	0.9	NS
36		3.981		0.18	9.6	62	0.95	2.8	mud and sand
37		7.638		0.17	28.1	73.9	1.13	3.7	mud
38	63	34.916		0.25	0.3	< 0.2	1.45	0.5	NS
39	31	34.971		0.27	0.3	< 0.2	0.24	0.6	sand
40	31	34.989		0.17	0.5	0.2	0.21	0.3	sand and shell
41	17	32.71		0.15	0.6	0.47	0.13	2.1	sand over mud
42	18	31.85		0.11	0.3	3.6	0.26	2.9	sand and shell
43	15	32.659		0.17	1.7	0.76	0.24	4.1	sand
44	15	31.877		0.32	0.9	0.65	0.15	2.3	sand
45	19	31.692		0.66	1.6	1.1	2.16	3.0	mud (anoxic below surface)
46	26	32.288		0.3	0.5	1.5	2.17	1.4	mud
47	32	33.18		NS	NS	NS	NS	NS	mud
48	38	NS		NS	NS	NS	NS	NS	mud
49	38	33.145		0.07	1.1	< 0.2	1.3	0.9	sand and mud
50	39	33.033		0.1	0.2	2.1	0.29	1.7	sand
51	45	34.111		0.05	1.0	0.78	0.32	1.0	sand and mud
52	38	34.728		0.12	0.5	< 0.2	0.36	0.5	sand
53	32	34.088	16.5	0.16	0.7	< 0.2	0.41	0.6	NS
54	35	34.609	14.7	0.24	1.0	2.2	0.28	1.3	sand
55	50	35.109	15.3	0.25	0.5	0.58	0.32	0.8	sand and shell
56	18	34.56	17.9	0.54	1.1	0.98	0.32	1.2	sand and shell
57	32	34.037	15.7	0.28	1.6	0.94	0.36	1.4	sand and shell
58	24	35.221	15.9	0.15	1.0	0.24	0.26	0.7	NS
59	24	35.252	15.5	0.07	1.2	0.47	0.23	1.6	NS
60	19	35.198	16.4	0.12	1.1	0.22	0.18	0.5	mud
61	38	35.154	16.8	0.22	1.0	0.71	0.19	0.9	NS
62	39	35.145	16.8	0.21	1.3	0.31	0.19	1.0	NS
63	43	35.338	15.4	0.17	1.9	0.25	0.15	0.7	sand, stones and shell
64	19	35.356	16	0.19	2.4	0.27	0.13	0.7	mud and sand
65	13	35.33	16.2	0.24	2.8	< 0.2	0.11	0.6	mud
66	47	35.402	14.8	0.14	1.7	< 0.2	0.12	0.6	mud and sand
67	55	35.381	14.9	0.16	1.7	< 0.2	0.14	0.5	sand and shell
68	69	35.441	13.9	0.15	1.3	< 0.2	0.14	0.5	NS
69	73	35.474	15.8	0.04	1.7	< 0.2	0.12	0.2	sand and shell
70	97	35.581	16.2	0.04	1.5	< 0.2	0.09	0.2	NS
71	119	35.539	16.2	0.05	0.8	< 0.2	0.08	0.3	NS
72	143	35.45	17	0.07	0.5	< 0.2	0.15	0.5	NS
73	173	35.499	16.3	0.14	0.5	< 0.2	0.16	0.4	NS
74	95	35.426	16.2	0.08	0.5	< 0.2	0.11	0.2	mud and sand
75	39	32.819	16.6	1.23	4.3	27.6	0.51	0.9	NS
76	25	33.301	16.9	0.55	1.3	9.0	0.53	2.3	NS
77	24	34.129	16.5	0.05	0.2	< 0.2	0.48	0.3	sand and mud
78	23	33.835	17	0.2	0.7	0.27	0.13	0.4	mud

Table 9. Continued

Station	Depth	Salinity	Temperature	Concentrations of nutrients (μmol l^{-1})					Sediment type
	(m)		°C	Phosphate	Silicate	Nitrate	Nitrite	Ammonia	
79	27	34.292	18	0.11	0.5	< 0.2	0.12	0.3	mud
80	75	33.949	13.9	0.24	0.5	0.54	0.16	0.6	sand and shell
81	105	NS		NS	NS	NS	NS	NS	mud
82	85	NS		NS	NS	NS	NS	NS	mud
83	57	33.849	13.1	0.23	0.2	< 0.2	0.16	0.4	mud
84	131	NS		NS	NS	NS	NS	NS	NS
85	114	NS		NS	NS	NS	NS	NS	NS
86	47	33.738	13.5	0.5	1.4	1.12	0.18	0.4	mud and sand
87	43	NS		NS	NS	NS	NS	NS	sand
88	51	NS		NS	NS	NS	NS	NS	stones and sand
89	40	NS		NS	NS	NS	NS	NS	shell and stones
90		31.224		0.68	0.2	0.94	0.16	0.4	sand
91	25	33.365	15.5	0.53	0.7	0.28	0.12	0.3	mud
92	21	33.006	15.7	0.72	3.0	0.35	0.15	1.0	mud and sand
93	37	33.254	15.7	0.62	0.7	< 0.2	0.11	0.5	sand
94	16	33.026	17.1	0.87	1.8	< 0.2	0.11	0.7	sand
95		NS		NS	NS	NS	NS	NS	sand
96		NS		NS	NS	NS	NS	NS	sand
97		NS		NS	NS	NS	NS	NS	mud
98		NS		NS	NS	NS	NS	NS	mud
99		NS		NS	NS	NS	NS	NS	mud
100		NS		NS	NS	NS	NS	NS	mud
101	22	33.033	16.6	0.76	0.9	0.28	0.12	0.4	sand
102		32.908		1.18	1.3	0.77	0.13	1.6	sand
103		33.049		0.8	0.4	< 0.2	0.12	0.4	sand
104		31.726		1.3	0.7	2.63	0.47	6.8	sand and mud
105		33.084		0.71	0.7	< 0.2	0.53	0.3	sand
106	70	33.937	14	0.52	0.8	2.42	0.28	0.8	NS
107	37	32.939	15.6	0.59	1.9	0.77	0.33	0.8	sand and shell
108	14	32.072	17.6	1.85	4.0	5.22	2.43	5.3	sand over anoxic mud
109		30.836		3.12	4.1	11.9	15.8	10.2	NS
110		30.009		4.09	5.4	24.7	25.3	12.5	sand
111		28.985		4.99	8.1	40	37	17.2	NS
112		27.906		5.82	9.6	53	48	22.1	NS
113		27.206		6.31	10.7	53	54	30	NS
114		24.593		7.76	10	79	76	52	mud
115	19	NS		NS	NS	NS	NS	NS	sand over anoxic mud
116		NS		NS	NS	NS	NS	NS	NS
117	20	32.137	16.6	1.74	3.2	7.91	3.39	7.3	NS
118		31.461		2.77	4.3	9.72	9.08	11.2	NS
119		30.563		3.54	4.7	36	17.9	9.0	NS
120		30.152		4.09	5.6	36	22.4	10	NS
121		29.201		4.85	6.6	57	32	13.3	NS
122		31.488		1.52	0.7	44.4	0.59	4.0	sand
123		NS		NS	NS	NS	NS	NS	mud over sand
124	97	34.19	13.3	0.55	0.7	20	3.97	2.7	NS
125	74	35.439	14.8	0.08	1.1	< 0.2	< 0.05	0.2	NS
126	31	NS		NS	NS	NS	NS	NS	mud and shell
127	30	NS		NS	NS	NS	NS	NS	stones and mud
128	29	NS		NS	NS	NS	NS	NS	sand
129	27	NS		NS	NS	NS	NS	NS	NS
130	28	NS		NS	NS	NS	NS	NS	stones and mud
131	28	NS		NS	NS	NS	NS	NS	stones
132	47	NS		NS	NS	NS	NS	NS	mud and sand
133	21	NS		NS	NS	NS	NS	NS	gravel
134	25	NS		NS	NS	NS	NS	NS	stones and sand
135	27	NS		NS	NS	NS	NS	NS	NS
136	35	NS		NS	NS	NS	NS	NS	mud
137	42	NS		NS	NS	NS	NS	NS	mud
138	47	NS		NS	NS	NS	NS	NS	mud and sand
139	45	NS		NS	NS	NS	NS	NS	mud
140	10	35.081	15.8	0.23	1.6	1.27	0.18	0.5	sand
141		29.449		0.88	3.4	15.5	1.02	1.2	mud
142		33.225		0.52	0.8	3.72	0.48	0.6	sand and stones
143		33.887		0.47	2.3	2.53	0.37	0.6	sand and stones
144		33.799		0.75	0.8	4.9	0.6	2.5	sand and stones
145	17	35.104		0.18	1.9	< 0.2	0.08	0.4	sand, shell and stones
+			17.7						sand
146		29.053		2.4	27.2	30.2	1.3	7.0	NS
147		33.653		0.8	7.7	5.36	0.54	2.3	sand
148		33.318		0.8	7.3	2.9	0.4	1.3	sand over mud
149	18	35.153	17.7	0.27	2.2	1.3	0.29	0.6	NS
150		32.438		1.75	18	22.6	1.38	5.6	anoxic mud
151		32.259		0.76	18	12.4	1.5	41	sand and mud
152		33.783		0.7	7.0	5.52	0.68	0.6	NS
153		34.686		0.4	4.3	2.94	0.36	0.8	NS
154		NS	17.1						NS

Table 10. Comparative nutrient data for stations sampled during 1991, 1992

Location	Concentrations of nutrients (µmol l^{-1})					
	Phosphate		Silicate		Nitrate+Nitrite	
	1991	1992	1991	1992	1991	1992
off R. Humber	2.0	0.6	5.4	4.9	39	43
R. Humber : no. 11A buoy	1.9	1.8	18	25	147	171
R. Humber : no. 10A buoy	0.56	1.1	17	19	134	145
R. Humber : Diffuser	0.66	1.4	15	21	125	159
R. Humber : no. 6B buoy	1.3	1.3	12	18	93	142
R. Humber : Pyewipe outfall	8.4	1.2	13	19	110	139
off R. Humber	0.81	0.13	3.0	1.6	4.6	0.5
off R. Tees	1.2	0.08	2.0	1.0	16	0.3
R. Tees : no. 8 buoy	5.2	1.3	8.0	4.0	82	101
R. Tees : no. 15 buoy	5.8	4.9	7.0	5.7	28	28
R. Tees : ICI No. 4 buoy	6.4	8.8	16	5.9	61	26
off Tees	0.17	0.11	1.3	0.6	0.8	0.35
off R. Wear	0.76	0.37	2.4	1.3	2.4	0.9
R. Wear : North Dock Basin	6.7	6.0	9.0	6.2	40	30
R. Wear : Wearmouth Bridge	19	23	17	13	118	155
off R. Tyne	0.91	0.36	1.9	1.0	4.2	3.9
R. Tyne : No. 2 buoy	3.4	3.0	9.6	1.9	8.4	9.1
R. Tyne : Jarrow Slake	9.8	3.1	44	5.5	30	17
R. Tyne : Swan Hunter	2.9	1.7	17	5	19	28
R. Tyne : Tyne Bridge	3.9	2.2	30	14	34	49
off R. Tyne	1.5	0.28	1.7	0.4	0.8	6.1
off R. Tweed	0.92	0.19	1.9	1.3	1.2	2.1
R. Tweed : Spittal Point	1.6	0.17	21	28	58	75
R. Tweed : Berwick/Tweed bridges	2.3	0.18	24	9.6	70	63
offshore Tyne/Tees	0.48	0.13	1.3	0.5	0.4	0.2
Dogger Bank	0.22	0.27	0.6	0.3	0.3	0.2
Silver Pit	1.2	0.18	0.9	0.9	0.4	0.3
the Wash	1.0	0.16	2.4	0.7	18	0.41
Smiths Knoll	0.52	0.24	1.7	1.0	1.7	2.5
Outer Gabbard	0.38	0.25	1.0	0.5	1.4	0.9
Warp (Thames)	2.4	0.54	1.4	1.1	18	1.3
South Varne	0.64	0.28	1.2	1.6	0.8	1.3
Isle of Wight	0.02	0.12	2.0	1.1	< 0.2	0.4
Central Channel	0.55	0.15	1.3	1.3	1.4	0.3
Western Channel	0.56	0.04	2.0	1.7	0.6	< 0.2
MAFF reference stn.	0.6	0.14	0.7	0.5	0.4	< 0.2
Celtic Deep	1.1	0.08	0.8	0.5	0.8	< 0.2
Bristol Channel	2.5	1.2	6.8	4.3	43	28
Swansea Bay	0.94	0.55	1.1	1.3	13	9.5
New Quay	1.1	0.2	4.1	0.7	0.6	0.4
offshore Cardigan Bay	0.21	0.24	1.1	0.5	1.0	0.7
Irish Sea offshore	0.73	0.5	1.1	0.8	0.6	2.7
off R. Mersey	1.3	1.9	1.9	4.0	4.1	7.6
R. Dee : Hilbre Swash	1.4	1.5	1.8	0.7	1.8	45
R. Mersey : C12 buoy	5.6	4.1	7.5	5.4	76	50
R. Mersey : C20 buoy	6.6	5.0	9.1	8.1	106	77
R. Mersey : Canada buoy	6.5	5.8	9.4	9.6	118	101
Morecambe Bay	1.0	0.8	1.4	0.9	0.6	0.4
Morecambe Bay : no. 7 buoy	3.9	1.2	2.1	1.3	0.4	0.9
off R. Lune	1.3	0.8	1.5	0.4	0.6	< 0.2
off R. Wyre	4.2	1.3	8.1	0.7	11	3.1
Plymouth Sound	1.5	0.23	4.6	1.6	5.1	1.5
R. Tamar : West Mallard buoy	0.98	0.75	8.3	0.8	10	5.5
R. Tamar : West Vanguard buoy	0.56	0.47	4.8	2.3	6.9	2.9
R. Tamar : South Rubble buoy	0.93	0.52	5.6	0.8	6.9	4.2
R. Tamar : Royal Albert Bridge	1.4	0.88	28	3.4	51	17
Solent : Mother Bank	0.35	0.27	3.3	2.2	1.3	1.6
Southampton Water : NW Netley buoy	0.66	1.8	1.8	18	0.8	24
Southampton Water : Cadland buoy	0.25	0.76	4.4	18	1.5	14
Southampton Water : Hamble Point buoy	0.32	0.7	5.1	7.0	3.3	6.3
Southampton Water : Calshot buoy	0.25	0.4	1.8	4.3	1.9	3.3

Table 11. Summary of concentrations of dissolved trace metals in offshore areas

Area	Year	Cadmium (ng l^{-1})	Copper (µg l^{-1})	Manganese (µg l^{-1})	Nickel (µg l^{-1})	Lead (ng l^{-1})	Zinc (µg l^{-1})
North Sea	1991	10 - 32	0.24 - 0.83	0.45 - 2.3	0.20 - 1.00	21 - 55	0.27 - 2.2
	1992	4 - 51	0.20 - 0.74	0.25 - 3.4	0.18 - 0.55	13 - 71	0.11 - 0.75
English Channel	1991	11 - 18	0.20 - 1.5	0.81 - 1.4	0.22 - 0.47	23 - 32	0.22 - 1.0
	1992	13 - 22	0.11 - 0.45	0.09 - 2.5	0.14 - 0.43	20 - 41	0.14 - 0.66
Irish Sea	1991	18 - 81	0.33 - 1.50	0.18 - 7.8	0.26 - 0.87	24 - 170	0.27 - 1.8
	1992	13 - 69	0.22 - 1.1	0.21 - 6.4	0.22 - 0.60	21 - 110	0.20 - 2.2

Table 12(a). Analysis of variance summary, dissolved metals between 1991 and 1992

Variable	Class mean (number)		P > CHISQ
	1991	1992	
Cd (ng l^{-1})	21 (21)	22 (21)	0.9494
Cu (ng l^{-1})	510 (21)	450 (21)	0.7435
Mn (ng l^{-1})	1700 (22)	1100 (21)	0.1279
Ni (ng l^{-1})	480 (22)	330 (21)	0.0162
Pb (ng l^{-1})	58 (20)	39 (21)	0.4035
Zn (ng l^{-1})	810 (21)	650 (21)	0.4206

Table 12(b). Analysis of variance summary, dissolved metals between North Sea (NS), English Channel (EC) and Irish Sea (IS)

Variable	Class mean (number)			P > CHISQ
	NS	EC	IS	
Cd (ng l^{-1})	17 (38)	16 (21)	35 (22)	0.0001
Cu (ng l^{-1})	440 (38)	310 (21)	740 (22)	0.0001
Mn (ng l^{-1})	1400 (37)	770 (21)	2500 (22)	0.0018
Ni (ng l^{-1})	420 (37)	270 (21)	490 (22)	0.0001
Pb (ng l^{-1})	44 (37)	24 (21)	59 (22)	0.0001
Zn (ng l^{-1})	600 (38)	470 (21)	1100 (22)	0.0001

6.3.2 Estuaries

Concentrations of dissolved trace metals in estuaries, as observed in 1992, are summarised in Table 13. They are compared with those obtained in 1991 in Figure 6. Concentrations of trace metals in estuaries are very variable and higher than those seen offshore. The lowest values are generally observed in the River Tweed; unlike the other estuaries studied this is not an industrialised river, and was included for comparative purposes (MAFF, 1992). For all metals, the highest values were seen in one or other of the industrialised estuaries: Humber (Cd), Tees (Cu), Tyne (Mn and Zn) and Mersey (Pb and Ni). None of the 1992 samples exceeded the EQSs for dissolved metals in coastal waters (5, 25, 30 and 40 µg l^{-1} for Cu, Pb, Ni and Zn respectively).

6.4 Conclusions

For offshore samples, the Irish Sea had generally higher concentrations of metals than the North Sea, while

Table 13. Summary of concentrations of dissolved metals in estuarine areas, 1992

Area	Cadmium (ng l^{-1})	Copper (µg l^{-1})	Manganese (µg l^{-1})	Nickel (µg l^{-1})	Lead (ng l^{-1})	Zinc (µg l^{-1})
Humber	76 - 220	1.8 - 3.6	6.8 - 59	1.6 - 4.9	24 - 620	3.6 - 15
Tees	15 - 97	0.49 - 4.0	4.0 - 120	0.30 - 0.88	54 - 820	0.7 - 12
Wear	17 - 58	0.35 - 1.4	2.7 - 110	0.30 - 1.40	58 - 550	0.5 - 7.5
Tyne	18 - 48	0.43 - 1.0	2.1 - 130	0.28 - 1.3	57 - 460	1.2 - 25
Tweed	12 - 33	0.49 - 2.6	3.0 - 13	0.23 - 0.35	40 - 150	0.43 - 1.6
Mersey	15 - 61	1.3 - 2.2	1.5 - 20	0.87 - 5.0	130 - 880	3.9 - 17
Morecambe Bay	13 - 49	0.58 - 1.2	2.1 - 23	0.40 - 0.81	27 - 260	1.4 - 3.8

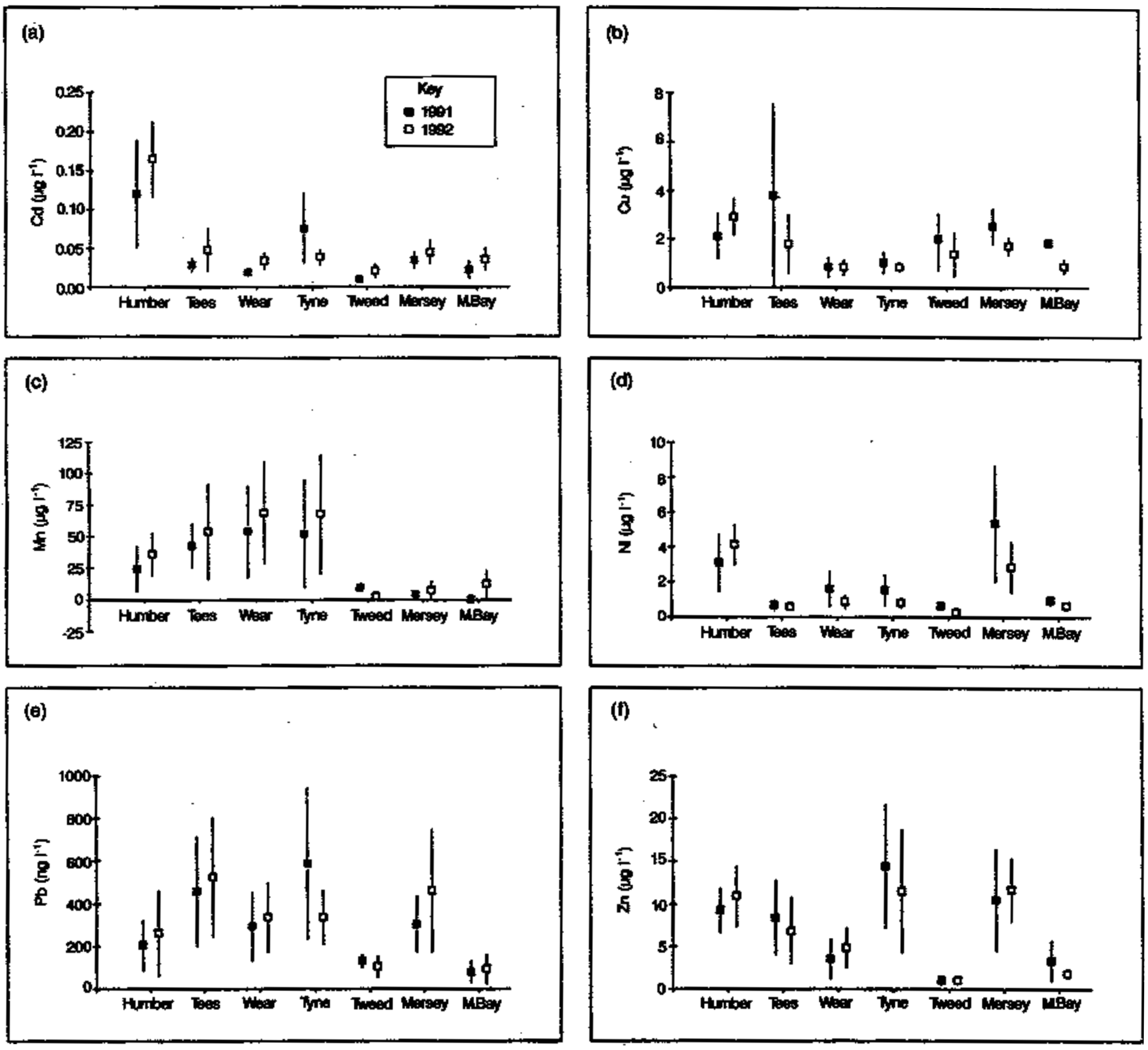

Figure 6. Concentrations of dissolved trace metals in estuaries (mean ± standard deviation) 1991 and 1992 data: (a) Cd; (b) Cu; (c) Mn; (d) Ni; (e) Pb and (f) Zn

lowest concentrations were observed in the English Channel. Highest concentrations were observed in estuarine waters, and attributed to anthropogenic inputs from land-based sources. Trace metal concentrations generally decrease rapidly with distance from the coast. None of the concentrations of dissolved metals measured in 1992 exceeded the established EQS values.

7. ALPHA- AND GAMMA-HEXACHLOROCYLO-HEXANES

A fourth full survey of alpha- and gamma-hexachlorocyclohexane residues in sea water was conducted during *RV CIROLANA* cruise 7, 26 June-19 July 1992. Results are shown in Table 14. Samples were also collected as a continuation of the studies undertaken following the loss of *MV PERINTIS* in the English Channel in March 1989 which carried a cargo of 5.8 tonnes of gamma-hexachlorocyclohexane (Lindane) that was never recovered.

The results of the samples taken in the Channel and southern North Sea were all typical of background concentrations and give no suggestion that there has been a significant release of Lindane from the lost container. This tends to confirm the assessment made at the time of the incident that any loss from the container would probably be very slow given the nature of the packaging (plastic sacks) and the presumed essentially intact structure of the container.

Samples at all stations were below the EQS value for estuarine and sea water of 20 ng l^{-1}. The recorded values were similar to the previous surveys (MAFF, 1993(a)) and those reported by German workers (Deutsches Hydrographisches Institut, 1985). The highest value for gamma-HCH (7.2 ng l^{-1}) was recorded at Station 24 in the River Wear with the River Mersey yielding values of up to 5.2 ng l^{-1}.

Table 14. Organic contaminants in sea water samples collected during CIROLANA 7, 26 June-19 July 1992. (Concentrations are in ng l^{-1}, except THC which is µg l^{-1} Ekofisk crude oil equivalents). Station details are given in Table 8, Section 4

Station	Location	Simazine	Atrazine	Alpha HCH	Gamma HCH	THC/Ekofisk
1	River Humber : Bull anchorage	4.1	8.6	0.14	0.41	1.4
2	River Humber : Pyewipe outfall	25	40	0.18	0.76	3.9
3	River Humber : Diffuser	23	27	0.14	0.64	4.7
4	River Humber : No. 6B buoy	33	40			4.2
5	River Humber : No. 10A buoy	27	38			12
6	River Humber : No. 11A buoy	23	38	0.25	1.4	6.5
7	Off River Humber	0.8	1.4	0.17	0.25	2.5
8		<0.6	0.95	0.22	0.66	
9	Off Humber/Wash (Outer Silver Pit)	<0.6	1.1	0.19	0.29	
10	Dogger Bank : SW Patch	<0.6	<0.6			0.3
11	Dogger Bank	<0.6	0.6			
12	Offshore Tyne/Tees	<0.6	<0.6			
13		<0.6	<0.6	0.19	0.25	
14	By mouth of River Tees	<0.6	<0.6	0.13	0.17	<0.2
15	River Tees : No. 8 buoy	<0.6	1.9	0.43	1.3	17
16	River Tees : Redcar jetty	<0.6	1.9	0.43	1.9	64
17	River Tees : ICI No. 4 buoy	<0.6	4	0.45	1.7	49
18	River Tees : No. 25 buoy	2.4	3.4	0.47	3	38
19	Off River Tees	<0.6	0.86	0.5	0.56	5.0
20	By mouth of River Wear	0.64	0.99	0.43	0.58	0.8
21	River Wear : Between South Piers	2.1	4.8	0.46	1.1	1.7
22	River Wear : North Dock Basin	3.8	8	0.46	1.7	5.0
23	River Wear : Wearmouth Bridge	1.9	5.3	0.43	3	7.4
24	River Wear : Hetton Staiths	6.9	16	0.44	7.2	
25	River Wear : Deptford Quay	8.6	19	0.38	5	7.6
26	River Wear : Queen Alexandra Bridge	11	23	0.16	4.1	9.1
27	By mouth of River Tyne	<0.6	<0.6	0.2	0.36	0.5
28	River Tyne : Lloyd’s Hailing Station	3.9	4.9	0.18	1.3	5.2
29	River Tyne : buoy off Jarrow Slake	4.5	5.7	0.22	2.7	8.2
30	River Tyne : Hebburn	4	5.1	0.19	2	6.8
31	River Tyne : Ouse Burn	7.4	10.6	0.21	2.2	7.0
32	River Tyne : Tyne Bridge	5.6	6.9	0.19	2.1	
33	River Tyne : Team Confluence	7.7	11.8	0.17	2.6	14
34	Off River Tyne	1.8	<0.6	0.18	0.41	0.4
35	By mouth of River Tweed	2.1	1.6	0.14	0.26	0.6
36	River Tweed : Berwick/Tweed Bridges	3.9	5.1	0.14	2.4	2.0
37	River Tweed : buoy off Spittal Point	2.6	4.3	0.12	1.4	3.1
38		1.9	<0.6	0.27	0.53	<0.2
39	Dogger Bank	1.5	<0.6			0.3
41	Off Esbjerg : No. 4 buoy	2.2	3.4	0.17	1	0.3
42	Off Sylt : by ODAS-S buoy	2	3.3	0.19	1.3	0.5
43	Amrumbank-W buoy	2	2.6	0.31	2.5	
44	North of Steingrund-O buoy	5.5	8.5	0.14	1.3	0.3
45	Bremerhaven Workshop transect : station 1	4.5	6.1	0.18	1	
46	Bremerhaven Workshop transect : station 2	<0.6	3.3	0.27	2	0.4
47	Bremerhaven Workshop transect : station 3	2	2.8			
48	Bremerhaven Workshop transect : station 4	2.6	3.4	0.16	1.4	
49	Bremerhaven Workshop transect : station 5	<0.6	4.4	0.25	1.7	
50	Bremerhaven Workshop transect : station 6	3.2	4.9	0.29	2.2	
51	Bremerhaven Workshop transect : station 7	2.5	3.2	0.13	0.8	
52	Bremerhaven Workshop transect : station 8	1.9	2.4	0.12	0.7	
53	Off the Wash	<0.6	3.3	0.23	1.4	1.5
54	Smiths Knoll	1.9	<0.6	0.38	2.7	14
55	Off Thames	5	3.8	0.14	1.5	0.8
56	Thames (Warp)	<0.6	<0.6	0.17	1.8	1.1
57	South Varne	1.2	1.4	0.13	1.8	0.2
58				0.12	0.85	
59				0.14	0.98	
60	Selsey Bill / Perintis 3	<0.6	<0.6	0.14	1	0.2
61				0.1	0.92	
62				0.1	0.54	
63				0.13	0.68	
64				0.18	1.1	
65				0.11	0.45	
66				0.13	0.47	
67				0.16	0.92	
68	Perintis 11 / Central Channel	0.8	<0.6	0.17	1.2	0.3
69	Perintis 12 / MBA Stn. E1	<0.6	1.1	0.14	0.55	<0.2
70				0.11	0.33	
71				0.17	0.46	
72				0.19	0.51	
73	Western Approaches reference station	<0.6	<0.6	0.2	0.41	
74	Celtic Deep	0.9	<0.6	0.19	0.48	
75	Severn Estuary	3	4.9	0.14	0.6	
76	Swansea Bay			0.17	0.55	
77	Carmarthen Bay	0.9	2	0.18	0.51	

Table 14. Continued

Station	Location	Simazine	Atrazine	Alpha HCH	Gamma HCH	THC/Ekofisk
78	New Quay			0.23	0.55	
79	Cardigan Bay	4.2	3.9	0.29	0.68	
80				0.19	0.47	
83		1.1	<0.6	0.24	0.45	0.3
86				0.34	0.72	0.2
90	Solway Firth	5.7	6.4	0.29	0.87	1.2
91	Offshore Solway	4	3.6	0.31	0.88	
92	Off Morecambe Bay	<0.6	<0.6	0.31	0.91	
93	SE Isle of Man	4	4.5	0.35	1	0.6
94	Off River Ribble	7	7	0.31	1	
101	Morecambe Bay	4.2	6.1	0.33	0.9	1.2
102	No. 7 buoy	6.8	9.3	0.18	0.55	
103	River Lune	3.3	6.4			1.7
104	River Wyre	5.9	11	0.35	1.7	7.0
105	Off Lune / Wyre			0.3	0.87	
106	Irish Sea			0.23	0.52	
107	Outer Liverpool Bay	9.6	8.3	0.26	0.88	
108	Inner Liverpool Bay	21	13	0.38	1.4	
109	River Mersey : C1 buoy	17	16	0.51	1.6	6.8
110	River Mersey : C12 buoy	22	22	0.79	2.5	
111	River Mersey : C20 buoy	28	31	0.95	2.8	8.7
112	River Mersey : Canada buoy	36	39	1.8	5.2	
113	River Mersey : Seacombe Ferry	37	42	2.9	5.1	11
114	River Mersey : E1 buoy	23	24	1.5	4.3	
141	River Tamar : Royal Albert Bridge					3.8
142	River Tamar : South Rubble buoy					3.0
143	River Tamar : West Vanguard buoy					2.9
144	Plymouth Sound : West Mallard buoy					6.2
146	Poole Harbour : No. 75 (Lake) buoy					6.4
147	Poole Harbour : No. 55 (Stakes) buoy					1.3
148	Poole Harbour : No. 42 (Brownsea) buoy					1.1

8. TRIAZINE HERBICIDES (SIMAZINE AND ATRAZINE)

8.1 Introduction

Simazine and atrazine have been found in many freshwaters at concentrations which exceed the EC standard for drinking waters of 1 µg l^{-1}. Both were identified for reduction in usage by the Third North Sea Conference of Ministers in 1990 and appear in the UK Red List. For this reason attention has been given to establishing the extent to which they may pose a threat to North Sea ecosystems.

