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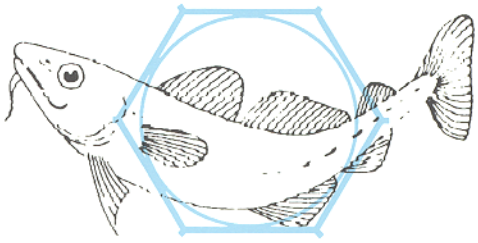
Number 26



Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment and Activities Regulating the Disposal of Wastes at Sea, 1988-89

MAFF

Directorate of Fisheries Research
Lowestoft, 1991

MINISTRY OF AGRICULTURE, FISHERIES AND FOOD
DIRECTORATE OF FISHERIES RESEARCH

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LOWESTOFT 1991

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

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FOREWORD

In 1990, MAFF published the first of its reports on monitoring of non-radioactive contaminants under its new title "Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment, 1984-1987" (MAFF, 1990). The revised version provides information to interested parties on a regular basis, in a single report covering all major on-going projects undertaken in this area of monitoring work. The report parallels that on monitoring of radioactivity in the aquatic environment, which is also published by MAFF in its Aquatic Environment Monitoring Report Series. This, the second report in the new style, covers work carried out during 1988-1989. It is envisaged that publication will be on an annual basis in future.

The bulk of the monitoring was undertaken by staff of the Directorate of Fisheries Research (DFR) Aquatic Environment Protection Division 2, at Burnham-on-Crouch; some work carried out by DFR's Fish Diseases Laboratory at Weymouth is also reported.

In addition to a general coverage of monitoring programmes, it has been decided that these reports should include information, previously published separately, on MAFF's activities in the field of the licensing of sea disposal, carried out under Part II of the Food and Environment Protection Act 1985. This has been presented as Section 19 of the present report. For the sake of completeness, Section 20 has also been added, describing MAFF's responsibilities as a statutory consultee in the consenting of pipeline discharges under the Water Act 1989 (formerly the Control of Pollution Act 1974).

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1. MONITORING OF CONTAMINANTS IN MARINE FISH AND SHELLFISH

1.1 Introduction

The results from a wide-ranging geographical survey of contaminants in marine biota were given in a previous report in this monitoring series (MAFF, 1990) which covered the work carried out from 1984 to 1987. Effort in 1988/89 (see Appendix 1 for staff involved in this and other projects throughout the Report) has been mainly concentrated in those areas where previous monitoring had indicated that contaminant levels were above the anticipated concentration ranges. The Joint Monitoring Group (JMG) of the Oslo and Paris Commissions 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. Results from samples taken in 1988 were required for this purpose and it is generally these data which are given in the tables which follow. Some analysis of fish for mercury was carried out in 1989 to indicate compliance with the Environmental Quality Standard (EQS) adopted by the European and Paris Commissions and these results have been given for this contaminant.

Also reported are the results from the latest (1987-89) survey of heavy metal concentrations in the edible crab (*Cancer pagurus*). Regular monitoring of this species was recommended by the MAFF Steering Group on Food Surveillance (MAFF, 1983), as metals such as cadmium can be accumulated to relatively high levels in some crab tissues.

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

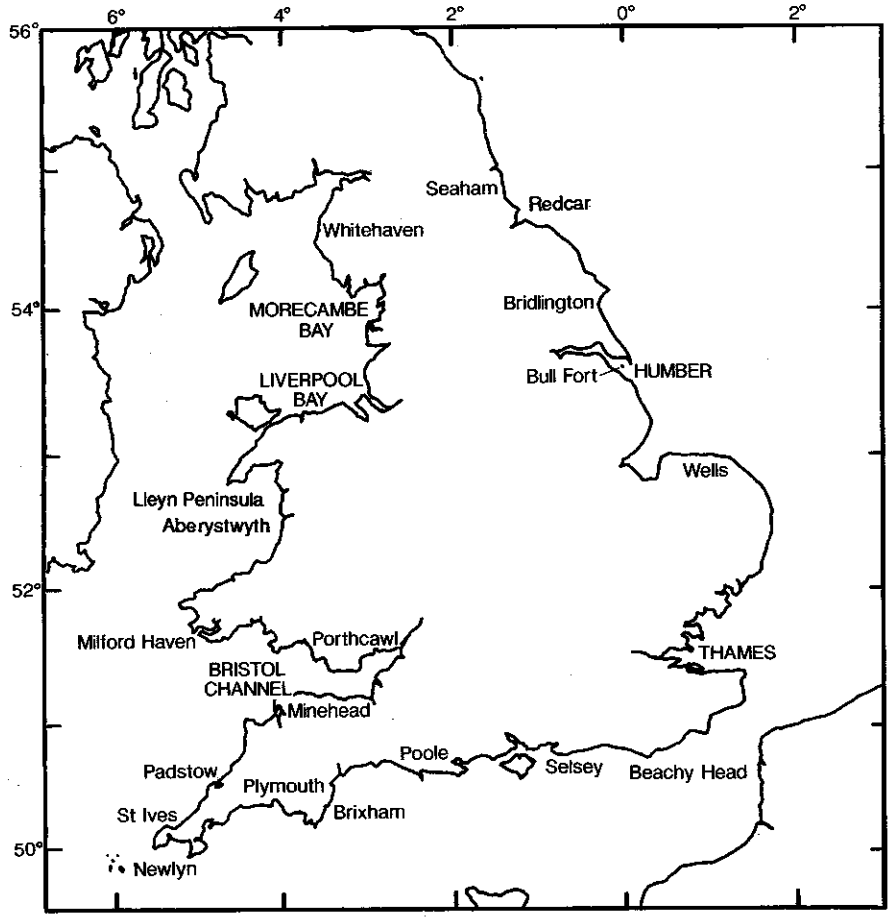


Figure 1. Sites and areas from which fish and shellfish samples were obtained

1.2 Methods

1.2.1 Sampling

There are seven 'core' species in the Burnham monitoring programme: two roundfish [cod (*Gadus morhua*) and whiting (*Merlangius merlangus*)], four flatfish [plaice (*Pleuronectes platessa*), sole (*Solea solea*), dab (*Limanda limanda*) and flounder (*Platichthys flesus*)] plus the blue mussel (*Mytilus edulis*). Samples of these were obtained, where possible, from areas where previous monitoring had indicated that contaminant levels were at or near the upper JMP categories, i.e. for mercury and PCBs in some species of fish from Liverpool and Morecambe Bays and cadmium in mussels from the Humber, Bristol Channel and near to Whitehaven (MAFF, 1990). Samples of crab were obtained from fishermen/ merchants at ports of landing around the coasts of England and Wales.

Individual analyses were undertaken of crab tissues and of fish muscle for mercury; duplicate samples of pooled tissue were used for all other analyses.

1.2.2 Analysis

Concentrations of mercury and cadmium were determined by cold vapour and conventional flame atomic absorption spectrophotometry respectively, following nitric acid digestion. PCBs and other organochlorine residues were determined by capillary gas chromatography, using electron capture detection after alumina and silica column 'clean-up' and separation. PCBs have been quantified as the formulation Aroclor 1254 to allow comparison with JMP guidelines; data will also be produced in future years on individual chlorobiphenyls. Full details of the methods employed by the Burnham laboratory are given in Harper *et al.* (1989) and Allchin *et al.* (1989).

1.3 Results

The results of the analyses are given in Tables 1-4. Where analysis of individual specimens was carried out, the range of contaminant values in the sample is given below the mean. Where analysis of duplicate bulked samples of tissue has been undertaken, both results are listed in the table below the mean.

1.3.1 Mercury in fish muscle

Concentrations of mercury found in fish taken from Liverpool and Morecambe Bays are given in Table 1. In addition, results are reported from 'check' monitoring carried out in the Thames Estuary, an area which has received considerable inputs of mercury in the past and for which data had to be supplied to the JMG until 1984.

Table 1. Concentrations of mercury in fish muscle (JMP 'upper' level >0.30 mg kg^{-1} wet weight)

Area	Species	Date of capture	Number of fish	Mean length (cm)	Concentration (mg kg^{-1} wet weight)
Liverpool Bay	Cod	Sep. 88	25	34.3	0.17 (0.06-0.27)
	Whiting	Sep. 89	25	32.6	0.32 (0.12-0.63)
	Plaice	Sep. 89	25	30.9	0.13 (0.02-0.28)
	Sole	Sep. 89	22	25.7	0.18 (0.09-0.27)
	Flounder	Sep. 89	31	32.2	0.16 (0.07-0.28)
	Dab	Sep. 89	34	25.0	0.24 (0.08-0.65)
Morecambe Bay	Cod	Nov. 89	25	42.1	0.13 (0.06-0.24)
	Whiting	Nov. 89	25	31.8	0.22 (0.11-0.49)
	Plaice	Nov. 89	25	31.5	0.12 (0.07-0.20)
	Sole	Jun-Aug. 88	29	28.2	0.18 (0.10-0.37)
	Flounder	Nov. 89	25	34.6	0.24 (0.14-0.35)
	Dab	Nov. 89	24	29.6	0.30 (0.16-0.50)
Thames Estuary	Cod	Nov. 88	25	45.7	0.11 (0.05-0.25)
	Whiting	Nov. 88	21	35.6	0.19 (0.09-0.39)
	Plaice	Feb. 88	25	26.5	0.06 (0.06;0.06)
	Sole	Jul. 89	15	26.8	0.07 (0.06;0.07)
	Flounder	Sep. 88	25	28.2	0.16 (0.10-0.33)
	Dab	Jul. 89	24	26.5	0.11 (0.04-0.22)

Concentrations in some fish species from both Liverpool and Morecambe Bays remain in or near the JMP 'upper' category for mercury. These areas are subject to discharges from chloralkali plants and thus to provisions of EC Council Directive 82/176 EEC, on Limit Values and Quality Objectives for Mercury Discharges by the Chloralkali Electrolysis Industry (see Appendix 2). Compliance has also to be demonstrated with agreed Paris Commission Environmental Quality Standards (EQSs) (see Appendix 2). Data have, therefore, to be submitted annually to indicate compliance with an EQS of 0.30 mg mercury kg^{-1} of representative wet fish flesh. For this purpose, each individual fish species' concentration is weighted according to the quantity of that particular species landed from the area in question and an overall representative area mean concentration is then calculated. The weighted mean mercury concentrations calculated for the years covered by this report were as follows:

	1988	1989
Liverpool Bay	0.20	0.16
Morecambe Bay	0.24	0.18

i.e. the standard was met at both sites.

Concentrations of mercury in fish from the Thames Estuary (Table 1) were all in the 'lower' or 'medium' JMP categories confirming, as reported previously (Franklin, 1987), that the situation has improved considerably since the early 1970s when the overall mean mercury concentration was around 0.40 mg kg^{-1} wet weight.

1.3.2 Cadmium in mussels

Concentrations of cadmium found in mussels taken from the Humber, the Bristol Channel and from Whitehaven are listed in Table 2. No commercial landings are recorded for these areas; the samples are used simply as an index of relative contamination.

The 1988 sample from the Humber (Bull Fort) indicated that the cadmium level was lower than that found previously (between 5.5-6.5 mg kg^{-1} dry weight in 1985-87) and was now in the 'medium' JMP category. A further sample was collected in 1989, and a similar value to that for 1988 was obtained. Samples were collected in 1988 from both the North (Porthcawl) and South shore (Minehead) of the Bristol Channel. The cadmium level in the mussels from Minehead remain in the 'upper' JMP category. The concentration in the sample from near to Whitehaven (Parton) was at a similar level to that found in earlier years, remaining well above that found elsewhere in MAFF surveys carried out in recent years in England and Wales [typical concentrations being around 2 mg kg^{-1} dry weight (MAFF, 1990)].

1.3.3 PCBs in fish

Concentrations found in fish taken from Liverpool and Morecambe Bays are given in Table 3. As in previous MAFF surveys, fish liver tissue was used as the primary monitoring medium. This was supplemented by analysis of the corresponding fish muscle tissue where, in the liver, PCB concentrations were found to be in or near the JMP 'upper' categories. Cod were not obtained in 1988 from Morecambe Bay, but results from a sample taken in 1989 have been provided.

Table 2. Concentrations of cadmium in 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)	(mg kg^{-1} dry weight)
Humber	Jul. 88	50	4.4	0.61 (0.64;0.57)	4.5 (4.6;4.3)
	Jul. 89	50	4.2	0.64 (0.65; 0.62)	4.5 (4.5;4.4)
Bristol Channel					
- Porthcawl	Feb. 88	50	4.2	0.47 (0.46;0.47)	3.7 (3.7;3.6)
- Minehead	Sep.88	50	4.1	1.2 (1.1;1.2)	8.1 (7.8;8.3)
Whitehaven	Jul. 88	50	4.1	4.3 (4.2;4.3)	33.4 (32.8;33.9)

Table 3. Concentrations of PCBs in fish (JMP upper level > 5.0 mg kg^{-1} wet weight for cod liver*; > 1.0 mg kg^{-1} wet weight for flounder liver+; > 0.05 mg kg^{-1} wet weight for fish muscle)

Area	Species	Date of capture	Number of fish	Mean length (cm)	Concentration (mg kg^{-1} wet weight)	
					Liver	Muscle
Liverpool Bay	Cod	Sep. 88	19	33.7	5.6 (5.6;5.6)	0.003 (0.002;0.003)
	Whiting	Sep. 88	30	35.9	6.1 (5.9;6.3)	0.012 (0.013;0.011)
	Plaice	Sep. 88	25	30.8	1.9 (1.8;2.0)	0.020 (0.021;0.019)
	Sole	Sep. 88	24	26.9	0.39 (0.38;0.39)	-
	Flounder	Sep. 88	29	33.2	0.88 (0.87;0.89)	0.010 (0.010;0.009)
	Dab	Sep. 88	25	25.2	2.2 (2.2;2.1)	0.066 (0.065;0.067)
Morecambe Bay	Cod	Nov. 89	25	42.1	1.6 (1.5;1.7)	-
	Whiting	Aug.88	18	28.0	8.2 (8.2;8.1)	0.017 (0.017;0.016)
	Plaice	Jun. 88	27	28.4	0.60 (0.58;0.62)	-
	Sole	Aug. 88	25	28.3	0.35 (0.33;0.36)	-
	Flounder	Aug. 88	25	34.0	0.39 (0.39;0.38)	-
	Dab	Aug. 88	23	27.2	1.7 (1.6;1.7)	0.044 (0.043;0.044)

**Value used for all roundfish*
+Value used for all flatfish

Concentrations in the JMP 'upper' category were found in cod, whiting, plaice and dab liver from Liverpool Bay and whiting and dab liver from Morecambe Bay, confirming that these areas are relatively highly contaminated. However, as found in previous MAFF surveys, concentrations in the fish muscle tissue were very much lower than those in liver; only in dab muscle tissue from Liverpool Bay was the level in the JMP 'upper' category (i.e. >0.05 mg kg^{-1} wet weight).

1.3.4 Organochlorine pesticides in fish

In addition to the JMP contaminants mentioned above, previous monitoring (MAFF, 1990) had indicated that pesticide concentrations above 'expected' ranges were also present in whiting liver, both from Liverpool Bay (dieldrin, DDE and TDE) and Morecambe Bay (DDE and TDE). No sample of fish liver taken in Liverpool Bay in 1988 contained pesticides above expected levels, but the levels of DDE and TDE in whiting liver from Morecambe Bay remained above the expected range (up to 0.50 mg kg^{-1} wet weight) at 0.64 and 0.96 mg kg^{-1} wet weight respectively.

1.3.5 Heavy metals in crabs

Analysis was carried out on both the 'white' (claw) meat and 'brown' (body) meat and results are summarised in Table 4(a) and (b) respectively.

Concentrations of mercury in both claw and body meat were in the JMP lower/medium categories and appeared to be slightly less than those found in 1975, the last time a large body of metals data were reported for this species (Murray, 1981).

Levels of copper and zinc were very similar to the 1975 values, and in both surveys lead concentrations were below the limit of determination. The Food Standards Committee guidelines state that relatively high values such as those found for copper in crab body and zinc in crab claw are 'permitted' if contamination is from natural sources (Appendix 2). Consideration of the geographical pattern of contamination, as shown by the results in Table 4(a-b), would indicate that such an assumption can be made. This appears also to be the case for cadmium, where levels found in brown meat (all were less than the limit of determination in white meat) do not appear to be related to known anthropogenic inputs. Results in Table 4(b) indicate that there is very considerable variation, in the levels of cadmium, both within and between samples of brown crab meat, and therefore some caution must be taken in comparing present results with those of earlier surveys. However, the overall mean concentration of cadmium in the samples taken during the 1987-89 survey (6.6 mg kg^{-1} wet weight) is of the same order as that reported previously from samples collected in the early and late 1970's (~8 mg kg^{-1} wet weight - MAFF, 1983; Franklin, 1987), though considerably less than in the 1975 survey (Murray, 1981) where some very high results were reported from western areas of England and Wales and where the overall mean concentration was 16 mg kg^{-1} wet weight.

Table 4(a). Concentrations of metals in crab 'white meat' (claw)

Port of landing	Date of capture	Mean length (cm)	Mean concentration in mg kg^{-1} wet weight (samples of 5 crabs analysed individually)					% dry matter
			Hg	Cu	Zn	Cd	Pb	
Seaham	Sep. 87	17.0	0.18 (0.11-0.26)	18 (12-23)	88 (82-97)	<0.2	< 1.0	27
Redcar	Sep. 87	17.8	0.16 (0.12-0.22)	14 (12-15)	84 (71-110)	<0.2	< 1.0	20
Bridlington	Sep. 87	16.1	0.15 (0.06-0.23)	17 (15-19)	50 (29-90)	<0.2	< 1.0	27
Wells/Cromer	Nov. 88	15.5	0.12 (0.07-0.21)	22 (17-29)	119 (92-165)	<0.2	< 1.0	22
Royal Sovereign/ Beachy Head	May 89	14.3	0.16 (0.11-0.21)	14 (10-18)	76 (62-100)	<0.2	< 1.0	25
Selsey	Oct. 89	17.4	0.14 (0.06-0.27)	13 (10-22)	61 (50-70)	<0.2	< 1.0	20
Poole	Jul. 89	15.1	0.08 (0.06-0.12)	11 (5-15)	65 (54-77)	<0.2	< 1.0	22
Brixham	Aug. 87	18.3	0.18 (0.05-0.24)	13 (10-15)	74 (69-86)	<0.2	< 1.0	23
Plymouth	Sep. 87	19.1	0.23 (0.12-0.33)	12 (7-15)	82 (67-98)	<0.2	< 1.0	24
Newlyn	Jul. 87	17.7	0.11 (0.08-0.14)	10 (6-15)	65 (56-75)	<0.2	<1.0	19
St Ives	Jul. 87	15.8	0.07 (0.06-0.08)	15 (14-16)	35 (30-38)	<0.2	<1.0	24
Padstow/ Newquay	Jul. 87	16.8	0.13 (0.08-0.22)	12 (10-14)	82 (74-88)	<0.2	<1.0	26
Milford Haven	Jul. 87	17.9	0.12 (0.10-0.17)	12 (10-15)	76 (67-98)	<0.2	< 1.0	23
Aberystwyth	Jul. 87	13.6	0.11 (0.09-0.13)	16 (11-21)	95 (93-100)	<0.2	<1.0	26
Lleyn Peninsula	Jul. 87	15.0	0.11 (0.09-0.15)	15 (6-25)	78 (60-91)	<0.2	< 1.0	24
Parton (Whitehaven)	May 88	12.3	0.09 (0.06-0.12)	17 (11-32)	84 (76-96)	<0.2	<1.0	25
Overall mean			0.13	14	76	<0.2	<1.0	24

Table 4(b). Concentrations of metals in crab 'brown meat' (body)

Port of landing	Date of capture	Mean length (cm)	Mean concentration in mg kg^{-1} wet weight (samples of 5 crabs analysed individually)					% dry matter
			Hg	Cu	Zn	Cd	Pb	
Seaham	Sep. 87	17.0	0.11 (0.09-0.16)	104 (54-140)	60 (49-69)	3.8 (1.9-7.3)	<1.0	38
Redcar	Sep. 87	17.8	0.14 (0.11-0.16)	91 (70-100)	27 (20-32)	2.7 (1.7-3.8)	<1.0	32
Bridlington	Sep. 87	16.1	0.14 (0.09-0.23)	65 (16-110)	45 (26-98)	3.6 (1.5-6.5)	<1.0	31
Wells/Cromer	Nov. 88	15.5	0.16 (0.07-0.24)	76 (16-240)	55 (35-88)	8.2 (1.4-12)	<1.0	28
Royal Sovereign/ Beachy Head	May 89	14.3	0.16 (0.13-0.19)	79 (24-120)	37 (24-60)	4.8 (2.6-5.6)	<1.0	32
Selsey	Oct. 89	17.4	0.09 (0.06-0.14)	33 (7-55)	41 (17-51)	3.8 (1.2-7.1)	<1.0	30
Poole	Jul. 89	15.1	0.09 (0.06-0.14)	52 (37-110)	36 (26-49)	9.8 (6.9-14)	<1.0	31
Brixham	Aug. 87	18.3	0.17 (0.12-0.21)	68 (37-98)	50 (41-60)	15.6 (6.1-26)	<1.0	31
Plymouth	Sep. 87	19.1	0.16 (0.13-0.23)	25 (7-48)	45 (26-79)	5.0 (1.8-9.7)	<1.0	34
Newlyn	Jul. 87	17.7	0.12 (0.09-0.16)	30 (12-50)	40 (26-51)	3.4 (0.09-8.6)	<1.0	30
St Ives	Jul. 87	15.8	0.06 (0.02-0.10)	71 (22-110)	40 (34-50)	4.8 (0.7-9.2)	<1.0	37
Padstow/ Newquay	Jul. 87	16.8	0.08 (0.05-0.11)	100 (87-110)	55 (41-77)	6.0 (3.2-8.7)	<1.0	31
Milford Haven	Jul. 87	17.9	0.10 (0.09-0.12)	64 (42-93)	51 (35-66)	6.2 (3.8-9.7)	<1.0	32
Aberystwyth	Jul. 87	13.6	0.09 (0.07-0.13)	96 (76-130)	38 (26-70)	6.8 (1.3-12)	<1.0	29
Lleyn Peninsula	Jul. 87	15.0	0.09 (0.06-0.14)	82 (48-110)	36 (23-49)	13.4 (9.0-17)	<1.0	35
Parton (Whitehaven)	May 88	12.3	0.08 (0.07-0.09)	56 (39-64)	33 (29-38)	7.7 (4.3-9.7)	<1.0	40
Overall mean			0.12	68	43	6.6	<1.0	33

1.4 Conclusions

The levels of mercury, PCBs and some organochlorine pesticides in some fish species taken from Liverpool and Morecambe Bays, and cadmium in mussels taken from non-commercial populations in the Bristol Channel and Whitehaven, remain relatively high and regular monitoring will be continued. There are indications that cadmium levels in mussels from the Humber are falling but for these too monitoring will continue for the time being.