Results from the first two annual surveys, 1990 and 1991, were presented in an earlier report in this series (MAFF, 1993(a)). The data presented here are the results of a further survey undertaken on *RV CIROLANA* Cruise 7, 26 June-19 July 1992.

8.2 Analytical methods

The analytical method remained broadly similar to that used for the previous surveys. Full details are to be published elsewhere. In brief, 2.7 l samples of sea water were collected from 1 m below the surface and extracted into dichloromethane. This extract was dried over anhydrous sodium sulphate and the solvent exchanged for 20% diethyl ether in n-hexane. The extracts were then subjected to column chromatography using 10% deactivated alumina to remove the majority of co-extractives, internal standards added and residues determined by capillary gas chromatography/mass spectrometry using an Ion Trap detector.

8.3 Results and discussion

The results are shown in Table 14. Concentrations and distribution patterns were broadly similar to previous surveys. Residues of both simazine and atrazine were detected at the majority of inshore stations with concentrations decreasing rapidly, but in some cases still at detectable concentrations, offshore. Again, as with previous surveys, the dealkylated breakdown products (des-ethyl and des-isopropyl atrazine) were not detected.

The highest concentration of simazine and atrazine were again found in the Humber and Mersey estuaries. The highest value for simazine was 37 ng l^{-1} at Station 113 in the River Mersey at Seacombe Ferry. This station also yielded the highest concentration of atrazine, 42 ng l^{-1}. In the River Humber the highest concentrations of simazine and atrazine were 33 and 40 ng l^{-1} respectively, found in both cases at Station 4 (No. 6B buoy).

The proposed EQS for the protection of saltwater life for simazine and atrazine is 2 µg l^{-1}. The concentrations reported here are well below this value and it is very unlikely that these concentrations would have any biological impact in the waters of England and Wales. Although the use of triazine herbicides has been restricted, this has not as yet resulted in any apparent decline in water column concentrations in the marine environment.

9. HYDROCARBONS

Total hydrocarbon concentrations (THCs) were determined at sea in subsurface (1 m) water samples from 61 stations, using ultra-violet 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 (Dl = 25 nm) were recorded for all samples. With the single exception of the sample from Smiths Knoll (off north east Norfolk) where the THC was found to be unusually high (14 µg l^{-1} Ekofisk crude oil equivalents), low THCs (generally <2 µg l^{-1}) were recorded both offshore and in coastal areas away from estuaries. Higher concentrations were found in some estuarine samples, e.g. up to 64 µg l^{-1} in the River Tees. The ranges of the estuarine concentrations determined were 2.0 to 3.1 µg l^{-1} in the Tweed; 5.2 to 14 µg l^{-1} in the Tyne; 1.7 to 9.1 µg l^{-1} in the Wear; 17 to 64 µg l^{-1} in the Tees; 1.1 to 6.4 µg l^{-1} in Poole Harbour; 2.9 to 6.2 µg l^{-1} in the Tamar; and 6.8 to 11 µg l^{-1} in the Mersey (Table 14). These concentrations are similar to those reported previously for samples taken in 1990 and 1991 (MAFF, 1992 and 1993(a)) ¾ as in 1992, in both previous years the highest value was observed in the River Tees, 55 and 48 µg l^{-1} respectively.

Extracts from samples for which the THC was greater than 10 µg l^{-1} (i.e. from Stations 5, 15 to 18, 33, 54 and 113) were also analysed in the laboratory using capillary gas chromatography. As for previous surveys conducted in 1990 and 1991 (MAFF, 1992 and 1993(a)), none of the traces showed evidence of any fresh oil inputs. An example is given in Figure 7, which shows the chromatogram obtained for Station 17 (River Tees, ICI no. 4 buoy) taken just downstream of the ICI North Tees works and across the river from Teesport. The chromatogram indicates the presence of a number of volatile organic compounds (eluting in the first five minutes), but although the remainder of the chromatogram is complex it does not show the sequential series of *n*-alkane peaks usually seen in fresh oils. Further data regarding the concentrations of volatile organic compounds in the waters around England and Wales are given elsewhere in this report (see Section 10). Considerable variability in the composition of effluent streams discharging to the River Tees has been demonstrated previously (Johnston *et al.*, 1991), and a wide range of organic compounds has also been tentatively identified in water samples from the Tees by means of computer matching of mass spectra (Law *et al.*, 1991(b)). Matthiessen *et al.* (1993) have also made an ecotoxicological assessment of the organic chemicals identified in a number of UK estuaries, using both published data on aquatic toxicity and data predicted using quantitative structure-activity relationships (QSARs) where experimental data were unavailable. In the case of the Tees, their conclusion was that poor water quality and acute toxicity observed using the oyster embryo bioassay (see Section 2 for 1992 data; for methodology see Thain, 1991) probably resulted from additive toxicity, as the concentrations of individual organic compounds measured or estimated, were all below toxicity thresholds.

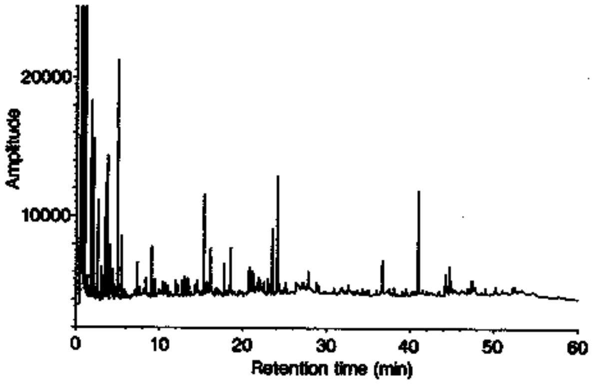

Figure 7. Gas chromatogram of a sub-surface water sample from Station 17 (River Tees, ICI no. 4 buoy)

10. VOLATILE ORGANIC COMPOUNDS

10.1 Introduction

A wide range of volatile organic compounds have an industrial use as solvents and chemical feedstock reagents, and can be found in many discharges to rivers and estuaries in the UK. For example, inputs of chloroform and carbon tetrachloride to the Mersey estuary from the Manchester Ship Canal and River Weaver, were estimated to be 5.7 and 11.7 tonnes per annum, respectively, from 1987 to 1990 (Rogers *et al.*, 1992).

Chloroform and carbon tetrachloride were included in the priority hazardous substance listing attached to the Final Declaration of the Third International Conference on the Protection of the North Sea (1990), with a recommendation that such discharges into rivers and estuaries should be reduced by 50% or more between 1985 and 1995. Consequently the UK National Monitoring Programme (NMP) incorporates volatile organics.

The concentrations of chloroform, carbon tetrachloride and eleven other volatile compounds were determined in water samples collected from NMP stations in October 1992 to provide more information on the distribution and fate of volatile organic contaminants present in UK waters.

10.2 Methods

All analyses were carried out using a commercially built purge and trap analyser. The system consists of an automatic sampling system capable of transferring 10 ml of water sample from a 40 ml vial to a purge vessel. The sample is then purged with helium for a pre-set time and the volatile compounds collected in a cryotrap composed of a length of fused silica which is cooled by means of liquid nitrogen. The volatile compounds are then desorbed by pulsed heating to a fused silica column in a gas chromatograph, and finally analysed using a quadrupole mass spectrometer in the multiple-ion detection mode.

10.3 Results and discussion

The highest concentrations of volatile organic compounds in water were detected in the Tees estuary, with lower concentrations in the Wear, Tyne, Humber and Mersey. In the Tweed estuary which is not industrialised and is considered likely to be only lightly contaminated, and at off-shore sites, few volatile components were detected. Figure 8 (a-e) shows concentrations of some volatile organics in the Tees estuary.

Chloroform was commonly detected in the estuaries; the highest chloroform concentration was found in water from the Tees (11 500 ng l^{-1}) whilst the concentrations found in the Wear, Tyne, Humber and Mersey estuaries were below 250 ng l^{-1}. These values are in general agreement with those found during previous surveys of the Mersey (Rogers *et al.*, 1992). Inputs of chloroform arise from miscellaneous industrial discharges, as the compound is used both in the manufacture of fluorocarbons and as an industrial solvent. Chloroform is also produced during the manufacture of other chlorinated bulk chemicals, including 1,2-dichloroethane, trichloroethylene and tetrachloroethylene (perchloroethylene). Another possible source of chloroform is as a by-product of the chlorination of drinking water.

Concentrations of 1,2-dichloroethane (EDC) in the Tees ranged from 720 ng l^{-1} to 4020 ng l^{-1}. The only other site in the survey where EDC was detected was the Mersey estuary, but only at concentrations below 50 ng l^{-1}. EDC is not thought to occur naturally, but is used in the industrial production of vinylchloride and polyvinylchloride (PVC). The Tees mid-estuary sample containing the highest concentrations of EDC was collected from a station near the Wilton industrial area. As this is a major site for production of both EDC and the vinylchloride monomer, this probably accounts for the detection of higher concentrations of EDC in the Tees estuary and in the waters of Tees Bay.

1,1,1-trichloroethane, trichloroethylene and tetrachloroethylene were detected in water samples from the Tees, Humber, Tyne, Wear and Mersey at concentrations of between 10 ng l^{-1} and 600 ng l^{-1}. These compounds are used industrially as metal degreasing and dry cleaning solvents. Therefore, they probably originate from various inputs of industrial effluent into the estuaries.

o-xylene was detected at a concentration of 1300 ng l^{-1} at the mid-estuarine site of the Tees. The most widespread use of xylene is as a component of petrol, although the isomer *o*-xylene is also an intermediate in the production of phthalic anhydride which is used to produce plasticizers for vinyl plastics.

Bromodichloromethane was found at concentrations of up to 1150 ng l^{-1} in the Tees, and at concentrations of <100 ng l^{-1} in the other industrialised estuaries. Other trihalomethanes (THMs), dibromochloromethane and bromoform, were also detected in the estuaries at concentrations of <150 ng l^{-1}. In the Tees the bromoform concentrations were fairly constant along the estuary, in the samples from Poole harbour the bromoform concentrations were seen to increase with increasing salinity, and in the Wear estuary bromoform was detected in samples taken inshore, but not in samples from the inner estuary. These results indicate that the detected bromoform was not necessarily due to bromoform discharged into the estuaries but probably resulted from diffuse bromination reactions.

Excluding bromoform, the concentrations of volatiles were generally found to decrease with increasing salinity to levels below the limits of detection in samples taken from the coastal in-shore sites. Sources of volatiles appear to be anthropogenic, originating mainly from the discharge of industrial effluent into estuaries.

An assessment of the biological impact of these concentrations of volatile compounds is difficult with so few data. Chloroform and EDC were detected in the Tees at concentrations of up to 11 µg l^{-1} and 4 µg l^{-1} respectively. These values approach EQSs proposed by EC Directive of 12 µg l^{-1} annual mean for chloroform in water and 10 µg l^{-1} for EDC. Oyster embryo bioassay data for whole water samples at the same sites but at a different time of year (see Section 2.3) showed no adverse effects. However, the assay was carried out in a test system that would not prevent loss of volatile compounds during the test. In view of the complex mixture of volatiles at some sites and the relatively high concentrations present, further work is planned for the Tees.

The full results of this survey will be reported elsewhere (Dawes and Waldock, in press.).

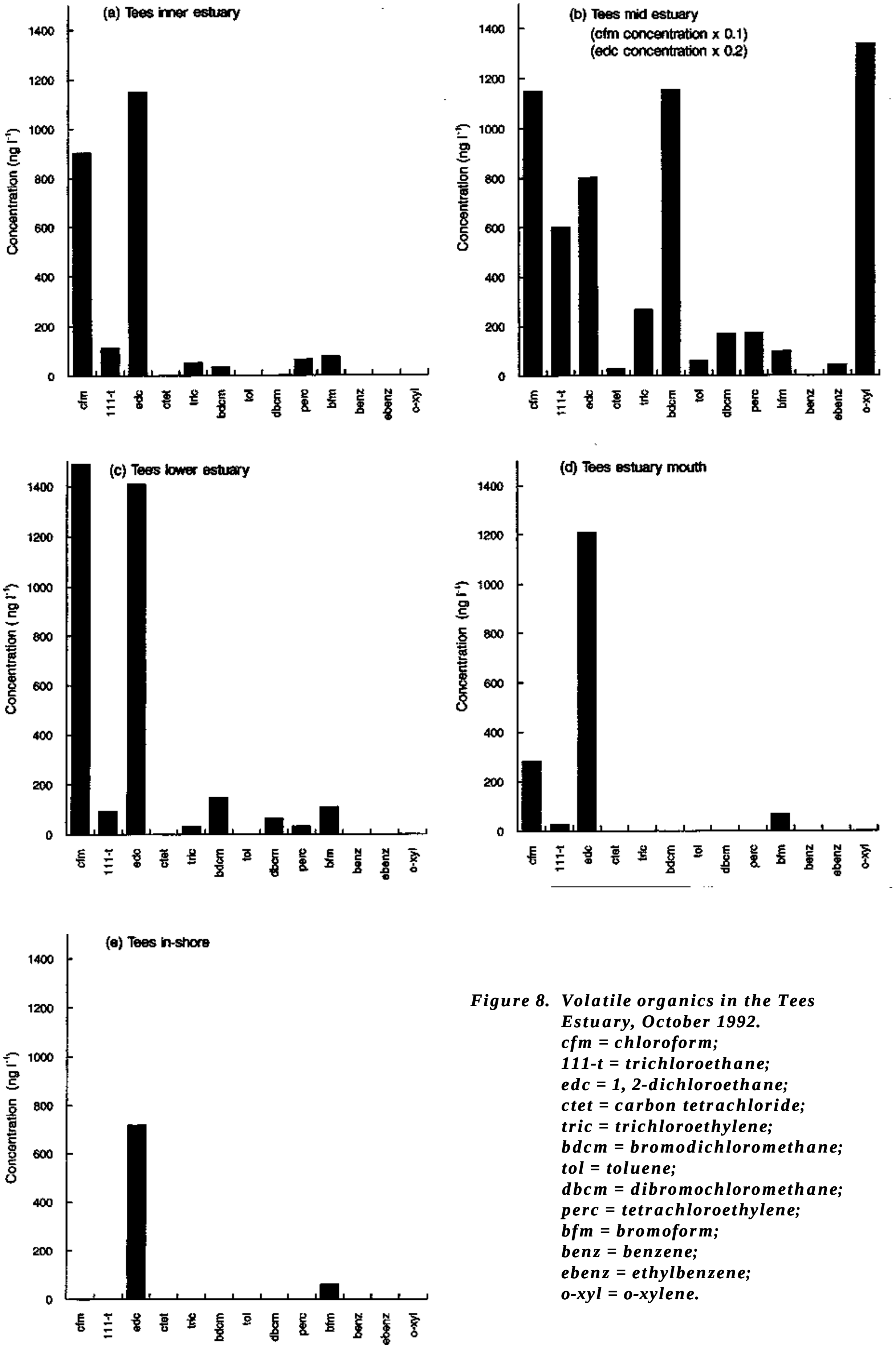

Figure 8. Volatile organics in the Tees Estuary, October 1992.
cfm = chloroform;
111-t = trichloroethane;
edc = 1, 2-dichloroethane;
ctet = carbon tetrachloride;
tric = trichloroethylene;
bdcm = bromodichloromethane;
tol = toluene;
dbcm = dibromochloromethane;
perc = tetrachloroethylene;
bfm = bromoform;
benz = benzene;
ebenz = ethylbenzene;
o-xyl = o-xylene.

SEDIMENTS

11. METALS IN SEDIMENTS AROUND ENGLAND AND WALES

11.1 Introduction

In the late 1980s, the Joint Monitoring Group (JMG) of the Oslo and Paris Commissions initiated an international programme to assess the spatial distribution of metals and organic substances in marine sediments throughout the Commission area. Seabed samples for this ‘JMG Baseline Sediment Study’ were collected in 1990 and 1991 by 12 countries. Data from analyses of the samples were submitted to ICES and combined with results produced for the North Sea Task Force (NSTF) assessment of the quality of the North Sea. The whole dataset has been assessed by various sub-groups of the JMG and NSTF and was used in the 1993 Quality Status Report for the North Sea (NSTF, 1993).

This section discusses the data produced from samples collected around England and Wales by the Burnham Laboratory.

11.2 Methods

Seabed sediments were sampled at 650 locations around England and Wales (Figure 9) between May 1990 and December 1991 using the MAFF ships *RV CORYSTES* and *RV CIROLANA* and the University College of North Wales (UCNW) vessel *PRINCE MADOG*. Most samples were collected using a Day grab fitted with stainless steel jaws, but where the substrate was either too hard or too gravelly for the effective use of this device, a stainless steel Shipek grab was employed.

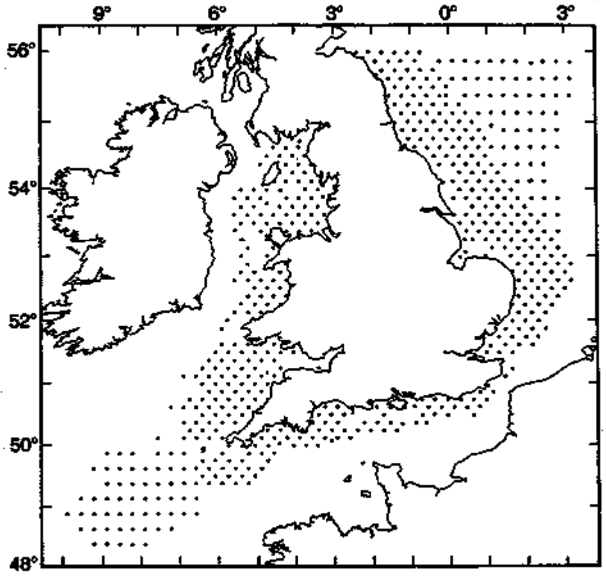

Figure 9. *Sampling sites used in the JMG baseline sediment survey, 1990/1991*

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

500 mg of the powder was completely digested in a microwave furnace using a mixture of hydrofluoric acid and *aqua regia* (Jones, in prep). The resulting solution was treated to minimise matrix effects before being analysed for a range of elements including lead, zinc, chromium and aluminium.

The collection and analytical methods used in this study followed ICES advice (ICES, 1987).

11.3 Results

Table 15 gives a statistical summary of the data obtained for Al, Cr, Pb and Zn.

Table 15. *Summary statistics for the variables measured in the JMG Baseline Sediment Survey. All concentrations as mg kg^{-1} except aluminium which is in per cent*

Element	Al	Cr^{+}	Pb	Zn
N	577	577	577	577
Mean	1.5		14.1	22
Std dev	1	17	15.7	19
Max	7.4	100	110	153
Median	1.4	18	11	15
Min	0.05	<5	3.1	3

Note: $^{+}$ *no mean value has been calculated for chromium as a significant proportion of the data are below the detection limit*

The sediments consisted largely of quartz sand and clay minerals with some shell material and rock fragments (especially near the coast). Clay minerals have a relatively high natural metal content compared to sand and also a greater active surface area which may adsorb metals. This causes concentrations of metals to be generally higher in areas with sediments rich in clays than in sandy regions. As clay particles are usually smaller than sand, the effect generally appears as one of grain size. The effect is demonstrated in Figure 10 where the muddier coastal and estuarine samples can be seen to contain higher concentrations of aluminium, an element present in clay, but not to a significant extent in quartz sand. Such mineralogical and grain size effects can confound attempts to compare concentrations of metals throughout a given area. The effect may be taken into account by using a normalisation procedure (ICES, 1987, Loring,1991).

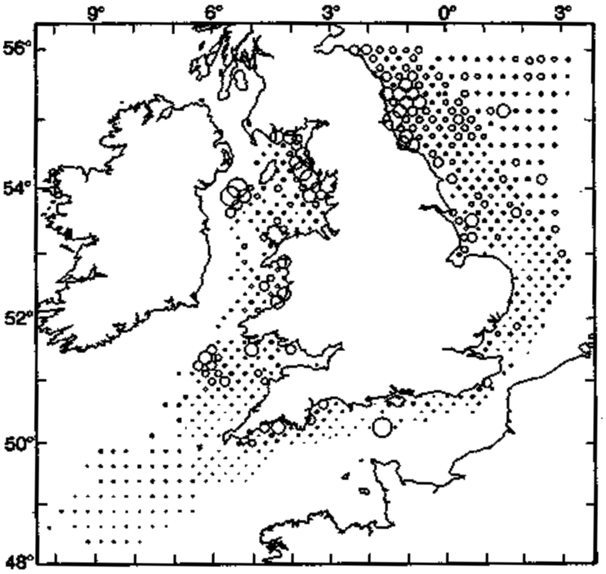

Figure 10. Distribution of concentrations of aluminium in sediments from around England and Wales expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent (The highest percentile groups are represented by the largest circles)

In the present context, normalisation requires the use of a non-contaminant element associated with clay minerals to account for mineralogical variations. Loring (1991) has questioned the usefulness of aluminium in high latitude areas which have been subject to glaciation. He has observed that such sediments may contain aluminium-bearing feldspars in the coarse sediment fraction which cause a reduction in the strength of the aluminium/clay relationship. In such situations he prefers the use of lithium. However, in the present study, tests have shown that aluminium gives similar results to lithium and, because more data are available for aluminium, it is therefore used as the normalising element.

Several methods of normalisation are possible, ranging from the use of simple metal/normaliser ratios to more complex methods based on regression analysis. As the metal/aluminium regressions in the present dataset have significant intercepts on the metal axis, the data show spuriously high metal/aluminium ratios at low aluminium values. Accordingly, rather than use the simple metal/aluminium ratio, it was decided to use a method less sensitive to the effects of the intercept value. The chosen technique is based on a metal/ aluminium regression model with calculation of residuals about the regression line (Figure 11). Most normalisation methods are only suitable for a limited range of elements; the one used here is appropriate for chromium, lead and zinc. For clarity and ease of identification of extreme values, the charts prepared from these residual data use symbols representing percentile classes with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent.

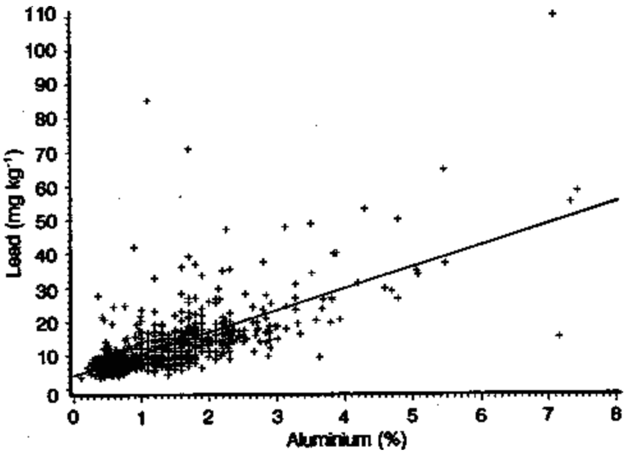

Figure 11. Relationship between concentrations of lead and aluminium in sediments from the coastal zone of England and Wales

Figures 12-14 show the distribution of the residuals for Cr, Pb and Zn. Individual high values, while of interest, need confirmation; only clusters of 2 or more are taken as indicating an effect.

Chromium values (Figure 12) are relatively high in the Tees area, at the centre of the Dogger Bank, along the south coast of England and in Morecambe Bay.

Lead values (Figure 13) are higher near the coast than offshore with the highest values occurring off the north-east coast of England and in the outer Bristol Channel.

Zinc values (Figure 14) are also relatively high near the north-east coast of England, in the north-east Irish Sea and the outer Bristol Channel.

Overall the picture is one of higher values near to shore and to estuaries.

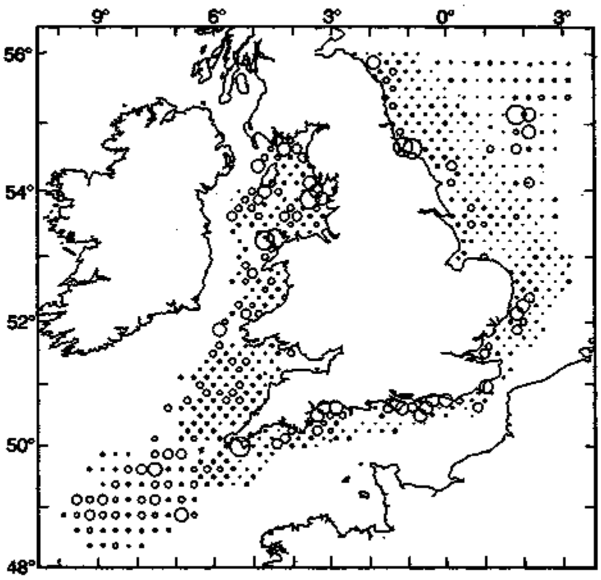

Figure 12. Residual values around the relationship between chromium and aluminium in sediments from around England and Wales expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent (The highest percentile groups are represented by the largest circles)

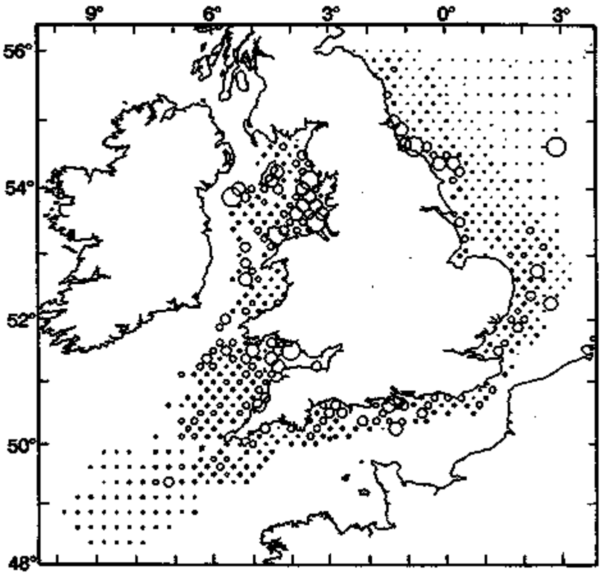

Figure 14. Residual values around the relationship between zinc and aluminium in sediments from around England and Wales expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent (The highest percentile groups are represented by the largest circles)

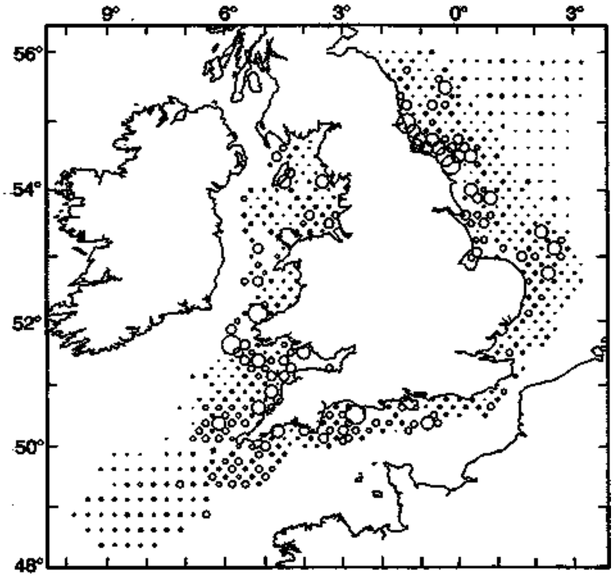

Figure 13. Residual values around the relationship between lead and aluminium in sediments from around England and Wales expressed as percentile groups with boundaries set at 99, 95, 90, 80, 75, 50, 25, 20,10, 5 and 1 per cent (The highest percentile groups are represented by the largest circles)

11.4 Discussion

Sediments are widely recognised as a suitable medium for assessing environmental quality as they reflect and integrate contaminant inputs to the marine environment, allowing sources and sinks to be identified (e.g. ICES, 1989). Detailed analysis of such information is only possible after compositional variations due to mineralogical and grain-size differences have been taken into account. In the present study this has been achieved by normalising the dataset using the concentration of aluminium.

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

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

As suitable bioassays are only in the early stages of development, it is not yet possible to comment on the biological significance of the elevated metal concentrations. The effects of metals will be investigated as soon as suitable sediment tests are available for routine use.

SPECIAL STUDIES

12. TRIBUTYLTIN CONTAMINATION OF MUSSELS AROUND THE UK COAST

Detailed surveys of concentrations of tributyltin (TBT) in shellfish growing areas, marinas and major estuaries and harbours used by ships have yielded comprehensive, but localised, information on the extent of TBT contamination (see MAFF, 1991 and 1992). In order to assess the geographical extent of TBT contamination, wider surveys have been undertaken using mussels, *Mytilus edulis*, as bioindicators of contamination.

Mytilus edulis have been collected each year from 1985-1992 from a variety of sites around the UK coast. Additionally, caged mussels deployed as part of a Plymouth Marine Laboratory (PML) study on 'scope for growth' at off-shore sites were also sampled and analysed. Generally a minimum of 20 mussels was collected from each site and the sample was then stored frozen at -20°C until analysis. Mussel samples were solvent extracted and analysed by gas chromatography (Waldock *et al*., 1989).

The results of the mussel surveys conducted between 1985 and 1992 are summarised in Table 16 and some of the data for 1985 and 1990 are represented in Figures 15(a) and (b).

The largest data sets were obtained for mussels sampled in 1985 and 1990. The range of concentrations of TBT was much greater in mussels sampled in 1985 (0.02-2.93 µg g^{-1} wet weight) than in 1990 (0.01-0.36 µg g^{-1} wet weight); however, it must be noted that the sampling locations were not all the same for the two years. Figures 15(a) and (b) illustrate the overall decrease in concentrations of TBT in mussel tissue between 1985 and 1990. For example, taking an arbitrary value of 0.1 µg TBT g^{-1} wet weight, in 1985 81% of the mussel samples contained >0.1 µg TBT g^{-1} wet weight but by 1990 (excluding the Scottish samples) only 25% contained >0.1 µg g^{-1} wet weight.

In 1990, 5 samples were taken from the east coast of Scotland, Orkney and the Shetlands. In the case of the Shetland and Orkney samples, the concentrations of TBT in the mussels (0.003-0.02 µg g^{-1} wet weight) are probably a result of inputs associated with shipping activity in and around the oil ports.

The 1991 mussel samples from the off-shore sites (see Figure 16) contained concentrations of TBT in the range 0.008-0.02 µg g^{-1} wet weight. However these mussels were originally collected from Exmouth and then relaid at the off-shore sites (which are light vessels). At the time of collection the mussels from Exmouth contained 0.02 µg g^{-1} wet weight.

Table 16. Concentrations of tributyltin in Mytilus edulis

Year	Number of samples	Range of TBT concentrations (µg g^{-1} wet weight)	Number of samples where TBT >0.1 µg g^{-1} wet weight
English/Welsh sites			
1985	27	0.02-2.93	22
1986	4	0.05-0.24	3
1987	6	0.18-0.88	6
1989	12	0.02-0.46	7
1990	36	0.005-0.36	9
1991	16	0.01-0.46	7
1992	4	0.005-0.06	0
Scottish sites			
1990	5	0.003-0.05	0
Offshore sites			
1991	10	0.008-0.02	0

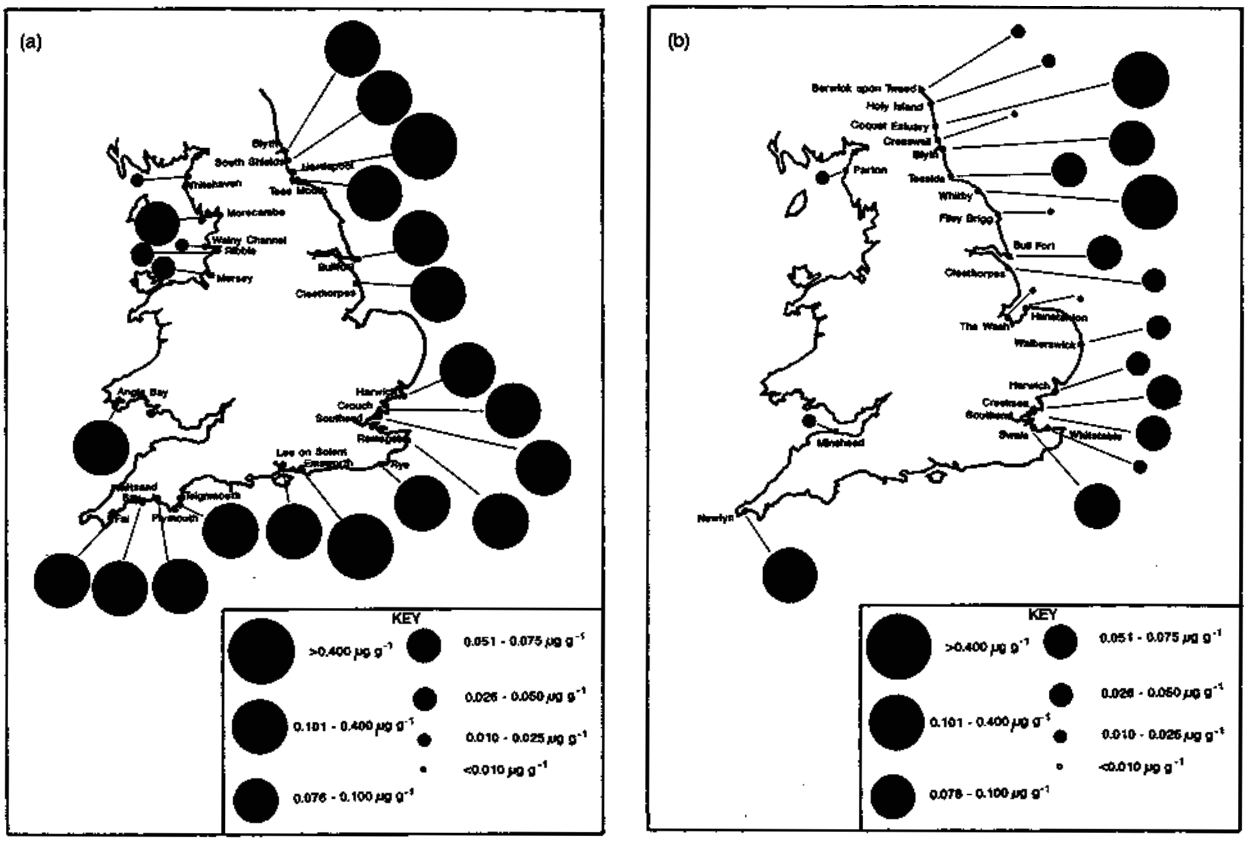

Figure 15. Concentrations of TBT (µg g^{-1} wet weight) in mussel tissue during: (a) 1985 and (b) 1990

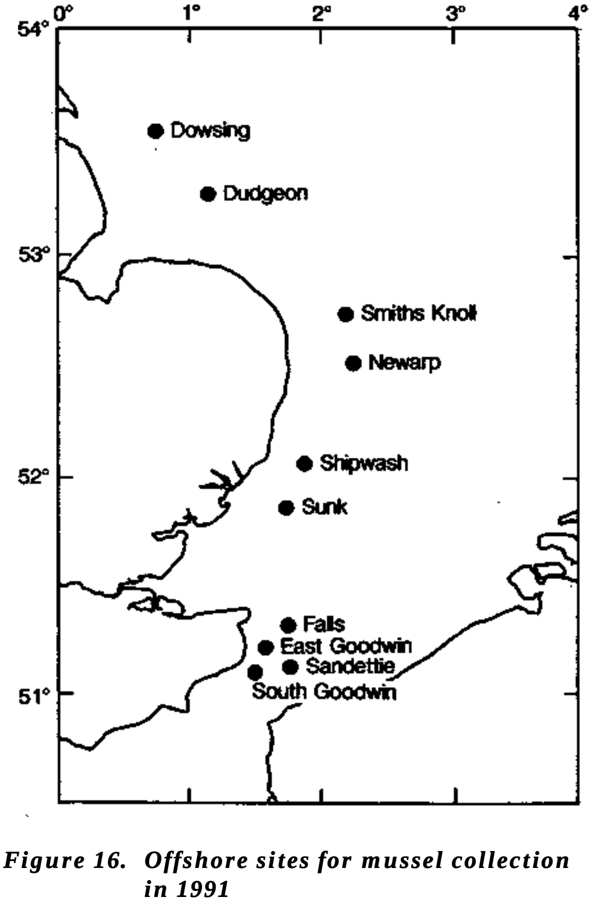

Figure 16. Offshore sites for mussel collection in 1991

Mussels collected from sites of shellfish cultivation, where the range of concentrations of TBT in sea water is 1-100 ng l^{-1}, have bioconcentration factors of approximately 20 000. Salazar *et al.* (1987) found that the bioconcentration factors for mussels in TBT contaminated sea water were generally in the range 20 000 to 40 000. The known tissue concentration and approximate bioconcentration factor can be used to estimate the concentration of TBT in the sea water that the mussels were exposed to. Using a bioconcentration factor of 20 000 it can be estimated that mussels which contained TBT at a level of 0.04 µg g^{-1} wet weight would have been exposed to 2 ng TBT l^{-1} sea water, i.e. the current UK EQS for sea water. Therefore, in 1985, 85% and in 1990, 56% (excluding the Scottish samples) of the mussel samples would have been exposed to TBT concentrations >2 ng l^{-1}.

The previously reported survey work in estuaries and harbours (MAFF, 1991 and 1992) has demonstrated that concentrations of TBT in waters used for recreational purposes dropped markedly following controls on use, introduced under the Food and Environmental Protection Act in 1987. By 1992, TBT values were reducing more slowly reflecting inputs from the reservoir of TBT present in sediments and the continued use on large vessels (>25 m).

The overall conclusions from the survey, are that significant environmental benefits have resulted from the FEPA controls. However, TBT concentrations will remain at low, but detectable, levels in most UK estuaries until further restrictions are introduced to control inputs from dry-docking activity and, the release of TBT from ships' paints is either reduced or discontinued.

13. PESTICIDE RUN-OFF AT ROSEMAUND FARM AND ITS BIOLOGICAL EFFECTS

13.1 Introduction

Earlier pesticide run-off data from Rosemaund Farm have been presented in previous reports in this series (MAFF, 1991 and 1992). The main purpose of the work has been to generate detailed profiles of concentrations of pesticides in the headwater stream which rises on the site. These are being employed to calibrate models of pesticide translocation which will eventually be used to make improved predictions of the concentrations of new pesticides likely to be found in surface waters. This type of prediction is an essential component in the evaluation of the risk to aquatic life which new agrochemicals may pose.

13.2 Results

In 1992, data were obtained on the triazine herbicides, atrazine, trietazine and terbutryn, and the carbamate nematicides/insecticides, carbofuran and aldicarb. These products were applied according to Good Agricultural Practice (Table 17) to fields near the upper end of the Rosemaund catchment, a site with crack-prone clay soils in which pesticides have been found to translocate quickly to the field drains and thence to the stream. This phenomenon is known as by-pass flow.

Table 17. Pesticide applications to the Rosemaund catchment

Pesticide	Area treated (ha)	Application rate (kg ha^{-1})	Date
atrazine	6.0	0.85	27/11/91
carbofuran	3.0	3.0	3/12/91
aldicarb	4.2	5.5	27/2/92
terbutryn	6.0	1.0	5/3/92
trietazine	6.0	1.0	5/3/92

As in previous years, concentrations of the pesticides in question were monitored in a field drain issuing from the treated fields, and in the stream water itself approximately 50 m downstream of the drain outflow. In common with the pesticides studied earlier, background concentrations in the stream tended to lie below the detection limits (0.01 µg l^{-1}), but peaked sharply during rainstorms exceeding approximately 10 mm in 24 h. The pesticide peak usually coincided with the peak in stream flowrate, but then declined rapidly (12-24 h) to background levels. The peak concentrations measured in streamwater are shown in Table 18 for each rainfall event that was monitored.

Table 18. Peak concentrations of pesticides in Rosemaund stream during several rainfall events in 1992

Date of event	Amount of rain (mm)	Atrazine (µg 1^{-1})	Carbofuran (µg 1^{-1})	Aldicarb* (µg 1^{-1})	Terbutryn (µg 1^{-1})	Trietazine (µg 1^{-1})
8/1/92	71.5	5.7	26.8	-	-	-
25/1/92	9.0	13.3	49.4	-	-	-
14/4/92	19.0	0.23	2.3	1.0 approx	0.16	1.3
28/5/92	18.0	0.13	0.02	<0.01	0.06	0.52

* *Aldicarb is biodegraded rapidly in soil to the sulphoxide and the sulphone. No unchanged aldicarb was detected in the stream, and the values given refer to sulphoxide and sulphone combined*

It can be seen that the highest concentrations were of the nematicide carbofuran which peaked in the stream at nearly 50 µg l^{-1} on 25 January. The highest level in the drain was 264 µg l^{-1} during the large rainfall event on 8 January, and concentrations in the drain of all pesticides were generally about ten times those seen in the stream. It had been expected that carbofuran might reach concentrations which would be of toxicological significance to certain sensitive stream fauna. For that reason, the amphipod crustacean *Gammarus pulex* was deployed as a bioassay in the stream from late October 1991 to 10 January 1992. One hundred animals were exposed in small individual cages and supplied weekly with pre-weighed leaf discs which were subsequently weighed again to determine feeding rate, a sensitive sublethal measure of pesticide poisoning. No significant changes in feeding rate were seen until the week of 3-10 January when feeding ceased and all the test animals died. The detailed course of the event on 8 January is shown in Figure 17, and the horizontal dotted line indicates the 24 h LC50 of carbofuran to *G. pulex* (21 µg l^{-1}) which was subsequently measured in the laboratory. Despite the transitory nature of the elevated pesticide concentrations, it is clear that they remained at or above the 24 h LC50 sufficiently long enough to cause the observed toxic effects. Atrazine is comparatively benign to crustacea at the concentrations seen (<10 µg l^{-1}), so there is little doubt that carbofuran was responsible for the kill. Although bioassays were not deployed during the event on 25 January, the concentration profile (not shown) would also have resulted in mass mortality of aquatic arthropods (insect larvae and crustacea).

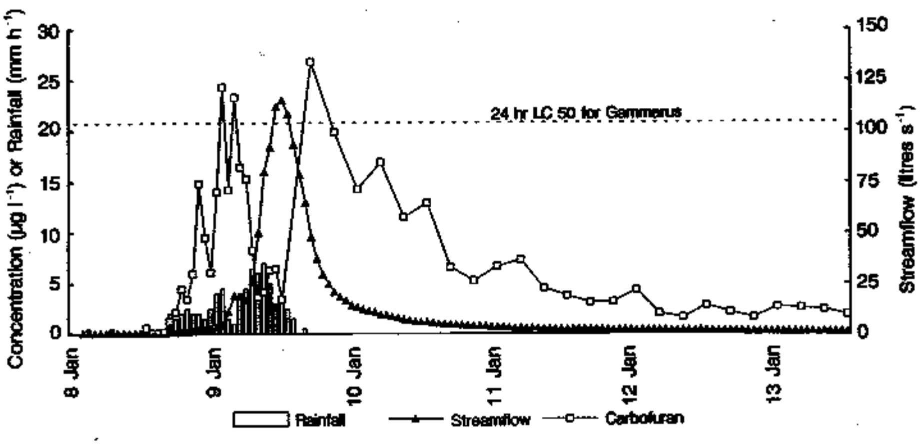

Figure 17. Carbofuran event in the stream at Rosemaund Farm

13.3 Discussion

These studies are some of the first to have deployed sensitive bioassays in headwater streams near agricultural fields, and they indicate that at least one pesticide is able to translocate rapidly through soil to appear in surface waters at harmful concentrations. Earlier *G. pulex* bioassays deployed at Rosemaund during spring 1990 had been unable to detect impacts from a runoff event of the herbicides mecoprop and dichlorprop (peak combined concentration = 2.4 µg l^{-1}), but this was not altogether surprising, because these substances are not very toxic to crustacea. The indigenous faunal community in the Rosemaund stream is in fact significantly impoverished for the type of habitat present. It therefore seems likely that carbofuran, and probably some other pesticides, are causing unforeseen impacts to aquatic life in some areas.

The value of the model calibration work is self-evident, and means the risks associated with this type of pesticide translocation process will have to be more accurately predicted in future. Although the particular soil types at Rosemaund are not very widespread in the UK, soils on which by-pass flow is an important component of the hydrological regime constitute approximately 28% of the UK total. The modelling results are therefore of considerable interest and will be described in detail in a future report in this series.

14. MEASUREMENT OF VITELLOGENESIS IN MALE AND FEMALE FISH EXPOSED TO TREATED SEWAGE EFFLUENTS AND RIVER WATER

14.1 Introduction

During the normal reproductive cycle in female fish, the steroid oestradiol, which is released by the ovaries, stimulates the liver to produce vitellogenin and release it into the bloodstream. Vitellogenin, a calcium rich lipoprotein, is incorporated into the yolk of developing eggs, providing much of the necessary nutritional requirement of the larval fish before they start to feed freely.

Vitellogenin can be measured in the plasma by use of a radioimmunoassay technique (Sumpter, 1985). Purified vitellogenin from rainbow trout is injected into rabbits in order to raise antibodies which are specific to its molecular structure. These antibodies (anti-rainbow trout vitellogenin) are used in the assay in combination with iodinated (radioactively labelled) trout vitellogenin. In a sample which is being assayed, the total amount of antibody which binds to vitellogenin can be determined. Measurement of the percentage of antibody which has bound to the radioactive vitellogenin allows calculation of the amount of vitellogenin present in the sample, this being equivalent to the residual bound antibody.

Studies of the reproductive cycle in trout, which were conducted at the MAFF experimental fish farm at Sacrewell (approximately 7 miles north-west of Peterborough) during 1986, measured vitellogenin unexpectedly in male trout, thus triggering the field investigations described below.

14.2 Initial studies of the effects of treated sewage effluents upon vitellogenesis

The Sacrewell site abstracts its water supply from a river only two miles downstream of a treated sewage discharge. A preliminary study was therefore conducted to ascertain whether treated sewage effluents could be responsible for the increases in vitellogenin observed at the Sacrewell site. Male and female rainbow trout (*Oncorhynchus mykiss*) reared in a spring-fed water supply, were exposed in cages to the final effluents from two sewage treatment works for periods ranging from 33 to 91 days. At both sites after 33 days, both sexes of fish had elevated levels of vitellogenin in their blood. The mean concentrations present in the males were 0.65 and 64 µg ml^{-1} for each site respectively by comparison to a control mean range of 0.02-0.15 µg ml^{-1}. After 67 days at one site, the concentrations of vitellogenin in the males had increased to a mean of 174 µg ml^{-1}. These concentrations of vitellogenin in the males at the treatment works sites are comparable to those found in immature female trout at the beginning of the reproductive cycle.

14.3 A nationwide survey of the induction of vitellogenesis in fish exposed to treated sewage effluents

As a direct result of the findings of the first survey, a nationwide survey was commissioned by the Department of the Environment (DoE) to determine whether all treated sewage effluents would induce vitellogenesis in trout. Out of 31 sites chosen initially, 16 were rejected because they could not support caged fish for the exposure period or were subject to vandalism (details of this survey are discussed in Purdom *et al.*, 1994). Both male and female rainbow trout were placed in cages at the remaining 15 sites. After three weeks exposure to the treated effluents (which derived from combined industrial and domestic sources), blood samples were taken and vitellogenin concentrations measured. The results for the males at all the sites are shown in Figure 18. The range of mean vitellogenin levels in the caged fish was 23 µg ml^{-1} to 65 000 µg ml^{-1}. This range spans the concentrations of vitellogenin which would normally only be found in female trout several months before spawning, up to the point at which ovulation occurs.

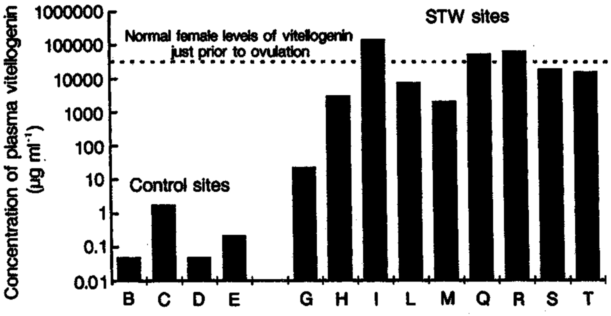

Figure 18. Mean concentrations of plasma vitellogenin in male rainbow trout held in sewage treatment works final effluent for three weeks

14.4 Studies of sites downstream of sewage effluent discharges on a single river system

In the summer of 1992, DoE commissioned a three year study to determine how far downstream of treated sewage effluent discharges, increases in vitellogenin could be detected in fish. This work will include measurement of vitellogenin in fish held in rivers and at drinking water abstraction points, as well as in raw water reservoirs and possibly in treated drinking water, although significant levels of oestrogens are not expected to be present in the latter. In the late summer of 1992, an initial survey was conducted on the River Lea in Hertfordshire. It should be noted that the River Lea was chosen for logistic reasons alone; many other rivers receive equivalent rates of sewage effluent inputs and would therefore be expected to show similar results to the Lea survey. Twelve male rainbow trout were allocated to each of nineteen cages which were placed at sites in the river immediately at the point of discharge of sewage effluents and at various distances downstream of these. A comparison of the initial vitellogenin concentrations measured in the blood of fish before placement in the river with those measured after three weeks exposure at each site, indicated that vitellogenin concentrations were highest in the immediate vicinity of sewage discharges but were still detectable up to 15 km downstream of a discharge (Figure 19). In the majority of cases the vitellogenin concentrations were in the low ng ml^{-1} range, with only those directly at the point of discharge reaching µg ml^{-1} levels.

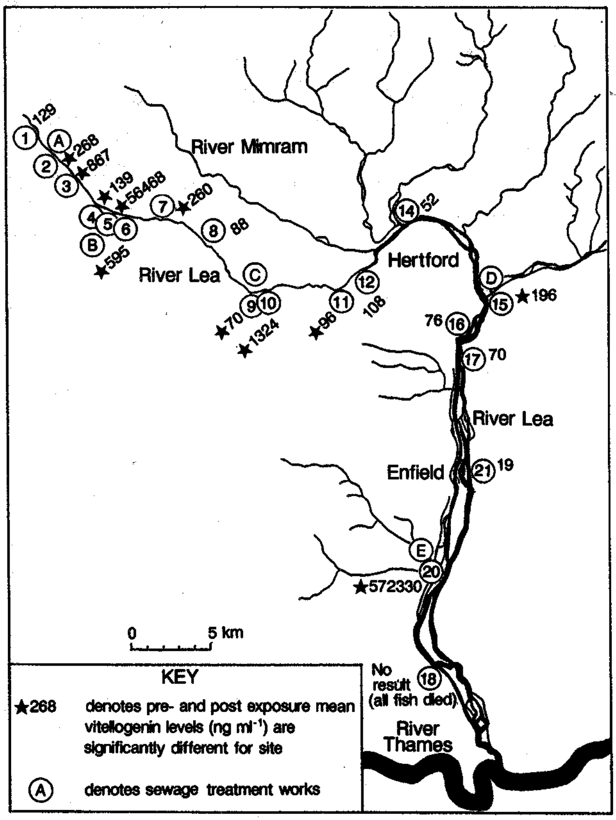

Figure 19. The River Lea, showing mean vitellogenin levels at each site after 3 weeks exposure

A follow-up study, looking in more detail at the Harpenden sewage discharge to the Lea, was conducted in autumn 1992. Two cages containing ten fish each were placed at each of nine sites at the point of discharge and at various distances downstream, for a period of four weeks. Only at the point of discharge were the vitellogenin concentrations significantly elevated by comparison to the initial values (Figure 20), although at the next two sites downstream there was an observed increase in mean vitellogenin concentration which did not prove statistically significant because of the variability in the response of the whole group of fish. The much lower mean vitellogenin concentration (0.38 µg ml^{-1}) at the point of entry of the Harpenden discharge to the Lea, compared to a mean of 56 µg ml^{-1} in the previous survey, was possibly a result of the greater dilution of the effluent at this time of year; comparison of 21 year monthly means for one flow gauging site on the Lea with actual recorded flow figures for November 1992, 1.1 and 2.0 cumecs respectively, illustrates that flows were unusually high at the time the survey was conducted. This is an important illustration of the influence of seasonal factors upon the potential impact of sewage effluents on fish present in the receiving waters.

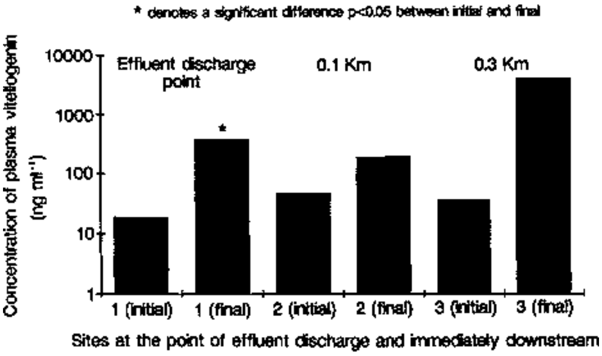

Figure 20. Concentrations of plasma vitellogenin measured in trout before and after a four week period in cages placed immediately down stream of the Harpenden sewage treatment works discharge to the River Lea

14.5 Likely causes of vitellogenesis in male fish

It has been clearly demonstrated that some component of sewage effluents is capable of causing the livers of male fish to produce vitellogenin. This chemical (or chemicals) acts in a manner analogous to oestradiol normally present in female fish and therefore the effect may be termed oestrogenic. Initially, it was considered that 17a ethynylestradiol (EE$_2$), a component of the oral contraceptive pill, was the most likely chemical candidate responsible for induction of vitellogenesis in trout. There is as yet no conclusive analytical data to indicate its presence in environmental samples from the UK, although steroidal oestrogens have been found in effluents and surface waters in Israel (Shore *et al.*, 1993). Laboratory studies at DFR have shown EE$_2$ to be an extremely potent inducer of vitellogenin in trout, positive responses may arise at concentrations of 0.1-0.5 ng l^{-1} (Purdom *et al.*, 1994), but a number of other chemicals which have been detected in sewage effluents (notably a group of non-ionic surfactants, the alkylphenols) have also been shown to induce vitellogenesis. From the results of laboratory studies conducted thus far, it would appear likely that a number of chemicals present in sewage effluents may act jointly to produce vitellogenesis in fish which are exposed to the effluents.

14.6 Future studies

A chemical which behaves like an oestrogen may have the potential to affect sex ratios in eggs or early juvenile fish exposed to it at sufficiently high concentrations, for periods of several weeks or months. The induction of vitellogenesis in immature fish, or males and females out of season, may also have a number of disrupting effects upon life-cycles and reproductive fitness in exposed populations. Future fieldwork will therefore have three main aims: identification of the chemical(s) responsible for oestrogenic effects of sewage effluents; determination of the distances downstream the effects extend, in a range of river systems (this will influence the direction of future laboratory studies); and clarification of whether or not there are consequences for fisheries, or for potable water supplies.

15. STUDIES AT AGGREGATE EXTRACTION AREAS

15.1 The effects of marine gravel extraction on benthos

15.1.1 Introduction

In October 1990, a research project was initiated jointly by MAFF and the Crown Estate Commissioners to investigate the impacts of marine aggregate extraction on the benthos of dredged areas. As part of the study, an offshore experimental site was selected off North Norfolk (Figure 21) following a wide-scale survey of gravel communities off the English coast (MAFF, 1993(a)).

15.1.2 Methods

Experimental dredging
During 5 days in April 1992, the *MV SAND HARRIER* an ‘H’ class commercial suction-trailer dredger, extracted a total of 52 000 tonnes of mixed aggregate,

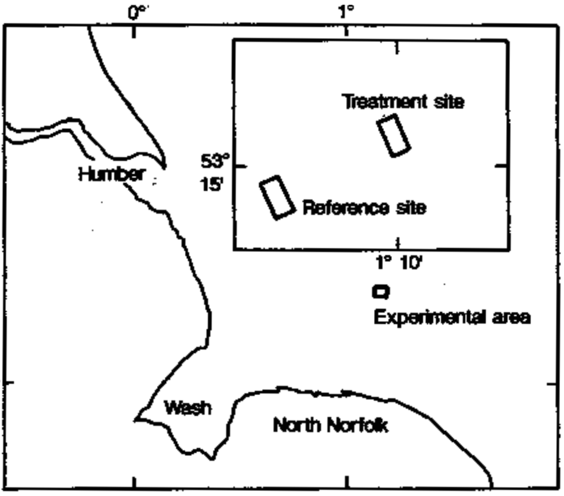

Figure 21. Location of offshore experimental area; inset shows position of reference and treatment sites

representing 11 hopper loads from an area measuring 270 by 500 metres. High navigational accuracy was achieved during the dredging operation using a Differential Global Positioning System (DGPS). This provided an almost constant accuracy of ±10 metres.

Pre- and post-dredging surveys
Remote sampling of the benthos was achieved using a Hamon grab. Both the treatment and reference sites were sampled before dredging in March 1992 and again after dredging in May, August and December. A total of 5 Hamon grab stations were located at random within the defined boundaries of each site during each survey. In order to provide an instant view of the seabed and provide information on the distribution and type of benthic organisms present, an underwater camera sledge was used. In addition, an acoustic map of the dredged site was generated using a dual frequency side-scan sonar.

Hamon grab samples were washed over 5 mm and 1 mm square mesh sieves to remove excess sediment and obtain all the colonial and solitary benthos. The benthos was fixed in 4-6% buffered formaldehyde solution with the addition of 'Rose Bengal' and stored for laboratory processing. In addition, a 1 litre, sub-sample of sediment was taken for particle size analysis.

Laboratory procedures
The samples were sorted and all individual and colonial benthos were enumerated and identified whenever possible to species level. For each species, a representative specimen was preserved and stored separately in a glass vial. Partial wet-weights for each species were determined using a five figure balance. Biomass estimates, in g ash-free dry weight (AFDW) m^{-2}, were calculated from generic conversion factors given in Eleftheriou and Basford (1989). Sediment sub-samples were analysed according to the procedures given in Kenny *et al.* (1991).

Benthic data were transformed (logx+1) before applying the Bray-Curtis distance index (Bray and Curtis, 1957) and then analysed using group average linkage cluster analysis (Lance and Williams, 1967) and presented in the form of a dendrogram.

15.1.3 Results

Physical observations
Particle size data for 6 samples taken from the treatment site in March 1992, four weeks before dredging, were compared to 6 samples taken two weeks after dredging in May. Results show that the gravel content (>2 mm) of the sediment had apparently increased, from 36% to 56% (Figure 22).

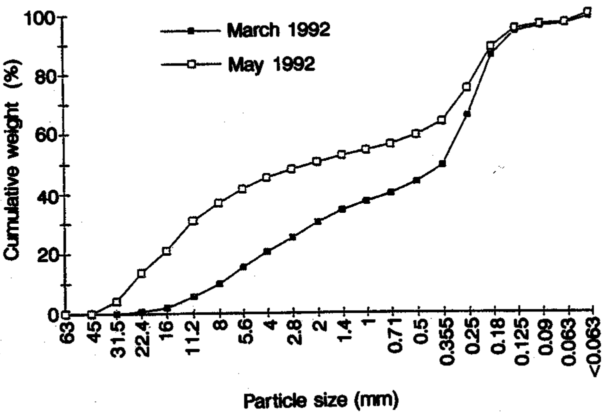

Figure 22. Average cumulative particle size distribution curves for samples taken before dredging (March, 1992) and post-dredging (May, 1992)

Upon examination of the seabed using side-scan sonar and underwater cameras, it was apparent that the dredge tracks had become filled-in with sand, suggesting a redistribution of sediment had occurred. The combined action of the draghead and the post-dredging effects of tides and waves had disturbed the sediment to such an extent that the gravel had consolidated to form ridges between furrows of sand. Inspection, by SCUBA divers, of the sand accumulations within the tracks, showed that the deposits were superficial sand-ripples, 1-2 cm deep. Despite this, it was concluded that the previous action of suction dredging, had led to disturbance, suspension and preferential removal of sand, probably accounting for an overall increase in the gravel content as stated above.