Monitoring of heavy metal levels in the edible crab, over the 1987-89 period, provides little evidence of any major change since the 1970s. As with previous surveys, the geographical pattern would indicate that contaminant levels are not generally directly related to known anthropogenic inputs.

2. SURVEYS OF CONTAMINANTS IN MARINE MAMMALS

2.1 Introduction

The 1988 seal epidemic, caused by the phocine distemper virus, resulted in the death of more than 17,000 harbour seals around the coasts of Scandinavia and Europe, (Harwood and Grenfell, 1990). In the wake of this event, attention was focused on the possible role played by contaminants in determining the severity of the outbreak. The situation remains unresolved but most experienced workers now consider the role of contaminants to have been, at most, a minor one. Nevertheless, there is evidence that a high body burden of some contaminants can adversely affect the health of marine mammals, and a greater degree of understanding of this role is desirable (ICES, 1990). In order to assess the current levels of contamination in marine mammals around the UK, and in collaboration with SOAFD (formerly DAFS)* and the NERC+ Sea Mammal Research Unit (SMRU), analyses of tissue samples (primarily blubber and liver) from seals, dolphins and porpoises have been carried out at the Burnham-on-Crouch Laboratory. The majority of these data have been published (Law *et al.*, 1989, 1991; Morris *et al.*, 1989;), but the main findings are summarised here.

2.2 Metals

Analyses for trace metals were carried out for the most part in samples of liver. The liver was chosen as the organ of study because concentrations of metals, other than cadmium, are generally higher in liver than in other organs. Higher concentrations of cadmium are found in renal tissue than in hepatic tissue of both pinnipeds and cetaceans, by a factor of 2 to 5. In all, 69 marine mammals were sampled, 42 seals, 20 porpoises and 7 dolphins. The results are summarised in Table 5. Detailed discussion of these data may be found in Law *et al.* (1991). Only in the case of mercury were levels in some individuals sufficiently high to give cause for concern. An apparent 'hot-spot' for mercury and lead was also identified in the Irish Sea. The analytical technique used for lead (flame atomic absorption spectrophotometry) yielded only 8 positive values but 7 of these, in the range 0.6 to 4.3 $\mu g\ g^{-1}$ wet weight, occurred in seals, a porpoise and a dolphin from the Wirral and the River Dee, in the region of Liverpool Bay. A lead concentration of 1.0 $\mu g\ g^{-1}$ was also found in a common dolphin from Carmarthen Bay, South Wales. The range of results for mercury was very wide, at 0.26 to 430 $\mu g\ g^{-1}$ wet weight (Table 5). The six higher concentrations (150 to 430 $\mu g\ g^{-1}$) were found in animals from the Irish Sea (Table 6).

Table 5. The range of concentrations of trace metals ($\mu g\ g^{-1}$ wet weight) found in samples of marine mammalian liver

	Seals	Dolphins	Porpoises
Cr	<0.5-1.0	ND	<0.4-0.5
Ni	ND*	ND	ND
Cu	5.2-26	4.2-12	6.5-160
Zn	25-89	26-65	25-140
Cd	<0.06-2.9	0.07-11	<0.06-1.2
Hg	0.26-430	10-44	0.6-150
Pb	<0.6-1.8	<0.6 to 3.8	<0.6-4.3

ND = Not detected in any sample
**Limit of detection generally 0.5-0.7*

Table 6. Details of the mammals exhibiting the highest concentrations of mercury ($\mu g\ g^{-1}$ wet weight) (and corresponding lead concentrations) in liver tissue

Species	Hg	Pb	Location
Grey seal	430	1.0	West Kirby*
Grey seal	370	1.4	Mostyn (R.Dee)
Grey seal	210	1.8	West Kirby
Common seal	170	<0.6	County Down#
Common seal	160	<0.6	County Down
Porpoise	150	4.3	Hoylake*

**On the Wirral*
#Northern Ireland

**Scottish Office Agriculture and Fisheries Department (formerly Department of Agriculture and Fisheries for Scotland)*
+Natural Environment Research Council

2.3 Organochlorines

Analyses for organochlorines were performed primarily on blubber samples, as high concentrations of this class of compounds may be found in this lipid-rich tissue. Analyses for polychlorinated biphenyls (PCBs) were carried out using capillary gas chromatography (Allchin *et al.*, 1989) and quantification was carried out both for a range of individual congeners, and on a formulation basis as Aroclor 1254. Experience would suggest, however, that it is extremely misleading to describe the assemblage of chlorobiphenyls seen in marine mammals on a formulation basis. Metabolic processes have so transformed the original mixtures released into the environment, during their passage through the food chain, that they render such figures almost meaningless. Their only value is for comparison with previous data sets, which do not include data for specific congeners. Detailed assessment of the results have been published by Law *et al.*(1989) and Morris *et al.*(1989). Concentrations of HCB and HCH were low and, in seals, concentrations of total DDT and PCBs were at the lower end of the range reported for seals from the North and Baltic Seas. However, concentrations higher than those expected of dieldrin, DDT and it's metabolites TDE and DDE, as well as PCBs, were found in dolphins and porpoises from Cardigan Bay, West Wales (Table 7). This occurred despite the fact that concentrations in local food-chain species were not elevated. Further collaborative work (coordinated by the Department of the Environment) is underway at SMRU, MAFF and elsewhere, to assess the hazard that these concentrations pose to coastal populations of small cetaceans.

Table 7. Concentrations of organochlorines (μg g^{-1} wet weight) in blubber of porpoises and a bottlenose dolphin calf from Cardigan Bay, West Wales. (Full details are given in Morris et al., 1989)

	D*	P#	P	P	P
HCB	0.65	0.37	0.59	0.37	0.46
ΣHCH	0.79	0.19	0.41	0.29	0.44
Dieldrin	74	2.7	9.1	7.5	9.2
ΣDDT	150	4.4	19	12	18
ΣPCB	290	24	93	45	61
Σ7CB	99	7.2	13	18	23

HCB Hexachlorobenzene.
ΣHCH α+β+γ hexachlorocyclohexane.
ΣDDT pp'-DDT + its metabolites pp'-DDE and pp'-TDE.
ΣPCB Total PCB as Aroclor 1254.
Σ7CB Sum of congeners 28, 52, 101, 118, 138, 153 and 180.

D Bottlenose dolphin (Tursiops truncatus).
P Common (harbour) porpoise (Phocoena phocoena).

**Mean of 3 determinations.*
#Mean of 2 determinations

3. MONITORING OF DISEASES IN FISH

3.1 Outline of studies on marine fish diseases

Investigations into the status of fish diseases and their possible relationships with contaminants in the marine environment have been undertaken by staff of the MAFF Fish Diseases Laboratory, Weymouth for the past 18 years (Bucke and Feist, 1990). Early studies were broad based, involving several fish species and a wide variety of disease conditions and parasites (Shelton and Wilson, 1973; Bucke *et al.* 1983). For the purposes of monitoring for the effects of pollution, results from such studies were considered to be too subjective for conclusive evidence of any relationship. Subsequently, through inter-departmental discussions and international collaboration, standardised sampling and recording procedures were adopted (ICES, 1989). Although recent studies have revealed fluctuations in disease prevalence between some areas of investigation, and even upward trends in disease levels between studies, no unequivocal evidence has been obtained for a link between diseases and pollution in UK coastal waters (Bucke and Nicholson, 1987; Bucke and Stokes, 1988; McVicar *et al.* 1988). In order to determine whether monitoring the prevalence of fish diseases is a useful indicator of biological effects, there has been a programme of research into the pathogenesis of certain disease conditions, including those occurring in livers of benthic flatfish species (Bucke *et al.*, 1984). Furthermore, laboratory studies have been instigated to investigate the health status of flatfish species exposed for long periods to highly-contaminated and less-contaminated bottom sediments (Bucke *et al.*, 1989).

3.2 Field studies

3.2.1 Objectives

The objectives of the fish disease studies undertaken by MAFF are two-fold. Firstly, by understanding the pathogenesis of fish diseases we can establish whether fish are safe for human consumption; and secondly, by monitoring the prevalence of diseases we are able to determine whether they are associated with anthropogenic activities. The approach adopted for fish disease surveys has been to select areas where anthropogenic activities occur and, for comparative purposes, to select control areas with similar hydrogeological characteristics. In both areas, the prevalence of selected

diseases is recorded for a small number of fish species. Thus, by comparing the results between areas, spatial trends can be recognised and, over a time period, temporal trends should be revealed.

3.2.2 Areas studied

For the period under report, investigations have included sewage-sludge disposal sites in the areas of the Tyne/Tees, Humber and Thames Estuaries and Liverpool Bay. Additionally, fish disease studies have been included in both the 1987 and 1988 summer groundfish surveys in the North Sea in order to provide overall background details about fish diseases. During 1988-1989, a total of 7 investigations were made on MAFF research vessels.

3.2.3 Sampling methods

Sampling methods were in accordance with the guidelines recommended by ICES (ICES, 1989). In summary, these methods involved selecting areas for sampling which had similar physical characteristics and using standard bottom-trawls for one-hour tows. The species selected for examination were flounder *(Platichthys flesus)*, dab *(Limanda limanda)* and cod *(Gadus morhua)*. The flounder and dab are benthic species; the former inhabits many estuarine and inshore coastal waters, while the latter is widespread and common in the shallow (<50 m) waters of the North Sea and Irish Sea. Both species exhibit a naturally-occurring number of easily-recognised gross lesions which are present in sufficient numbers to be statistically significant. The cod was selected because, as an important commercial species, it has over the years been the subject of disease reports by many workers (Bucke, 1988). Catches of the selected species were randomly sampled into stratified groups by length and a minimum number of fish per length group was examined for disease. Theoretically, sample sizes were based on statistical requirements (a total sample of 250 allows detection of disease prevalence of 1.5% within a 95% confidence limit). Catch weights (by species) and in the case of dab, condition factor, were recorded on samples from groundfish surveys. Table 8 presents the sample sizes for dab, flounder and cod, required by the guidelines.

For internal examination of flounder and dab it was not always possible to obtain sufficient numbers from the upper size group and samples were examined from the next size group down until the optimum number was reached.

After weighing, measuring and sexing, the dab and flounder were examined for gross lesions representing the following diseases: skin ulcers, lymphocystis and epidermal hyerplasia/papilloma. Internal examination was limited to the livers which were removed and all gross lesions present (e.g. nodules more than 2 mm in diameter) were recorded and sampled for later diagnosis by histological examination. Additionally, a total of 10 'normal' liver and spleen samples were taken from each station for histological examination. Fixation was in 10% neutral-buffered formalin. For cod, gross lesions recorded included skin ulcerations, pseudobranchial tumours and skeletal deformities. Any other unusual lesions from dab, cod and other species were subjected to histological examination.

Table 8. Sample numbers and length /size groups of dab, flounder and cod (as recommended by ICES, 1989)

Size group (cm)	Disease examination	
	External	Internal
Dab		
15-19	100	-
20-24	100	(50)
>25	50	50
Flounder		
20-24	100	-
25-29	100	(50)
>30	50	50
Cod		
<29	100	-
30-44	100	-
>45	50	-

Figures in parentheses indicate that fish from a smaller size group were used for internal examination when insufficient numbers of the larger group were present.

3.2.4 Results of field studies

(a) Outer Thames Estuary, 16-18 February, MV JODALEE 1/88

Tows were made at 3 stations in the vicinity of the Barrow Deep disposal site and a further 5 in the control area at Swire Hole (15 km distant from the disposal site). The depths at stations in Barrow Deep were from 14-29 m; those at stations in Swire Hole were from 4-18 m (Figure 2). A total of 263 flounders and 157 dabs of all sizes were examined. The prevalence of all three diseases was low (Table 9).

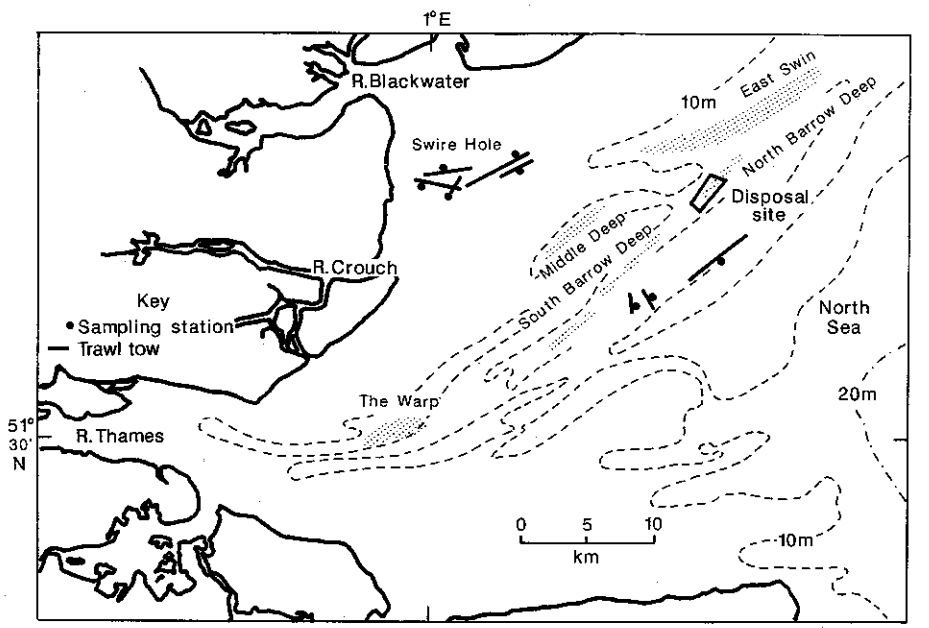


Figure 2. The distribution of stations trawled in the outer Thames Estuary in February 1988

Table 9. Summary of disease data for dab and flounder in the Thames Estuary in February 1988

(a) Total number of fish examined

	Barrow Deep	Swire Hole
Dab	92	65
Flounder	89	174

(b) Prevalence of external disease (% figures in parentheses)

		Barrow Deep		Swire Hole	
		Normal	Diseased	Normal	Diseased
Dab	Male	8	0	5	0
	Female	79	5(5.9)	59	1(1.7)
Flounder		82	7(7.9)	165	9(5.2)

(c) Disease breakdown (% figures in parentheses)

	Barrow Deep			Swire Hole		
	Lymph.	Ulcers	Hepatic nodules	Lymph.	Ulcers	Hepatic nodules
Dab	1(1.1)	4(4.3)	0	0	1(1.5)	0
Flounder	4(4.5)	3(3.4)	1(1.1)	5(2.9)	4(2.3)	0

Lymph. = Lymphocystis

(b) *North Sea groundfish survey, 12 August-10 September, RV CIROLANA 7/88*

This survey was made in conjunction with a routine groundfish survey in the North Sea (Figure. 3). Priorities limited the fish disease aim to examination of dabs from 21 stations for gross lesions, plus the examination of dab livers from one station, and screening of samples of commercial fish for gross diseases. The results are detailed in Table 10 and the details of distribution are illustrated in Figure. 4. The highest prevalences of combined external gross lesions in dab were recorded from the Firth of Forth, Moray Firth and Dogger Bank. Lymphocystis and skin ulcerations were equally present, but epidermal hyperplasia/papilloma were almost non-existent on fish from the 21 stations investigated. Twenty-five samples of liver examined histologically were from station 71 (Firth of Forth). Histological results revealed one liver showing an adenoma and 2 livers showing megalocytic changes; other livers appeared to be normal.