Biological observations
The combined output from cluster analysis on transformed abundance data, pre- and post-dredging, at the treatment site is shown in Figure 23. It is apparent that stations have been grouped according to their respective sampling times (with the exception of Station 15 taken in August) reflecting the change in the abundance and composition of taxa, pre- and post-dredging. Samples taken immediately after dredging in May show the

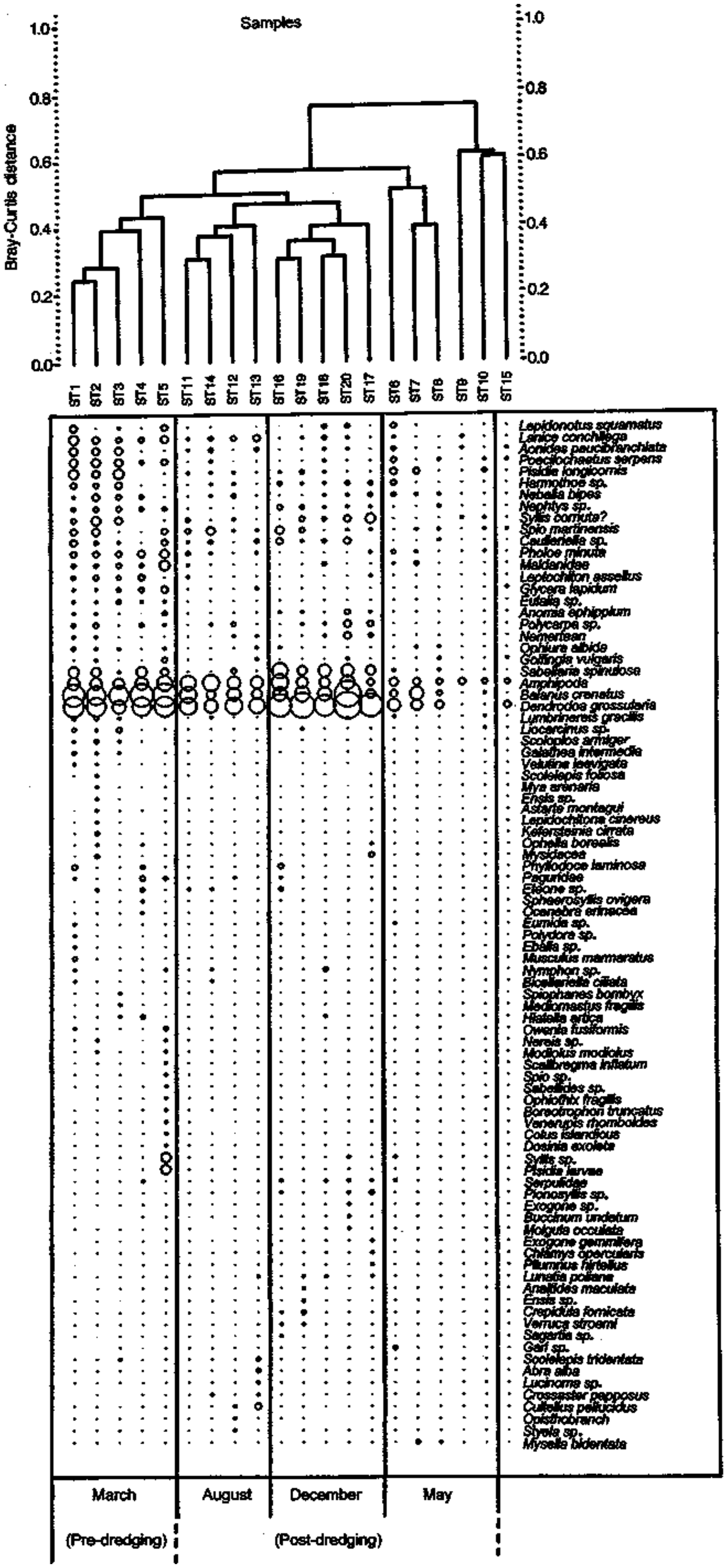

Figure 23. Combined cluster analysis for samples obtained pre-dredging in March (Stations 1-5) and post-dredging in May (Stations 6-10), August (Stations 11-15) and December (Stations 16-20)

greatest variability as indicated by the increased distance between sample groups in the dendrogram. Samples taken in March had large numbers of species with relatively large counts, whereas in May the number of species was greatly reduced, as were the counts. However, when the August samples were compared with those taken in May it was apparent that the overall number of species had increased and that certain dominant taxa, present immediately after dredging, had increased in their abundance due to the recolonisation by juvenile species. Notable among these were, amphipods, barnacles (*Balanus crenatus*) and sea squirts (*Dendrodoa grossularia*). To a lesser extent a number of polychaetes had also increased in density, for example, *Spio martinensis, Lanice conchilega* and *Caulleriella sp*. Samples obtained in December (seven months post-dredging) showed a further increase in the abundance, though not biomass, of *Dendrodoa grossularia, Balanus crenatus* and *Sabellaria spinulosa* in excess of pre-dredging levels. However, there were a number of taxa which had not returned to their pre-dredging abundance such as *Pholoe minuta, Leptochiton assellus, Glycera lapidum, Liocarcinus sp, Galathea intermedia* and *Eulalia sp*.

A nearby reference site was sampled in order to assess the spatial and temporal variation in the abundance and number of species present post-dredging, relative to an undisturbed area.

The total numbers of species recorded from 5 Hamon grab samples taken at the treatment and reference sites pre- and post-dredging are shown by major phyla in Figure 24. The reference site exhibited an almost constant number of species (60) during the 8 month sampling period, whereas at the treatment site, an immediate post-dredging reduction from 68 to 28 species occurred. A limited increase in the number of species during the 7 month post-dredging period from 28 to 48 species indicates that some readjustment or recolonisation had occurred.

Densities (excluding *D. grossularia* and *B. crenatus*) at the treatment and reference sites are shown in Figure 25. An 85% reduction in the abundance of benthos had occurred post-dredging, from a total of 594 individuals per m^2 in March to 87 individuals per m^2 in May, whereas at the reference site the total abundance was essentially identical with 554 individuals per m^2 in March and 568 individuals per m^2 in May.

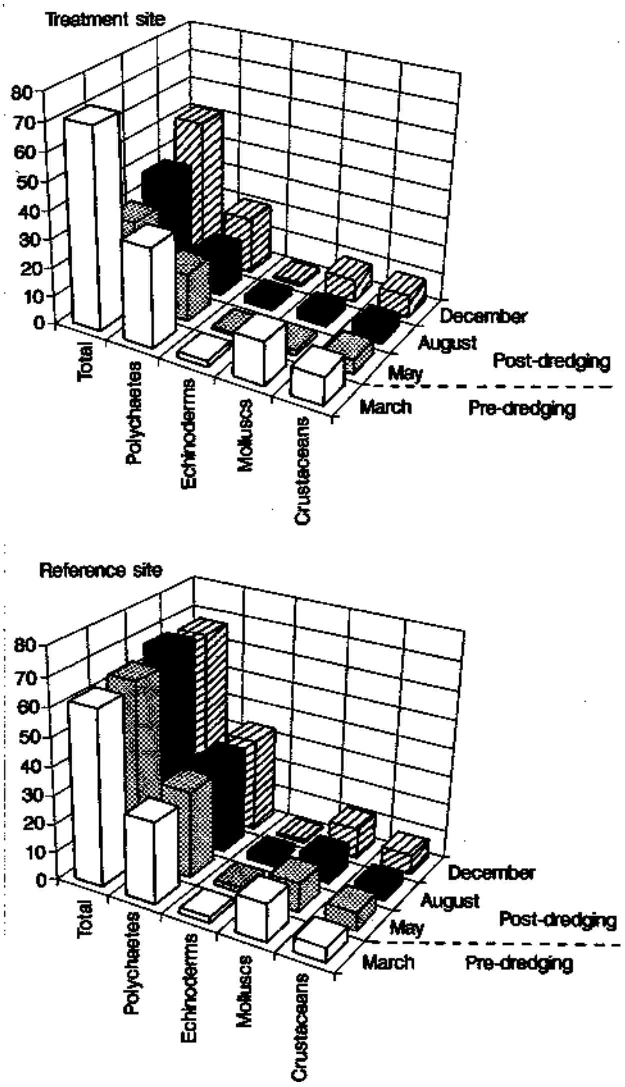

Figure 24. The number of species, recorded by major phyla, from 5 combined samples taken from each site during each survey

Some seasonal changes were observed in the number of individual polychaetes and crustaceans present at the reference site in December. However, although seasonal increase in abundance of certain taxa occurred at the treatment site the nature of the change is different to that observed at the reference site. This is highlighted by comparing the abundance of *D. grossularia* and *B. crenatus* (Figure 26). Biomass data for *D. grossularia* and *B. crenatus* at each site support the observations made on the abundance data (Figure 27). In addition, the relatively low biomass of *D. grossularia* at the treatment site in December indicates that the populations are dominated by small individuals, suggesting they are new recruits. Similarly, the relatively large biomass of *B. crenatus* at the reference site compared to the treatment site indicates the presence of a mixed population of adults and juveniles, whereas at the treatment site the majority of individuals are juveniles.

15.1.4 Discussion

Dredging at the experimental site in April 1992 was deliberately timed to precede the natural summer recruitment of benthos. The ‘opportunists’ *D. grossularia* and *B. crenatus,* which were numerically dominant before dredging, showed the greatest increase in abundance post-dredging. However, the increase was

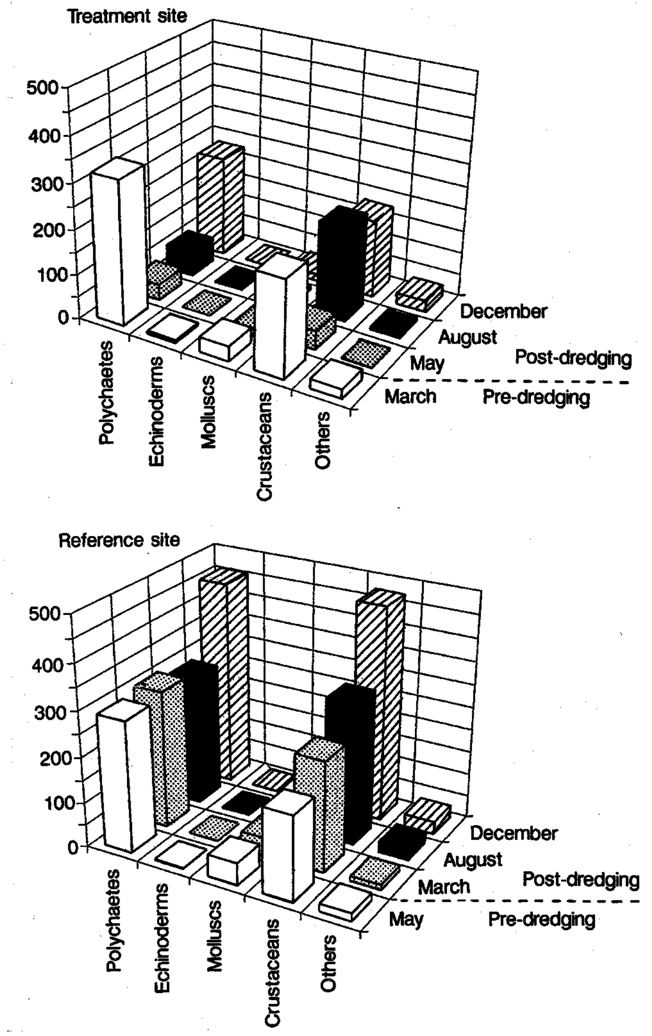

Figure 25. ***Abundance (m^{-2}) of taxa, by major phyla, from 5 combined samples taken from each site during each survey***

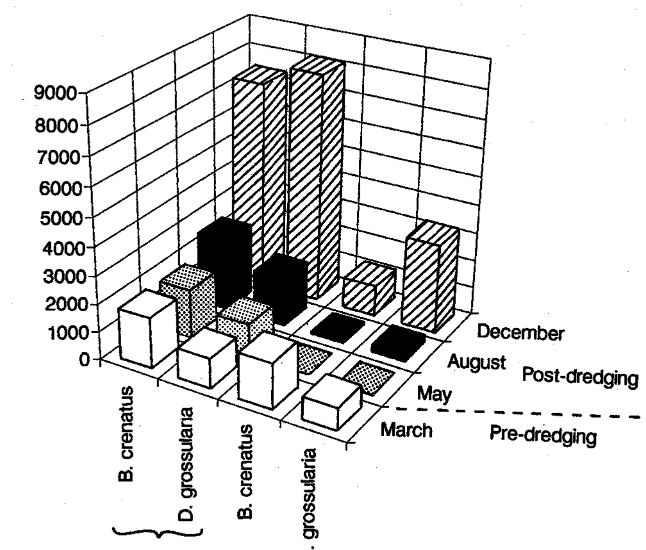

Figure 26. ***Abundance (m^{-2}) of D. grossularia and B. crenatus at the treatment and reference sites for each survey***

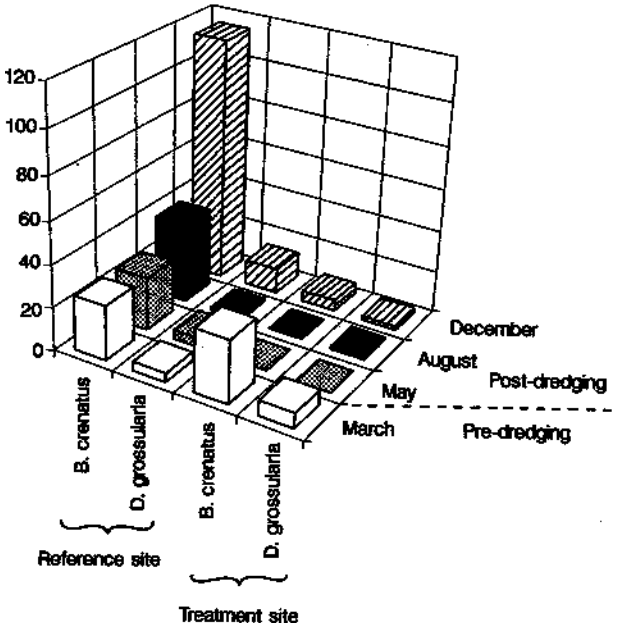

Figure 27. ***Biomass g(AFDW)m^{-2} of D. grossularia and B. crenatus at the treatment and reference site for each survey***

greatest at the reference site. This may be explained by a combination of the following:

(i) the treatment site is physically stressed compared with the reference site, due to deposits of mobile sand being present within the dredge tracks, thereby reducing the recruitment success of sessile epibenthos such as *Sabellaria spinulosa, D. grossularia* and *B. crenatus*;

(ii) there may be spatial differences in recruitment success between the reference and treatment sites, such that a larger settlement has occurred at the reference site;

(iii) the loss of adult sessile populations at the treatment site may have reduced the potential for juvenile recruitment, since the ‘cues’ to settle are no longer present. For example, *B. crenatu*s may require the presence of adult populations in order to stimulate settlement, as has been observed for *B. balanoides* (Stubbings, 1975). In addition, as many adults have been removed by dredging, the source of juveniles which recruit locally (within 100 m) is reduced; for example, *D. grossularia* larvae are not transported in the plankton but settle within a few metres of their parents, often on the tunics of other ascidians (Svane and Young, 1989).

The effects of seasonality should be borne in mind. The data cover a period of seven months post-dredging and mortalities will have occurred during the winter of 1992, thereby reducing the observed gains in abundance at the treatment site.

It remains to be seen whether the populations at the treatment site adjust to the newly-created physical regime by shifting from a relatively stable community to one characteristic of a more mobile sediment, or whether the sediment restabilises and the benthos returns to a community similar to that which existed pre-dredging.

The results from further surveys, undertaken in 1993 and planned for 1994, will help to further clarify the processes of recolonisation, and the resultant long-term community structure, post-dredging.

DISPOSAL AT SEA: ENVIRONMENTAL ASSESSMENT STUDIES

16. STUDIES AT SEWAGE-SLUDGE DISPOSAL SITES

16.1 Thames transect survey

Between 1887 and 1967, sewage sludge and dredged material were disposed of at two sites in the outer Thames Estuary, one in the Black Deep and one in the Barrow Deep. The Black Deep site received sewage sludge between 1915 and 1967 and dredged material between 1887 and 1920. The Barrow Deep site has been used exclusively for sewage sludge disposal since 1967, with approximately 5 million wet tonnes being disposed there each year. Monitoring to assess the impact of this material has been undertaken regularly since then and it is incumbent on the licensing authority, under the Food and Environment Protection Act (FEPA) (1985) (Great Britain - Parliament, 1985(a)), to satisfy itself that the effects are minimal.

In accordance with this duty a survey was undertaken in May 1992. Sediment samples were collected with a 0.1 m^2 Day Grab from 49 sites in the outer Thames Estuary. Transects were sampled in the Middle, Barrow, Black and Knock Deeps. The Barrow Deep transect was extended offshore as far as the Outer Gabbard, and inshore as far as the Oaze Deep to include ‘background samples’. The samples were wet sieved at 63 µm fraction using *aqua regia* digestion and analysed by atomic absorption spectrophotometry (Harper *et al.*, 1989). Concentrations of metals in the sediments are shown in Figures 28-31.

In general, concentrations of metals do not vary greatly over the survey area, with few areas exhibiting elevated concentrations for all metals. The highest concentrations of metals were generally found along the Barrow, Black and Oaze Deeps. Figure 32 shows the range, and mean concentrations of lead present in the Middle, Barrow, Black, Knock and Oaze Deeps. Although the ranges overlap, the mean values for the Barrow, Black and Oaze Deeps are higher than for the Middle and Knock Deeps. The southernmost sites in both the Knock and Middle Deeps also show relatively high concentrations (Figure 28). With the exception of mercury, concentrations of metals at sites in the eastern part of the survey area, furthest from the influence of anthropogenic inputs, do not differ greatly from many of the sites further inshore.

Concentrations of metals at the Barrow Deep disposal site are not high compared to those at other sites away from the immediate area of disposal.

Sediment transport studies carried out previously (Talbot *et al.*, 1982) suggest that the elevated concentrations of metals along the Barrow Deep, and at the southernmost site in the Middle Deep, may be attributable to sewage-sludge disposal operations at the Barrow Deep disposal site. Higher concentrations of metals in the Black Deep, particularly at the southern end, might be associated with the dredged material and sewage-sludge disposal operations carried out at the Black Deep disposal site prior to 1967. Elevated concentrations of metals in the Oaze Deep may be a result of their closer proximity, relative to the rest of the survey, to the industrialised River Thames.

The results of this survey suggest that heavy metals associated with sewage sludge are dispersed away from the immediate area of disposal to become part of the general sediment metal load for the Thames Estuary. The results also show that concentrations of metals in sediments in the inshore part of the survey closest to both land-based and marine anthropogenic input do not vary greatly from those furthest offshore.

16.2 Nab Tower

Each year approximately 250 000 wet tonnes of sewage sludge, and between 600 000 and 1 500 000 tonnes of dredged material are deposited at the Nab Tower disposal site. As is the case for the Thames sewage-sludge disposal sites, field surveys are carried out at regular intervals to assess whether any adverse effects are occurring as a consequence of this disposal operation.

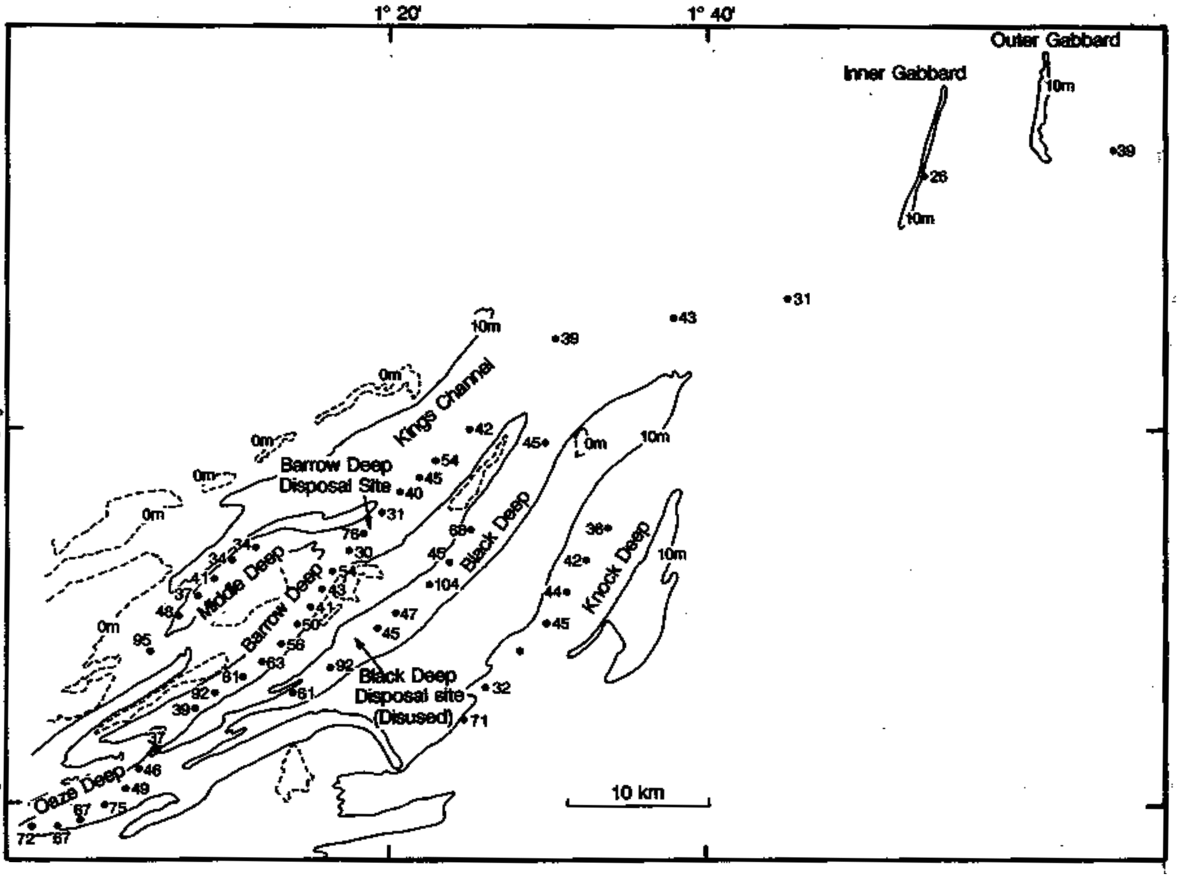

Figure 28. Distribution of the concentrations of lead (mg kg^{-1}) in the fine (<63 µm) fraction of sediment samples collected from the outer Thames Estuary

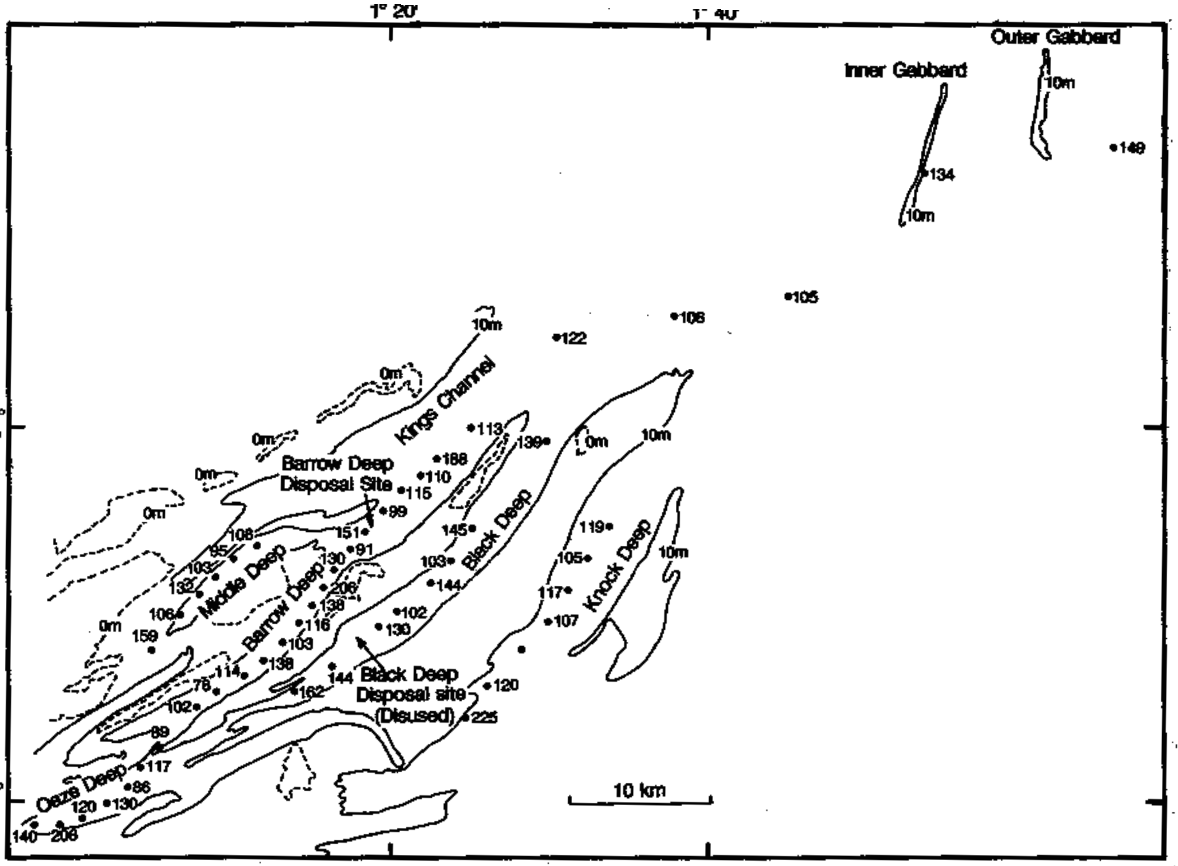

Figure 29. Distribution of the concentrations of zinc (mg kg^{-1}) in the fine (<63 µm) fraction of sediment samples collected from the outer Thames Estuary

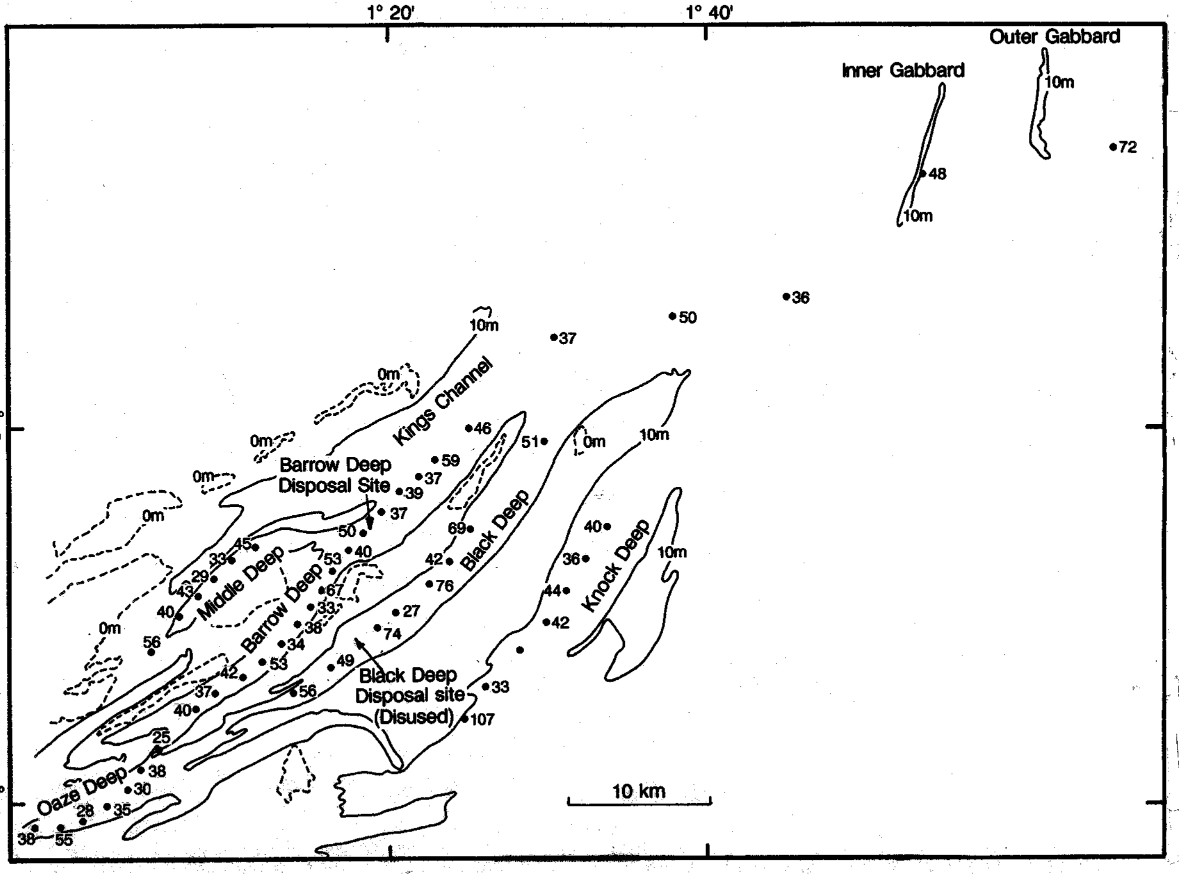

Figure 30. Distribution of the concentrations of copper (mg kg^{-1}) in the fine (<63 µm) fraction of sediment samples collected from the outer Thames Estuary

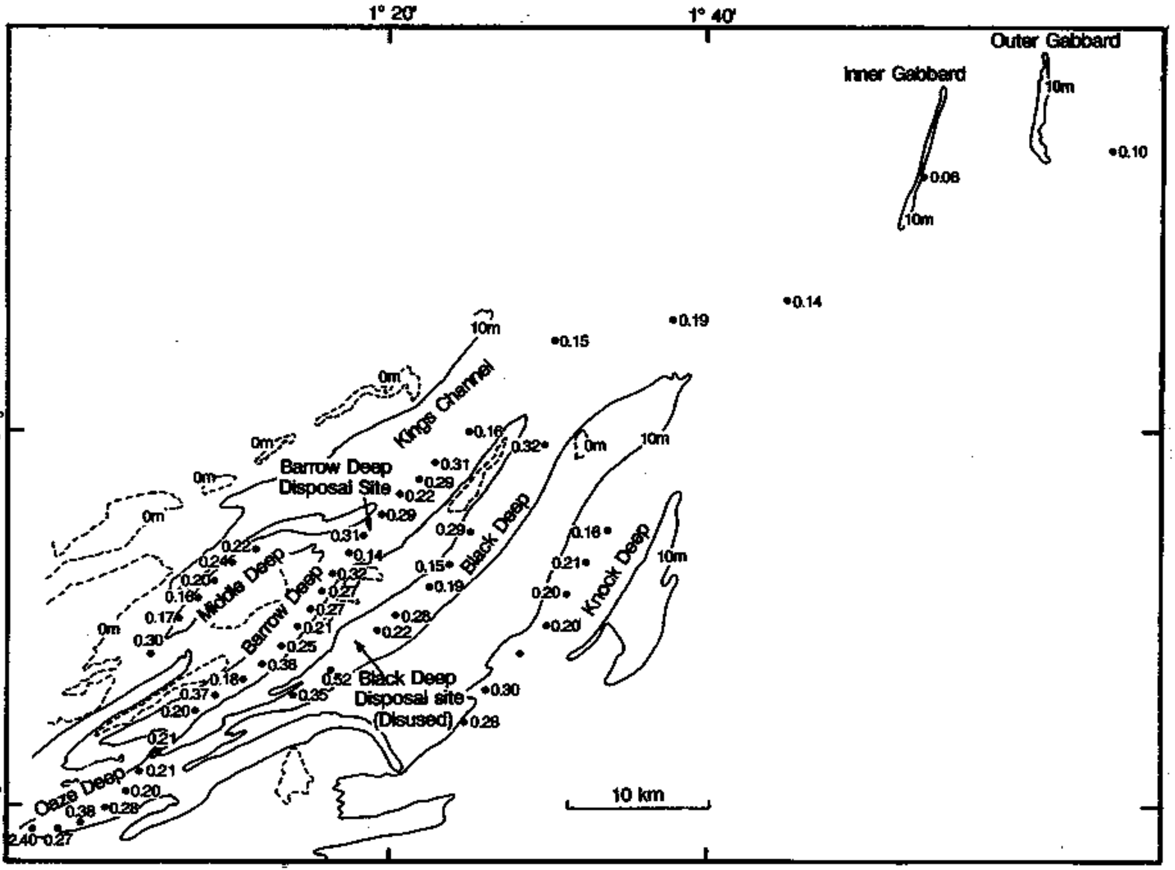

Figure 31. Distribution of the concentrations of mercury (mg kg^{-1}) in the fine (<63 µm) fraction of sediment samples collected from the outer Thames Estuary

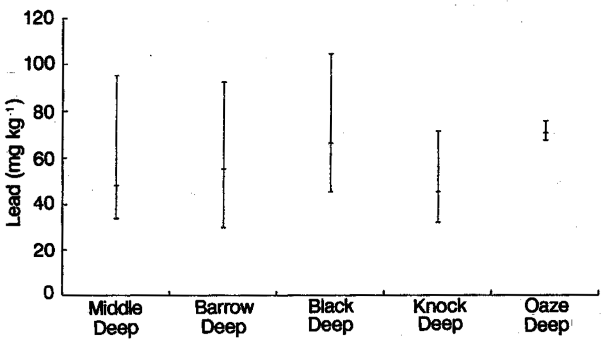

Figure 32. Concentrations of lead (mean + range) present in sediments from the Middle, Barrow, Black, Knock and Oaze Deeps

Surveys were carried out in 1989, 1991 and 1992 to monitor changes in sediment quality over time. They were based on a random stratified design centred on the area of likely impact. Samples were sieved at 63 µm and the fine fraction digested using *aqua regia*, and analysed by flame atomic absorption spectrophotometry.

Table 19 shows the mean values and standard deviations for the concentrations of metals in the <63 µm fraction in the three years. While some elements appear to show consistent changes, for example increasing chromium and decreasing mercury, the changes for other metals are less consistent and it is not possible to draw firm conclusions without many more years' data such as were presented for Liverpool Bay and the Thames in previous reports in this series (MAFF, 1991 and MAFF 1992).

Table 19. Concentrations of metals (mg kg^{-1}) in the <63 µm fraction of sediment from the area of the Nab Tower sewage-sludge disposal site 1989, 1991 and 1992

Year		N	Cr	Cu	Hg	Ni	Pb	Zn
1989	mean	23	42.3	27.2	0.41	38.9	62.6	106.7
	SD		10.2	22.6	0.39	31.0	98.8	46.8
1991	mean	21	55.9	60.9	0.29	27.1	72.6	163.0
	SD		11.4	48.3	0.28	8.4	55.0	117.0
1992	mean	23	63.8	33.9	0.15	27.5	55.0	98.3
	SD		12.5	11.0	0.07	7.0	45.5	17.5

16.3 Plymouth

Each year approximately 60 000 wet tonnes of sewage sludge is deposited at the Plymouth disposal site. The impact of this disposal operation is assessed at regular intervals as part of the licensing authorities obligations under FEPA.

Surveys were carried out in 1986, 1988 and 1992 to monitor changes in sediment quality over time. As with the studies at Nab Tower they were based on a random stratified design centred on the area of likely impact. Samples were sieved at 63 µm and the fine fraction digested using *aqua regia*, and analysed by flame atomic absorption spectrophotometry.

Table 20 shows the mean values and standard deviations for the concentrations of metal in the <63 µm fraction in the three years. While some elements appear to show consistent change, for others the pattern is one of little change. However, it must be noted that, as with the Nab Tower data (see Section 16.2), it is not possible to draw firm conclusions without many more years' data.

Table 20. Concentrations of metals (mg kg^{-1}) in the <63 µm fraction of sediment from the area of the Plymouth sewage-sludge disposal site in 1986, 1988 and 1992

Year		N	Cr	Cu	Hg	Ni	Pb	Zn
1986	mean	15	38	32	0.23	22	79	100
	SD		3.9	10.7	0.07	2.2	45.3	9.5
1988	mean	15	41	33	0.26	27	52	111
	SD		2.8	7.7	0.05	1.1	7.1	7.8
1992	mean	26	54.7	39.0	0.24	25.2	53.6	132.6
	SD		5.3	9.2	0.04	1.6	9.6	23.2

17. SURVEY OF LITTER AT VARIOUS SEWAGE-SLUDGE DISPOSAL SITES

Each year approximately 7 million wet tonnes of sewage sludge are disposed of at sea to various sites around the English and Welsh coasts. The sludge is generated from the treatment of contaminated water arriving at sewage treatment works from three main sources; domestic and industrial waste discharges and land run off. The proportional make up of the sludge with respect to these sources varies widely, both between treatment works and temporally at each treatment site. A large variety of litter is associated with these sources of sewage and is liable to find its way through to the sewage sludge, especially when that sludge is of the undigested type arising from primary treatment. Due to the unpleasant qualities of some litter artefacts and the non-biodegradable nature of many of them, efforts are made to remove this material before the sewage sludge is disposed to sea. A proportion of the larger litter items are removed when the sludge is screened at the sewage treatment works. The efficiency of screening at the various treatment works varies considerably, depending mainly on the size of screen used. However, some litter items are small enough or flexible enough to pass through even the smallest meshed screens and it is likely, therefore, that at least some sewage-derived litter enters the marine environment at all of the designated sewage-sludge disposal sites around the English and Welsh coasts. The amount of sewage-derived litter present at the seabed at any disposal site depends on several factors, including the original make up of the sludge, the screening efficiency of the treatment plant, the hydrography of the site and the amount of rainfall within the catchment area of the treatment plant.

Not all litter present at sewage-sludge disposal sites is derived from sewage sludge. The disposal of dredged material and the past disposal of household rubbish at some sewage-sludge disposal sites are additional sources of anthropogenic artefacts. Rubbish discarded from ships of all types is also commonly found at the seabed and may therefore be present at these sites. It is also possible that land-based sewage discharges either via rivers or directly from pipelines are responsible for a proportion of the sewage-derived litter found at the seabed. With such a variety of possible sources of litter, it is not always easy to associate some artefacts with a specific method of disposal. Each site must be looked at individually with respect to the quantity and origin of the litter found there.

During 1992 and early 1993, a survey was carried out at various sewage-sludge disposal sites around the English and Welsh coasts to assess the quantity of sewage-derived litter present (Figure 33). At sites where the nature of the sediment allowed, a 2 metre beam trawl was used to collect samples from at least three stations at, or close to, the disposal site. At sites where the seabed was too rough to allow the use of a beam trawl, a single tow with a Newhaven scallop dredge was carried out within the area of the disposal site. Tows were between 5 and 10 minutes long, and the distance covered by the sampling gear on each tow was recorded. The samples were sorted on deck, the litter separated and each item of litter recorded. It was possible to accurately quantify some litter items such as cigarette butts or tampon strings. Artefacts such as fragments of sanitary towel were assessed semi-quantitatively, and other litter items, for example human hair, were only recorded on a presence or absence basis.

For the purposes of this report, at disposal sites where more than one station was worked successfully, the most contaminated sample with respect to the amount of sewage-derived litter contained, has been selected for consideration.

Table 21 lists the nature and quantity of litter items found in the samples for each site. All the sites sampled had some degree of litter contamination, and all but one of the samples (Humber) contained litter which was clearly associated with sewage sludge. The total amount of litter varied from site to site, with the Tyne, Thames and Liverpool Bay sites the most contaminated.

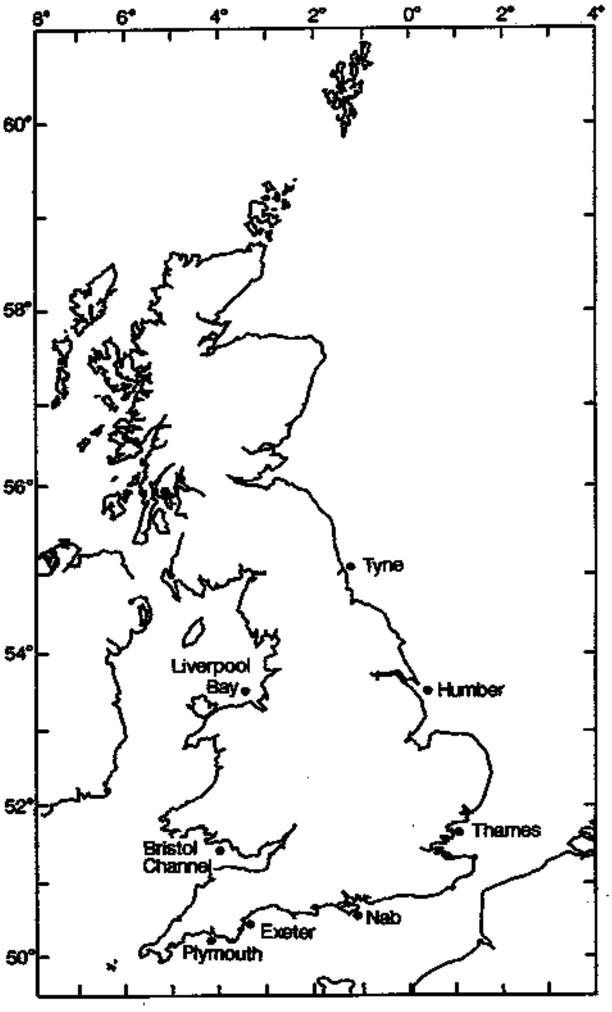

Figure 33. ***UK sewage sludge disposal sites surveyed for litter during 1992 and 1993***

Relatively large quantities of glass and china dating from earlier this century were found at the Liverpool Bay site. The Bristol Channel site produced only a small quantity of litter items, but a large proportion (70%) of those items were fragments of sanitary ware and therefore very obviously sewage-derived. Using the numbers of fragments of sanitary towel present in the samples (see Table 21), the sites were ranked according to their degree of contamination with sewage-derived litter. The Tyne disposal site which receives only undigested sludge was by far the most contaminated in this respect and this observation is confirmed by the presence of large numbers of other sewage-derived items, such as cigarette butts and vegetable matter. The Thames site was the next most contaminated site yielding similar types of sewage-derived items to those found at the Tyne site, but in considerably lower quantities. The remaining sites produced low numbers of fragments of sanitary towel and only limited quantities of other sewage-derived artefacts. A group average linkage cluster analysis, using the Dice coefficient (Field *et al*., 1982) on presence/absence total litter data, is shown as a dendrogram (Figure 34). The dendrogram shows that the Liverpool Bay, Nab, Thames and Tyne sites are linked together, but are

Table 21. ***Total quantity of litter found in trawl/dredge samples at sewage-sludge disposal sites***

Litter description	Disposal site							
	Tyne	Humber	Thames	Nab	Exeter	Plymouth	Bristol Channel	Liverpool Bay
Wood	P		P	P				P
Leaf Litter	P		P	P	P	P		P
Straw			P					
Grass	P							
Coal						P		
Clinker				P				P
China								5
Brick				1				
Glass				P				5
Shoe				1		1		1
Industrial Rubber Glove				1				
Ring Pull				1				
Monofilament Nylon(Fishing Line)		P	P					
Nylon Twine	1		P		P	P	1	2
Natural Fibre String					P			
Bone				2				1
Skull (Small Mammal)			1					
Cereal Seed			1					
Onion Peel			1					
Tomato Stalk	1							
Potato Peel	3		3					
Pea Skin	1							
Pea Pod	1							
Broad Bean	1							
Piece of Carrot	1							
Cabbage Leaf	1							
Fruit Pip/Stone	2		9					
Nut Shell			2					
Orange Peel	1							
Tea Bag	1							
Chewing Gum	1							
Fragment of Sanitary Towel	**36**		**10**	**6**	**1**	**1**	**1**	**2**
Fragment of Tampon	6						4	2
Hormone Patch	1					1		
Human Hair	P		P					P
Condom Packet			1					
Cigarette Filter	32		12					
Matchstick	3							
Plastic Bags								1
Plastic Unwrap Strip	3							
Amorphous Plastic Fragments	3		4			1	1	1
Plastic Solid	1			4				
Plastic Tie-Wrap								1
Plaster	3							
Plaster Tear Strip	1							
Make-Up Sponge								1
Button								1
Adhesive Tape								
Elastic Band	2							1
Silver Foil	1		1					
Milk Bottle Top				1				
Identified Sweet/Drinks Wrapper	1			1				
Piece of 'J' Cloth								1
Rag			1					1
Piece of Net Curtain	1							
Strand of Wool	1		1					
Tissue Paper						1		
Rubber 'O' Ring								1
Piece of Rubber				1				
Puncture Repair Kit Patch	1							

Note: P = Present (not quantified)

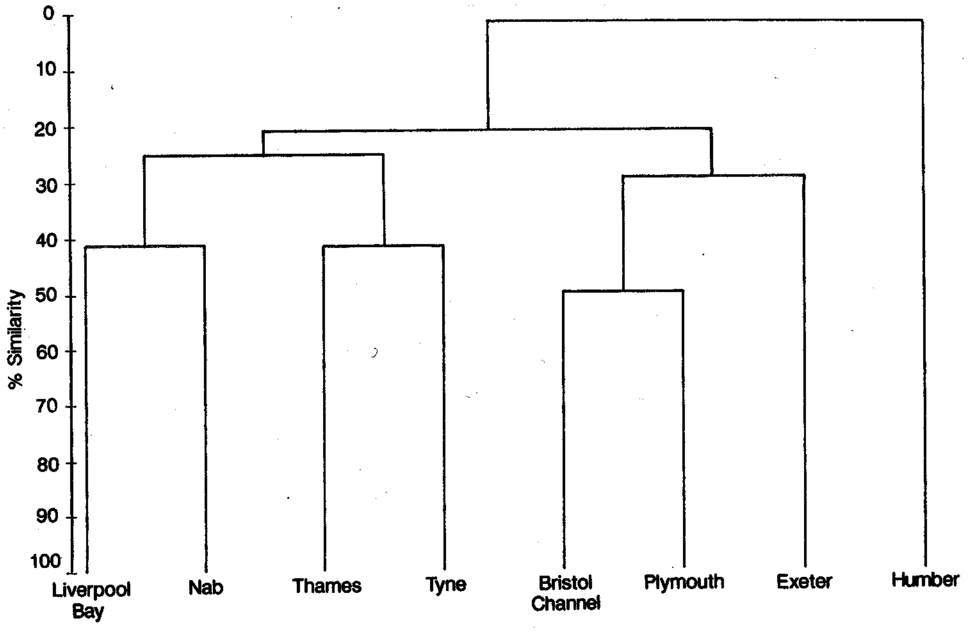

Figure 34. Dendrogram showing the linkage between sites with respect to their total litter content

separated from the Bristol Channel, Plymouth and Exeter sites. All these sites are clearly separated from the least contaminated Humber site. This indicates that the sites within each of the three groups are similar to each other with respect to the type (not quantity) of litter found there and that each group of sites is different from the others with respect to the type of litter associated with each of the groups.

As the Tyne site does not receive a large quantity of sewage sludge compared to the Thames and Liverpool Bay sites (Table 22), it may be deduced that the comparatively large quantities of sewage derived litter found at the Tyne site are not a result of larger amounts of sludge disposal. Hydrographically the Tyne site is dispersive and litter surveys carried out at the site over several years suggest that quantities of litter, although large compared to other sites, are not increasing. Surveys carried out at the site in previous years had noted that large quantities of sewage-derived litter were present in the sludge slick immediately after disposal, indicating that screening procedures at the treatment sites were not effectively removing these items prior to disposal. Subsequent improvements in the screening procedures at the treatment sites increased the amount of litter removed from the sludge, and observations of the sewage slick made during this survey showed that very few litter items were now present. The apparently small amount of sewage-derived litter currently entering the marine environment at the Tyne disposal site suggests that the large quantity of litter found on the seabed during this survey was due mainly to disposal operations carried out before improved screening procedures were introduced.

Table 22. Quantity of sewage sludge deposited during 1991

Disposal Site	Quantity Deposited (t)
Thames	3983592
Liverpool Bay	1879500
Tyne	601035
Bristol Channel	278430
Nab	262194
Humber	114738
Plymouth	57322
Exeter	38456

The relatively small amounts of litter found at the Humber, Exeter, Plymouth, Bristol Channel and Nab disposal sites may be a reflection of the smaller quantities of sewage sludge entering these sites compared to the inputs at the Thames, Liverpool Bay and Tyne sites.

18. STUDIES AT SOLID WASTE AND DREDGED MATERIAL DISPOSAL SITES

18.1 Benthic studies at solid waste disposal sites off the north east coast of England

18.1.1 Introduction

There are a number of sites off the north east coast of England designated for the disposal of solid wastes from ships and earlier extensive studies in this area by

MAFF were reported by Eagle *et al.* (1979). The material consists of minestone (shale), tailings from collieries and dredged material from estuaries. Fly ash from coal-burning power stations was also deposited at certain sites up until 1992, when the last station was closed. Three disposal sites, located off the Tyne and Wear estuaries at about 40 m depth, were investigated by MAFF in 1992 (Figure 35: note that the disposal of fly ash ceased earlier than 1992 in this region). The prevailing natural substrate type in the area is a mud/ sand admixture, but in addition to the disposal sites themselves, the wider inshore area is modified by inputs of solid materials from a long history of waste disposal, much of which pre-dates statutory controls. For example, Buchanan (1963, 1964) described an inshore 'polluted' zone off the Tyne which was characterised by the presence of fly ash, along with 'wood, tin cans, oil drums and general rubbish'. Similar observations were made by Rees *et al.* (1985). As this zone extends as far out as the present study, some influence on the outcome of sampling at stations away from the designated disposal sites is to be expected. The possible influence of sewage sludge disposal nearby (Figure 35) must also be considered.

All three solid waste disposal sites receive combinations of different types of inputs; these vary with site and with time (Figure 36). Maintenance dredgings from the Tyne estuary, together with minestone (i.e. coarse material rejected before the coal-washing process) accounts for most of the material deposited at the 'North Tyne' site in recent years. The 'Souter Point (inner)' disposal site has consistently received large quantities of minestone, together with a significant amount of tailings since 1988. (Tailings consist of fine material generated from the crushing of rock during the coal-washing process). The 'Souter Point (outer)' site is used largely for maintenance dredgings from the Tyne. The total quantity of material deposited at all three sites in 1992 amounted to some 1.3 million wet tonnes.

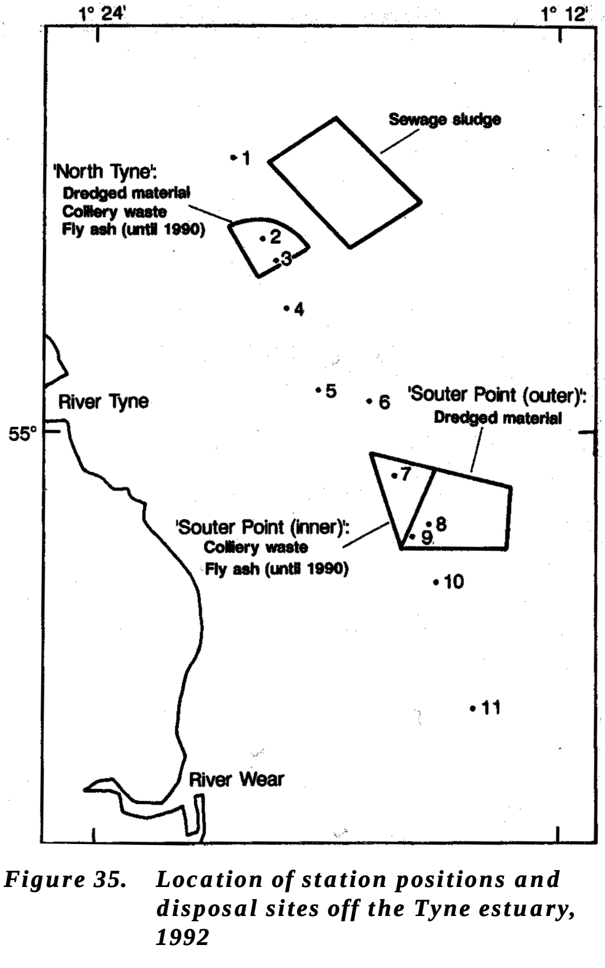

Figure 35. Location of station positions and disposal sites off the Tyne estuary, 1992

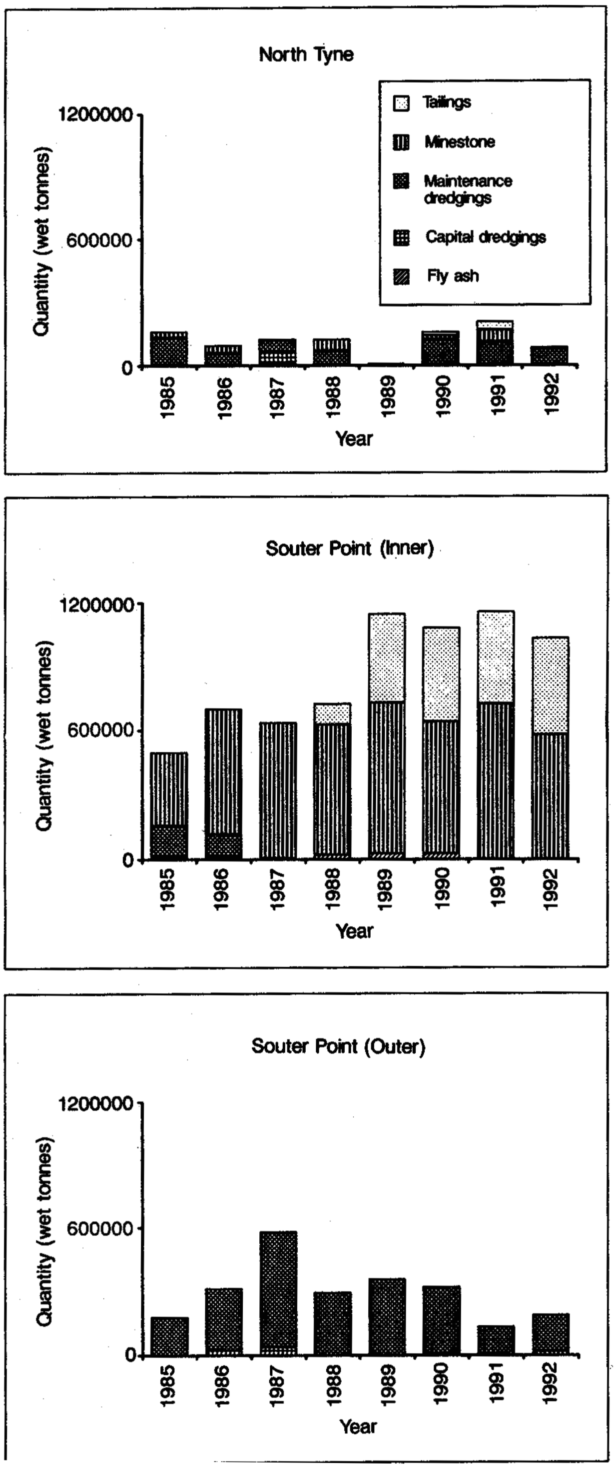

Figure 36. Quantities and nature of solid wastes disposed of at three locations off the north east coast of England

18.1.2 Methods

1. Field sampling
Eleven stations were sampled by Day grab in May 1992 from the MAFF research vessel *RV CORYSTES*. The sampling design took the form of a transect running approximately parallel with the coastline at about 40 m depth (Figure 35); this is approximately in line with the direction of tidal current flow. Three samples were taken at each station, with the exception of Station 2, where only one could be obtained due to the coarse substrate. After measurement of the depth of sediment retained in the grab, and the removal of a sub-sample for particle size analysis, the first two samples were washed over a 1 mm mesh sieve, and the contents preserved in 5% buffered formaldehyde with Rose Bengal, for later macrofauna analysis. A sample of the surface layer of the third grab was removed and frozen, for later analysis of concentrations of trace metals.

2. Laboratory and data analysis
Sediment samples for particle size analyses were first wet-sieved to separate the <63 µm fraction. Material coarser than this was then dry-sieved at 0.5 phi intervals. The <63 µm fraction was analysed using a sedigraph. Medians and sorting coefficents were calculated from the resulting cumulative curves; only one sample was analysed from each station.

Sediments for analyses of trace metal content were first wet-sieved to separate the <63 µm fraction. This fraction was then analysed for cadmium, chromium, copper, mercury, nickel, lead and zinc, following procedures outlined in Harper *et al*. (1989) and Jones *et al*. (in prep.).

Macrofauna samples were analysed to species level, as far as possible, using a standard range of taxonomic keys (see e.g. Rees *et al*., 1990). Taxonomic nomenclature followed that of Howson (1987). In addition, the biomass (as wet blotted weight) was determined for individual taxa. These were then converted to ash-free dry weights (AFDW), using conversion factors from the literature, notably Rumohr *et al.* (1987).

Analyses of the macrofaunal and environmental data employed a combination of SAS (Statistical Analysis System) and PRIMER (‘Plymouth Routines in Multivariate Ecological Research’) software packages. Cluster analyses employed the Bray-Curtis dissimilarity measure (Bray and Curtis, 1957) and group-average sorting (Lance and Williams, 1967). ‘Multidimensional Scaling’ (see e.g. Field *et al*., 1982) also employed the Bray-Curtis measure.

Two samples from each station were pooled for multivariate analyses, with the exception of Station 2 at the centre of the ‘North Tyne’ disposal site, where counts for the single sample obtained were doubled. As a result, numbers of taxa would be expected to be under-represented at this station. However, the fauna and sediments here (and at Station 7 to the south) were so markedly different as a result of waste disposal, that such an approach was felt unlikely to seriously affect the outcome of analyses.

18.1.3 Results

Water depths, sample volumes and sediments
Water depths at the time of sampling did not vary greatly between stations (Table 23). However, a depth of 39 m at Station 7a, which is notably shallower than at adjacent stations, suggests a degree of shoaling due to solids deposition at the apex of the ‘Souter Point (inner)’ disposal site (see Figure 35).

Table 23. Station positions and summary measures of sediment characteristics off the Tyne estuary, 1992

Station	Latitude	Longitude	Water depth (m)	Sample volume (l)	Median (mm)	Sorting (sd)	% silt/clay	Tomato pips	Field description
1a	55 04.07N	01 20.28W	45	6.8	1.01	2	17.1	0	Muddy sand over coarse sand
1b	55 04.09N	01 20.33W	45	7.7				0	Coarse shelly muddy sand
2	55 02.88N	01 19.53W	43	1.8	4.25	1.25	2.1	0	Minestone
3a	55 02.56N	01 19.18W	44	2	3.29	2.03	8.5	2	Minestone and muddy sand
3b	55 02.58N	01 19.22W	44	2.4					Minestone
4a	55 01.85N	01 18.91W	43	1.8	3.59	1.6	17.5	4	Sand and coal
4b	55 01.88N	01 18.96W	43	5.5				0	Sandy mud and colliery waste
5a	55 00.65N	01 18.07W	41	2.8	0.39	2.05	8.7	1	Muddy sand with colliery waste
5b	55 00.68N	01 18.15W	41	3.1				1	Colliery waste in sandy mud
6a	55 00.49N	01 16.74W	46	9.1	0.12	2.25	33.7	2	Sandy mud
6b	55 00.53N	01 16.79W	46	8.6				23	Muddy sand
7a	54 59.38N	01 16.09W	39	4.7	1.8	1.3	8.1	0	Muddy colliery waste
7b	54 59.41N	01 16.14W	39	3.1				0	Colliery waste and sandy mud
8a	54 58.62N	01 15.16W	48	10.4	0.08	2.28	50.1	58	Sandy mud
8b	54 58.69N	01 15.21W	48	7.7				13	Sandy mud
9a	54 58.48N	01 15.63W	46	7.7	0.11	1.85	33.5	2	Sandy mud
9b	54 58.54N	01 15.72W	46	6.3				1	Sandy mud
10a	54 57.81N	01 15.00W	46	7.2	0.09	1.35	37.2	27	Sandy mud
10b	54 57.86N	01 15.01W	46	7.2				15	Sandy mud
11a	54 55.97N	01 14.03W	46	10	0.11	0.9	25.8	0	Sandy mud
11b	54 56.03N	01 14.11W	45	7.7				1	Sandy mud

There was significant variation in the volumes of sediment retained in grabs (Table 23). This is a direct consequence of difficulties in obtaining adequate samples on the very coarse substrates present at, or near to, the ‘North Tyne’ and ‘Souter Point (inner)’ disposal sites, which have received large amounts of mine waste. This variability may affect the numbers of animals retained, but gross differences in the nature of the substrates themselves, along with the recent history of disposal, are expected to have a much greater influence on the presence or absence of benthic species.

Large differences in sample volume are matched by those in sediment statistics, and in the field descriptions of the grab contents (Table 23). The trend towards finer sediments to the south is interrupted by the presence of varying amounts of colliery waste in and around the disposal sites. The nature of this variation is more clearly seen in Figure 37, where particle size distributions are plotted for one sample from each station. The highest percentage of silt/clay was found at Station 8, within the ‘Souter Point (outer)’ disposal site, although the overall sediment structure at this and all stations to the south are very similar. Sediment data are further discussed below, in relation to macrofauna distributions.

Small numbers of tomato pips were noted in samples at and to the south of the ‘North Tyne’ disposal site (Table 23). Considerably larger numbers were found at Station 8, within the ‘Souter Point (outer)’ disposal site, and at Stations 6 and 10 immediately to the north and south, respectively. Their presence can be associated with the disposal of relatively large quantities of maintenance dredgings from the Tyne estuary, where they arise from pipeline inputs of sewage. The occurrence of pips on either side of this disposal site provides an indication of the dispersal of finer material in response to water movements. Accumulations of large numbers of tomato pips have also been found in the immediate vicinity of the sewage-sludge disposal site to the north (Rowlatt *et al*., 1989 and Rees *et al*., 1992(a)); a contour plot of counts in each grab sample revealed that these could be distinguished from elevated numbers at dredgings disposal sites.