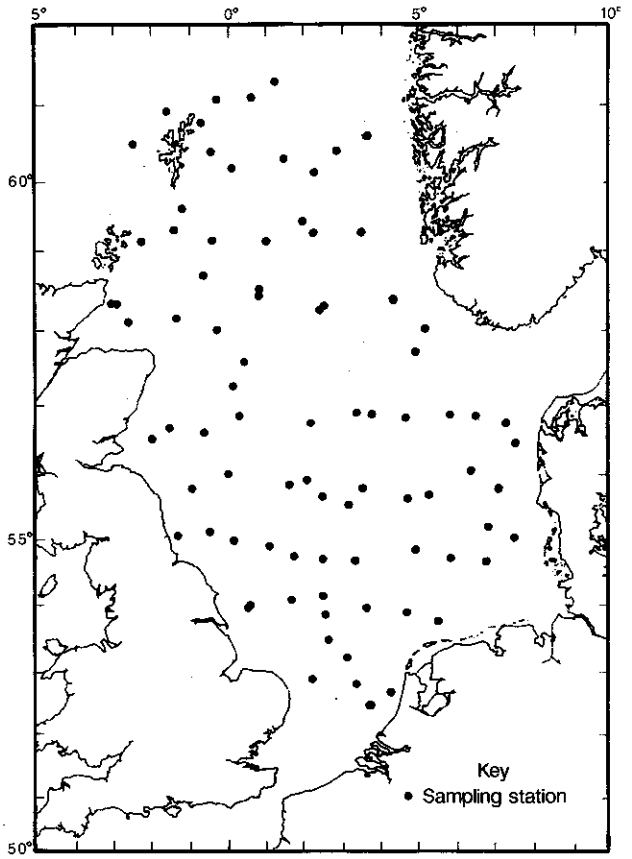


Figure 3. Standard stations trawled on MAFF's annual groundfish survey of the North Sea

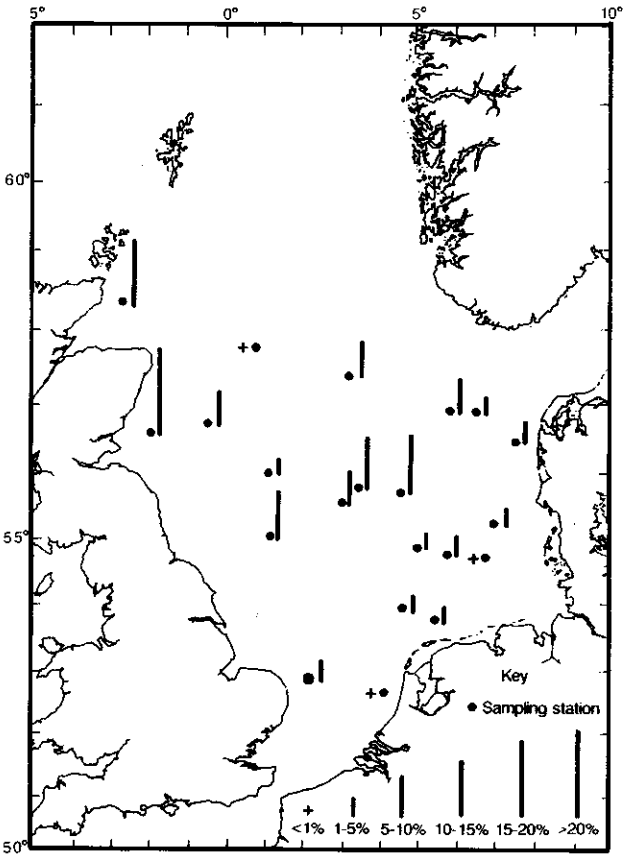


Figure 4. The prevalence (%) of disease in dab in the North Sea in August-September 1988

Table 10. Disease data for dab in the North Sea in August-September 1988

Station	Catch		Catch weight (kg)		Condition factor		Size group (cm)	Sample		Lymph.	Epi. Hyp.	Ulc.	Prevalence according to size (%)	Prevalence according to sample (%)
	M	F	M	F	M	F		M	F					
2	241	276	14.5	21.2	0.006	0.007	15-19	50	50	0	0	0	0.00	
							20-24	42	58	0	0	0	0.00	0.44
							>25	6	22	1	0	0	3.57	
13	70	136	5	13.4	0.007	0.01	15-19	39	61	0	0	0	0.00	
							20-24	20	27	1	0	1	4.26	1.15
							>25	1	26	0	0	0	0.00	
42	10133	4964	381	167.7	0.004	0.003	15-19	36	64	9	0	0	9.00	
							20-24	38	62	3	0	8	11.00	12.00
							>25	2	23	2	0	5	28.00	
52	1248	1039	41.9	44.3	0.003	0.004	15-19	50	50	5	0	3	8.00	
							20-24	14	21	4	0	7	31.43	13.4
							>25	0	14	1	0	0	7.14	

Table 10. Continued

Station	Catch		Catch weight (kg)		Condition factor		Size group (cm)	Sample		Lymph.	Epi. Hyp.	Ulc.	Prevalence according to size (%)	Prevalence according to sample(%)
	M	F	M	F	M	F		M	F					
54	1842	2255	54	86.8	0.003	0.004	15-19	50	50	4	0	3	7.00	
							20-24	14	86	4	0	7	11.0	9.13
							>25	0	30	1	0	2	10.0	
62	692	426	23.6	18.2	0.003	0.004	15-19	50	50	3	0	0	3.00	
							20-24	0	0	0	0	0	0.00	3.00
							>25	0	0	0	0	0	0.00	
71	1578	1308	124	155	0.008	0.02	15-19	50	50	8	0	7	15.00	
							20-24	58	42	9	1	17	27.00	26.66
							>25	48	52	15	0	23	38.00	
74	661	267	29.6	15	0.004	0.006	15-19	50	50	2	1	2	5.00	
							20-24	6	35	2	0	5	17.07	8.51
							>25	0	0	0	0	0	0.00	
95	126	111	4.2	5.1	0.003	0.006	15-19	50	50	0	0	0	0.00	
							20-24	0	12	1	0	0	8.33	0.89
							>25	0	0	0	0	0	0.00	
106	4036	2540	147	104.5	0.004	0.004	15-19	50	50	1	0	7	8.00	
							20-24	31	69	7	0	17	24.00	15.74
							>25	4	31	0	0	5	14.29	
217	850	1147	27	60.2	0.003	0.005	15-19	50	50	1	0	0	1.00	
							20-24	7	93	8	0	1	9.00	5.45
							>25	1	19	2	0	0	10.00	
225	749	1039	20.1	37.3	0.003	0.004	15-19	34	66	3	0	0	3.00	
							20-24	10	58	9	0	0	13.24	7.06
							>25	0	2	0	0	0	0.00	
227	512	874	17.3	42.8	0.003	0.005	15-19	34	66	1	0	1	2.00	
							20-24	18	76	6	0	0	6.38	4.20
							>25	2	19	1	0	0	4.76	
233	4161	3644	105	111.9	0.003	0.003	15-19	50	50	0	0	0	0.00	
							20-24	29	52	3	0	5	9.88	4.87
							>25	6	39	2	0	1	6.67	
235	909	1310	26.2	56.7	0.003	0.004	15-19	47	53	2	0	7	9.00	
							20-24	0	100	4	0	9	13.00	11.64
							>25	2	30	1	0	4	15.63	
245	890	2132	25.3	71.1	0.003	0.003	15-19	38	62	2	0	0	2.00	
							20-24	20	60	0	0	1	1.25	2.13
							>25	0	8	0	0	1	12.50	
252	1564	2230	49.6	78.2	0.003	0.004	15-19	64	36	0	0	0	0.00	
							20-24	50	12	1	0	0	1.61	0.62
							>25	0	0	0	0	0	0.00	
256	595	1214	23.2	59	0.004	0.005	15-19	50	50	1	0	0	1.00	
							20-24	24	76	0	0	1	1.00	1.78
							>25	3	22	0	1	1	8.00	
259	1749	2896	85.6	154.6	0.005	0.005	15-19	55	45	1	0	0	1.00	
							20-24	34	66	0	0	1	1.00	1.40
							>25	1	13	0	0	1	7.14	
265	1288	2131	53.1	93.9	0.004	0.004	15-19	50	50	0	0	1	1.00	
							20-24	27	39	0	0	0	0.00	1.10
							>25	1	14	0	0	1	6.67	
268	679	1363	22.4	44	0.003	0.003	15-19	50	50	1	0	2	3.00	
							20-24	9	29	1	0	2	7.89	4.26
							>25	0	3	0	0	0	0.00	

M = Male
F = Female
Lymph. = Lymphocystis
Epi. Hyp. = Epidermal hyperplasia
Ulc. = Ulcers

(c) *Irish Sea, 5-8 September, MV PRINCE MADOG 38/88*

Sampling was carried out at 15 stations covering 5 areas in the eastern Irish Sea. A total of 1003 dabs were examined for diseases, of which there was a 6.5% overall prevalence rate (Table 11). The disease prevalence for areas sampled is illustrated in Figure 5.

Table 11. Summary of the prevalence of disease in dab in the eastern Irish Sea in September 1988 (% figures in parentheses of combined lymphocystis, ulcers and epidermal hyperplasia)

(a) Total number of dab examined

Area	No. dabs	No. diseased
Disposal site (DS)	178	8 (4.5)
South-west of DS	110	15 (13.6)
South of DS	377*	25 (6.6)
Burbo Bight (BB)	248	10 (4.0)
Morecambe Bay (MB)	90	7(7.8)
Totals	1003	65 (6.5)

(b) Prevalence of external disease according to sex

	Female		Male	
Area	Normal	Diseased	Normal	Diseased
DS	98	1 (1.01)	72	7 (8.9)
South-west of DS	68	9 (11.7)	27	6 (18.2)
South of DS	154	15 (8.9)	85	10 (10.5)
BB	164	8 (4.7)	74	2 (2.6)
MB	70	6 (7.9)	13	1 (7.1)

*Of total sample (377), 264 only were sexed.

(d) *North Sea, 22 October-3 November, RV CORYSTES 8/88*

This study was part of a joint survey where the main aim was to study predation rates and prey selection in cod and whiting. The programme was conducted in two parts, with a survey for dab diseases on the Dogger Bank with reference stations off the Dutch coast, and a survey of cod diseases in the south-western area of the North Sea. A total of 1158 dabs were examined from 7 stations. Table 12 shows a breakdown of disease data for dabs. The prevalence of external diseases in dabs was higher at stations on the Dogger Bank (Figure 6). Liver nodules were only identified in samples from the westerly stations. Histological examination of those samples confirmed that the nodules represented hepatic adenoma. A total of 749 cod (between 30 cm and 100 cm length) were examined for disease prevalence from 38

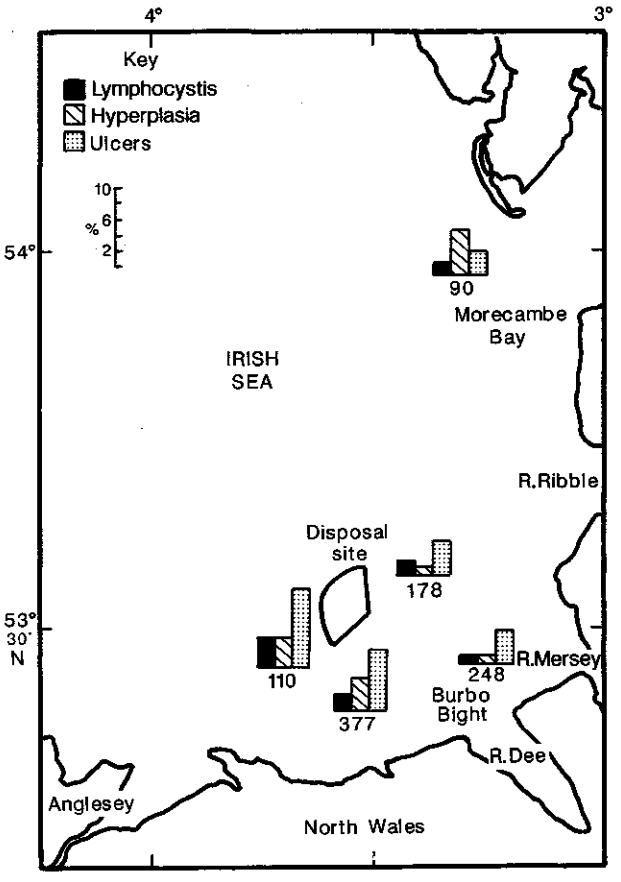


Figure 5. The prevalence (%) of disease in dab in the Irish Sea in September 1988

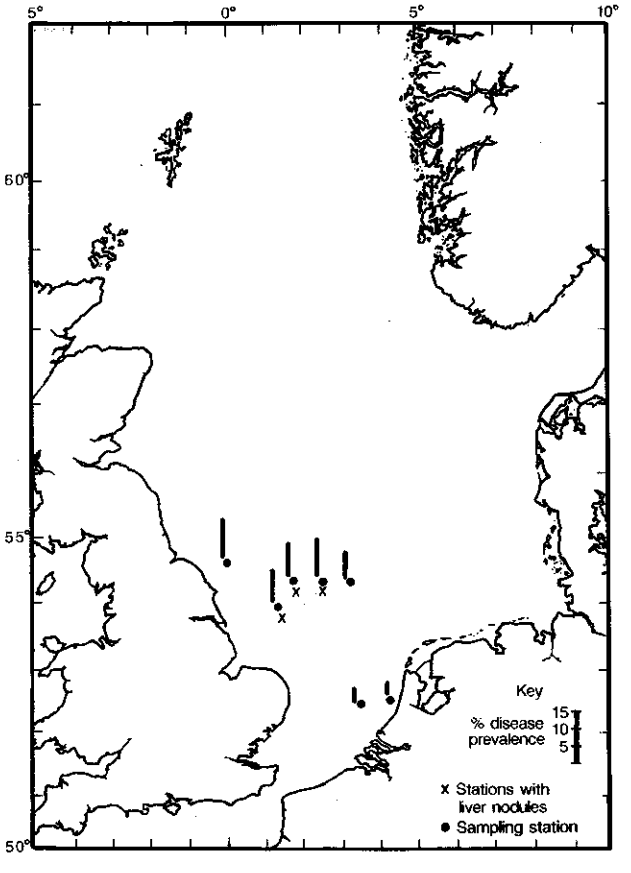


Figure 6. The prevalence (%) of disease in dab in the North Sea in October-November 1988

Table 12. Analysis of the prevalence of disease in dab in the North Sea in October-November, 1988

Station	Catch		Size group (cm)	Sample		Lympho-cystis	Epi. Hyp.	Ulcers	Prevalence according to size (%)	Prevalence according to sample(%)
	M	F		M	F					
			15-19	32	53	7	0	3	11.7	
2	62	140	20-24	30	76	7	0	3	9.4	10.4
			>25	0	11	0	0	1	9.0	
			15-19	75	28	4	3	3	9.7	
8	87	37	20-24	11	8	1	0	1	10.5	9.7
			>25	1	1	0	0	0	0.0	
			15-19	44	56	3	1	5	9.0	
9	80	118	20-24	36	47	3	3	3	10.8	9.8
			>25	0	0	0	0	0	0.0	
			15-19	25	45	0	1	4	7.1	
10	40	119	20-24	15	66	3	2	7	14.8	11.3
			>25	0	0	0	0	0	0.0	
			15-19	76	24	5	3	0	8.0	
11	113	43	20-24	27	18	1	2	1	8.9	8.2
			>25	0	1	0	0	0	0.0	
			15-19	23	28	0	0	0	0.0	
54	40	91	20-24	14	44	1	2	1	6.9	3.8
			>25	3	19	0	0	1	4.5	
			15-19	36	64	0	0	0	0.0	
57	76	145	20-24	35	54	1	2	2	5.6	3.2
			>25	5	27	1	0	1	6.3	

M = Male
F = Female
Epi. Hyp. = Epidermal hyperplasia
Note: Liver nodules > 2 mm in diameter were observed in 5 dab samples > 25 cm in length, on Stations 2(1), 9(2) and 10(2). All were confirmed by histology as being adenoma.

Table 13. Analysis of the prevalence of disease in cod in the North Sea in October-November 1988

	I	II	III	IV	V
No. of cod examined	44	460	56	164	25
No. showing nodular lesions in viscera	3 (6.8%)	39 (8.5%)	6 (10.7%)	13 (7.9%)	0
No. showing skin ulcers	1	4 (0.9%)	2	0	1
No. showing other liver abnormalities	2	9 (2.0%)	1	1	0
No. exhibiting skeletal deformities	0	1	1	0	0

Table shows the prevalence of cod diseases from 749 fish examined from 29 hauls.
Insignificant numbers of cod were caught for examination on the Humber/Dogger Bank stations.
I-V represent sampling stations illustrated in Figure 7.

stations grouped into 5 areas. Of these, 61 (8.1%) had white/brown, raised nodular lesions in either or all hearts, livers, spleens and kidneys. This condition is referred to as visceral granulomatosis (Bucke, 1989). The degree of incidence per fish varied from 1 or 2 small (2 mm) nodules to masses of nodules 1 cm in diameter. Other diseases recorded in these cod included 8 fish (1%) showing skin ulcerations, 13 fish (1.7%) with other liver anomalies, and 2 fish (0.3%) exhibiting skeletal deformities. Details of diseases for cod are shown in Table 13 and the distribution of visceral granulomatosis is illustrated in Figure 7. Visceral granulomatosis in cod has been reported previously (Van Banning, 1987). The results from this survey are similar to those results. Histological and bacterial examination of the abnormal tissues did not reveal the aetiological agent, but the lesions were typical for those associated with infectious agents.

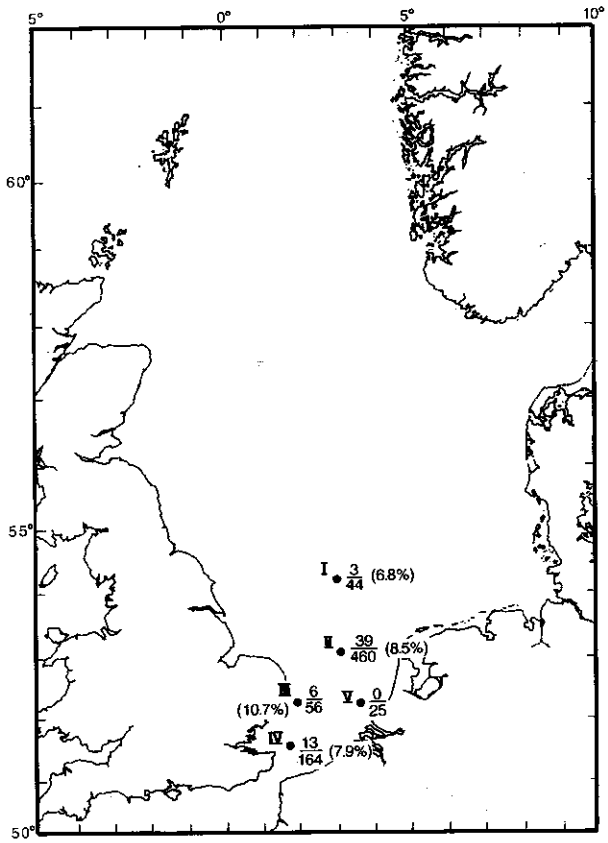


Figure 7. The prevalence (%) of visceral granulomatosis in cod in the North Sea in October-November 1988: I-V are sampling stations (see also Table 13); overall cod examined = 749; visceral granulomatosis = 61.

(e) North Sea, 25-30 April, RV CORYSTES 5/89

This cruise was instigated to examine dabs and other fish species for prevalence of disease on disposal sites and reference areas (Figure 8). A series of trawls were worked between Blyth and the Tees. The majority of these were associated with recognised disposal sites. Stations further offshore in deeper water were used as reference areas. Attempts to investigate disease prevalence on a proposed disposal site (Station 7) were unsuccessful, since the adverse ground conditions resulted in serious trawl damage. A total of 1,548 dabs were examined; 170 dabs exhibited lesions representing external gross diseases (Tables 14 and 15). The results revealed no significant differences in disease prevalence between stations. There was no macroscopical evidence in dabs > 25 cm in length, of liver nodules over 2 mm in diameter but, of the 189 livers subjected to histological examination, there were changes present associated with cell storage variability in 7 livers. A total of 514 cod were examined, but evidence of disease was restricted to one fish with a small ulcer and 4 fish with skeletal deformities. Prevalence of disease in cod was considered to be insignificant for comparative purposes, as the overall percentage of prevalence was less than 1% (Table 16).