Concentrations of trace metals

Concentrations of a range of trace metals (Table 24) provide no clear evidence of gradients in the data which might be attributable to disposal activity. This is to be expected for localities in receipt of colliery waste, which is a natural by-product of mining and not subject to chemical contamination during treatment. However, a relatively high mercury content at Station 8 might be explained by the disposal of maintenance dredgings from the Tyne estuary. These data are further examined below, in relation to macrofauna distributions.

Table 24. Trace metal content in the <63 µm fraction of sediments off the Tyne estuary, 1992 (mg kg^{-1} dry weight)

Station	Cr	Cu	Hg	Ni	Pb	Zn
1	76	52	0.3	51	103	158
2	63	66	0.07	49	22	64
3	63	63	0.24	50	95	133
4	67	60	0.27	45	77	150
5	60	126	0.22	93	105	167
6	68	58	0.28	46	91	138
7	49	86	0.15	41	97	74
8	68	63	0.7	43	121	53
9	65	59	0.24	47	75	129
10	61	50	0.23	44	71	123
11	56	40	0.29	36	87	126

Macrofauna

Summary statistics for the macrofauna are plotted in Figure 38(a-d). Reduced abundances occur at and to the south of the ‘North Tyne’ disposal site, and at the apex of the ‘Souter Point (inner)’ site (Figure 38(a)). By contrast, there is no indication of depressed abundances at the outer site. A similar pattern is evident for numbers of taxa (Figure 38(b)), with lowest numbers occurring at the centres of disposal activity at the inner sites. Biomass values are inflated at Stations 1 and 4 (Figure 38(d)) on account of one specimen of the hagfish *Myxine glutinosa* at each. These large and mobile organisms are unlikely to be effectively sampled by grab, and the effect of their exclusion (Figure 38(d)) provides a result which is similar in kind to those for densities and numbers of taxa.

Overall, there is a suggestion of marginal enrichment in the fauna at and to either side of the ‘Souter Point (outer)’ disposal site, possibly as a result of the dispersion of organically-rich fine particulates arising from the disposal of maintenance dredgings from the Tyne estuary. Values of all three variables are relatively low at the southernmost station (11), which is furthest from the disposal sites, although it may still be in receipt of dispersing fine material in response to a net southward residual drift (see Eagle *et al.*, 1979).

The outcome of ‘Multidimensional Scaling’ (MDS) of the log (x+1)-transformed macrofauna data is expressed in a 2-dimensional plot in Figure 39(a). Four station groups are recognisable (which were also evident from cluster analyses) and these are circled on the plot. Stations 2 and 7, where the fauna is most depleted as a result of colliery waste disposal, are clearly separated on the left of the plot. From left to right, the next group comprises Stations 3 and 4 which are located at and just to the south of the ‘North Tyne’ disposal site, and both show evidence of the effects of disposal activity. Stations 1 and 5 (the third group) are located at either end of the inshore transect. These show little evidence of effects arising from <u>recent</u> disposal activity, but the

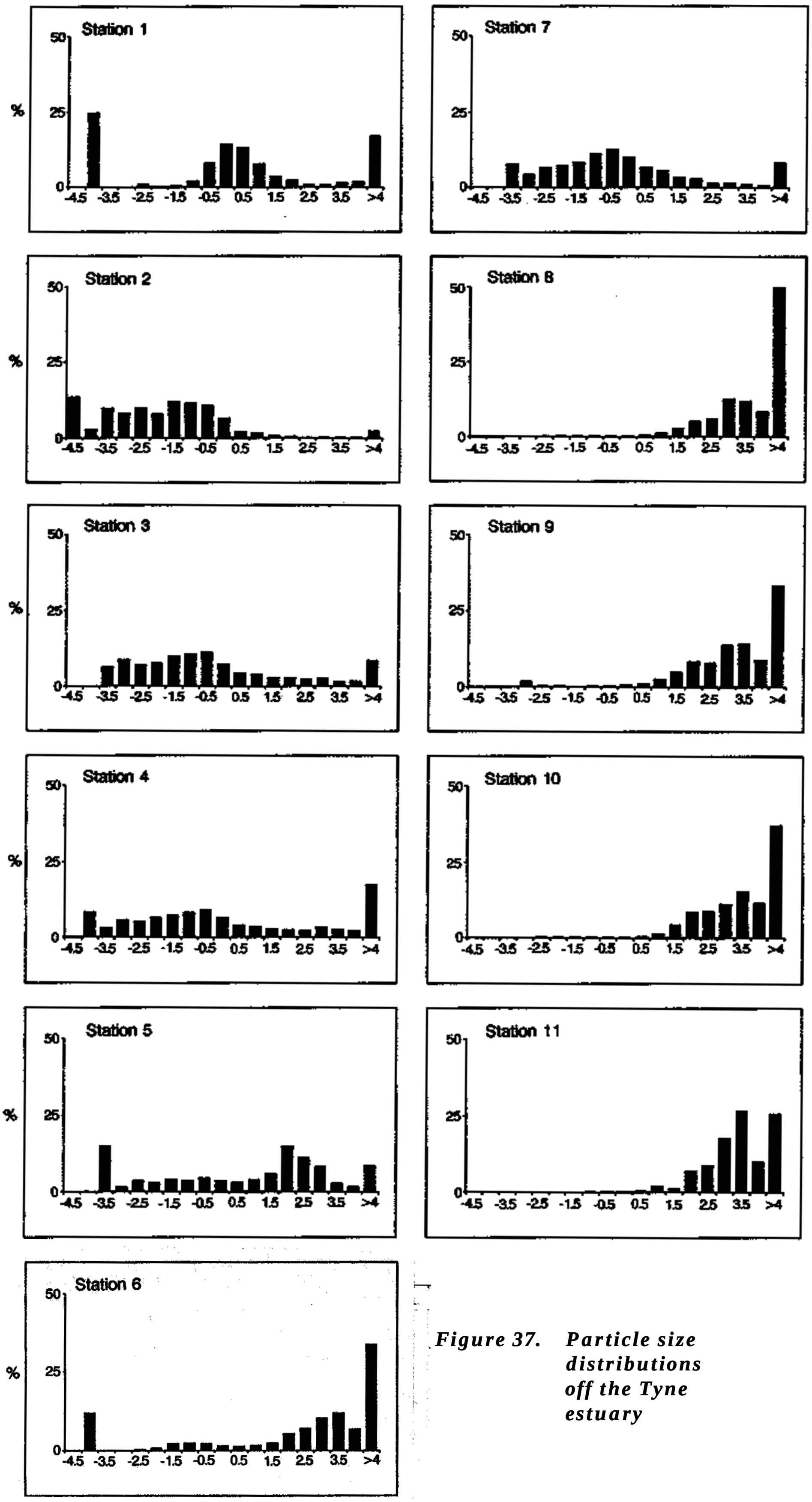

Figure 37. ***Particle size distributions off the Tyne estuary***

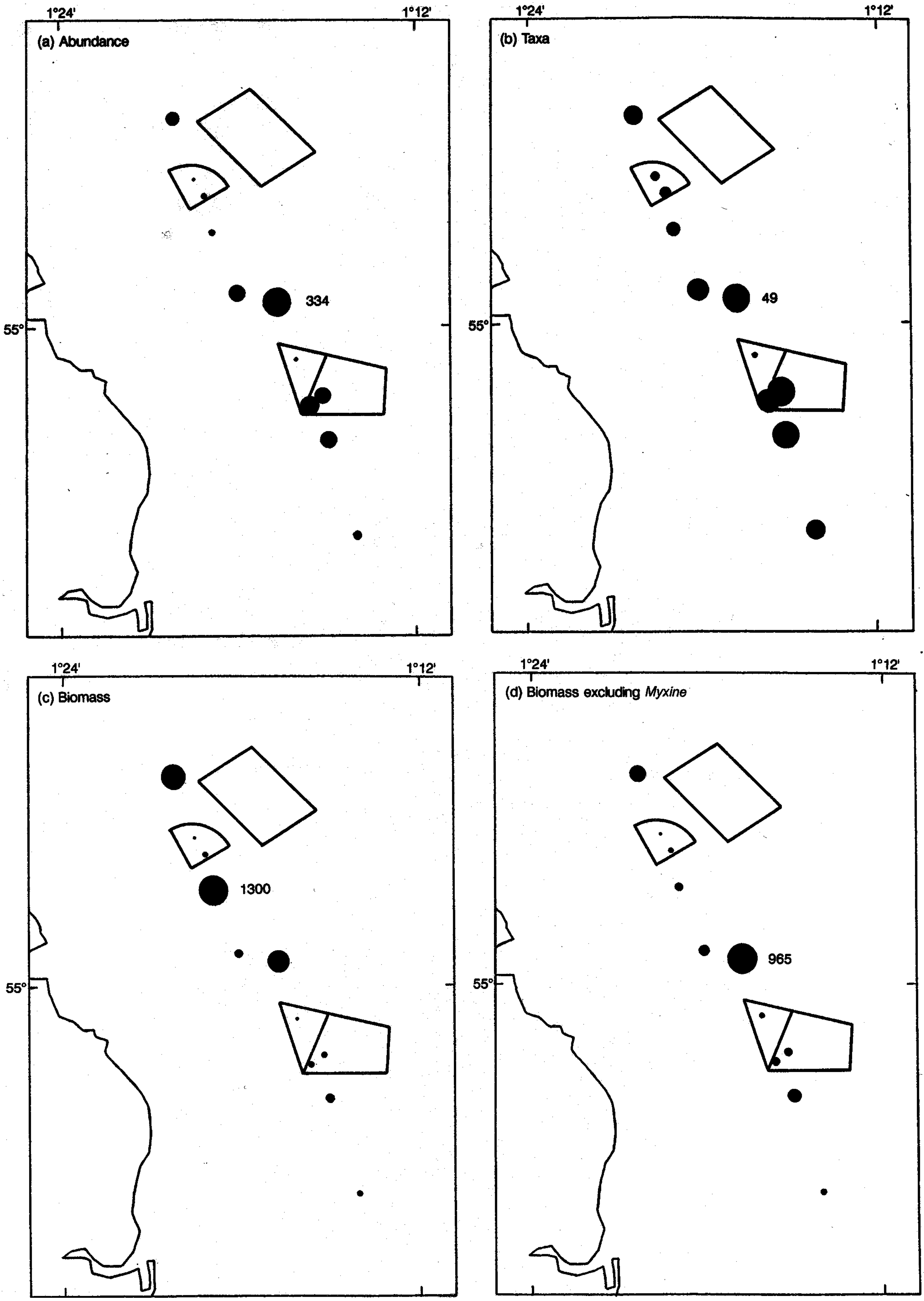

Figure 38. Changes in: (a) abundance; (b) numbers of taxa; and (c-d) ash-free dry weight biomass off the Tyne estuary, 1992. Circles are scaled relative to the highest values encountered in pooled samples from each station, and these are specified on the plots

sediment structure appears to reflect past inputs (see Figure 37). The tight clustering of stations at and to either side of the 'Souter Point (outer)' disposal site suggests that the benthos are not substantially affected by recent inputs of maintenance dredgings to this site.

A similar result emerged from the analysis of AFDW biomass (Figure 39(b); in this case, the data were fourth root-transformed to further reduce the effect on the outcome of analysis, of the occurrence of one or two overwhelming biomass dominants would otherwise have had), although Stations 1 and 5 are not so clearly distinguishable from the southern group of stations.

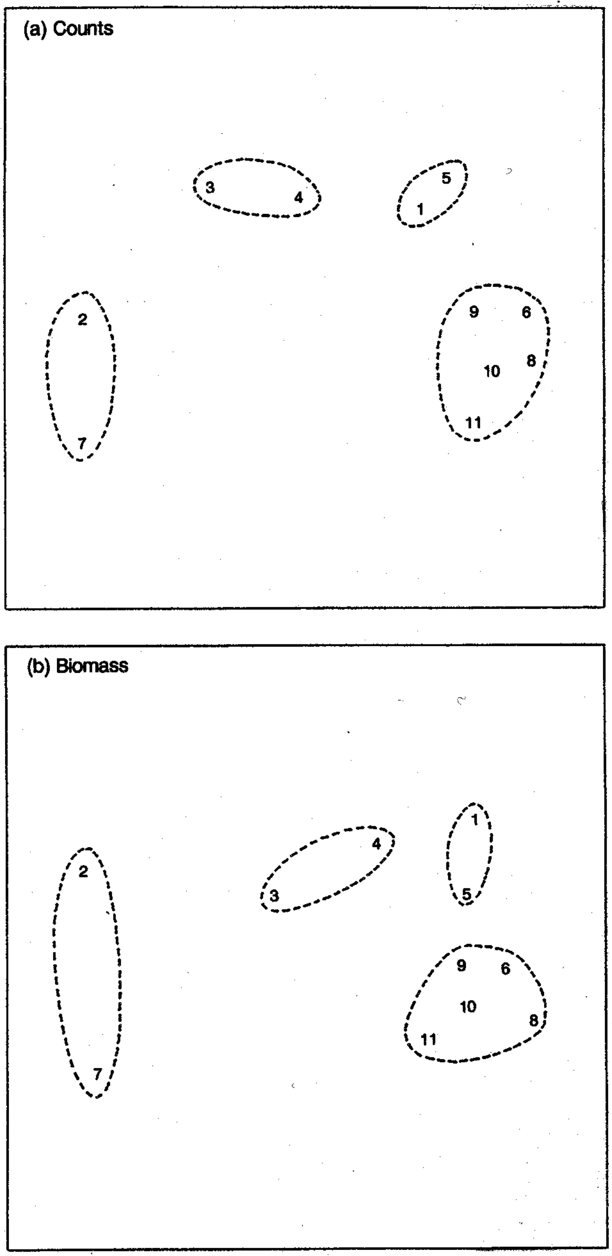

Figure 39. Two-dimensional plots of the outcome of: (a) Multidimensional Scaling of macrofaunal abundances and (b) ash-free dry weight biomass. Samples from each station were pooled prior to analysis

The contributions of the dominant taxa to these station groups are shown in Table 25. The polychaete worm *Lumbrinereis* spp is numerically common across all station groups. Lower counts of all taxa are evident at Stations 2 and 7, which are the most severely affected by solid waste disposal, and the free-swimming cumacean *Diastylis bradyi* is the most common organism here (Table 25(a)). Other than relatively high counts of *Lumbrinereis* spp, abundances for the group represented by Stations 3 and 4, which are located at and just to the south of the 'North Tyne' disposal site, are also depressed. Neither station group can be characterised by the occurrence of 'pollution indicator' species which are not present elsewhere across the transect. However, biomass rankings for Stations 2 and 7 (Table 25(b)) include juveniles of the swimming crab *Liocarcinus pusillus* and the spider crab *Hyas,* which might be expected to occur on coarser substrates. Other than for this station group, bivalve molluscs, notably *Dosinia lupinus* and *Chamelea gallina,* are more prominent in these rankings.

The relationship between the ranked distribution of numbers and biomass across species in a sample has been proposed by Warwick (1986) as a means to identify the effects of disturbance on the benthos. An undisturbed (or 'unpolluted') fauna is characterised by low % numerical dominance. However, the biomass dominants are typically large (and longer-lived) species, which results in the curve for ranked biomasses lying above that for ranked abundances. In contrast, a severely disturbed (or 'polluted') fauna is characterised by high numbers of short-lived, opportunistic species of small individual body size. In the absence of heavier, longer-lived species which are not favoured in such circumstances, the positions of the curves are reversed. An intermediate level of disturbance is indicated by intersecting curves.

The outcome of such a comparison for the above station groups is shown in Figure 40(a-d). (The hagfish *Myxine* was excluded from the data at Stations 1 and 4). There is a progressive increase in % dominance by the first-ranked species, from the 'less disturbed' group (Stations 6, 8-11) to the 'more disturbed' groups (Stations 3, 4 and 1, 5). Intersecting curves for the latter two station groups also suggest an intermediate level of disturbance, although the % biomass dominance for Station group 3 and 4 has increased, against expectation. The result for Stations 2 and 7 clearly does not conform with the above model. This may be explained by the the fact that **regular** inputs of inert solid wastes do not find favour with any species, and hence there is a proportional decline in all.

Table 25. The top ranked taxa for station groups identified by multivariate analyses: (a): counts m^{-2}; (b): ash-free dry weight biomass in mg m^{-2}

a. Counts

Station 2,7		Station 3,4		Station 1,5		Station 6,8-11	
Diastylis bradyi	100	*Lumbrinereis spp*	285	*Lumbrinereis spp*	538	*Thyasira flexuosa*	273
Lumbrinereis spp	75	*Nucula nitidosa*	35	*Ampharete spp*	80	*Prionospio spp*	218
Ophiura albida	35	*Prionospio spp*	33	*Tanaidacea spp*	68	*Lumbrinereis spp*	174
Pholoe minuta	30	*Ampharete spp*	25	*Cirratulid spp*	58	*Ophiura albida*	99
Nemertine spp	23	*Heteromastus filiformis*	20	*Amphiura filiformis*	58	*Rhodine gracilior*	70
Prionospio spp	20	*Amphiura filiformis*	20	*Prionospio spp*	53	*Trichobranchus glacialis*	63
Cirratulid spp	20	*Glycera spp*	18	*Ophiura albida*	53	*Nemertine spp*	58
Chaetozone setosa	20	*Anobothrus gracilis*	18	*Mysella bidentata*	50	*Diastylis bradyi*	55
Glycera spp	18	*Pholoe minuta*	15	*Pholoe minuta*	45	*Cirratulid spp*	53
Nucula nitidosa	10	*Myriochele oculata*	15	*Terebellides stroemi*	45	*Anobothrus gracilis*	52
Thyasira flexuosa	10			*Diastylis bradyi*	45		

b. Biomass

Station 2,7		Station 3,4		Station 1,5		Station 6,8-11	
Nephtys spp	491	*Myxine glutinosa*	5727	*Myxine glutinosa*	3054	*Dosinia lupinus*	529
Lumbrinereis spp	118	*Lumbrinereis spp*	423	*Amphiura filiformis*	674	*Nephtys spp*	308
Glycera spp	116	*Amphiura filiformis*	154	*Dosinia lupinus*	658	*Chamelea gallina*	256
Nemertine spp	70	*Lunatia poliana*	130	*Lumbrinereis spp*	540	*Glycera spp*	232
Ophiura albida	32	*Terebellides stroemi*	103	*Ophiura albida*	386	*Ophiura albida*	184
Diastylis bradyi	30	*Chamelea gallina*	100	*Terebellides stroemi*	297	*Owenia fusiformis*	168
Cirratulid spp	26	*Glycera spp*	91	*Nemertine spp*	229	*Leptosynapta bergensis*	135
Liocarcinus pusillus	25	*Corbula gibba*	67	*Gari fervensis*	208	*Lumbrinereis spp*	112
Hyas sp	22	*Nucula nitidosa*	56	*Chamelea gallina*	186	*Anobothrus gracilis*	98
Chaetozone setosa	10	*Ophiura albida*	29	*Sabellidae*	184	*Amphiura chiajei*	84

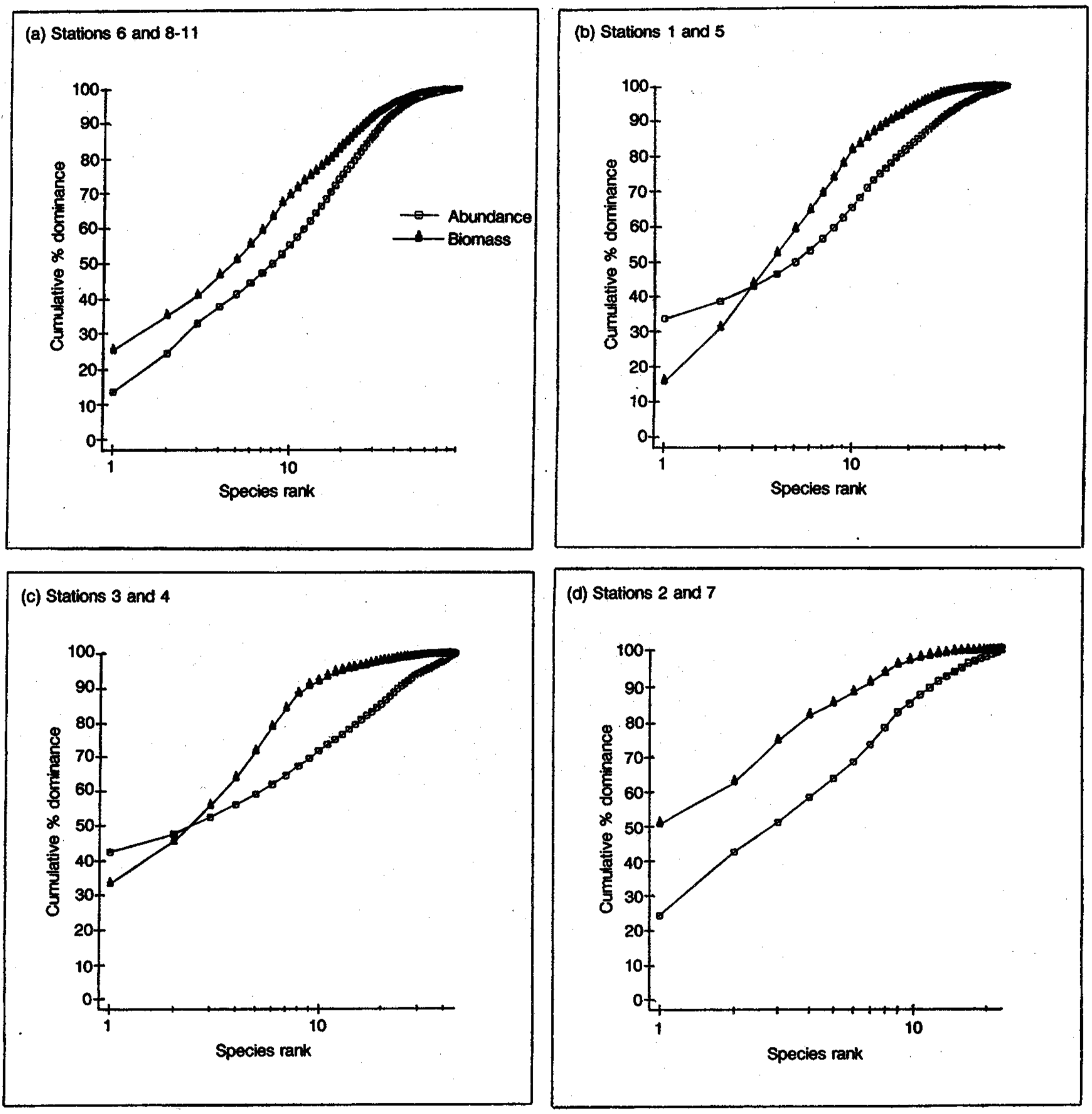

Figure 40. Cumulative curves for ranked abundances and biomass at station groups identified from multivariate analysis

Relation of macrofauna to environmental variables
Figure 41(a) shows the outcome of a correlation-based Principal Components Analysis of particle size data. The stations are arrayed in a similar fashion to the output from MDS analysis (Figure 39). This reinforces the conclusion that the physical nature of the substrate is the most important variable affecting benthic populations in the study area. Of the two most affected Stations (2 and 7), the median particle size at Station 7 is smaller, which may be accounted for by the input of significant quantities of finer as well as coarser mine waste.

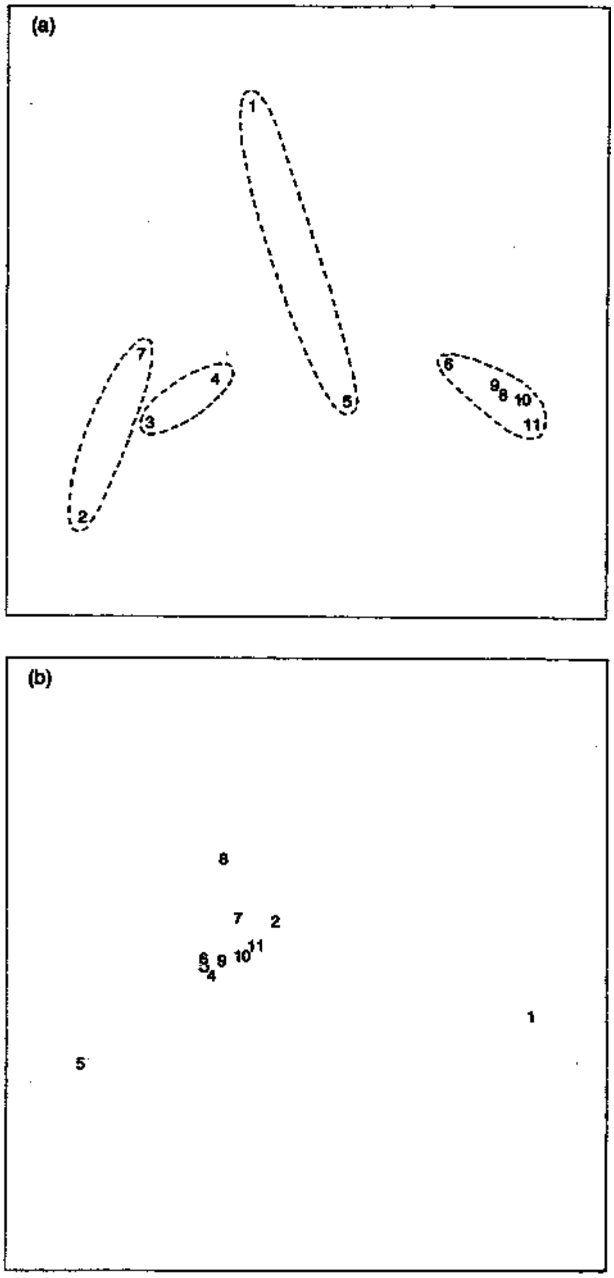

Figure 41. Two-dimensional plots of the outcome of: (a) correlation-based Principle Components Analysis for particle sizes and (b) concentrations of trace metals of sediments. Encircled stations in (a) correspond with groups identified from multivariate analysis of faunal data

By contrast, the distribution of stations arising from correlation-based Principle Components Analysis of concentrations of trace metals (Figure 41(b)) is notable for the absence of any clear gradient which might be attributable to solid waste disposal. The poor match with Figure 39 therefore indicates that trace metals are unlikely to have had any direct influence on the distribution of the benthic fauna.

18.1.4 Discussion

The benthic fauna sampled in the present study is similar in character to that described by Bamber (1984) just to the north of the area. He investigated the effects of the disposal of fly ash; as with mine waste, the relatively ‘inert’ nature of this material has no special attraction for benthic species, and the degree of faunal depletion was directly related to the proportion of ash in sediments. No new infaunal species were found at the disposal site, i.e. there was no preferential colonisation by any recognised ‘pollution indicators’. Similarly, Eagle *et al.* (1979) observed that the benthos at stations around a colliery waste disposal site off the Wear was composed of species present elsewhere in the area, but at much reduced densities.

A notable feature of samples near to the disposal sites which contained significant quantities of coarse material was the absence of a well-developed epifauna, which could not be accounted for by the smaller sample sizes at these locations. This might indicate an inhibitory effect of dispersing fine particulates on this component, or that the surfaces are in some other way unattractive for larval settlement. It might be expected that ‘diversification’ of the habitat arising from the mixing of coarse with fine material would increase the variety of benthic species. For example, Nicolaidou *et al.* (1993) report such an increase in response to the disposal of coarse metalliferous wastes. In the present study, only the occasional presence of small crabs and the brittle-star *Ophiothrix fragilis* accompanied the change in substrate type. In contrast, there was an indication of enhanced infaunal populations at stations surrounding the offshore dredgings disposal site. This included the occurrence of elevated numbers of the bivalve mollusc *Thyasira flexuosa*, which has been cited as an indicator of organic enrichment (e.g. Lopez-Jamar *et al.*, 1987).

It may be concluded that the study area is one of considerable complexity, arising from both natural and anthropogenic influences. In addition to physical changes in bottom sediments as a result of solid waste inputs, sewage-sludge disposal and efflux from the Tyne estuary also have the potential to influence the distribution of the benthic fauna throughout the area. Nevertheless, it was the former factor that provided the closest association with faunal gradients. The most severe physical effects were manifested by smothering in the immediate vicinity of disposal. As there was no evidence for southward transport of coarse material from the ‘Souter Point (inner)’ site (or for a depleted benthic fauna), this might suggest that changes to the sediments and fauna at central Stations 4 and 5 could at least partly be ascribed to historical inputs to the general inshore area, as noted above. However, the survey design was not intended to accurately define the spatial extent of these effects.

There was no clear evidence of successional changes in the fauna in the direction of the northern disposal site, but rather of a net depletion of organisms from within the species-pool. However, super-imposed upon any physical consequences of mine-waste disposal to the south, substantial inputs of organically-enriched maintenance dredgings from the Tyne estuary may account for a net enhancement of the benthic fauna nearby. Such an effect would be comparable with that observed in response to sewage-sludge disposal (Rees *et al.*, 1992(a)), and similarly localised in extent (and intensity) to about 2 km beyond the boundary of the licensed area.

18.2 Benthic studies at dredged material disposal Site ‘Z’, Liverpool Bay

18.2.1 Introduction

The disposal site for maintenance dredgings arising from the River Mersey has been the subject of a number of recent studies by MAFF (e.g. Rowlatt, 1988; Rowlatt *et al.*, 1986; Rees *et al.*, 1992(b)). It is located just to the north of the Mersey estuary in shallow water of about 10 m depth (Figure 42), where the bottom sediments are typically sandy or muddy in nature. The combination of tidal current action and wave effects at the seabed (especially arising from westerly winds) characterises the area as ‘dispersive’, and a net movement of a proportion of disposed sediments back towards the Mersey mouth can be anticipated.

The dredged material ranges from sand to mud depending on the predominant dredging location (channels or dock approaches) at different times of the year; in 1991 the site received some 2 million wet tonnes. The purpose of the present survey was to extend the spatial coverage achieved in previous years, so as to include the possible influence of the Mersey estuary on faunal and contaminant distributions. An additional aim was to collect specimens of the dominant benthic species for later analyses of contaminant concentrations in flesh. Contaminants associated with the particulate component of waste material (e.g. sewage sludge, dredgings) disposed of at sea may accumulate in the tissues of marine benthic organisms in the vicinity of the disposal area. This may occur in a variety of ways, for example, through filter- or deposit-feeding activities, predation of one benthic species on another, or by direct absorption through the body wall. There is a potential risk that such contaminants could be further bio-accumulated by fish (including commercial species), notably through predation on the benthos.