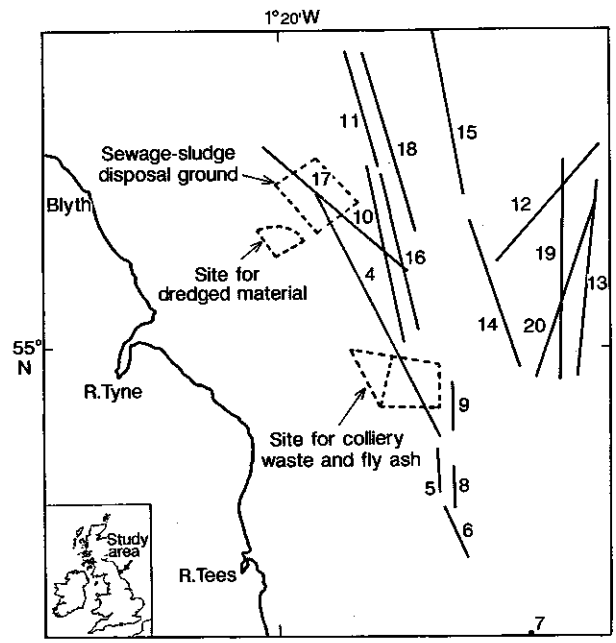


Figure 8. The distribution of trawl tows at stations sampled for fish diseases in the North Sea in April 1989: Stations 4-11 and 16-18 are in the area of the disposal site and 12-15 and 19-20 are in the reference area. (Note: Station 7 was unsuccessfully sampled)

Table 14. Analysis of the prevalence of disease in dab in the North Sea in April 1989

Station	Sample M	Sample F	Size group (cm)	Lympho-cystis	Epi. Hyp.	Ulcers	Liver nodules	Prevalence according to size (%)	Prevalence according to sample (%)
	128	105	15-19	10	8	3	0	9.0	
16-20	46	137	20-24	12	7	3	0	12.0	11.4
	0	67	> 25	4	4	4	0	17.9	
	140	117	15-19	10	7	5	0	8.6	
10-15	60	188	20-24	21	10	6	0	14.9	13.1
	5	70	> 25	6	4	7	0	22.7	
	59	69	15-19	4	0	1	0	3.9	
6-9	18	69	20-24	8	2	2	0	13.8	7.9
	0	26	> 25	2	0	0	0	7.7	
	18	27	15-19	4	1	0	0	11.1	
5	6	24	20-24	0	0	1	0	3.3	8.2
	1	9	> 25	1	0	0	0	10.0	
	45	29	15-19	0	0	2	0	2.7	
4	11	53	20-24	5	3	1	0	14.1	9.7
	0	17	> 25	2	0	1	1	23.5	

M = Male
F = Female
Epi. Hyp. = Epidermal hyperplasia

Table 15. Distribution of epidermal diseases in dab in the North Sea in April 1989

Station	15-19 cm size group				20-24 cm size group				> 25 cm size group			
	No. exam.	No. diseased			No. exam.	No. diseased			No. exam.	No. diseased		
		Lympho-cystis	Epi. Hyp.	Ulcers		Lympho-cystis	Epi. Hyp.	Ulcers		Lympho-cystis	Epi. Hyp.	Ulcers
Disposal site												
4-11 and 16-18	532	22 4.1+	9 1.7+ (7.1%)*	7 1.3+	367	31 8.4+	13 3.5+ (13.9%)*	7 1.9+	103	10 9.7+	3 2.9+ (16.5%)*	4 3.9+
Reference area												
12-15 and 19-20	205	6 2.9+	7 3.4+ (8.3%)*	4 2.0+	247	15 6.1+	9 3.6+ (12.1%)*	6 2.4+	94	5 5.3+	4 4.3+ (18.1%)*	8 8.5+

+ = Individual percentage figure
** = Combined percentage figure*
Epi. Hyp. = Epidermal hyperplasia

Table 16. Analysis of the prevalence of disease in cod in the North Sea in April 1989

Station	Total sample	Size group (cm)	Skin ulcer	Pseudobranch tumour	Skeletal deformity	Prevalence according to size (%)	Prevalence according to sample (%)
16-20	31	< 29	0	0	0	0	
	64	30-44	0	0	0	0	0
	178	> 45	0	0	0	0	
10-15	32	< 29	0	0	0	0	
	46	30-44	0	0	1	2.2	1.3
	82	> 45	1	0	0	1.2	
6, 9	3	< 29	0	0	0	0	
	6	30-44	0	0	0	0	0
	14	> 45	0	0	0	0	
5	2	< 29	0	0	1	50	
	5	30-44	0	0	0	0	8.3
	5	> 45	0	0	0	0	
4	0	< 29	0	0	0	0	
	4	30-44	0	0	1	25	4.3
	42	> 45	0	0	1	2.4	

(f) *North Sea groundfish survey, 11 August-12 September, RV CIROLANA 7/89*

This survey (Figure 9) was made in conjunction with a routine groundfish survey in the North Sea, similar to that described previously . The results are detailed in Tables 17 and 18. In summary, over 6,000 dabs and other fish species were examined from 28 stations. The highest prevalences of combined external diseases in dabs were recorded from the Firth of Forth (Figure 10). Liver nodules in dab were found to be widespread but of low prevalence on the Dogger Bank. Disease prevalence in other species was insignificant. The results from this survey revealed that the overall distribution of prevalence rates of disease in dabs were similar to those observed in studies in previous years.

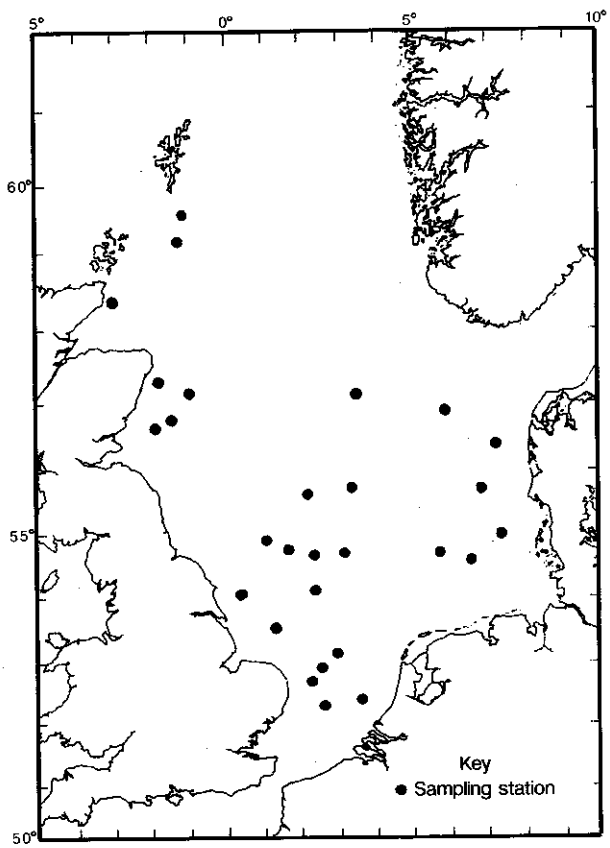


Figure 9. Stations from which dabs were examined for disease in the North Sea in August-September 1989

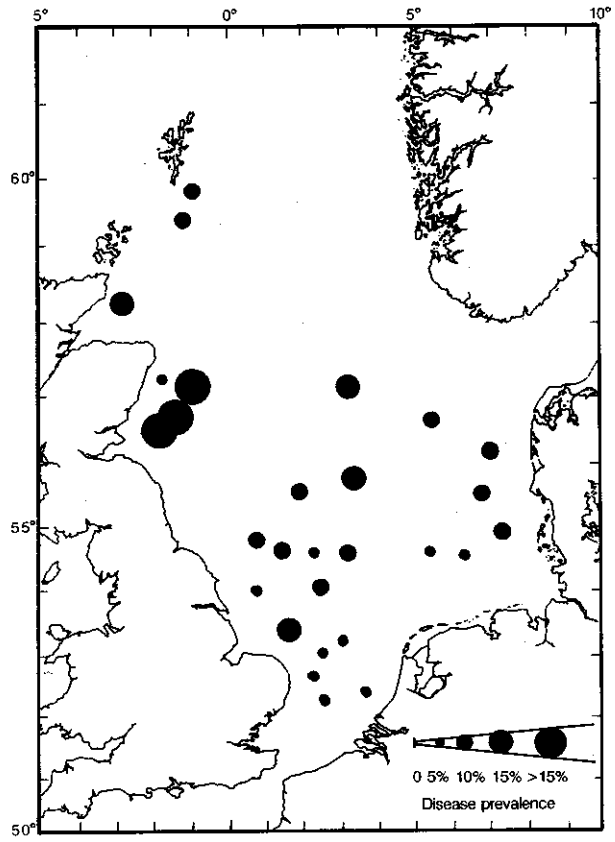


Figure 10. The prevalence (%) of disease in dabs in the North Sea in August-September 1989

Table 17. Disease data for dab taken in the North Sea groundfish survey in August-September 1989

Station	Catch		Catch weight (kg)		Condition factor		Size group (cm)	Sample		Lymph-cystis	Epi. Hyp.	Ulc.	Liver nodules	Prevalence according to size (%)	Prevalence according to sample (%)
	M	F	M	F	M	F		M	F						
1	34	104	2.7	11.8	0.010	0.010	15-19	19	28	1	0	2	0	6.4	2.3
							20-24	13	57	0	0	0	0	0	
							>25	0	14	0	0	0	0	0	
4	236	388	14.8	32	0.010	0.010	15-19	48	52	0	0	0	0	0	1.2
							20-24	27	73	0	0	2	0	2.0	
							>25	9	38	0	0	1	0	2.1	
8	76	101	6.6	12.5	0.010	0.011	15-19	35	26	0	0	0	0	0	0
							20-24	19	33	0	0	0	0	0	
							>25	6	25	0	0	0	0	0	
10	126	157	7.8	13.4	0.010	0.010	15-19	36	64	0	0	0	0	0	2.7
							20-24	17	49	0	1	2	0	4.5	
							>25	5	15	0	1	1	0	10.0	
19	947	1032	37	42.4	0.010	0.009	15-19	44	56	0	0	0	0	0	2.6
							20-24	12	25	1	0	0	0	2.7	
							>25	2	15	1	1	1	0	17.6	
23	372	524	10.8	26.8	0.009	0.009	15-19	34	42	2	0	2	0	5.3	11.4
							20-24	9	33	0	1	0	0	2.4	
							>25	3	37	2	0	3	8	32.5	

Table 17. Continued

Station	Catch		Catch weight (kg)		Condition factor		Size group (cm)	Sample		Lympho-cystis	Epi. Hyp.	Ulc.	Liver nodules	Prevalence according to size (%)	Prevalence according to sample (%)
	M	F	M	F	M	F		M	F						
35	858	584	41.5	32.1	0.010	0.010	15-19 20-24 >25	35 39 0	65 49 15	1 2 0	0 3 1	0 0 0	0 0 2	1 5.7 20	4.4
41	773	771	38.3	41.6	0.010	0.010	15-19 20-24 >25	85 10 0	76 27 3	9 0 0	7 0 0	1 2 0	0 0 0	10.6 5.4 0	9.5
43	258	234	13.5	15.1	0.009	0.009	15-19 20-24 >25	65 16 1	61 20 9	7 2 2	0 0 0	2 0 1	0 0 1	7.1 5.5 40	8.7
50	4335	2531	163	103	0.010	0.009	15-19 20-24 >25	75 7 2	42 10 42	2 1 1	1 0 0	1 1 1	0 0 0	3.4 11.7 4.3	4.6
58	4441	3001	162	119.3	0.008	0.009	15-19 20-24 >25	65 15 0	68 28 8	6 0 1	1 0 0	4 3 0	0 0 0	8.3 7 12.5	8.2
62	2689	3945	116	215.2	0.010	0.011	15-19 20-24 >25	44 33 8	67 86 92	4 2 3	1 0 3	0 2 7	0 0 0	4.5 3.4 13	6.7
63	1186	2352	47.7	126.1	0.010	0.010	15-19 20-24 >25	41 8 1	77 98 38	1 0 0	0 1 0	4 6 1	0 0 0	4.2 6.6 2.6	4.9
65	1267	1082	46.7	44.2	0.009	0.009	15-19 20-24 >25	92 6 0	95 18 5	8 1 1	2 0 0	0 0 0	0 0 1	5.3 4.2 40	6.0
70	1583	1560	71	95.2	0.010	0.010	15-19 20-24 >25	80 18 1	78 93 39	3 8 4	1 2 0	1 8 1	0 0 4	3.2 16.2 22.5	10.4
80	2121	1782	53.3	66.9	0.009	0.009	15-19 20-24 >25	34 46 10	48 107 45	1 3 2	0 1 3	1 3 1	0 0 3	2.4 4.6 16.4	6.2
82	2502	1332	118	67.9	0.010	0.010	15-19 20-24 >25	89 30 2	97 51 11	3 1 1	1 1 0	0 2 0	0 0 2	2.2 4.9 23.1	3.9
84	2097	2133	109	119.1	0.010	0.010	15-19 20-24 >25	50 88 6	50 40 45	1 0 3	0 1 0	3 4 1	0 0 4	4 3.9 15.7	6.1
91	1785	1895	42.3	61	0.009	0.009	15-19 20-24 >25	46 20 6	74 80 49	3 2 1	0 0 0	2 1 2	0 0 9	4.2 3 21.8	7.3
110	277	523	10.7	34.3	0.009	0.009	15-19 20-24 >25	50 15 1	50 85 11	2 5 0	0 1 0	2 3 0	0 0 1	4 9 8.3	6.6
164	119	67	6	4.4	0.009	0.009	15-19 20-24 >25	50 5 0	50 11 2	7 0 0	1 0 0	1 1 1	0 0 0	9 6.3 50	9.3
166	65	51	3.3	3.5	0.009	0.010	15-19 20-24 >25	50 3 0	42 10 0	7 1 0	1 0 0	1 0 0	0 0 0	9.8 7.7 0	9.5
178	2664	1994	93.1	78.9	0.009	0.009	15-19 20-24 >25	50 18 1	50 59 15	4 7 1	2 3 0	5 2 1	0 0 0	11 15.6 12.5	12.9
210	962	1591	36.5	84	0.010	0.009	15-19 20-24 >25	50 4 2	50 96 31	14 8 0	0 2 0	3 2 2	0 0 1	17 12 9.1	13.7
228	586	878	14	14	0.009	0.009	15-19 20-24 >25	50 12 0	50 88 50	6 5 1	1 1 0	10 8 6	0 0 2	17 14 18	16
230	155	292	13	14	0.009	0.009	15-19 20-24 >25	50 12 1	50 80 7	1 1 1	0 2 0	0 1 0	0 0 0	1 4 12.5	2.9
232	750	569	9	12	0.009	0.010	15-19 20-24 >25	50 30 5	50 65 16	7 6 1	0 0 0	5 7 7	0 0 3	12 13.7 52.4	16.7
234	1288	1022	10	11	0.008	0.010	15-19 20-24 >25	50 50 80	50 50 70	3 4 15	1 2 4	21 19 41	0 0 5	25 25 43.3	32.9

M = Male; F = Female; Epi. Hyp. = Epidermal hyperplasia; Ulc. = Ulcers.

Table 18. Analysis of the prevalence of disease in dab by size group, showing the range and mean percentage values in August-September, 1989

Size group (cm)	Lymphocystis		Epidermal hyperplasia		Ulcers		Liver nodules/ tumours	
	Range (%)	Mean (%)	Range (%)	Mean (%)	Range (%)	Mean (%)	Range (%)	Mean (%)
15-19	0-14.0	3.3	0-4.3	0.5	0-21.0	2.6	not examined	
20-24	0-9.1	2.9	0-3.9	0.9	0-19	3.4	not examined	
> 25	0-20.0	4.8	0-6.7	1.0	0-50	6.7	0-20.0	5.4

Table 19. Summary of the prevalence of disease in dab in the eastern Irish Sea in September 1989

(a) Total number of dab examined (% figures in parentheses are combined external diseases)

Area	No. dab	No. diseased	Liver samples	Nodules
Morecambe Bay	64	1 (1.56)	10	0
Disposal site (DS)	12	0	3	0
Off Rhyl (South of DS)	589	55 (9.33)	20	1
South-west of DS	40	0	10	1
Total	705	56 (7.9)	43	2

Figures include 12 healed ulcers; disease percentage excluding healed ulcers = 6.34

(b) Prevalence of external disease according to sex (including healed ulcers) (% figures in parentheses are combined external diseases)

Area	Female		Male	
	Normal	Diseased	Normal	Diseased
Morecambe Bay	34	0	29	1 (3.3)
Disposal site (DS)	7	0	5	0
Off Rhyl (South of DS)	307	28 (8.4)	227	27 (10.6)
South-west of DS	25	0	15	0

(c) Breakdown of disease by area (% figures in parentheses for individual diseases)

Area	Total dab	Lymphocystis	Ulcers	Healed ulcers	Hyperplasia	Hepatic nodules
Morecambe Bay	64	0	1 (1.56)	0	1 (1.56)	0
Disposal site (DS)	12	0	0	0	0	0
Off Rhyl (South of DS)	589	4 (0.68)	22 (3.74)	12 (2.04)	18 (3.06)	1 (0.17)
South-west of DS	40	0	0	0	0	1 (2.5)

(g) Eastern Irish Sea (Liverpool Bay), 11-14 September, MV PRINCE MADOG 33/89

This study was a repeat of the previous year's investigation of the prevalence of disease in dabs from a disposal site in Liverpool Bay and reference areas in Morecambe Bay. The distribution of the dab samples differed considerably from those of the previous year. A total of 705 dabs were examined and the results are presented in Table 19. As insufficient numbers of dabs were caught from the selected areas, no inter-area comparison could be made from this data. However, the overall disease prevalence (7.9%) was only slightly higher than in 1988 (6.5%).

3.3 General conclusions of the field studies

The standardised methods of sampling (recommended by ICES, 1989) for monitoring fish diseases were instigated during the period covered by this report. However, some problems were encountered in complying with the recommended methods. These mainly related to numbers of fish per age-length sample, which frequently fell below the recommended size, due to the shortage of time available at each station. Ideally, at least 3 hauls (and preferably 6 hauls) should have been made per station. This would have increased the sample numbers and allowed inter-haul variabilities to be measured. Allowing for the fact that the sample numbers were not always satisfactory, the overall conclusion is that prevalence and levels of disease were comparable with those obtained by other workers in the North Sea.

4. MONITORING OF PARALYTIC SHELLFISH POISONING TOXIN IN MUSSELS

4.1 Introduction

Paralytic shellfish poisoning (PSP) in humans is caused by the consumption of shellfish which have been feeding on certain marine dinoflagellates. These naturally-occurring dinoflagellates can increase in numbers dramatically under certain environmental conditions and, as they grow, several species produce toxic compounds. Bivalve molluscs, such as mussels, filter-feed and may accumulate these toxins. If sufficient toxin is ingested in a meal of bivalves, numbness, paralysis and possibly death can occur in humans. Because these toxins are not eliminated rapidly during cleansing in shellfish depuration tanks, or effectively denatured during cooking, the only practical measure is to prevent the consumption of shellfish during periods of toxic algal 'blooms'. As these 'blooms' do not occur every year or on the same scale, it is necessary to monitor during the likely period of bloom alerting public health authorities as toxin levels approach those which could affect humans. In Great Britain, the problem of toxic blooms has so far only been recognised on the north-east coasts of England and Scotland in the period between April and July, when the dinoflagellate *Alexandrium tamarense* can occur in large numbers.

The monitoring programme was set up by MAFF in 1968 when, as the result of a major unmonitored bloom, 78 people were taken ill after consumption of mussels. Since then, Environmental Health Officers on the north-east coast of England have collected samples of mussels at weekly intervals for analysis for PSP. In the late 1980s, regular monitoring has been carried out at 14 sites on the English coast and, in addition, at 3 sites on the east coast of Scotland. More recently, on behalf of the Department of Agriculture and Fisheries for Scotland (DAFS), additional sites on the west coast of Scotland have been included in the monitoring programme.

4.2 Methods

Once the samples arrive in the laboratory, they are washed and approximately 50 g of meat and shell liquor is weighed into a beaker. An equivalent volume of dilute HCl is added and the whole macerated. The liquor is adjusted to a pH of less than 4, boiled for 5 min and then allowed to cool, when the pH is checked again and, if necessary, re-adjusted to pH 4. After clarification by centrifugation, the supernatant is ready for use in the mouse bioassay.

Female mice of between 18-20 g are injected intraperitoneally with the extract and observed for reaction for a period of 30 min. Death time and weight are recorded for each mouse showing a reaction and a calculation is then made with a reference to published tables which give a level of toxins in mouse units per 100 g (mu/100 g). The established action level at which shellfish are considered unfit for human consumption is 400 mu/100 g.

4.3 Results

Results for the 1988 and 1989 monitoring programmes are given in Table 20(a-b). In 1988, PSP was first detected at Elie Beach in Southern Fife on 6 May, appearing further south on 13 May. By 20 May, all sites between Elie Beach and Redcar were positive for the presence of PSP and by 27 May a slight further extension beyond Redcar was recorded. Peak levels of PSP were detected at Redcar on 27 May, with 374 mouse units (i.e. the action level was not exceeded in 1988). By 10 June, PSP had disappeared everywhere except from Trow Rocks in south Tyneside, which cleared evidence of PSP by 17 June after approximately 5 weeks of intoxication.

In 1989, the effects of PSP were more dispersed geographically and blooms appeared to be more localised. Again, Elie Beach in Fife was the first site to become positive on 5 May, followed by Budle Bay and Holy Island a week later on 12 May, with Cresswell being affected on 19 May, and Trow Rocks and Redcar on 26 May. Budle Bay was clear after 9 June, and Redcar by 16 June. North of Elie Beach, Montrose was positive on 19 May, but had cleared by 2 June. In the week of 19 May, both Montrose and Budle Bay exceeded the action level with 518 and 401 mouse units, respectively. The Department of Health and Environmental Health Departments were informed accordingly.

4.4 Discussion

The areas affected by PSP, in 1988 and 1989, followed a general pattern observed in previous years in which first evidence of toxicity occurred in Fife and in the Holy Island/Budle Bay area and then moved slowly down the coast over the following 2 weeks, clearing from the Redcar area in early June. In 1989, action levels were exceeded briefly at two sites after several years in which only very low levels of PSP were detected.

It is believed that the occurrence of *A. tamarense* blooms is associated with a natural hydrographic frontal system and certain weather conditions favourable to the bloom becoming established. Therefore, the potential for a major *A. tamarense* bloom to occur, with resultant high levels for PSP toxin in shellfish, remains very real, despite the relatively low levels recorded in recent years. Because of the risks to shellfish consumers, this monitoring programme must be continued every year.

Table 20(a). Results of monitoring for PSP toxin in 1988

Station	Week ending 1 Apr	8 Apr	15 Apr	22 Apr	29 Apr	6 May	13 May	20 May	27 May
Scotland : West									
Loch Spelve	*	*	0	*	0	*	*	*	0
Cairndow	*	*	*	0	*	0	*	0	*
Loch Ryan	*	*	*	0	*	0	*	0	*
Scotland: East									
Montrose	*	*	0	*	0	*	0	*	0
Elie Beach	*	*	*	0	*	207	*	227	*
Musselburgh	*	*	*	*	0	*	0	*	0
England: North-East									
Berwick Harbour	*	*	*	*	*	*	*	*	*
Holy Island	*	*	*	*	0	*	0	0	0
Budle Bay	*	*	0	0	*	0	220	209	194
Coquet Estuary	*	0	0	0	*	0	*	231	0
Cresswell	*	*	*	0	0	0	0	255	247
E Staithes	*	0	0	0	0	0	174	226	320
Trow Rocks	*	*	*	0	*	*	209	250	338
Sunderland	*	*	*	0	*	*	344	201	217
Hartlepool	*	*	*	0	*	0	*	208	*
Redcar	0	0	0	0	*	0	0	216	374
Saltburn	0	0	0	0	0	0	0	0	211
Whitby	*	*	*	*	*	*	0	0	*
Scalby Mills	*	*	*	0	*	0	0	0	0
Cornelian Bay	*	*	*	0	*	0	*	0	0

Table 20(a). Continued

Station	Week ending: 3 Jun	10 Jun	17 Jun	24 Jun	1 Jul	8 Jul	15 Jul	22 Jul	29 Jul	5 Aug	12 Aug	19 Aug
Scotland: West												
Loch Spelve	*	0	*	*	*	0	0	*	*	*	*	0
Cairndow	*	Trace	*	*	*	*	*	*	*	*	*	*
Loch Ryan	0	*	0	*	0	*	0	*	0	*	0	*
Scotland: East												
Montrose	*	0	*	0	*	0	*	0	*	*	0	*
Elie Beach	*	0	*	0	*	0	*	0	*	*	*	*
Musselburgh	*	0	*	0	*	0	*	0	*	*	*	*
England: North-East												
Berwick Harbour	*	*	*	0	0	0	*	*	*	*	0	*
Holy Island	0	0	0	0	0	*	*	*	*	*	*	*
Budle Bay	*	*	0	*	*	0	0	*	*	*	*	*
Coquet Estuary	*	0	0	*	*	*	*	*	*	*	*	*
Cresswell	221	0	0	0	*	0	*	0	*	*	0	*
E Staithes	201	0	0	0	0	*	0	0	*	0	*	*
Trow Rocks	246	211	0	0	0	0	0	0	*	*	0	*
Sunderland	0	*	0	*	0	*	*	*	*	*	0	*
Hartlepool	*	*	0	*	*	0	*	0	*	*	*	*
Redcar	209	0	0	0	0	0	0	0	*	*	*	*
Saltburn	0	0	0	0	0	0	0	0	*	*	*	*
Whitby	*	*	0	0	*	0	*	0	*	0	*	*
Scalby Mills	*	0	0	0	0	0	*	0	*	0	*	*
Cornelian Bay	*	0	0	0	0	*	*	0	*	0	0	*

** Not tested*
0 No toxin detected
Toxin levels are expressed in mouse units per 100g

Table 20(b). Results of monitoring for PSP toxin in 1989

Station	Week ending 7 Apr	14 Apr	21 Apr	28 Apr	5 May	12 May	19 May	26 May
Scotland: West								
Dundomell	*	*	0	*	0	*	215	*
Loch Spelve	*	*	0	*	0	*	0	*
Cairndow	*	*	0	*	*	*	0	*
Loch Ryan	*	*	0	0	*	0	*	0
Scotland: East								
Montrose	*	*	0	*	*	*	518	*
Elie Beach	*	*	*	*	202	*	0	*
Musselburgh	*	*	0	*	*	*	0	*
England: North-East								
Berwick Harbour	*	*	*	0	*	*	*	0
Holy Island	*	*	*	*	*	222	0	275
Budle Bay	*	*	*	*	0	195	401	0
Coquet Estuary	*	*	0	0	*	Trace	*	0
Cresswell	*	0	0	0	*	0	209	0
E Staithes	*	*	0	0	*	0	0	0
Trow Rocks	0	0	0	0	*	0	0	250
Sunderland	*	*	*	*	*	*	*	0
Hartlepool	*	*	0	0	*	0	*	0
Redcar	0	0	0	0	0	0	0	207
Saltburn	0	0	0	0	0	0	0	0
Whitby	*	*	*	0	*	*	0	*
Scalby Mills	0	0	0	0	0	0	0	0
Cornelian Bay	0	0	0	0	*	0	0	0

Table 20(b). Continued.

Station	Week ending 2 Jun	9 Jun	16 Jun	23 Jun	30 Jun	7 Jul	14 Jul	21 Jul	28 Jul	4 Aug
Scotland: West										
Dundomell	*	0	*	0	*	*	*	*	*	*
Loch Spelve	*	*	*	*	0	*	*	*	0	*
Cairndow	*	*	*	*	*	*	*	*	*	*
Loch Ryan	*	0	*	0	*	*	*	*	*	*
Scotland: East										
Montrose	0	*	0	*	0	*	*	0	0	*
Elie Beach	*	0	*	*	0	*	0	*	0	*
Musselburgh	0	*	0	*	0	*	*	*	*	*
England: North-East										
Berwick Harbour	0@	*	0	0	0	*	0	*	0	0
Holy Island	*	0	*	*	0	*	*	*	*	0
Budle Bay	228	0	0	0	0	0	*	0	0	*
Coquet Estuary	*	0	0	*	*	*	*	*	0	*
Cresswell	*	0	0	0	*	*	*	*	*	*
E Staithes	*	0	0	0	*	*	0	*	0	0
Trow Rocks	0	0	0	0	0	0	0	*	0	0
Sunderland	*	0	*	*	*	*	*	0	*	0
Hartlepool	*	0	*	0	*	*	*	*	0	0
Redcar	275	228	0	0	0	*	*	*	*	0
Saltburn	0	0	0	0	0	*	*	*	*	0
Whitby	*	*	*	0	*	*	*	0	*	0
Scalby Mills	*	*	0	0	0	0	0	0	*	0
Cornelian Bay	*	*	*	0	0	0	0	0	*	0

** Not tested*
0 No toxin detected
@ Scallops tested on this occasion
Toxin levels are expressed in mouse units per 100g

5. USE OF THE OYSTER EMBRYO BIOASSAY TO ASSESS WATER QUALITY

5.1 Introduction

This bioassay has been used by MAFF since 1976, primarily to measure biological water quality on industrial waste and sewage-sludge disposal grounds in coastal waters around England and Wales (Lloyd and Thain, 1981; Thain and Watts, 1984; Utting and Helm, 1985; Byrne *et al.*, 1988). The main benefits of the assay are that it can be deployed at sea, it has a high sensitivity to a wide range of contaminants, and large numbers of samples can be assayed to give results within 48 h. The organismal response used is the ability of the oyster embryo to develop normally and reach the 'D' shaped larval stage within 24 h. Impaired development indicates poor water quality which may arise as a consequence of anthropogenic inputs or, in some instances, be associated with natural dinoflagellate blooms (Thain and Watts, 1984).

5.2 Methods

The bioassay is deployed on-board MAFF's research vessels at sea. Usually 2.5 l water samples are taken from an extensive sampling grid and assayed within 6 h. For each assay, oysters pre-conditioned for spawning are opened, the gametes stripped and the eggs artificially fertilised. The embryos are placed in the water samples and their development after 24 h is assessed by microscopic examination (see Thain and Watts, 1984). The embryos' development in the test water is compared to the development in a control water and the result expressed as Percent Net Response (PNR), i.e.

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

5.3 Results

The technique was utilised at a disposal site area on the north-east coast (Figure 11) during the period covered by this report. Seawater samples were taken from a depth of 3 m below the surface at each of the 47 stations shown. At the time of sampling, a liquid industrial waste was being discharged in the disposal area (944 m^3 at a rate of 92.4 $m^3\ h^{-1}$, at a speed of 6-8 knots). The results show that biological water quality

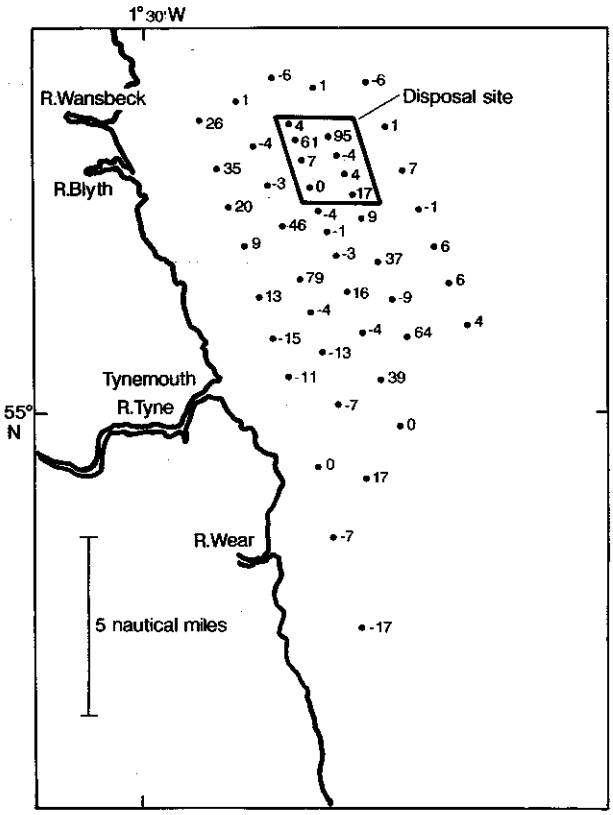


Figure 11. Oyster embryo bioassay of the north-east coast in May 1989. Percent Net Response for surface water samples: values above 21 are significantly different from the controls

was reduced at 9 of the 47 stations (sampled on 2 May - the same day that the waste was released; Figure 11). Two of these stations are in the disposal area and five are to the south of it. The remaining two stations are off the Rivers Blyth and Wansbeck and may be associated with estuarine plumes. It is considered that the bioassay was measuring the effects of the waste; conditions were very calm, the waste could be seen on the surface 4 h after disposal and water samples taken in the slick 30 min after disposal measured 100 PNR (no development). Clearly, the sea conditions were exceptional and mixing by dilution and dispersion minimal. Under normal circumstances mixing occurs rapidly (see Byrne *et al.*, 1988). In the survey described in the present report, the concentration of waste was not determined chemically. However, it is desirable that wherever possible chemical analyses of expected active components of the waste be carried out alongside biological testing.

The oyster bioassay is one of the biological effects monitoring techniques to be adopted for the North Sea Task Force Monitoring Master Plan (NSTF, 1990) and for the UK National Monitoring Plan.

6. MONITORING OF MERCURY IN SEA WATER

6.1 Introduction

Between 1985 and 1987, the distribution of the trace metals mercury, lead, cadmium and copper was established for coastal waters around England and Wales (MAFF, 1990). Offshore concentrations of all of the metals were found to be low outside the mixing zones associated with inputs from large estuaries. During 1988 and 1989, major estuaries in the UK were targeted for further research on the fate of metals, and comprehensive studies have been carried out on mercury.

In a series of 5 cruises, information was collected on the behaviour of dissolved and particulate mercury in the Tyne, Tees, Humber, Thames and Tamar estuaries and in the Bristol Channel and Liverpool Bay. In order to establish the relative concentrations of different forms of mercury present, differential extraction procedures were adopted.

6.2 Methods

The analytical methods used have been described in detail elsewhere (Harper *et al.*, 1989; Fileman *et al.*, 1990). The final detection step was the same in all cases, (cold vapour atomic absorption spectrophotometry, CVAAS); however, several operational distinctions were made to define the mercury compounds which were determined. 'Total mercury' is that measured when all mercury species in the sample are converted into a form which can be detected by CVAAS. 'Reactive mercury' is that detected without prior oxidation, i.e. ionic and weakly complexed mercury, but not the strongly complexed inorganic or organic forms. The term 'Non-reactive' describes the total mercury content minus the reactive component. The term 'Dissolved' describes that passing through a 0.7 µm glass-fibre filter, and the term 'Particulate' that which is retained.

6.3 Results and discussion

Table 21 lists the concentrations of mercury found in

Table 21. Concentrations of mercury in major UK estuaries

Location		Total (ng l^{-1})	Particulate (µg g^{-1})	Total dissolved mercury (ng l^{-1})	Dissolved reactive mercury (ng l^{-1})
Thames	Mean	29	0.63	1.2	0.75
	Range	1.4-310	0.12-2.3	0.54-2.8	0.39-1.8
	SD	54.5	0.51	0.40	0.22
Tamar	Mean	ND	0.71	0.60	0.40
	Range	ND	0.13-3.2	0.40-1.3	0.23-0.85
	SD	ND	0.67	0.22	0.16
Humber	Mean	28	ND	0.90	0.61
(1988)	Range	3.4-100	ND	0.43-1.7	0.35-1.5
	SD	24.1	ND	0.27	0.28
(1989)	Mean	8.2	ND	0.76	0.61
	Range	0.56-39	ND	0.33-1.5	0.14-1.1
	SD	9.86	ND	0.24	0.24
Tyne	Mean	7.9	ND	1.3	0.97
	Range	3.2-15	ND	0.47-3.5	0.31-3.5
	SD	3.5	ND	0.88	0.83
Tees	Mean	3.1	ND	0.68	0.52
	Range	2.1-6.0	ND	0.44-1.4	0.39-1.0
	SD	1.1	ND	0.22	0.11

ND = Not determined

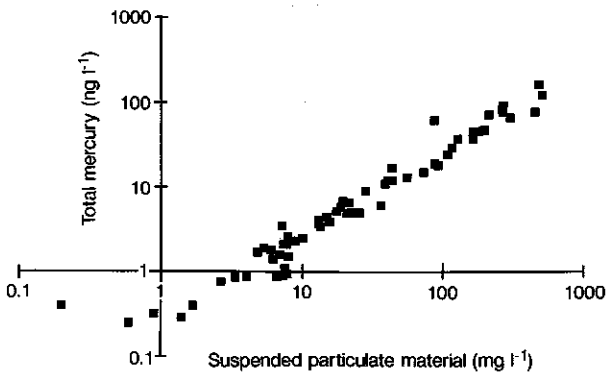


Figure 12. The relationship between concentrations of total mercury and concentrations of suspended particulates in the Bristol Channel

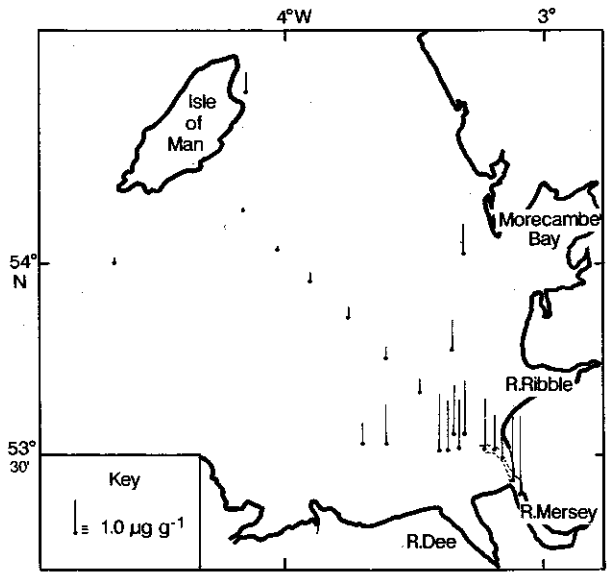


Figure 13. Concentrations of particulate mercury in Liverpool Bay and the north-east Irish Sea

the water column in the estuaries under study. The range in concentrations of total mercury was generally large, the Thames representing the most extreme case (1.4-310 ng l^{-1}). In the Thames and Tamar, the mean concentration of particulate mercury in both estuaries was similar (0.63 and 0.71 µg g^{-1} respectively). Dissolved forms of mercury were low and relatively constant for all estuaries (0.33-3.5 ng l^{-1}). Within the dissolved fraction, 'reactive' mercury comprised approximately two-thirds to three-quarters of the total. Clearly, absorption to the particulate phase regulates the concentration of mercury dissolved in the water column. There was a well-defined positive correlation (r^2=0.91) between the relationship of suspended particulate material and concentration of total mercury in the Bristol Channel (Figure 12). Concentrations of dissolved mercury in the same samples did not exceed 1.8 ng l^{-1}. High concentrations of total mercury were only associated with turbid waters. Such plumes of total mercury have been described in the past for the outer Humber (Baker, 1977), but the data collected on these more recent surveys demonstrate that the dissolved forms of mercury comprise approximately one-tenth of the total. As concentrations of suspended particulates decreased away from the estuaries, so too did total mercury concentrations. Figure 13 shows the attenuation of concentrations of particulate mercury as turbid waters of the Mersey are influenced by settlement of the particulate load, mixing with particulates with lower concentrations of mercury in Liverpool Bay and dilution with clearer water in the north-eastern Irish Sea.