In order to test for such an occurrence, preliminary work was conducted on the common deposit-feeding polychaete *Pectinaria (Lagis) koreni.* This worm may reach very high abundances in inshore soft sediments, especially those with a moderately high organic content;

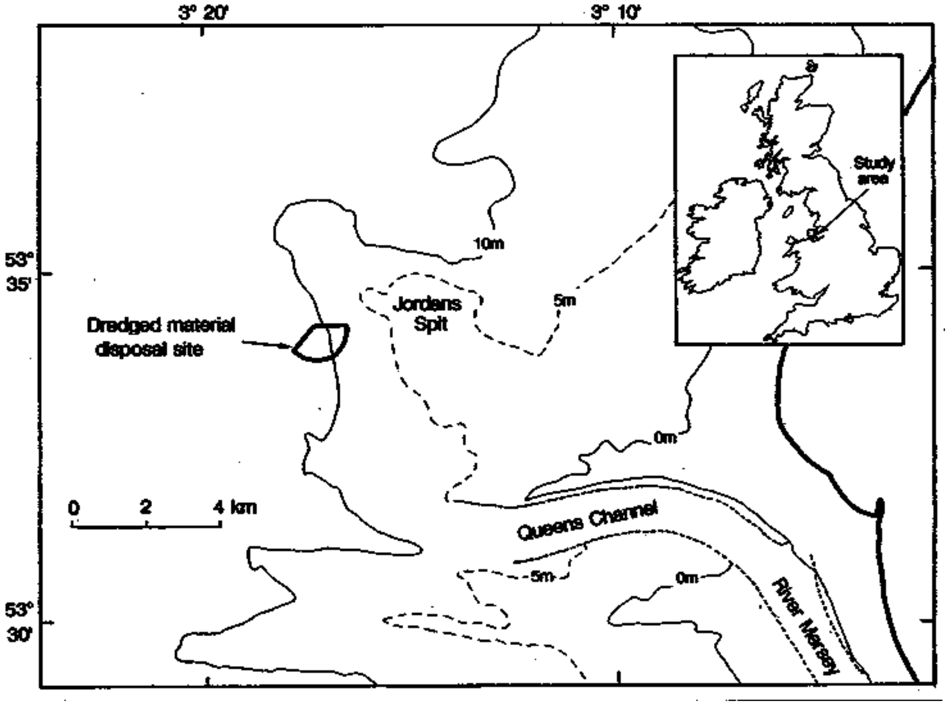

Figure 42. Location of dredged material disposal Site ‘Z’ in inner Liverpool Bay

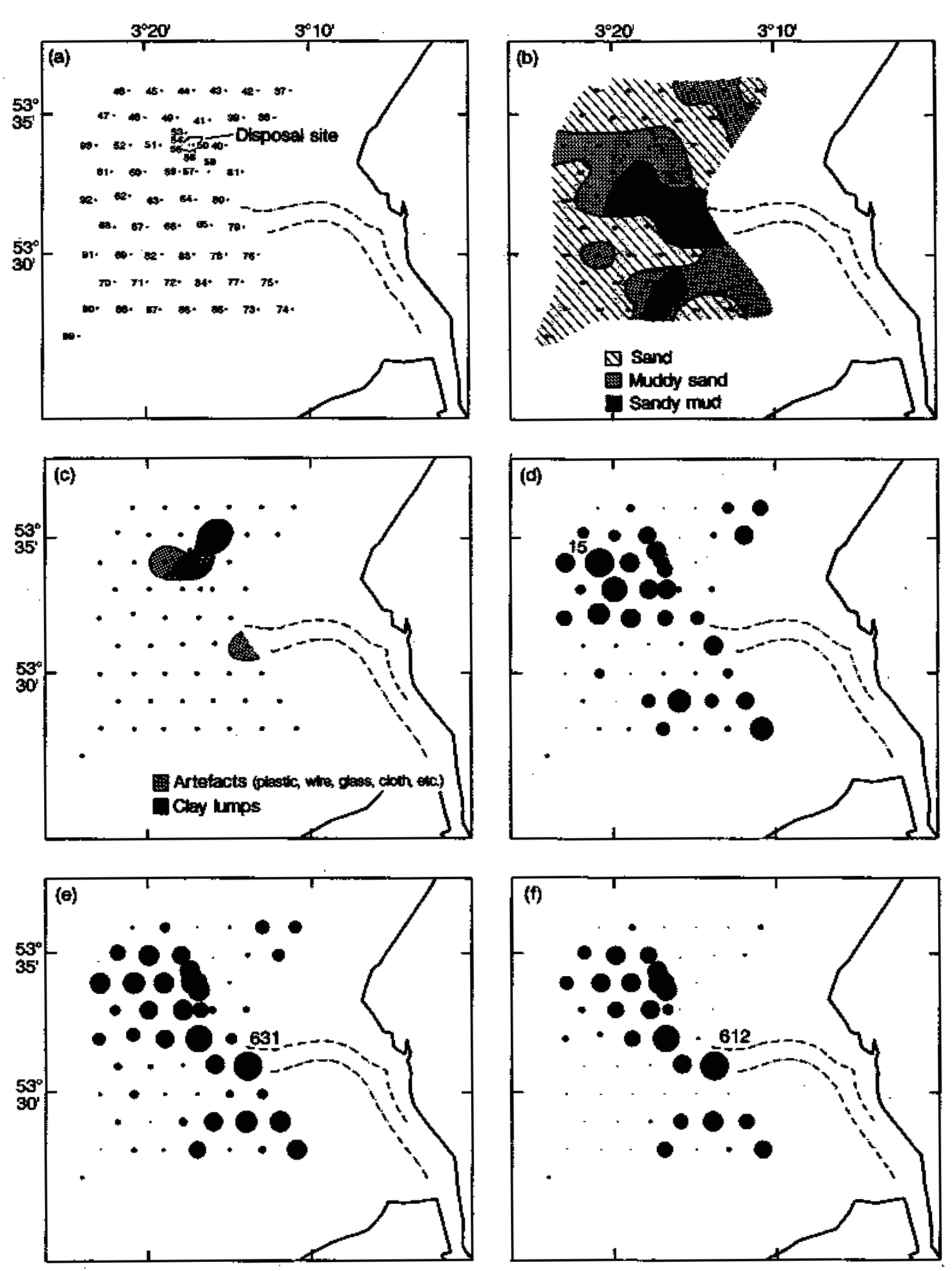

Figure 43. ***(a) Station positions for the September, 1992 survey; (b) distribution of superficial sediments; (c) distribution of artefacts and clay lumps in Day grab samples; (d) total numbers of taxa per grab. Circles are scaled relative to the highest value (specified); (e) total numbers of individuals per grab. Circles are scaled logarithmically relative to the highest count (specified); and (f) numbers of Pectinaria per grab. Circles are scaled logarithmically relative to the highest count (specified)***

it is also a significant component in the diet of many inshore flatfish populations (e.g. Macer, 1967; Lockwood, 1980; Basimi and Grove, 1985).

18.2.2 Methods

Field sampling

Sampling was conducted in September, 1992 from the University College of North Wales Research Vessel *PRINCE MADOG.* The station grid (Figure 43(a)) encompassed the dredged material disposal site, and extended southwards across the Mersey mouth into a muddy depositional area (the ‘Burbo Bight’). At each station, a Day grab sample was collected, and ¾ after sub-sampling sediments for later analyses of trace metal content ¾ the residual material was gently washed over a 5 mm mesh sieve for on-board identification and enumeration of the larger benthic fauna.

At stations where sufficiently high densities of *Pectinaria* were present, 30-50 intact specimens were carefully removed, and transferred into aerated plastic containers containing acid-washed fine sand topped up with sea water. (Commercially-available sand in the size range 0.1-0.3 mm was found to be suitable). The animals were left for 12 hours in this state, during which time the majority quickly burrowed and began deposit-feeding. Care was taken to distribute animals evenly onto the sand initially, to facilitate this process. Containers were checked from time to time during this period, and any dead or moribund animals which remained horizontally on the surface were discarded. The purpose of this treatment was to allow any sediment (and associated contaminants) present within the guts of the worms at the time of sampling to be replaced by clean sand.

After 12 hours, animals were removed from the acid-washed sand, and transferred onto 2 mm woven mesh sieves with circular supports, which were held within plastic bins containing aerated sea water. They were maintained in this condition for a further 12 hours, to allow voiding of most of the gut contents, which now consisted largely of acid-washed sand. At the end of this period, animals together with their tubes were carefully transferred into labelled plastic bags (padded with tissue paper), and then frozen prior to subsequent laboratory analysis.

Laboratory processing of samples
In order to establish variability in size distributions between samples of *Pectinaria*, the maximum tube width of each animal was measured to the nearest 0.1 mm using a micrometer eyepiece fitted to a binocular microscope. This measure is a good indicator of animal size, as may be deduced from the linear relationship: $y = -3.555 + 3.772x$ ($n = 90$; $r = 0.957$), where $y = \log_{10}$ wet blotted wt in g and $x = \log_{10}$ maximum width of tube in mm. Animals were then removed from tubes and the total wet and dry weight of each sample determined, prior to microwave digestion in nitric acid and subsequent analysis for cadmium, copper, zinc, lead and chromium content by Atomic Absorption Spectrophotometry (see Harper *et al*., 1989) and of mercury content by Atomic Fluorescence (Jones *et al*., in prep.). Results are expressed as ppm dry weight.

18.2.3 Results

Sediments and benthic fauna
The distribution of superficial sediments from visual descriptions of Day grab samples (Figure 43(b)), shows a band of muddy sediments extending from the mouth of the Queen's Channel (Mersey estuary), grading to muddy sands, notably in the central and southern parts of the sampling area. Sediments with a mud component are interspersed with extensive areas of clean fine sand. Physical indicators of dredgings disposal (or other recent anthropogenic) activity were limited to the immediate vicinity of the disposal site and off the mouth of the estuary (Figure 43(c)). In contrast to earlier surveys (e.g. Rees *et al*., 1992(b)), no underlying anoxic muddy layer was found in samples at the disposal site, which may reflect disposal of a high proportion of sandy material in the period immediately preceding the present survey. Data on the concentrations of trace metals in sediments are not yet available.

Numbers of taxa and individuals encountered in each grab (Figure 43(d and e)) show a clear association with sediment type, with the lowest densities occurring in the sandy areas. Somewhat fewer taxa occur at the disposal site compared with nearby muddy sands, but the numbers of individuals are not noticeably depressed. This accords with earlier evidence (e.g. Rees *et al*., 1992(b)) that newly deposited dredgings are rapidly recolonised. Densities of *Pectinaria* (Figure 43(f)) demonstrate that these worms are the dominant numerical component, at least of the larger (5 mm) size fraction of the benthos. They are largely confined to sediments with an appreciable mud component, i.e. muddy sands or sandy muds in Figure 43(b). Densities of adults of the bivalve *Abra alba* were conspicuously low in the survey area as a whole, compared with earlier surveys (e.g. Rees *et al*., 1992), but this probably simply reflects natural variation in recruitment success between years.

Trace metal concentrations in Pectinaria
Mean tube widths were generally similar across the sampling grid, but noticeably larger animals occurred at three southern stations (Figure 44(a)). The size-frequency distribution for all specimens encountered (Figure 44(b)) has a modal width of about 4 mm, but there is an indication that the distribution comprises a series of overlapping cohorts. This might be due to small differences in settling times or, alternatively, subtle differences in growth rates of the same cohort between locations. The right-hand tail is attenuated and ¾ although ill-defined ¾ appears to represent an older age-class which accounts for the three high mean widths in Figure 44(a).

Such differences between stations have the potential to confound interpretations of contaminant concentrations in the flesh of animals. Concentrations of a range of trace metals in samples of *Pectinaria* are shown in Figure 44(c-h). With the exception of mercury, there is no significant elevation in concentrations at the dredgings disposal site, though the content of cadmium and chromium was marginally higher nearby (Figure 44(c) and (d)). Overall, there is a tendency for higher concentrations of metals to occur at southern stations in the Burbo Bight area and, in all cases except mercury, highest concentrations corresponded with the location at which the largest mean animal width was found (Figure 44(a)). A degree of size dependance is therefore indicated.

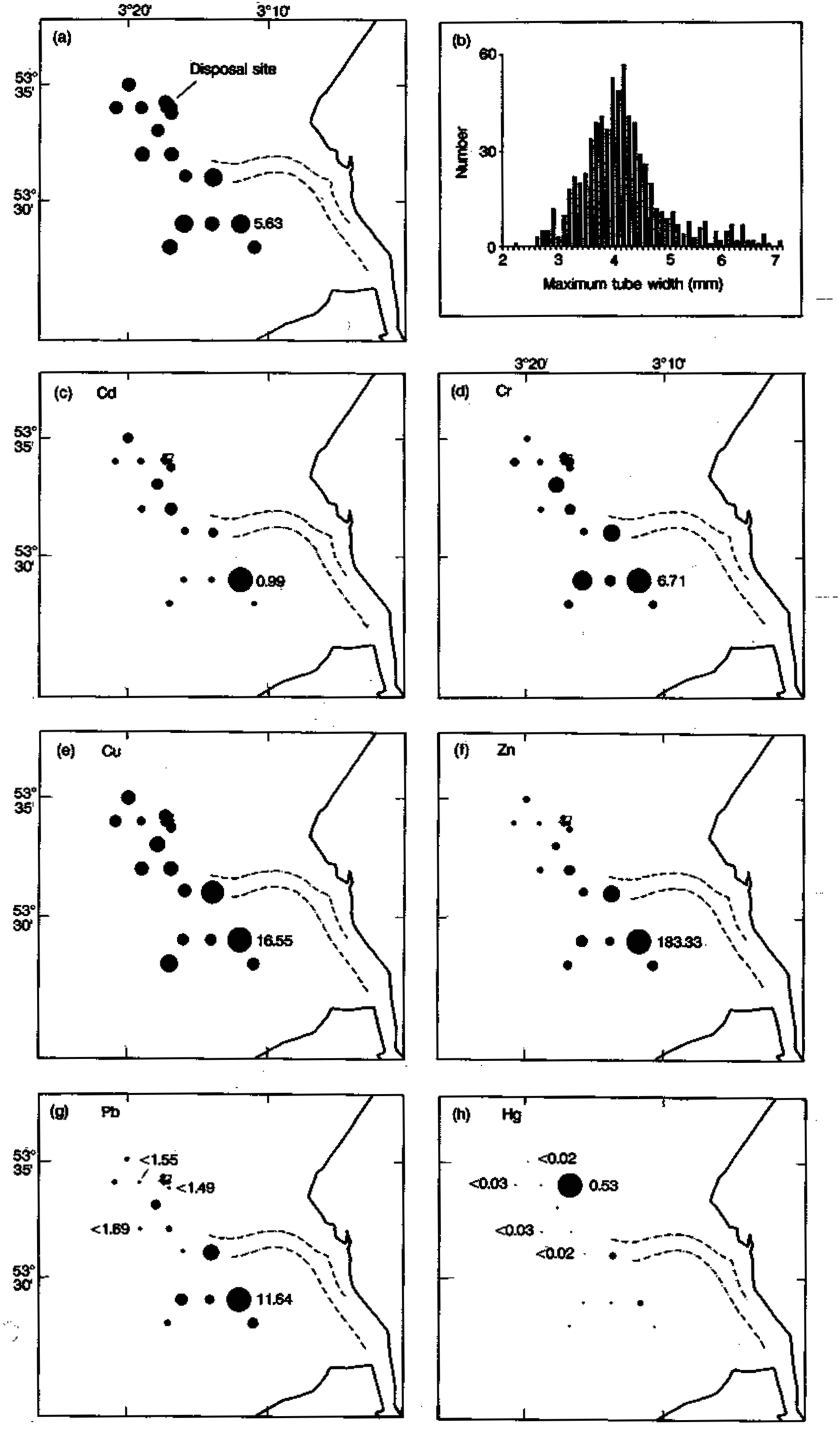

Figure 44. (a) Pectinaria : mean tube widths (mm) from samples collected in September, 1992. Circles are scaled relative to the highest value (specified); (b) frequency distribution of maximum tube widths (mm); and (c-h) concentrations of trace metals in the flesh of Pectinaria (ppm dry wt). Circles are scaled relative to the highest value in each plot (specified)

Concentrations of mercury (Figure 44(h)) were below the detection limit at a number of stations, including two at the disposal site. However, a concentration at least an order of magnitude greater than at most other stations was also encountered at the disposal site, demonstrating the capacity of *Pectinaria* to locally accumulate contaminants from newly-deposited sediments. There is no evidence to suggest poor exchange of original with clean sediment at this station during the gut voiding procedure, and it is notable that no other metals were significantly elevated, compared with elsewhere. A marginally elevated concentration (of 0.14 ppm) was also found at the mouth of the Queen's Channel.

Plots of concentrations of metals against mean tube widths (Figure 45(a-f)) confirm the relationship between the two. The solid circles in Figure 45 correspond with stations at the disposal site and again demonstrate that, with the exception of mercury, concentrations are not elevated here, compared with elsewhere on the grid.

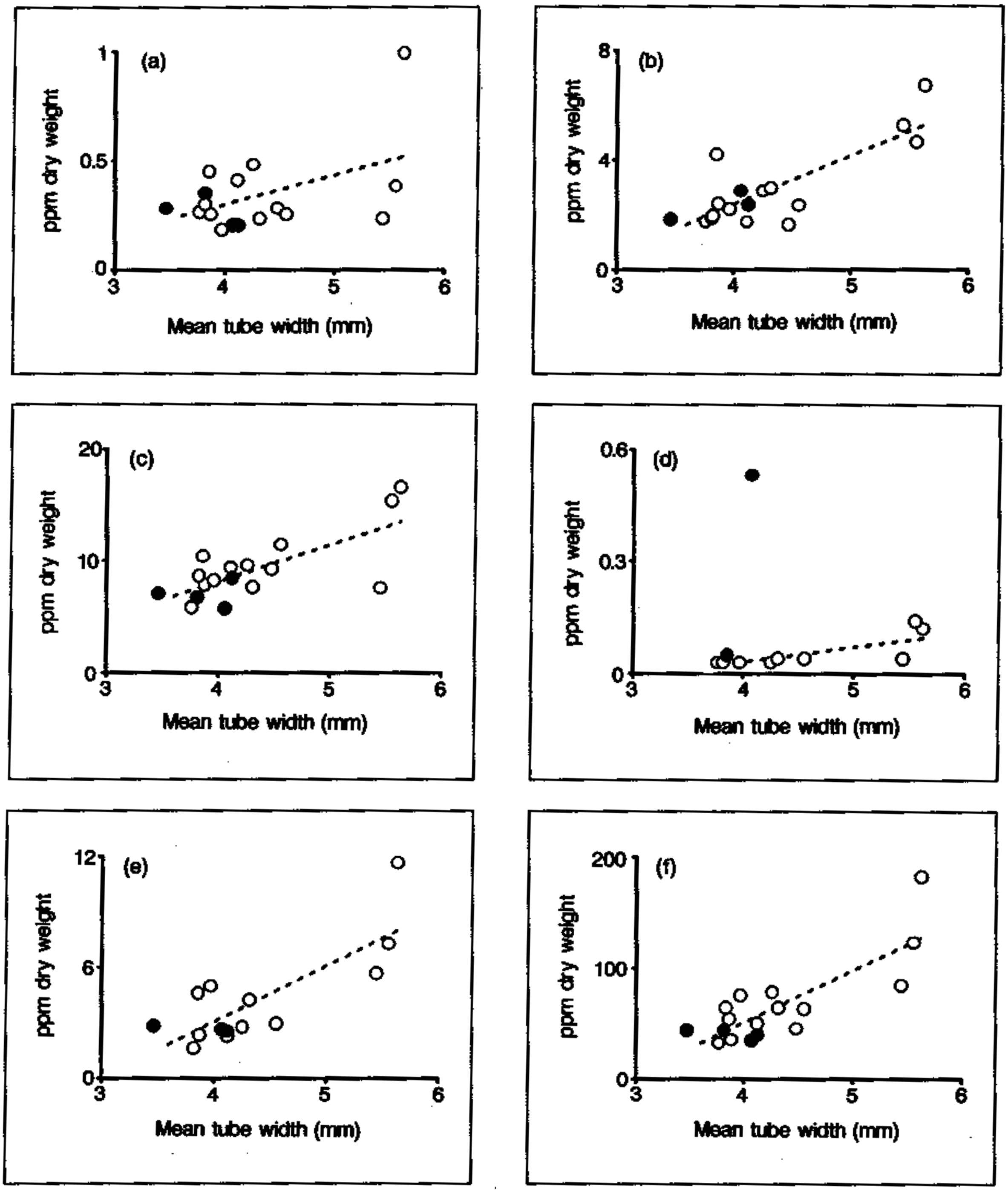

Figure 45. ***Relationships between mean tube widths and trace metal concentrations of Pectinaria in inner Liverpool Bay, September 1992: (a) cadmium; (b) chromium; (c) copper; (d) mercury; (e) lead and (f) zinc; solid circles represent stations at the dredgings disposal site***

Figure 45(b) is notable for one relatively high value of chromium (against a small mean width) which is located just to the south-west of the disposal site. Concentrations of cadmium, lead and zinc are particularly high at a station in the Burbo Bight area which also supported the largest mean animal size. Despite these anomalies, the comparative strengths of the linear relationships between concentration and size suggest an overall degree of uniformity in the bio-availability of metals to *Pectinaria* across stations. This would support the contention of Rowlatt (1988) that sediments in the inner Bay (and much of the Mersey estuary) are dominated by a single population of fines, with which trace metal contaminants are mostly associated.

18.2.4 Discussion

The results from field assessment of the distribution and abundance of larger benthic species confirmed the outcome of previous findings, namely that the effects of the disposal of maintenance dredgings to this shallow-water area are transient and localised. Populations of the polychaete *Pectinaria* appear to preferentially colonise newly-deposited material in a short space of time (at least in the summer months), though there was no evidence of enhanced population sizes at the disposal site, compared with nearby, on this occasion. This may be due to the disposal of dredgings with a reduced mud component in the period immediately preceding the present survey.

The procedure for clearing the gut contents of *Pectinaria* appeared to be successful, and may be adapted for other species. It is recognised that the use of a 5 mm mesh selects only relatively large animals. However, there are practical advantages to the handling of larger specimens, as the delicate tubes are much less prone to breakage. (Selecting for animals still retained in tubes is essential, as in their absence they are unable to burrow and hence deposit-feed: see Schafer, 1972).

Pectinaria is a short-lived polychaete (see e.g. Rees and Dare, 1993), and populations in this area probably rarely survive for more than one year, so that the relatively large differences in animal sizes encountered may be assumed to have developed over periods of, at most, months. In these circumstances, it is perhaps surprising that contaminant concentrations were generally found to increase linearly as a function of body size.

The concentrations of a range of trace metals in *Pectinaria* provided no evidence of a general enhancement at the disposal site. However, one station here yielded a high concentration of mercury, suggesting that the animal is capable of accumulating contaminants from newly-deposited material from the Mersey estuary.

Although no comparative data for this polychaete from other areas are presently available, the concentrations encountered do not appear to be particularly high. In Table 26, data from the present study are given alongside those for the polychaete *Nereis diversicolor* sampled by Langston (1986) in the Mersey estuary. As this is a longer-lived species than *Pectinaria*, the potential for bio-accumulation may be higher; also, samples were collected from substrates with historically high contaminant levels. However, the comparison is useful as an indication of differences in concentrations that may be found between two polychaetes from the general area, both of which are preyed upon by fish. It can be seen that, with the exception of chromium, concentrations of all metals are substantially lower in *Pectinaria*.

Leah *et al*. (1991) found a mean mercury content of 0.1 ppm wet weight in the stomach contents of dab sampled around the sewage-sludge disposal site in Liverpool Bay in 1988. This represented a decline over previous years, and was similar to values found elsewhere in the Bay. Furthermore, the broadly comparable concentrations in stomach contents and fish muscle tissue at that time suggested a process of simple bio-accumulation of mercury through the food chain, rather than of bio-magnification, assuming that this represented the major route for contaminant transfer.

Individual food items were not identified by Leah *et al.* (1991), so that the contribution of *Pectinaria* to the diet cannot be ascertained. The concentrations of mercury

Table 26. Comparison of concentrations of metals (ppm dry weight) in Pectinaria koreni (present study) and Nereis diversicolor (Mersey estuary; Langston, 1986)

Species		Cd	Cr	Cu	Hg	Pb	Zn
	Mean	0.34	2.9	9.1	0.1	3.8	66
Pectinaria koreni	S.D.	0.19	1.5	3	0.15	2.7	38
	Range	0.18-0.99	1.6-6.7	5.8-16.6	0.03-0.53	1.6-11.6	32-183
	Mean	0.7	0.6	46	0.91	9.5	196
Nereis diversicolor	S.D.	1	0.4	22	0.62	4.2	45
	Range	0.06-3.8	0.07-1.6	19-97	0.26-2.8	3.2-21.3	130-294

in *Pectinaria* at all but one station in the present study were considerably below 0.1 ppm wet weight, suggesting that the animal would not alone constitute a significant source of mercury to dab populations in this area **at the time of sampling**. However, as mercury concentrations are related to body size, then the potential for bioaccumulation by fish through feeding is likely to be greatest at those times of year when populations of *Pectinaria* are dominated by larger sizes while still being present in significant densities.

Future work will examine the relationship between concentrations of metals in sediments and animals. A comparison will also be made with data from a distant reference station, in order to establish whether there is any evidence of significantly higher levels of contamination in populations inhabiting the inner Liverpool Bay area as a whole. By these means, a better understanding will be gained of the routes for contaminant transfer and their potential significance in a coastal area subject to historically high volumes of waste inputs, notably via the Mersey estuary.

DEPOSITS IN THE SEA: LICENSING AND RELATED ACTIVITIES

19. LICENSING OF DEPOSITS IN THE SEA

19.1 Introduction

This section gives information about the licensing of deposits in the sea during 1992 under Part II of FEPA (1985) (Great Britain - Parliament, 1985(a)). It fulfils an undertaking by the Government to report on the licensing, enforcement and monitoring of activities related to the disposal of wastes at sea.

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

For the first time this report also includes statistics of other deposits in the sea; these are principally for construction purposes.

19.2 Legislation and licensing authorities

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

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

19.3 Enforcement

Scientists from the Burnham-on-Crouch Laboratory have powers to enforce licence provisions by visits to production units, storage sites and disposal vessels. They may take samples, and check records including logbooks. They carried out 70 inspections in 1992. The Sea Fisheries Inspectorate, with staff based on the coast, detects unlicensed disposal operations and enforces licence conditions relating to the disposal of the wastes in the designated disposal area. They made 188 inspections in 1992.

In Scotland, similar enforcement powers are held by staff of the SOAFD Marine Laboratory, Aberdeen and by the Scottish Sea Fisheries Protection Agency (SFPA). The Marine Laboratory made 10 inspections in 1992 and a further 66 visits were made by the SFPA. In Northern Ireland, enforcement duties are carried out by advisory officers and suitably trained outside contractors. Eight inspections were made in 1992.

MAFF investigated two alleged infringements in 1992. One case involved allegations that a vessel had deposited its cargo of dredgings before it reached the authorised site at Roughs Tower, off the Essex coast. The other involved a case where a licence was being used to dispose of material from a dredging site in Kent which was not listed on the licence. In both cases, there was insufficient evidence to mount a prosecution but warning letters were sent reminding the licensees to adhere strictly to the terms of their licences. A further two investigations, relating to allegations of the disposal of farmed salmon carcasses, were carried out by SOAFD. In one case, it was considered that there was insufficient evidence to proceed with a prosecution. In the second case, no fish carcasses were found in the alleged disposal area.

19.4 Report on licensing activities

Tables 27-32 give details, over the period 1988-1992, of the number of sea disposal licences issued, the quantity of waste licensed, and the quantity actually deposited, together with information on those contaminants in the wastes which the UK is required to report internationally to meet obligations under the Oslo and London Conventions (Great Britain - Parliament, 1972, (a-b)).

19.5 Licensing of liquid industrial waste disposal

Table 27(a) shows details of the two remaining liquid industrial wastes licensed for disposal at sea which remained in the period covered by this report Table 27(b) summarises the quantity of wastes licensed and the amounts deposited, including metallic contaminants, for the period 1988-1992. The locations of the disposal sites and the quantities deposited at each site are given in Figure 46. Sterling Organics were granted an extension of their licence for the early part of 1993 but in practice did not dispose of waste after the end of 1992. Thus, the UK fulfilled its international and other commitments to cease sea disposal at the end of 1992.

Table 27(a). Liquid industrial wastes licensed for disposal at sea in 1992 (1)

Licensed Quantity (t)	Company	Description of waste	Disposal sites	Quantity Deposited (t)
34,804	Imperial Chemical Industries Ltd	Ammoniacal liquor	Tees	148,658
40,800	Sterling Organics Ltd	Phenolic liquor	Tyne/ Spurn Head	32,067

Notes: (1) *No liquid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report*
Licensed quantities: licences were issued throughout the calendar year 1992 and were generally valid for twelve months
Tonnages deposited: relate to quantities deposited in the calendar year 1992, which may be covered by two licences, including one issued in 1991

Table 27(b). Summary of liquid industrial wastes licensed and disposed of at sea in 1992 (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	1988	19	311,411	249,744	0.03	0.29	0.88	0.01	0.41	0.63	0.76
	1989	16	292,968	248,454	0.02	0.19	0.78	0.00	0.27	0.81	0.60
	1990	5	228,000	209,961	0.02	0.10	0.46	0.00	0.19	0.56	0.27
	1991	2	205,000	191,945	0.03	0.32	0.10	0.00	0.45	0.49	0.07
	1992	2	75,604	180,725	0.03	0.37	0.12	0.00	0.30	0.18	0.05

Notes: (1) *No liquid industrial wastes were licensed or disposed of in Scotland or Northern Ireland in the period covered by this report*
For information on licensed quantities and tonnages deposited see footnote to Table 27(a)

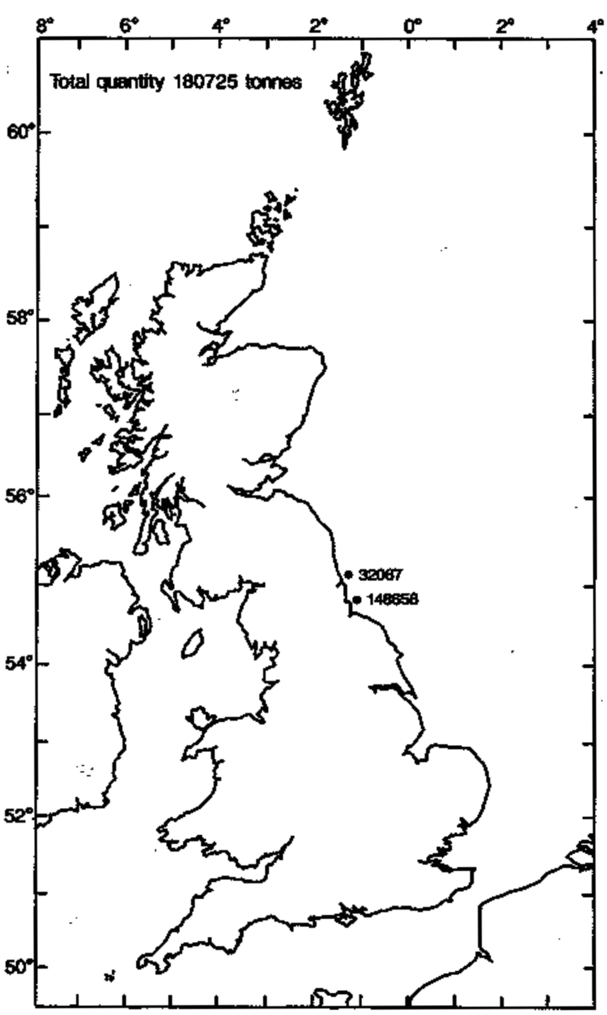

Figure 46. Liquid industrial material deposited at sea in 1992

19.6 Licensing of solid industrial waste disposal

Table 28(a) gives details of licences issued for the disposal of solid industrial wastes, and Table 28(b) details the quantities of metal contaminants in the material deposited. Figure 47 shows the locations of the relevant disposal sites together with the quantities deposited at each site. The bulk of the material licensed was stone extracted during coal mining. The material is accepted internationally as inert in the marine environment and as such, the presence of the metal contaminants is of no significance in terms of potential ecological impact. National Power's remaining licence to dispose of power station ash to sea was terminated at the end of 1992.