The relationship of the various forms of mercury with changes in salinity were examined in the Tamar and Thames estuaries. Data on particulate mercury for the Tamar (Figure 14(a)) showed reasonably uniform concentrations throughout the whole salinity range, whereas the riverine inputs of total dissolved and dissolved reactive mercury decreased towards the seaward end of the estuary (Figure 14(b)). The rapid decreases in concentration in dissolved forms of mercury in the 0-10 practical salinity units (psu) (UNESCO, 1981) range indicate rapid partitioning of mercury to the particulates. In the Thames, however, the profiles for total mercury, particulate mercury, and total dissolved and dissolved reactive mercury were more complex. Both the total amount of mercury and

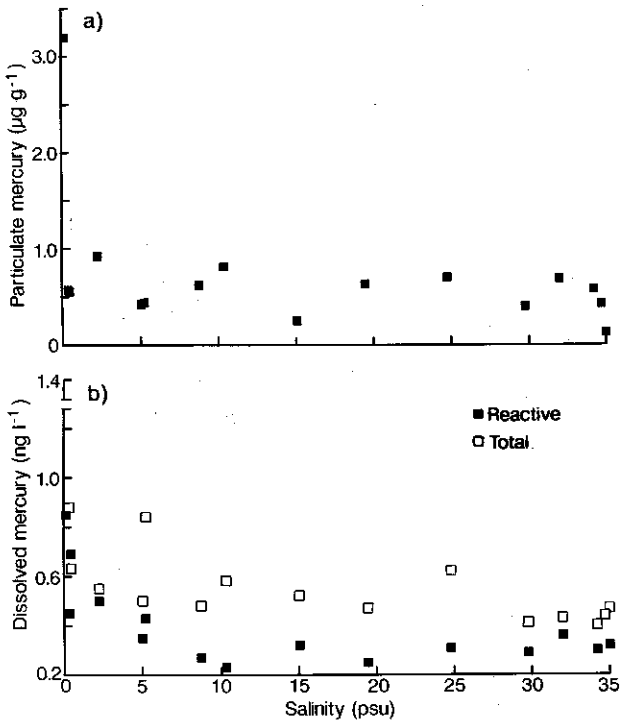


Figure 14. Changes in concentrations of: (a) particulate mercury, and (b) dissolved mercury, with increasing salinity in the Tamar

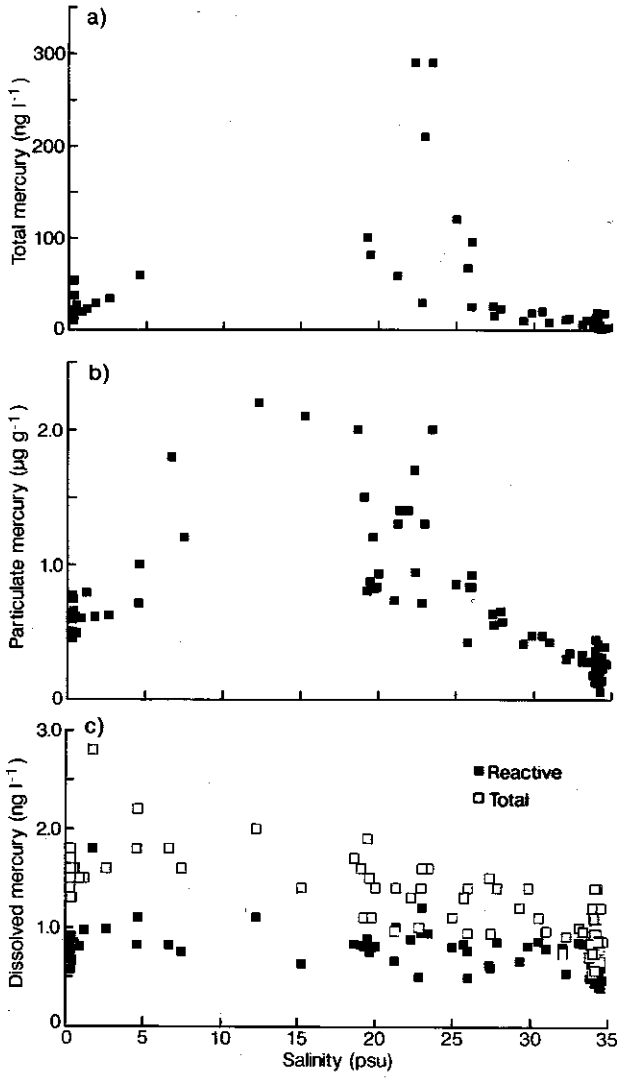


Figure 15. Changes in concentrations of: (a) total mercury, (b) particulate mercury, and (c) dissolved mercury with increasing salinity in the Thames

the concentration associated with the particulates were elevated in the mid-salinity range (5 - 25 psu) resulting from inputs from the London area (Figure 15(a) and (b)). Concentrations of total dissolved mercury hint at a decrease in concentration as salinity increases, whereas concentrations of dissolved reactive mercury remained constant (Figure 15(c)). It appears that any scavenging of the dissolved forms of mercury to particulates are offset by inputs in the mid-part of the estuary.

The environmental quality standard (EQS) set by the European Community (EC), for protection of marine and other life in coastal waters, is a concentration of 0.3 µg l^{-1} (300 ng l^{-1}) of dissolved mercury, and up to 0.5 µg l^{-1} (500 ng l^{-1}) for waters affected by the chlor-alkali industry (European Communities, 1982). Such values were not approached in any of the study areas.

7. MONITORING OF SPECIFIC ORGANIC COMPOUNDS IN SEA WATER:

(i) *XYLENE, STYRENE, CHLOROBENZENE AND PHTHALATE ESTERS*

7.1 Introduction

Studies involving the determination of concentrations of individual organic chemicals in marine waters around the UK have become more common in recent years, as techniques for the measurement of low concentrations (generally in the nanogram per litre range) have improved (Waldock, 1983; Preston and Al-Omran, 1986, 1989; Rogers *et al.*, 1989). Recent work at the Burnham-on-Crouch Laboratory has concentrated on discharges from chemical tankers, and explored the need for designation of the North Sea as a Special Area for the purposes of Annex II of the Marine Pollution Convention (MARPOL 73/78) (Hurford *et al.*, 1989a and b). These studies showed that concentrations of phthalate esters and a range of specific organic chemicals were low or undetectable in offshore waters of the North Sea, but detectable concentrations of all compounds analysed were found near the mouths of estuaries. In 1988 and 1989, further studies were conducted during research cruises aboard *RV CIROLANA*, with the aim of extending sampling into a number of estuaries in the North and Irish Seas.

7.2 Methods

Details of the sampling and analytical techniques used have been reported elsewhere (Hurford *et al.*, 1989a). Gas chromatography/mass spectrometry (GC/MS) analyses were conducted using a Finnigan Incos 50 mass spectrometer, operated in the electron-impact mode at 70 eV.

7.3 Results

The results of the analyses are given in Table 22. Figure 16(a-c) shows the positions of sampling sites in the vicinities of Tees Bay, Plymouth Sound and Liverpool Bay.

The highest concentrations of xylene, styrene and chlorobenzene were found in the River Tees (29000, 1700, and 500 ng l^{-1} respectively). Unfortunately, no comparative data have been found for these compounds. Although Rogers *et al.* (1989) have reported data for eleven chlorobenzene isomers (such as di- and tri-chlorobenzenes) in water column samples from the Firth of Forth, they have not determined monochlorobenzene itself.

Table 22. Concentrations of 'target' chemicals in sub-surface (unfiltered) water samples (ng l^{-1}). Samples 1 to 29 were collected in the period 20 May to 6 June 1988, and samples 30 to 38 between 15 November and 3 December 1989

Location	R Humber	off R Tees			Tees Bay			off R Tyne		
Sample number	1	2# (1430h)	3# (1630h)	4# (1730h)	5	6	7	8 (0630h)	9 (0730h)	10 (0930h)
Compound										
Xylene	67	54	34	11	350	3200	29000	6	75	25
Styrene	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Chlorobenzene	3	<1	<1	<1	25	320	500	<1	<1	<1
Dimethyl phthalate	11	<1	<1	<1	<1	<1	<1	<1	<1	<1
Diethyl phthalate	17	14	11	11	61	230	430	15	14	3
Di-*iso*-butyl phthalate	150	330	60	74	660	890	1100	110	89	54
Di-*n*-butyl phthalate	200	710	150	240	470	500	550	100	99	4300
Di- (2-ethylhexyl) phthalate	580	600	480	350	980	1800	2200	350	480	280

Table 22. Continued

Location	R Mersey				R Dee			DSDG						SSDG
Sample number	11	12	13	14	15	16	17	18+	19+	20	21	22 (1430h)	23 (1830h)	24
Compound														
Xylene	<1	<1	<1	<1	7	51	<1	120	140	6	330	<1	51	<1
Styrene	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Chlorobenzene	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Dimethyl phthalate	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Diethyl phthalate	24	<1	47	51	<1	10	<1	10	45	39	160	5	27	<1
Di-*iso*-butyl phthalate	97	110	99	81	67	120	90	110	86	200	220	79	210	66
Di-*n*-butyl phthalate	250	4800	550	140	210	430	240	300	140	9600	5000	200	240	140
Di- (2-ethyl-hexyl) phthalate	1500	930	390	480	190	350	340	350	300	1400	750	280	1800	290

Table 22. Continued

Location	Plymouth			Solent	Seine	R Tees	R Mersey			Plymouth				R.Thames
Sample number	25	26	27	28	29	30	31	32	33	34	35	36	37	38
Compound														
Xylene	24	38	<1	<1	<1	1700	<1	320	350	<1	<1	<1	<1	<1
Styrene	NA	NA	NA	NA	NA	1700	<1	<1	<1	<1	<1	<1	<1	<1
Chlorobenzene	<1	<1	<1	<1	<1	51	<1	<1	<1	<1	<1	<1	<1	<1
Dimethyl phthalate	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Diethyl phthalate	<1	<1	<1	<1	<1	39	<1	3	1	18	<1	<1	<1	<1
Di-*iso* -butyl phthalate	15	16	25	<10	18	75	60	38	36	49	31	20	<10	<10
Di-*n*-butyl phthalate	50	42	<50	160	160	<50	670	14	12	150	200	53	<50	<50
Di- (2-ethyl-hexyl) phthalate	99	110	280	<50	230	180	550	270	260	8400	3300	680	70	120

NA= Not analysed. +Samples 18 and 19 are duplicates. #Samples for which times are given were taken during sampling at anchor stations. DSDG = Dredged-spoil disposal ground. SSDG = Sewage-sludge disposal ground.

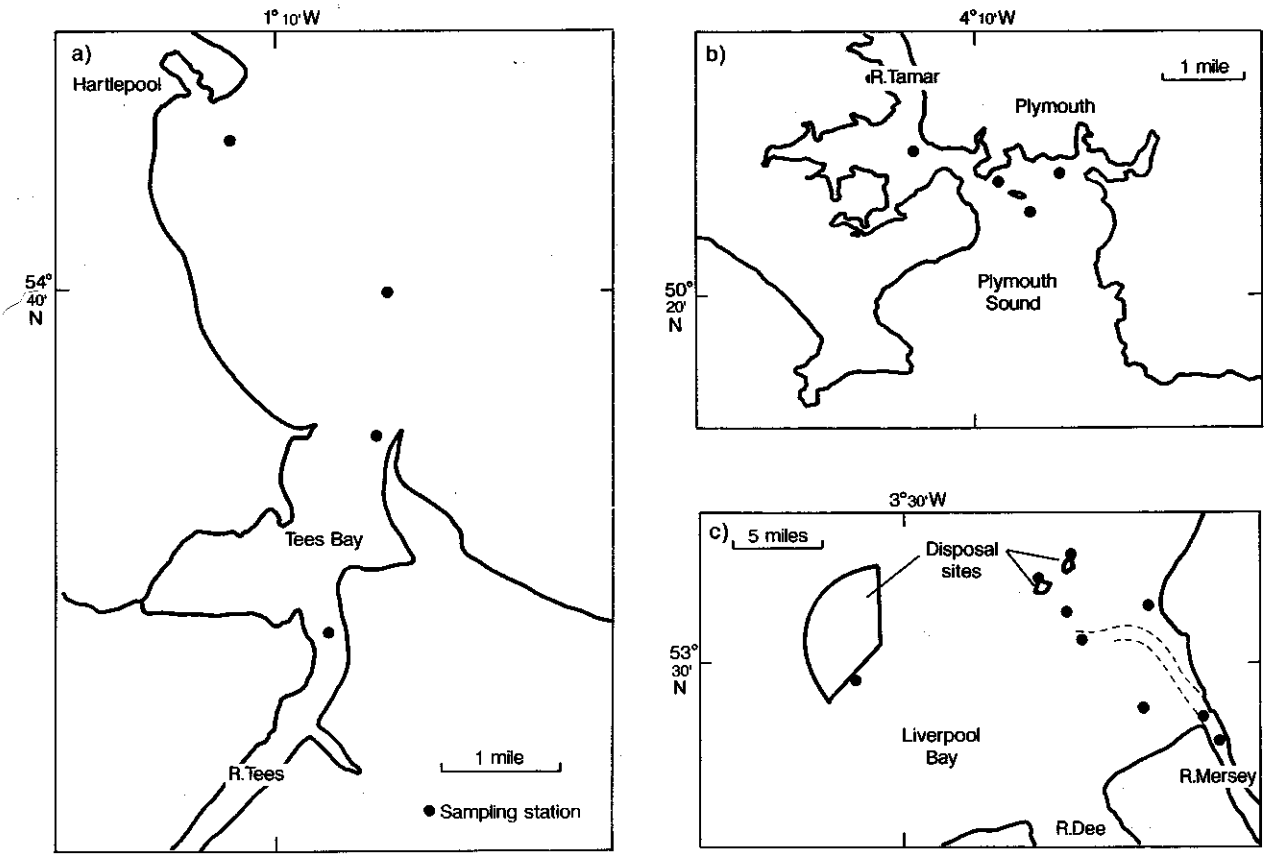


Figure 16. Sampling stations in: (a) Tees Bay; (b) Plymouth Sound; and (c) Liverpool Bay

Table 23. Comparison of the higher concentrations (ng l^{-1}) of phthalate esters found in this study with those reported in the literature

Location	DMP	DEP	DiBP	DnBP	DEHP	Ref
R. Crouch	7.3	100	30	30	60	a
R. Tyne	<1	15	110	4300	480	b
R. Tees	<1	430	1100	550	2200	b
R. Humber	11	17	150	200	580	b
R. Mersey	<1	51	110	4800	1500	b
R. Mersey	970	67	680	2100	340	c
R. Mersey	700	240	660	1800	700	d
Plymouth Sound	<1	<1	25	50	280	b
R. Meuse	NA	NA	NA	1000	4200	e
R. Rhine	NA	NA	NA	1700	1400	e
R. Rhine	200	430	NA	1200	50000	f
R. Ijssel	NA	NA	NA	2800	2600	e
Lake Ijssel	<10	80	NA	400	300	f

References:

a. *Waldock, 1983*
b. *This study*
c. *Preston and Al-Omran, 1986*
d. *Preston and Al-Omran, 1989*
e. *Schouten et al., 1979*
f. *Ritsema et al., 1989*

NA = *Not analysed.*
DMP = *Dimethyl phthalate*
DiBP = *Di-iso-butyl phthalate*
DEHP = *Di-(2-ethylhexyl) phthalate*
DEP = *Diethyl phthalate*
DnBP = *Di-n-butyl phthalate*

The range of concentrations of phthalate esters found in the estuaries sampled (Table 23) was broadly similar to those reported previously for the Rhine, Meuse and Ijssel in the Netherlands (Schouten *et al.*, 1979; Ritsema *et al.*, 1989). Figure 17 illustrates a representative chromatogram from this work.

Of the eight compounds analysed quantitatively, a reasonably comprehensive body of ecotoxicological data was found only for xylene. Estimated lower chronic 'safe' levels were only exceeded for xylene, di-*n*-butyl phthalate and di (2-ethylhexyl) phthalate in 2, 1 and 7 samples respectively of the 38 analysed. The estimated upper 'safe' levels were not exceeded in any sample. A more detailed ecotoxicological assessment of the data will be made, and published later.

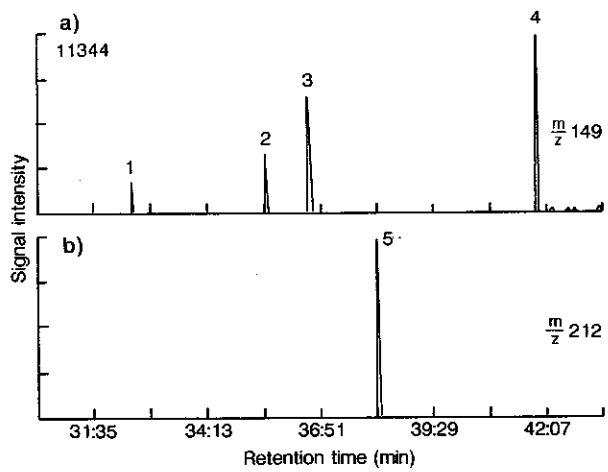


Figure 17. ***GC/MS mass chromatograms showing : (a) m/z 149 (1) diethyl phthalate, (2)di-iso-butyl phthalates, (3) di-n-butyl phthalate, (4) di-(2-ethylhexyl) phthalate; and (b) m/z 212 (5) pyrene-d_{10} internal standard***

(ii) ALPHA AND GAMMA HEXACHLOROCYCLO-HEXANE

7.4 Background

Gamma hexachlorocyclohexane (γ-HCH, lindane) is one of a series of HCH isomers, and is widely used as a pesticide in the U.K. In the past, pesticide formulations consisted of technical grade HCH which was composed of a mixture of isomers, but in recent years the majority of formulations have contained only the insecticidally active gamma isomer. Its use in a variety of agricultural and domestic applications has led to significant quantities being released into the marine environment. Lindane is one of a number of substances on the U.K. Red List (Anon., 1990) and for which the U.K. Government is committed to reduce inputs to the marine environment by 50% by the year 1992. Six tonnes of lindane were lost from the *M.V. PERINTIS* in the English Channel in the spring of 1989. Few data on concentrations of lindane in U.K. waters were available at the time and therefore the establishment of good quality baseline data was identified as a research priority.

7.5 Results

Detailed surveys of α-HCH (still commonly found from earlier technical usage of HCH and possibly as a result of photoisomerisation of the gamma isomer) and lindane have been carried out.

An account of the *PERINTIS* incident is fully documented in Section 10 and elsewhere (Law *et al.*, 1990).

Table 24. ***α HCH in sea water***

Location	n	Mean (ng l^{-1})	Range (ng l^{-1})
Humber	3	0.23	0.2-0.25
Dogger Bank	14	0.22	0.15-0.34
Thames	6	0.11	<0.1-0.13
Northern Channel	29	0.11	<0.1-0.16
W.Approaches	2	<0.1	<0.1
Channel Isles	7	<0.1	<0.1
Baie de Seine	8	<0.1	<0.1
*MV PERINTIS**	13	0.11	<0.1-0.2

**Relates to a grid of stations worked at the last known position of the MV PERINTIS*

Table 25. ***γ HCH in sea water***

Location	n	Mean (ng l^{-1})	Range (ng l^{-1})
Humber	3	0.95	0.56-1.3
Dogger Bank	14	0.53	0.3-0.73
Thames	6	0.57	0.32-0.81
Northern Channel	29	0.31	0.13-0.65
W.Approaches	2	<0.1	<0.1
Channel Isles	7	0.12	<0.1-0.22
Baie de Seine	8	0.51	0.14-0.98
*MV PERINTIS**	13	0.26	<0.1-0.81

**Relates to a grid of stations worked at the last known position of the MV PERINTIS*

Full results of the recent work on α and γ HCH (including the methodology) are also published elsewhere (Allchin, 1990) together with a detailed assessment of the results. However, a brief outline of the main findings and a summary of the data are reproduced here (Tables 24 and 25).

The levels of lindane were generally low (<1 ng l^{-1}) Concentrations tended to decrease along the east coast of England and further decrease westwards along the Channel until values fell to limits below detection. Samples from the area of the Channel, where the container from the *PERINTIS* was lost, were not significantly higher than those seen on the north side of the Channel. Samples from this area will continue to be taken periodically, as too will samples of fish and shellfish.

The levels of α HCH were also low, often at or below the limits of detection.

7.6 Conclusions

Given that the levels of lindane found were considerably below the safe level set in the U.K. of 10 ng l^{-1}, it is unlikely that similar, extensive surveys will be conducted again in the near future. In those areas isolated as being significantly above the baseline (e.g. the Dogger Bank and the Humber) further work will be carried out to verify the initial results and to establish the sources.

8. MONITORING OF ORGANOCHLORINE CONTAMINANTS IN SEDIMENTS

8.1 Introduction

The main emphasis in monitoring of organic contaminants has, in the past, been on fish and shellfish (Section 1). The use of sediments as a medium for monitoring levels of organic contaminants is however to receive greater attention under the North Sea Task Force Monitoring Master Plan (NSTF, 1990) and it is expected that these criteria will also be applied to the Irish Sea. The initial sampling programme for this major initiative is now under way and further data will be presented in subsequent reports.

Prior to the NSTF initiative, some limited work had been undertaken by the Burnham-on-Crouch Laboratory (during 1988 and 1989) in order to establish a baseline level of organic contaminants (primarily PCBs) and these limited data are presented here.

8.2 Methods

Samples of sediment were collected and analysed using published methodologies (Allchin *et al.*, 1989).Various techniques have been used by different workers in attempts to normalise any contaminant levels found; in this initial survey, however, the total sediment was examined (after passage through a 1mm sieve to remove larger stones, shells, etc) without any attempt at normalisation and results are expressed on a dry weight basis. It may well be that, in future, data will be normalised, for example to the <63 μm fraction, total organic carbon or a particular metal (such as aluminium). However, considerable analytical problems need to be resolved before this can be undertaken. Because this initial work was primarily a screening exercise, results for PCBs have been expressed on an Aroclor formulation basis. Future, more detailed, work will concentrate on those specific chlorobiphenyl congeners that are important in a marine sediment matrix.

8.3 Results

The results are presented in Table 26.

Generally, levels were low, often at or below the limits of detection (5ng g^{-1} dry weight) and most of the concentrations are considered to represent baseline levels as they now typically stand. The one exception to this was the samples taken in the vicinity of the Rame Head dredged-spoil disposal ground. Here, some elevation of PCB levels can be seen but, although some individual samples appear to have been contaminated, this effect is very localised and relates to the presence of fine dredged material overlaying the natural sediments occurring at this site.

Table 26. Concentrations (ng g^{-1}) of PCBs and pp DDT in whole sediments

Location	No. of samples	PCBs*	pp DDT
Eastern Irish Sea	5	<5	<5
Liverpool Bay	5	<5 <5-9	<5 <5-5
Cardigan Bay	5	<5	<5
Off the R.Tyne	12	5 <5-15	8 <5-16
Dogger Bank (Silver Pit)	15	5 <5-17	ND
Off Rame Head	14	44 <5-140	ND

**As Aroclor 1254*
ND = Not determined

9. MONITORING THE EFFECTS OF GAS AND OIL EXPLORATION : (i) *STUDIES IN THE RAVENSPURN GAS FIELD*

9.1 Introduction

The Ravenspurn gas field development being undertaken by British Petroleum (BP) is a major one, with plans to drill more than 40 wells using oil-based drill muds during the development phase. For this reason, it has been decided to supplement the essential surveys to be carried out by the operators as required by the Department of Energy with additional study by MAFF. Whenever possible, this is being coordinated with the statutory surveys so that comparable samples can be obtained. Discharges of cuttings derived from the use of oil-based drill muds are of interest as they represent a major source of input of hydrocarbons to the UK sector of the North Sea (Bedborough, Blackman and Law, 1987).

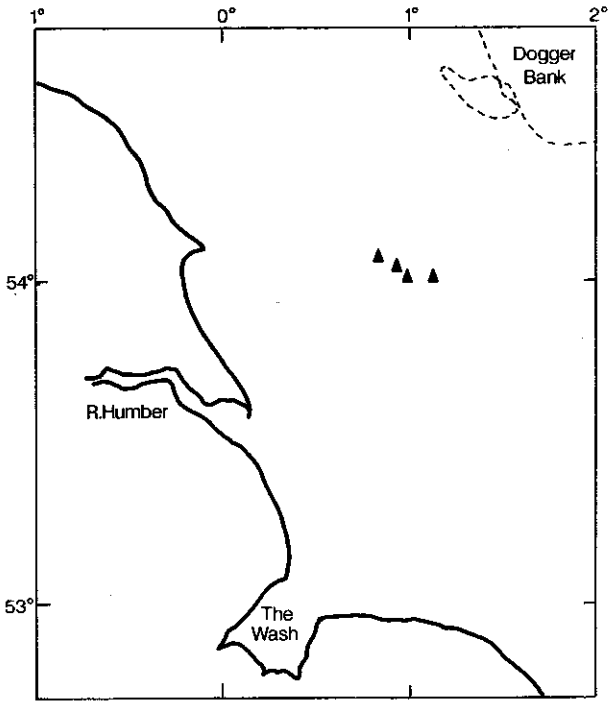


Figure 18. The location of the Ravenspurn gas field in the North Sea. The platforms marked (▲) are (L to R) Ravenspurn C, B, A and the adjacent Ravenspurn North platform operated by Hamilton Brothers (oil and gas)

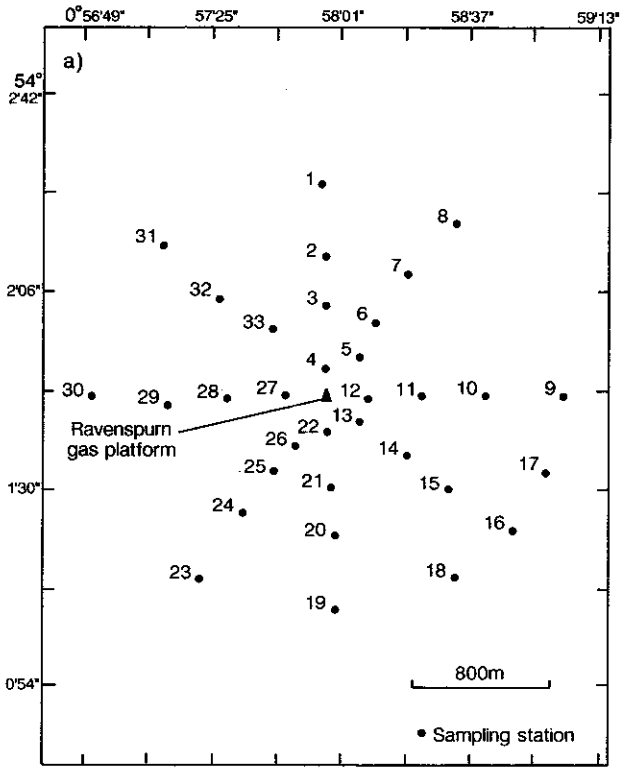


Figure 19(a). Locations of sampling stations on the cruise of NORTH SEA SURVEYOR around Ravenspurn A in March 1988

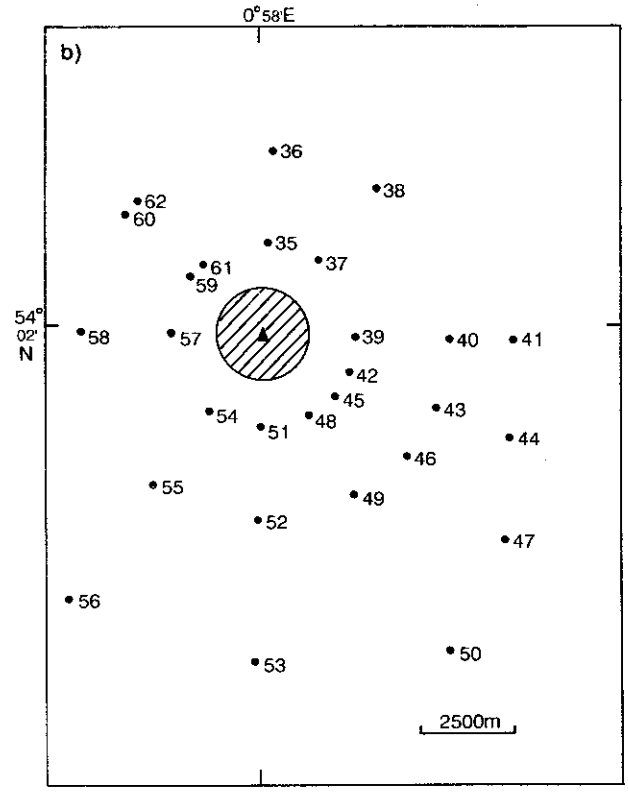


Figure 19(b). Locations of sampling stations on the cruise of RV CIROLANA around Ravenspurn A in May 1988). (Figure 19(a)) illustrates the shaded area in the centre of this grid)

The Ravenspurn gas field is situated about 40 miles north-east of the River Humber, approximately half-way between the coast and the Dogger Bank (Figure 18). The area was surveyed by BP in 1986, prior to the start of the development. This showed that the platforms are situated on a sand sheet, with a sandbank to the north of the field and sandwaves occurring in the northern half of the field. The water depth in the area is 40-60m.

Samples of surface sediments were collected in 1988 and 1989 during one MAFF research cruise and two cruises aboard vessels chartered by BP. These were *RV CIROLANA* cruise 5/88 (21 May 1988, 28 stations), the *NORTH SEA SURVEYOR* (16-19 March 1988, 33 stations) and the *PACIFIC CHAMPION* (8-9 July 1989, 28 stations) respectively. Drilling of development wells from the Ravenspurn A platform began in April 1988 and it was for this reason that sampling of the innermost stations was undertaken from a charter vessel in March, prior to the planned full survey from *CIROLANA* in May 1988. As the surveys had different aims, the grids sampled in 1988 and 1989 were very different. The pre-development surveys of 1988 covered the whole area around Ravenspurn A (Figure 19(a and b)), whilst the first post-development survey undertaken in 1989 concentrated on the south-eastern sector of the previous grid, oriented around the major current axis (Figure 19(c)).

c)

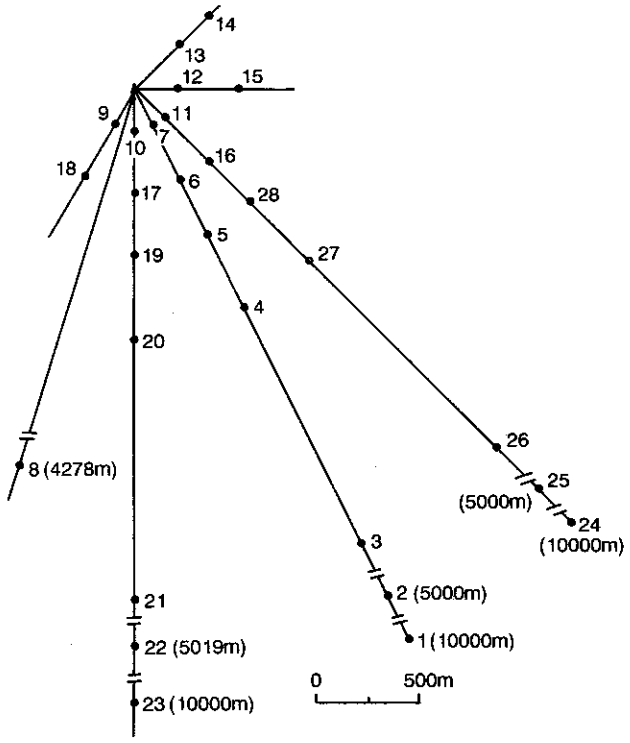


Figure 19(c). Locations of sampling stations on the cruise of PACIFIC CHAMPION around Ravenspurn A in July 1989. (Note that the indicated scale does not apply to stations 1, 2, 8, and 22 to 25, which are shown individually)

9.2 Methods

Samples of surface sediment were collected and analysed as described previously (Law *et al.*, 1988), using both fluorescence spectrometry (UVF) and capillary gas chromatography (GC). Squalane was used as the internal standard for quantification by gas chromatography, as it's retention time (42.73 min) is beyond the boiling range of the base-oil in use at Ravenspurn.

9.3 Results

The positions of the sites sampled during the 3 cruises are shown in Figure 19(a-c), and the results of analyses conducted are given in Tables 27 and 28. Table 27 lists results obtained from the samples taken during the pre-development surveys of 1988. Concentrations were generally fairly low, with a tendency for slightly higher values to be found at stations 35 to 61. The concentration range found (5.8-18 μg g^{-1} dry mass Ekofisk crude oil equivalents) is a little higher than that observed previously in an area to the North of Ravenspurn (1.0-11 μg g^{-1}; Law and Fileman, 1985). The first post-development samples taken from the *PACIFIC CHAMPION* in 1989 show only a small change in the results obtained by UVF (Table 28). Elevated values (25-44 μg g^{-1}) are seen at stations 6, 7 and 9-11.

Table 27. Total hydrocarbon concentrations (μg g^{-1} dry mass Ekofisk crude oil equivalents) in surface sediments, taken from CIROLANA cruise 5 (May 1988) and that of NORTH SEA SURVEYOR (March 1988) by UVF

Station	THC	Station	THC	Station	THC
1	6.8	22	6.6	43	18
2	9.1	23	7.2	44	10
3	7.1	24	8.6	45	13
4	6.5	25	9.4	46	11
5	7.7	26	6.2	47	11
6	13	27	7.2	48	14
7	9.4	28	11	49	12
8	12	29	7.8	50	11
9	6.6	30	7.6	51	14
10	7.0	31	6.9	52	12
11	6.1	32	7.0	53	17
12	5.8	33	9.9	54	16
13	11	34	NS	55	15
14	6.6	35	11	56	17
15	9.5	36	11	57	16
16	11	37	12	58	14
17	7.4	38	15	59	14
18	9.2	39	12	60	12
19	9.3	40	11	61	16
20	8.0	41	14	62	9.1
21	6.6	42	16		

NS = Not sampled

Table 28. Total hydrocarbon concentrations (μg g^{-1} dry mass Ekofisk crude oil equivalents) in surface sediments taken from the July 1989 cruise of PACIFIC CHAMPION by UVF/GC

Station	THC by UVF	THC by GC
1	13	6.9
2	11	9.3
3	15	23
4	10	43
5	10	280
6	30	6630
7	40	7710
8	19	13
9	25	3020
10	34	6140
11	44	18300
12	17	1580
13	16	250
14	18	65
15	14	67
16	13	495
17	17	174
18	13	109
19	13	102
20	11	25
21	12	89
22	14	2.2
23	15	2.0
24	6.0	2.4
25	11	9.2
26	14	13
27	13	76
28	13	164

UVF detects only aromatic hydrocarbons and, unlike the diesel oils and gasoils originally used in oil-based drilling fluids, modern low toxicity base oils contain very low concentrations of aromatics. Diesel oils are toxic to marine life, and it was in order to reduce the toxic effects of discharged oil on cuttings that de-aromatised 'alternative' base-oils were developed. Whilst UVF methods may be applied very usefully in general surveys of oil pollution and those directed towards polycyclic aromatic hydrocarbons (PAH), they are no longer wholly appropriate in seabed surveys around drilling installations, other than for background surveys at the pre-development stage. They would, of course, indicate contamination of sediments by crude oil and its usual refined products. Sediments contaminated by base-oils and cuttings must be analysed instead by GC.

The GC results reveal a very different picture, with the highest concentration observed being 18300 μg g^{-1} at station 11 (compare 44 μg g^{-1} by fluorescence spectrometry). A selection of gas chromatograms are shown in Figure 20, where the unresolved complex mixture (UCM) and resolved peaks seen in the trace for station 7 (Figure 20(a)) can be seen to match the boiling range profile of the base-oil (Figure 20(b)). The chromatogram for station 8 (Figure 20(c)) is much more typical of the uncontaminated situation seen in samples from the 1988 surveys, with no UCM and relatively few peaks. In general, the degree of weathering seen increases with distance from the platform. The elevation of sediment concentrations is in line with that seen at many other installations (Davies *et al.*, 1984 and 1989), with more than 1000 x background levels being seen within 500 m of a discharging platform and background concentrations being re-established some 3000 m downstream, in the direction of the major current. In the case of Ravenspurn, the major current direction is 154/334°, along the central transect in Figure 19(c). Total hydrocarbon concentrations (THCs) by GC (Table 28) show very high values >3000 μg g^{-1} to occur at 200 m from the platform (stations 7, 9, 10 and 11) and at 500 m along the major current axis (station 6). The concentrations decline rapidly with distance from the platform. Effects on benthic communities around platforms discharging cuttings derived from the use of oil-based drilling fluids are generally confined to within 2 km of the platform (Davies *et al.*, 1984), where hydrocarbon concentrations are more than 10 times greater than background values (Zone II in Davies *et al.*, 1984). Serious effects (an impoverished and highly modified benthic community, with in some cases no benthic fauna beneath and close to the platform) are limited to an area within 500m of the discharging platform (Zone I in Davies *et al.*, 1984).

Further studies will be conducted during the life of the Ravenspurn gas fields, and the results will be published in future reports in this series.

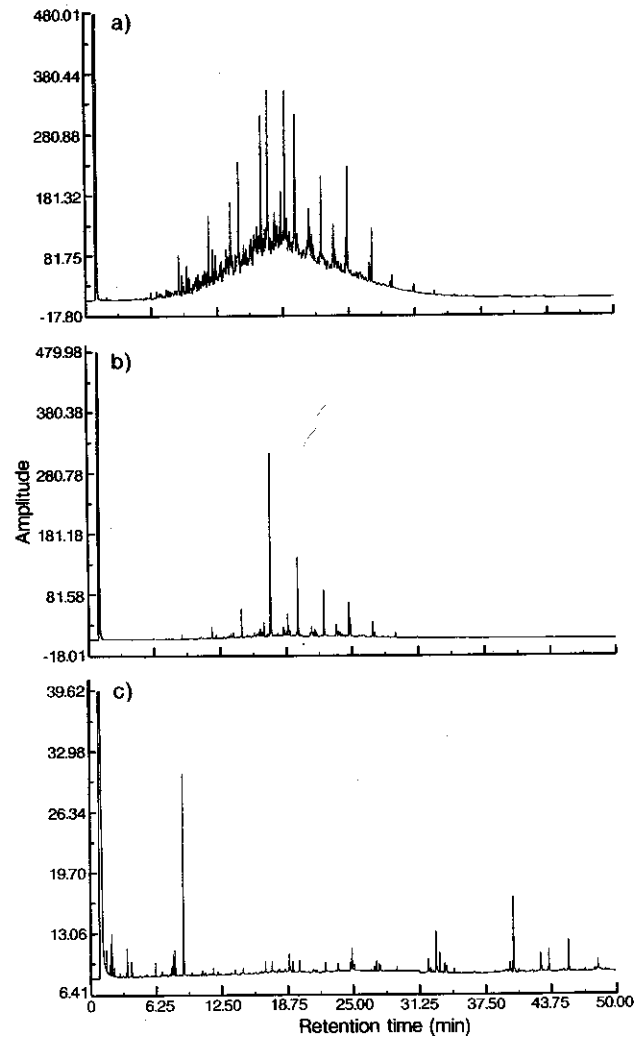


Figure 20. ***Gas chromatograms showing: (a) an extract of surface sediment from PACIFIC CHAMPION's station 7; (b) the base-oil in use at Ravenspurn A; and (c) an extract of surface sediment from PACIFIC CHAMPION's station 8***

(ii) STUDIES AROUND THE SOUTHERN GAS FIELDS

9.4 Background

The previous monitoring report in this series (MAFF, 1990) gave the results of a survey of the Leman and Thames gas fields (off N. Norfolk) carried out in 1987. This survey was intended to study the distribution of sedimentary hydrocarbons, and in particular those resulting from discharges of drill cuttings. In 1982, a survey of the wider gas field area of the Southern Basin was carried out (*RV CLIONE* cruise 9, 12-16 July 1982) and a proportion of these sites (20 of 80) were re-sampled in 1989 (*RV CIROLANA* cruise 9, 1-2 December 1989) for comparison purposes. Sampling and analysis were carried out as described previously (Law *et al.*, 1988).

Table 29. Total hydrocarbon concentrations (µg g^{-1} dry mass Ekofisk crude oil equivalents) in surface sediments

CLIONE 9/82			*CIROLANA* 9/89		
Station	Sediment type	THC	Station	Sediment type	THC
1	Sand & clay	14	214	Sand	2.1
5	Sand & shell	5.5	224	Sand	6.7
7	Clay & sand	15	223	Sand & shell	6.9
11	Moorlog & sand	87	215	Mud, sand & shell	8.3
13	Sand & anoxic mud	10	216	Sand & mud	3.7
18	Sand, mud & shell	5.4	217	Sand	7.5
23	Sand, mud & shell	7.8	218	Sand	3.5
25	Sand & shell	21	219	Sand	11
26	Sand	6.5	220	Sand	3.5
27	Sand & shell	16	222	Sand & shell	4.3
32	Sand	1.7	221	Sand	3.0
55	Sand	3.1	225	Sand	5.5
57	Sand & shell	1.6	212	Sand & shell	1.7
60	Sand & shell	1.6	210	Sand & shell	6.3
69	Sand & shell	8.4	209	Sand & shell	4.8
71	Sand, shell & stones	20	208	Sand & shell	6.5
74	Shell & sand	8.1	211	Sand & shell	6.4
77	Sand & mud	2.4	213	Sand	0.9
79	Sand & shell	1.6	207	Sand & shell	1.2
80	Sand & shell	1.6	206	Sand & shell	0.1

9.5 Results

The quantitative results obtained by fluoresence spectrometry (UVF) are given in Table 29. The total hydrocarbon concentrations (THCs) ranged from 1.6 to 87 µg g^{-1} in 1982, and from 0.1 to 11 µg g^{-1} in 1989. Only 1 value >10 µg g^{-1} was recorded in 1989, whereas 7 of 20 samples in 1982 had THCs >10 µg g^{-1}. The 5 samples with the highest THCs in 1989 (stations 215, 217, 219, 223 and 224) were also analysed by gas chromatography (GC). As modern low-toxicity base-oils used to formulate oil-based drilling fluids can have very low aromatic hydrocarbon contents, they may exhibit very little fluorescence and are better analysed by GC techniques. The gas chromatograms of these extracts did not show evidence of any base-oil or other fossil fuel inputs to the sediments, either by the presence of an unresolved complex mixture (UCM), which is a characteristic of old weathered oil, or by the presence of resolved peaks. Although not analysed quantitatively, concentrations measured by GC would be expected to be similar to those obtained by UVF. A typical example is given in Figure 21.