19.7 Licensing of sewage sludge disposal

Table 29(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 29(b). Figure 48 shows the location of the disposal sites for sewage sludge and the quantities deposited at each site.

At the 1987 Second International Conference on the Protection of the North Sea, (Department of Environment, 1987), the Government indicated it was taking urgent action to reduce the contamination by persistent, toxic or bioaccumulatable materials present in sewage sludge deposited in the North Sea and to ensure that the

Table 28(a). Solid industrial wastes licensed for disposal at sea in 1992 [1]

Licensed Quantity (t)	Company and source of waste	Description of waste	Disposal sites	Quantity Deposited (t)
	British Coal Collieries			
700,000	Dawdon/Seaham	Minestone	Bankside, Seaham	362,012
850,000	Easington	Minestone	Foreshore at Easington	682,632
650,000	Ellington	Minestone	Foreshore at Ellington	511,169
100,000	Point of Ayr	Minestone	Foreshore at Point of Ayr	99,169
800,000	Wearmouth	Minestone	Sunderland/Souter Point	424,040
800,000	Westoe	Minestone	Souter Point/North Tyne/Sunderland	578,899
1,100,000	Wearmouth/Westoe	Mine tailings	Souter Point/North Tyne/Sunderland	760,153
	National Power			
80,000	Blyth Power Station	Pulverised fuel ash	Blyth A	231,653

Notes: [1] *No solid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report*
For information on licensed quantities and tonnages deposited see footnote to Table 27(a)

Table 28(b). Summary of solid industrial waste licensed and disposed of at sea in 1992 [(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	1988	10	5,756,200	4,211,615	0.26	21	193	0.25	63	244	470
	1989	10	5,928,917	4,835,508	0.28	24	200	0.24	68	245	481
	1990	9	6,575,000	4,919,654	0.29	24	182	0.21	64	223	441
	1991	8	5,750,000	4,530,860	0.28	23	172	0.21	60	212	414
	1992	8	5,080,000	3,649,727	0.22	18	133	0.16	46	163	319
Scotland	1988	3	570	30	0.00	0	0	0.00	0	0	0
	1989	0	0	102	0.00	0	0	0.00	0	0	0
	1990	0	0	0	0.00	0	0	0.00	0	0	0
	1991	1	300	460	0.00	0	0	0.00	0	0	0
	1992	0	0	0	0.00	0	0	0.00	0	0	0
UK total	1988	13	5,756,770	4,211,645	0.26	21	193	0.25	63	244	470
	1989	10	5,928,917	4,835,610	0.28	24	200	0.24	68	245	481
	1990	9	6,575,000	4,919,654	0.29	24	182	0.21	64	223	441
	1991	9	5,750,300	4,531,320	0.28	23	172	0.21	60	212	414
	1992	8	5,080,000	3,649,727	0.22	18	133	0.16	46	163	319

Notes: [(1)] *No solid industrial wastes were licensed or disposed of in Northern Ireland in the period covered by this report*
For information on licensed quantities and tonnages deposited see footnote to Table 27(a)

quantities of such contaminants disposed of in the North Sea did not increase above 1987 levels. Earlier reports explained that to apply this control, limits were set for a series of key contaminants deposited under each licence. Table 30 compares the aggregated figures (in tonnes) authorised for disposal in the North Sea in 1992 with the estimated quantity at 1987 licensed levels.

Although the bulk of these contaminants comes from general domestic sources rather than industry, the licence levels for 1992 were below the 1987 baseline figures and, with the exception of chromium, below the 1991 levels.

Welsh and Wessex Water Companies ended sea disposal in 1992, the first companies to do so following the Ministerial announcement in March 1990 that sea disposal of sewage sludge must end in 1998. Welsh Water have opted for increased disposal of sludge on agricultural land while Wessex Water have built a plant to dry and pellet sludge, for sale as a soil conditioner.

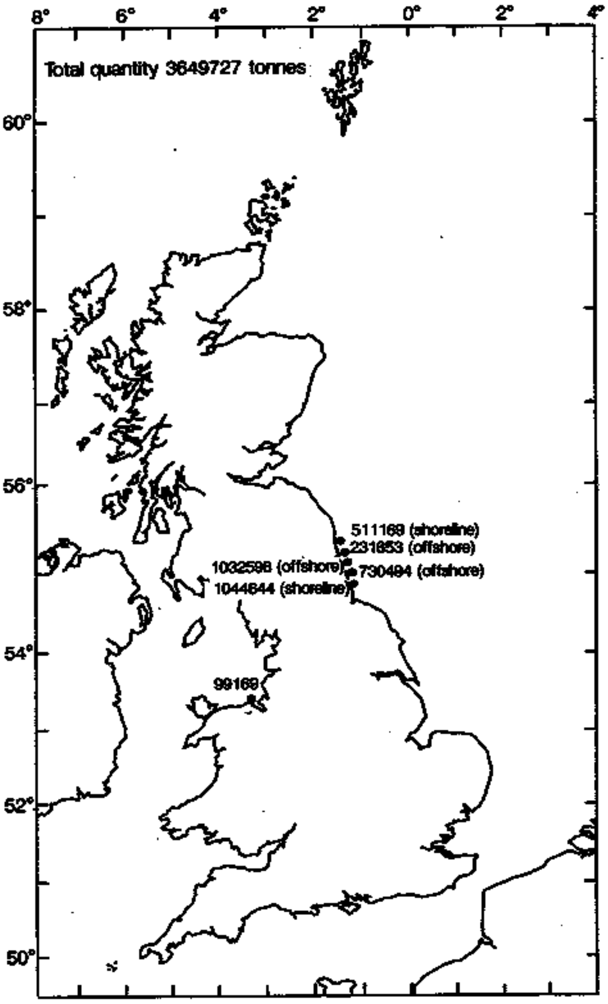

Figure 47. Solid industrial material deposited at sea in 1992

19.8 Licensing of dredged material disposal

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, for example, from channel development and deepening. Table 31 shows the numbers of licences issued, the quantity licensed, and the quantity deposited, together with figures for the quantity of a range of trace metals which enter the sea in the dredged materials. A proportion of the trace metals associated

Table 29(a). Sewage sludge licensed for disposal at sea in 1992

Country	Licensed quantity (t) (1)	Company and source of waste	Disposal sites	Quantity deposited (t) (1)
England and Wales	80,000	Anglian Water (Cliff Quay STW, Ipswich)	Roughs Tower	73,904
	65,000	Anglian Water (Colchester STW)	Roughs Tower	45,698
	150,000	Anglian Water (Tilbury STW)	Roughs Tower	145,922
	554,000	Northumbrian Water (Howdon, Chester-le-Street,Cramlington,Washington STWs)	Tyne/Spurn Head	495,000
	105,000	Northumbrian Water (Portrack, Billingham, Guisborough, Ayton STWs)	Tyne/Spurn Head	73,986
	1,965,000	North West Water (Davyhulme, Liverpool, Warrington STWs)	Liverpool Bay	1,897,684
	5,000	North West Water (Walney Island)	Liverpool Bay	3,763
	300,000	Southern Water (Woolston, Portswood, Millbrook, Slowhill Copse STWs)	Nab Tower	262,593
	58,000	South West Water (Countess Wear STW)	Lyme Bay	55,665
	63,000	South West Water (Plympton, Radford, Camel's Head, Ernesettle, Ivybridge, Saltash, Newton Ferrers STWs)	Plymouth	41,969
	4,500,000	Thames Water (Beckton, Crossness, Riverside, Deephams STWs)	Barrow Deep	4,279,686
	140,000	Yorkshire Water (Knostrop STW)	Spurn Head	132,892
Scotland	500,000	Lothian Regional Council	St Abb's Head/Bell Rock	298,525
	2,500,000	Strathclyde Regional Council	Garroch Head	1,686,000
Northern Ireland	300,000	Dept. Environment (Northern Ireland)	Belfast Sludge	261,000 (2)

Notes: *(1)All figures are for tonnage in wet weight*

(2)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 27(a)

Table 29(b). Summary of sewage sludge licensed and disposed of at sea in 1992

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	1988	16	7,503,580	7,267,935	2.45	92	130	0.95	18	144	355
	1989	15	8,321,305	7,373,212	2.17	85	128	0.94	18	157	297
	1990	14	8,553,579	7,098,944	1.76	68	113	0.88	18	109	238
	1991	14	8,373,569	7,449,084	1.88	67	121	0.92	17	103	269
	1992	12	7,985,000	7,739,369	1.70	61	123	0.85	17	93	273
Scotland	1988	2	2,300,000	2,004,963	0.42	45	33	0.19	3	26	56
	1989	2	2,300,000	1,940,575	0.46	53	32	0.17	3	24	58
	1990	2	3,000,000	1,946,430	0.27	34	30	0.16	3	18	39
	1991	2	3,000,000	1,984,035	0.24	35	41	0.10	3	20	44
	1992	2	3,000,000	1,984,525	0.21	29.91	36.26	0.10	2.38	17.81	40.35
Northern Ireland	1988	1	90,000	291,904 *	0.06	3	3	0.03	1	2	18
	1989	1	80,000	329,060 *	0.05	3	4	0.03	1	3	12
	1990	1	80,000	290,030 *	0.03	2	3	0.03	0	2	11
	1991	1	80,000	302,370 *	0.04	2	3	0.04	0	3	11
	1992	0	0	261,000 $	0.04	2	3	0.02	0	3	10
UK total	1988	19	9,893,580	9,564,802 *	2.92	140	166	1.17	22	172	429
	1989	18	10,701,305	9,642,847 *	2.68	141	164	1.14	22	184	367
	1990	17	11,633,579	9,335,404 *	2.06	104	147	1.06	21	129	288
	1991	17	11,453,569	9,735,489 *	2.15	104	165	1.06	20	125	325
	1992	14	10,985,000	9,984,894 $	1.95	93	163	0.97	19	114	323

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

$ *Includes 261,000 t disposed of by DoE(NI) Water Services under an administrative authorisation*

For information on licensed quantities and tonnages deposited see footnote to Table 27(a)

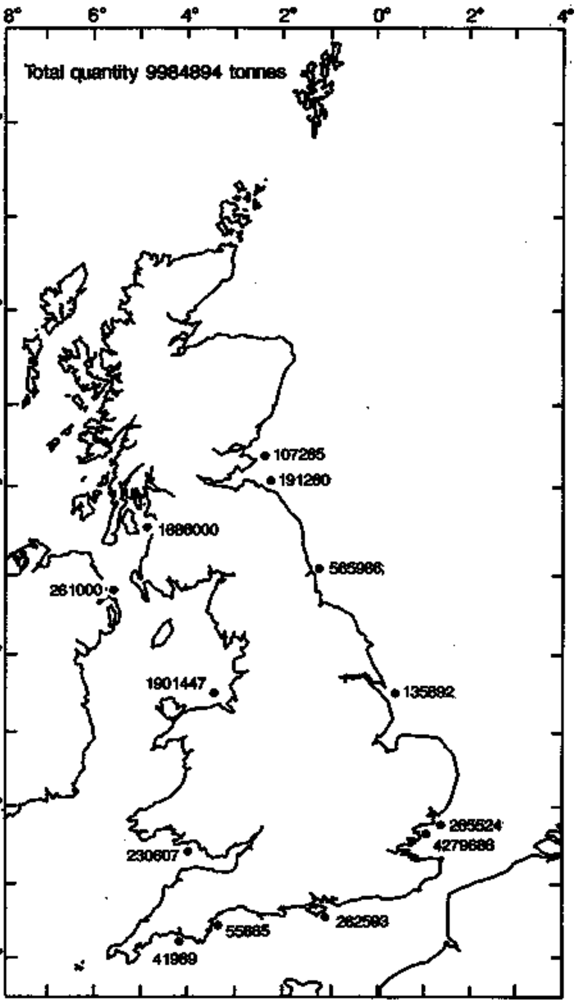

Figure 48. Sewage sludge deposited at sea in 1992

Table 30. Contaminants in sewage sludge authorised for disposal in the North Sea in 1992 compared against estimated quantities in 1987 (tonnes)

Year	Hg	Cd	Cr	Ni	Cu	Zn	Pb
1987	1.2	3.7	56.2	19.4	133.6	468.2	146.4
1992	1.1	2.3	43.8	15.8	127.1	339.4	125.3

with dredged material occurs within the mineral structure or is tightly bound, such that generally it will not be available to marine organisms. Figure 49 shows the main disposal sites used in 1992 and the quantities deposited at each site.

In its Report on Coastal Zone Protection and Planning, the Select Committee on the Environment (House of Commons, Environment Committee, 1992) recommended (among other things) that ‘there should be better co-ordination between those authorities responsible for dredging, disposing of, and utilising marine aggregates in order to avoid the unnecessary disposal of dredged material that could be used elsewhere’. The Food and Environment Protection Act already charges licensing authorities ‘in determining whether to issue a licence’ to ‘have regard to the practical availability of any alternative methods of dealing with’ the material which is the subject of the application. Nevertheless, responding to the Committee’s recommendations and recognising the pressure on aggregate resources and the growing interest in soft beach replenishment schemes, all licence applicants are now required to consider in detail all the available disposal options, including, the possibility of beneficial uses of the material.

Table 31. Summary of dredged material licensed and disposed of at sea in 1992

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	1988	131	61,645,223	34,691,093	23.2	1,165	1,091	11.0	753	2,199	5,191
	1989	138	66,408,100	40,810,718	18.6	1,234	1,037	9.3	638	1,877	4,938
	1990	135	63,983,920	33,728,978	12.2	1,023	834	6.8	484	1,426	3,724
	1991	108	57,782,520	39,886,812	7.4	1,189	773	7.0	518	1,263	3,394
	1992	123	55,741,813	24,243,998	6.0	812	511	4.2	291	877	2,271
Scotland	1988	25	4,148,690	3,506,685	1.2	114	89	1.5	43	123	259
	1989	27	4,252,950	3,154,756	1.1	106	106	1.3	40	141	313
	1990	21	3,031,960	2,109,114	0.8	61	59	0.8	36	116	210
	1991	26	5,147,245	2,788,611	0.6	70	53	0.6	22	79	167
	1992	35	5,920,005	4,026,861	0.5	83	71	0.8	28	91	197
Northern Ireland	1988	9	1,534,200	1,077,023	0.1	10	7	0.1	8	11	26
	1989	6	383,300	338,521	0.1	2	2	0.0	2	2	7
	1990	6	261,700	317,082	0.1	3	3	0.0	3	5	8
	1991	10	807,400	519,049	0.1	2	2	0.1	3	2	6
	1992	6	1,097,800	891,087	0.3	2	3	0.2	2	3	10
UK total	1988	165	67,328,113	39,274,801	24.6	1,289	1,187	12.5	805	2,333	5,476
	1989	171	71,044,350	44,303,995	19.8	1,343	1,145	10.6	679	2,021	5,258
	1990	162	67,277,580	36,155,174	13.2	1,086	895	7.6	523	1,546	3,942
	1991	144	63,737,165	43,194,472	8.1	1,260	828	7.8	543	1,344	3,566
	1992	164	62,759,618	29,161,946	6.9	897	585	5.2	321	970	2,479

Notes: For information on licensed quantities and tonnages deposited see footnote to Table 27(a)

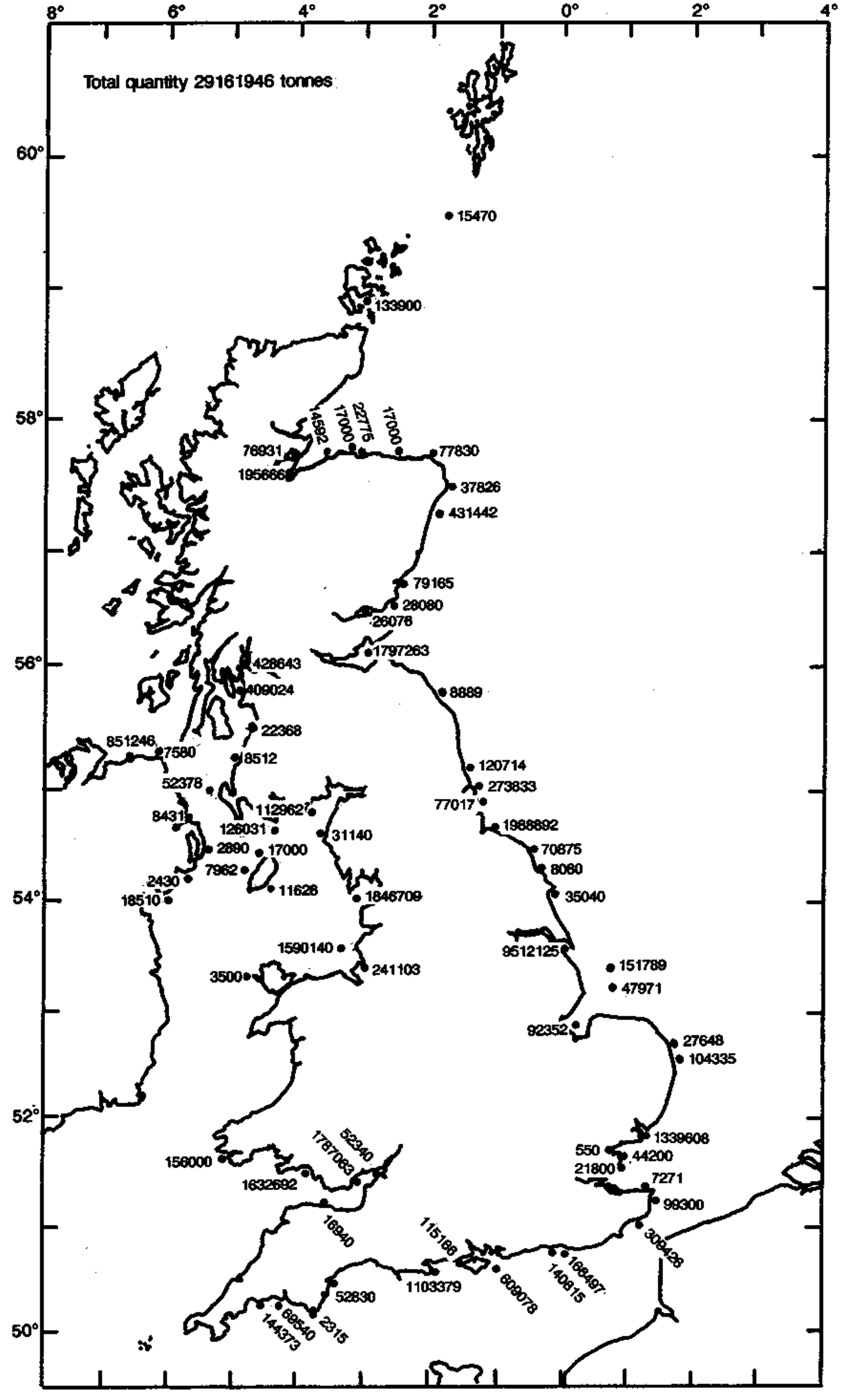

Figure 49. Dredged material deposited at sea in 1992

19.9 Other materials deposited at sea

Under Part II of FEPA, licences are also required for certain activities which do not involve the disposal of wastes, but nevertheless involve the deliberate and permanent deposition of substances in the sea. Each request for such a licence is carefully considered, but generally the anticipated impact is minimal and no monitoring is required. Specifically, such activities involved construction work below mean high water, the use of tracers, the application of biocides and burials at sea. Table 32 shows the numbers of such licences issued in 1992.

Table 32. Other categories of licence issued in 1992

Licence	England and Wales	Scotland	Northern Ireland	Total
Construction New and Renewal	183	101	5	289
Tracers, Biocides, etc	19	1	0	20
Burial at Sea	16	2	0	18

20. ADVICE ON FISHERY IMPLICATIONS OF PIPELINE DISCHARGES

This section gives a brief summary of activities carried out during 1992 in connection with provision of advice on fishery implications of pipeline discharges. The background to this work in relation to MAFF's responsibilities as a statutory consultee under the Water Resources Act 1991 (Great Britain - Parliament, 1991) was described in previous reports in this series (MAFF, 1991, 1992 and 1993(a)).

Table 33 shows the number of applications for each type of effluent sent to MAFF for comment since the implementation of statutory controls.

Table 33. Discharge consent applications assessed by MAFF

Year	Sewage	Trade effluent	Surface water	Total
1985	42	3	12	57
1986	38	21	19	78
1987	136	36	51	223
1988	132	53	60	245
1989	214	32	53	299
1990	88	28	30	146
1991	128	55	15	198
1992	812	94	35	941

The increase in annual totals does not reflect a rise in number of new discharges to the environment, but is mainly due to the phased implementation of the legislation. National control of non-radioactive discharges began in July 1984 with the implementation of Part II of the Control of Pollution Act 1974 (COPA) (Great Britain - Parliament, 1974). Initially only new discharges, or existing discharges containing certain hazardous substances, required consent, with the remainder being exempted. The exemption order was lifted in October 1986 and each of the previously exempt discharges (between two and three thousand) was then legalised by an unconditional 'deemed' consent which ignored any previously applicable conditions and simply reflected existing effluent quality. The original aim was to replace these by October 1992 with full, positively determined, consents which took account of the needs of the receiving water. In September 1989, the Water Act 1989 (Great Britain - Parliament, 1989) replaced COPA as the legislation controlling discharges, led to creation of the National Rivers Authority (NRA) and gave this body responsibility for control of discharge consents. This and other factors resulted in a delay in determination of deemed consents, as a consequence of which the process did not begin until the end of 1991. On 1 December 1991, the control of discharges passed to Schedule 10 of the Water Resources Act 1991.

About 15% of the applications received during 1992 were for discharges which had no potential for adverse fishery implications and therefore required no comment from MAFF. A further 20% were accepted subject to the inclusion of certain conditions such as freedom from hazardous materials, more stringent treatment or tighter consent limits. For the remainder (mainly determination of deemed consents for existing discharges), approval was given for temporary consents pending the acquisition of additional monitoring data, or cessation of the discharge (for example, as part of a major re-sewerage scheme).

As in previous years, the majority of the applications received during 1992 were for sewage discharges. Advice in relation to these was generally aimed at reducing inputs of material which could foul fishing gear (particularly persistent plastics) and improving the microbial quality of shellfish production areas. In November 1992, shellfish areas were given preliminary classifications under EC Directive 91/492, laying down the health conditions for the production and the placing on the market of live bivalve molluscs (European Communities, 1991(a)). This Directive, due to take effect on 1 January 1993, requires all commercial live bivalve mollusc production areas to be classified according to the level of bacterial contamination present in the mollusc flesh. The classification categories are given in Table 34.

Because of practical difficulties with relaying of shellfish in clean areas, a classification of less than Category B is likely to pose severe problems for the industry. MAFF's aim in advising on sewage discharge applications for such areas has therefore been to encourage schemes which will achieve a minimum of Category B as soon as possible, with a long term

Table 34. Classification of live bivalve mollusc production areas under Directive 91/492

Category	*E. coli.* $(100\ g)^{-1}$ flesh	Faecal coliforms $(100\ g)^{-1}$ flesh	Treatment required
A	<230	<300	None, provided end product standard is met
B	<4600 (in 90% of samples)	<6000 (in 90% of samples)	Depurate, heat treat or relay to meet Category A
C		<60000	Relay or heat treat using an approved process
Unclassified		>60000	Harvesting prohibited

objective of Category A for all shellfish production areas. In the absence of a statutory requirement to improve shellfish quality, such schemes have had to rely on the importance attached by the Water Company concerned to the continued existence of a local shellfish industry and the elimination of a health hazard (shellfish would still be liable to collection by individuals for their own use). Fortunately, most of the Water Companies have been very supportive and improvements are now scheduled for some shellfish areas.

During the year, MAFF provided advice in connection with determination of UK policy on implementation of EC Directive 91/271 concerning urban waste water treatment (European Communities, 1991(b)). Under this Directive, all significant discharges will generally require a minimum of secondary treatment, i.e. settlement followed by, for example, biological filtration or treatment with activated sludge. However, primary treatment (settlement only) may be allowed if the area has been designated as ‘less sensitive’ and the population equivalent (PE) is between 2000 and 10 000 (for discharge to an estuary) or 10 000 and 150 000 (for discharge to coastal water). Figures 50 and 51 give an indication of the number and location of discharges

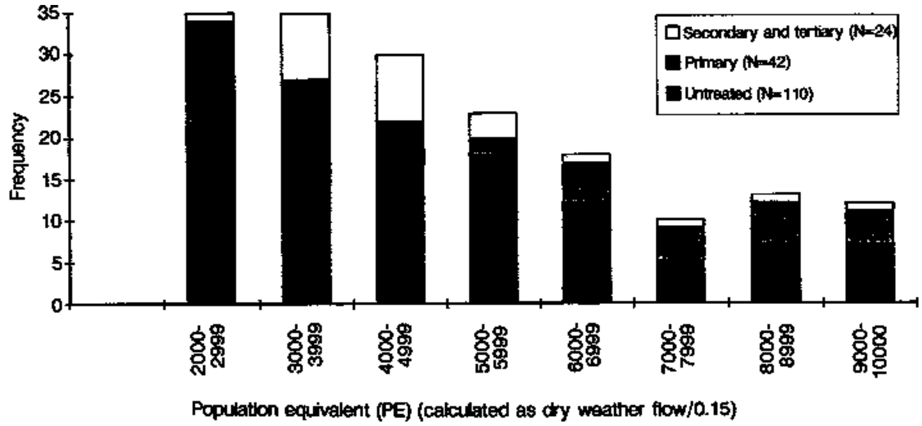

Figure 50. Number of estuarine sewage discharges which would qualify for primary treatment if the receiving water is designated as ‘less sensitive’ (i.e. population equivalent between 2000 and 10 000)

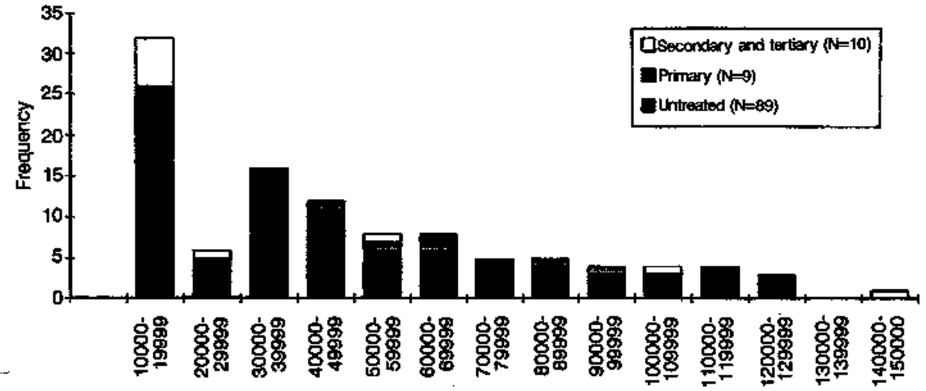

Figure 51. Number of coastal sewage discharges which would qualify for primary treatment if the receiving water is designated as ‘less sensitive’ (i.e. population equivalent between 10 000 and 150 000)

which could be regarded as falling within this category if the DoE allow the relaxation clauses to apply. This information, which includes the existing level of treatment, was taken from the MAFF discharge database (which is based on information provided by the NRA); the PE was calculated from dry weather flow data based on an assumption of 0.15 m^3/person/day.

Article 6 of Directive 91/271 states that primary treatment will only be allowed for significant discharges to ‘less sensitive’ areas, where comprehensive studies indicate that such discharges will not adversely affect the environment. However, the Directive does not define adverse effects or the nature of the comprehensive studies which are required. A task team was set up by the Marine Pollution Monitoring Management Group (MPMMG) under its Group Co-ordinating Sea Disposal Monitoring (GCSDM) to define such effects and the nature of the studies which will be necessary. Its report and recommendations were produced in December 1993 (MAFF, 1993(b)). The report makes passing reference only to microbiological determinands as these are not specifically covered by the Directive concerning urban waste water treatment. However, the importance of recognising the use of an area in determining actual consent conditions and level of treatment is acknowledged.

Acknowledgement

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

BIOTA

Contaminants in marine fish and shellfish	A Franklin J Jones
Bioassay techniques	J E Thain D A Sheahan S Bifield
Contaminants in marine mammals	R J Law

SEA WATER

Nutrients	R J Law
Dissolved trace metals	R E Laslett
Hexachlorocyclohexanes	C R Allchin
Triazine herbicides	C R Allchin
Hydrocarbons	R J Law
Volatile organic compounds	V Dawes

SEDIMENTS

Metals	S M Rowlatt D R Lovell

SPECIAL STUDIES

TBT	M J Waldock
Pesticides	P Matthiessen R Rycroft
Vitellogenesis in fish	D A Sheahan J Harries
Aggregate extraction areas	A J Kenny

DISPOSAL AT SEA: ENVIRONMENTAL ASSESSMENT STUDIES

Sewage sludge disposal sites	H L Rees S M Rowlatt
Litter	D S Limpenny
Solid waste and dredged material disposal sites	H L Rees S M Rowlatt

DEPOSITS IN THE SEA: LICENSING AND RELATED ACTIVITIES

Licensing	G Boyes C M G Vivian
Pipeline discharges	F L Franklin

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). [See also addendum on page 83.]

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 food which naturally contain more than 50 mg kg^{-1}, such as herring and shellfish).’

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

A2.2 Pesticides/PCBs

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

(a) HCB

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

(b) HCH

Codex Alimentarius Commission’s maximum residue limit (MRL) (FAO/WHO, 1987) is 2 mg kg^{-1} in meat fat for g-HCH. The ‘expected’ values are up to 0.05 mg kg^{-1} wet weight for each of a- and g-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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A2.4 Addendum for mercury

Community Decision 93/351/EEC will apply to samples of fishery products collected after 1992. This states that the mean total mercury content of the edible parts of fishery products must not exceed 0.5 mg kg^{-1} fresh weight, increased to 1.0 mg kg^{-1} fresh weight for some species listed in an annex.

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AQUATIC ENVIRONMENT MONITORING REPORT Number 40