The sediments in this area are predominantly sandy, and so would not be expected to show high THCs. Law and Fileman (1985) analysed fine sand and sand/shell sediments from the Dogger Bank and obtained THCs of 0.4 - 2.0 µg g^{-1}, with other fine sands from the central North Sea yielding 1.0 to 11 µg g^{-1}. As would be expected, sediments containing a proportion of mud gave higher values. As was concluded following the 1987 survey reported earlier (MAFF, 1990), if there is any trend in concentration with time over this general area then it is downwards. Developmental drilling continues at many of the gas fields in this area, however, and so it is possible that localised high concentrations may occur close to certain installations drilling wells with oil-based drilling fluids (see Sub-sections 9.1-9.3).

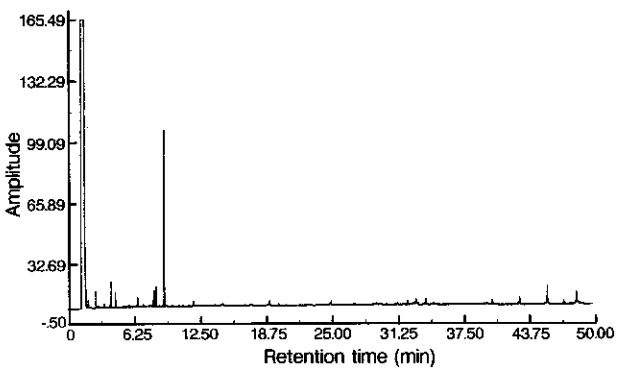


Figure 21. Gas chromatogram of the sediment extracted from station 219 on CIROLANA 9, 1-2 December 1989. Peaks prior to a retention time of about 9 min are also present in the blank, and appear prominent because of the low concentration of hydrocarbons in the sample. An internal standard (squalane) can be seen at a retention time of 42.73 min

10. THE LOSS OF THE *MV PERINTIS* IN THE ENGLISH CHANNEL IN MARCH 1989

10.1 Introduction

The *MV PERINTIS*, an Indonesian owned and Panamanian registered cargo ship, was *en route* from Antwerp to Indonesia when it sank around midday on 13 March 1989, at 49°55.4'N, 3°1.42'W, 35 miles south-east of Brixham in the English Channel. The ship carried a mixed cargo, including electrical equipment, stainless steel sheets and cigarette papers. Amongst it's general cargo was a tank containing 18.5 t of dichlorodifluoromethane, a freight container loaded with 5.8 t of lindane, plus 1 t of permethrin and 0.6 t of cypermethrin in steel drums. A copy of the cargo manifest was obtained shortly after the sinking, and a first assessment was that this incident posed a grave and imminent threat to the marine environment. The area in which the *PERINTIS* sank is actively fished, by both British and French fishermen. The British effort alone involves beam trawlers (24 at this time, from Brixham) and other boats taking plaice, sole and other demersal fish, and 30 vessels or more from Brixham, Poole and the Channel Islands potting for crab and lobsters. The value of the UK catch from this area, in March had been estimated at £800,000. When the vessel was first lost, there was a fear of a rapid release of the three insecticides to the water column in the event of their containers being ruptured, and the prohibition of sale of fish and shellfish caught within 10 miles of the wreck was actively considered as a precautionary measure, until monitoring could be undertaken to ascertain the level of contamination. No fishing in fact took place on 13-14 March because of bad weather. The area of interest is shown in Figure 22.

10.2 Stowage of cargo

All of the insecticides were packed in 50 kg batches, the lindane in cardboard drums with polythene liners and the cypermethrin and permethrin in steel drums. The 116 drums of lindane were carried in a freight container on the deck. In all, 12 freight containers were loaded as deck cargo, but the cargo manifest identified the lindane container by the serial number painted on the ends. The permethrin and cypermethrin were carried in the hold of the ship, and comprised 20 and 12 drums respectively.

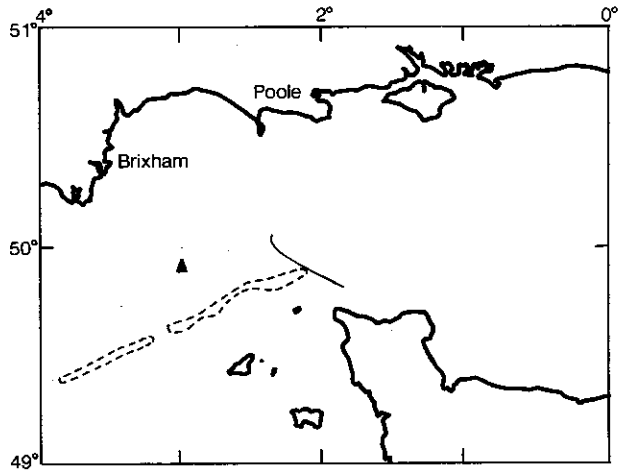


Figure 22. The central portion of the English Channel. The triangle indicates the location at which the MV PERINTIS sank. The dotted line indicates the 100 m depth contour around the Hurd Deep, and the solid line the course of the tug towing the container of lindane during the night of 15-16 March

10.3 Accumulation and toxicity of the insecticides carried

10.3.1 Lindane

The maximum solubility of lindane in water is 1 to 10 mg l^{-1}, and the safe level set in the UK is 10 ng l^{-1}. Effects on marine life may be expected at concentrations greater than 100 ng l^{-1}. From a human health standpoint, internationally agreed limits set an upper limit of 0.1 to 0.6 mg kg^{-1} for fish consumed regularly. An instantaneous release of 5.8 t could theoretically generate levels up to 10 mg kg^{-1} in fish flesh over an area of more than 300 km^2.

10.3.2 Permethrin and cypermethrin

The maximum solubility of permethrin in water is 70 µg l^{-1}; that for cypermethrin is a little less. Both of these compounds are extremely toxic to crustacea and fish and effects on marine biota may be expected to occur at concentrations of 10 ng l^{-1} and over. The Environmental Quality Standard (EQS) for protection of marine species is set at 1 ng l^{-1}. A full release could theoretically contaminate a very considerable area, up to 40 000 km^2, to 1 ng l^{-1}. Although re-population of an affected area by fin-fish could be rapid, crustaceans would take longer to recover. Thus the concern over these compounds was not for consumers of fish, but for the possible scale of damage to fishery resources.

The major uncertainties in these assessments relate to the solubility of these compounds. The maximum solubility data provided presumably relate to distilled water at 20 or 25°C, in a well stirred system. How these may relate to sea water at approximately 8°C is unclear. In addition, by 16 March, it was clear that none of the compounds had been formulated, but all were being shipped as pure crystalline compounds. This would make a large-scale, instantaneous release unlikely.

10.4 Further developments

Late in the afternoon of 16 March, the lindane container was located floating to the north of the Channel Islands. Positive identification was made, and the container was taken in tow by a naval tug *en route* to Cherbourg. Unfortunately, the tow parted sometime during the night, and the container was lost in the vicinity of the Hurd Deep, to the north-east of Alderney. It was not known whether the container sank or remained afloat after the tow was lost, but attention was focused on it's likely movement if afloat.

The near-surface water circulation in the English Channel shows a major seasonal shift, with a summer and winter circulation changing in July. Both are shown in Figure 23, which is derived from Dietrich (1950).

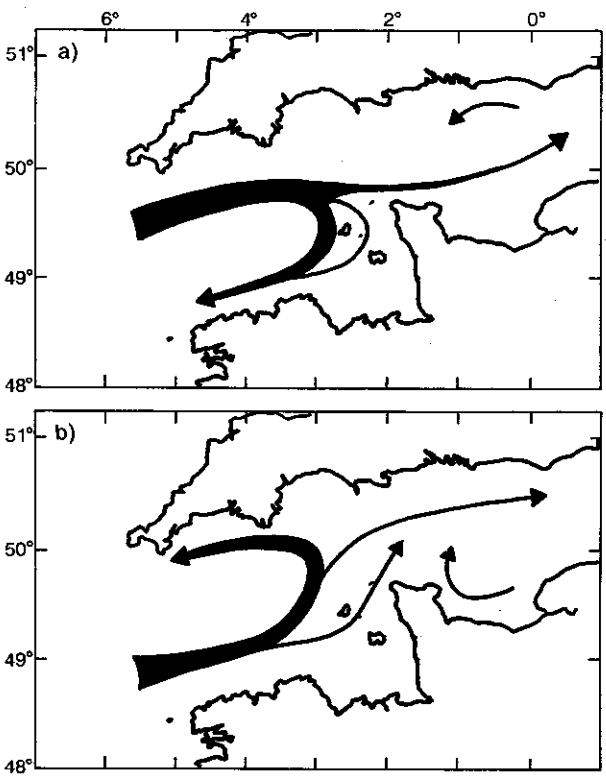


Figure 23. The near-surface water circulation pattern in the English Channel (after Dietrich, 1950): (a) winter; and (b) summer

The winter circulation pattern obtains in March. Hydrographers based at MAFF's laboratory at Lowestoft used a computerised water transport model (NORSWAP) to carry out drift simulations, with a release date in mid-March. The results of this modelling are given in Figure 24. Both the generalised circulation and the computerised water transport model suggest a likely movement to the north and east, towards the Strait of Dover, but the exact release position is critical to the outcome of the modelling of the drift , as it is possible for a parcel to be caught in a gyre to the north of the Channel Islands (track A, Figure 24) rather than be carried to the east.

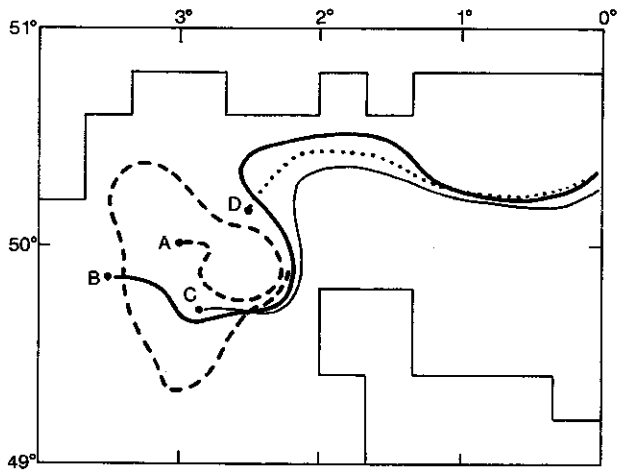


Figure 24. Results of the computerised drift modelling. Circles labelled A-D indicate different starting points. All tracks begin 16 March and represent transit times of 70 to 180 days. (Tracks shorter than 180 days are those that continue to the east of the meridian)

10.4.1 Control and monitoring

The French Government instituted a ban on anchoring, dredging and trawling in an area 12 km^2 around the presumed site of the container on 20 March. This was primarily to prevent the operation of other vessels in the area hampering the search for the container then being undertaken by the French navy.

On 21 March, the UK applied a precautionary fishing ban in an area of radius 12 km around the presumed position of the container, assumed to be 49°53.2'N 2°11'W. The ban was applied under Section 1 of the Food and Environment Protection Act, 1985, (FEPA) (Great Britain — Parliament, 1985(a)) and prohibited both fishing in the area and the landing or sale in the UK of fish caught in the prohibited area. These controls did not extend to the Channel Islands in respect of fish landed at ports other than in the UK, as Section 1 of FEPA does not apply there.

France and the UK had by this time set up a monitoring programme involving the analysis of water samples and fish and shellfish from the area. Analyses of water samples collected by UK scientists yielded concentrations of lindane of <0.1 to 4 ng l^{-1} (n=28), believed to represent the background for the area although no monitoring had been carried out previously. Sampling by UK scientists was undertaken from a chartered fishing vessel, as neither of the MAFF research vessels was available at this time (one was working in the Atlantic and the other was in dry-dock in Grimsby) and work was frequently seriously curtailed by bad weather.

Between 20 March and the end of May, French scientists, sampling from a naval vessel, collected and analysed over 400 water samples (surface and near-bottom). These yielded very variable concentrations of lindane , from <0.5 to 30 ng l^{-1}, but with no clear pattern of distribution of high values. It was concluded that these represented background values, although there may also have been some methodological bias between English and French scientists. In addition, and continuing to the present time, many samples of fish and (particularly) crustaceans have been analysed for lindane in England, France and the Channel Islands. To date, no elevations over the low background levels seen prior to the *PERINTIS* incident (0.001 mg kg^{-1} or less) have been recorded.

Given the maximum bioconcentration factor of 1000, the levels found in biota would suggest a mean concentration of lindane in water in this area of the order of 1 ng l^{-1}, which agrees well with the determined values. Further water samples were taken during MAFF's research vessel cruises in this area in late 1989 and mid-1990. The first of these yielded concentrations in surface water of <0.02 to 0.98 ng l^{-1}; analysis of the second set gave concentrations from 0.23-0.56 ng l^{-1}.

The controls imposed by both the French and British authorities remained in force until 7 April, when the search for the lindane container was abandoned, since by that time more detailed assessment of the actual risks posed to consumers had taken account of the true local conditions and likely dissolution rates rather than assumed the worst circumstances (see Sub-section 10.4.3 below).

10.4.2 Permethrin and cypermethrin

These compounds were carried in the No. 1 hold of the *PERINTIS*, and were presumed to be in or near to the wreck. Responsibility for search and recovery operations at the wreck site falls to the UK, under a geographical split of responsibility agreed previously with the French authorities. The responsible body in the UK is the Marine Pollution Control Unit (MPCU) of the Department of Transport. Poor weather delayed the start of operations, but by 29 March a vessel was on site and a remotely operated vehicle had established that the hatch covers of the ship were missing, and that some at least of the cargo was scattered on the sea bed adjacent to the wreck. Visibility at the site of the wreck was poor, sometimes as low as 0.6m. Despite these difficulties, five drums had been located on the sea bed by 31 March. A number of sonar contacts, consistent with the presence of other drums, were made by a Royal Navy minesweeper, but these could not be confirmed at the time. Discussions between MAFF and MPCU confirmed the desirability of recovering as many drums as possible from the sea bed. Although the drums would be expected to survive for at least one year in sea water, corrosion and release of the contents would inevitably occur in time. The maximum concentration reached in sea water would be dependent on the *rate* of solution, and experiments were undertaken in 'flow-through' tank systems at Burnham-on-Crouch to establish what this might be in sea water at the temperatures prevailing in the English Channel in April. When the recovery operation was terminated, on 23 April, all twelve drums of cypermethrin and sixteen of the twenty drums of permethrin had been recovered, along with the tank of dichlorodifluoromethane which had also been carried in the hold. One of the drums of permethrin recovered was transported to the Burnham Laboratory, where experiments on dissolution rate were repeated with the actual batch of cargo. The permethrin recovered from the sea bed proved to be a waxy solid with a much slower rate of dissolution than had been assumed in initial hazard assessments. Calculations suggested that exposure of the contents of one drum would result in concentrations exceeding 1 ng l^{-1} over an area of < 0.25 km^2, and that the area within which fish, shellfish or their larvae would be damaged would probably be < 0.03 km^2. These areas were considered to be insignificant, and further search and recovery of the remaining drums was considered to be unjustifiable.

10.4.3 Lindane - reassessment

A meeting of Anglo-French experts was held on 4 April in Cherbourg, at which the original assessment of the threat posed by the unrecovered lindane to marine resources and human consumers was reconsidered in the light of studies conducted in the two countries since the loss of the *PERINTIS* and of the container carrying the lindane. Their summary was as follows:

(1) analyses conducted to date showed no tangible sign of pollution by lindane;

(2) a concentration of 100 ng l^{-1} in sea water would indicate that release of the compound had occurred;

(3) the maximum bioconcentration factor for fish is 1000 times. Thus a concentration of lindane of 1 μg l^{-1} would lead to a concentration in fish flesh of about 1 mg kg^{-1}, or 1000 times higher than both the concentrations found in these studies and background studies carried out prior to the incident. Even these values would be less than those permitted by British and French authorities; and

(4) sampling should continue. If concentrations of 1 mg kg^{-1} were to be seen they would indicate contamination, but not until higher concentrations were seen (a few milligrams per kilogram) would they represent any risk to even regular consumers of fish from the affected area.

10.5 Summary and conclusions

The loss of the *MV PERINTIS* was a novel problem for authorities on both sides of the English Channel but, given the volume of shipping traffic passing along this waterway, it is unlikely to be the last. A good scheme for reporting lost cargo seems desirable with a means of locating the lost chemical containers in the event of an accident.

In the present case, cooperation between English and French authorities was generally very effective, and the recovery efforts mounted by the MPCU were certainly successful. However, it proved impossible to relocate the freight container carrying the lindane after the tow was lost. If the freight container had been fitted with an acoustic 'pinger' or transponder when loaded as deck cargo, this could have been a relatively simple task. Such transponders are fitted, for instance, on diving and rescue bells used in North Sea oil operations.

11. TECNAZENE IN THE AQUATIC ENVIRONMENT

11.1 Background

Tecnazene, 2,3,5,6-tetrachloronitrobenzene (2356-TCNB), a sprouting inhibitor and fungicide, is applied, after harvesting, to potatoes to keep them in good condition during storage. An increasing trend in the demand for washed, clean potatoes has meant that large-scale washing plants have been established to process potatoes prior to distribution to retail outlets. This practice has meant that ecotoxicologically significant quantities of tecnazene, and its associated breakdown products (2,3,5,6-tetrachloroaniline (2356-TCA), 2,3,5,6-tetrachlorothioanisole (2356-TCTA)) ,may find their way to surface waters, either by direct discharge or via sewage treatment works. Prior to this change in work practice, potatoes were generally washed by the consumer and any residual tecnazene underwent substantial dilution before entering the sewerage system.

The Burnham-on-Crouch Laboratory has been involved in laboratory investigations into the aquatic toxicity of tecnazene and its breakdown products, the results of which have already been published (Whale *et al.*, 1988). This section describes the occurrence of tecnazene and its associated breakdown compounds in the effluent, the water column and the tissue of fish from a small river local to this laboratory receiving a direct discharge from a small-scale commercial vegetable washing plant. Sampling of the effluent and water column took place periodically during 1988 and 1989. The fish were sampled in the spring of 1989. Using data supplied by the Potato Marketing Board and the then Anglian Water Authority, a number of washing plants, within an approximate 25 mile radius of the laboratory, were visited and samples of effluent taken for analysis.

Potato washing plants appear to use a variety of techniques for effluent disposal. The most common found during the course of this survey was the use of soakaways, although the largest plant investigated discharged to the public sewerage system, after on- site treatment to remove starch (produced by high-pressure steam cleaning) and sediment fines. The smallest plant had a direct discharge to a watercourse after only primary screening to remove vegetable matter. The soakaways used are of variable efficiency depending on the type of substrate. Those on less permeable ground were often overflowing into nearby ditches. Because of the often very high suspended particulate material content of the effluents, most soakaways are periodically dug-out and the (contaminated) spoil generally applied to agricultural land.

One of the smaller plants investigated was discharging its effluent, after only rudimentary treatment, direct to the River Widd near Writtle on the outskirts of Chelmsford, Essex. The river at this point supports a mixed coarse fishery, exploited by a local angling club, and seemed a suitable site for further investigations.

Samples of effluent were taken for analysis, as were samples of the water column for approximately 1km downstream. With the assistance of a fisheries team from the Anglian Water Authority, samples of chub, *Leuciscus (Squalius) cephalus* and pike, *Esox lucius* were taken, using electrofishing techniques, for residue analysis from both upstream and downstream of the point source input. The section downstream of the input of tecnazene was isolated from the upstream control site by a weir and it was thought likely that this barrier would inhibit the movement of fish upstream

and that the upstream population could therefore be considered as separate from those downstream. At the time of sampling, the only species that was represented both upstream and downstream of the discharge was chub. Pike were caught downstream and a sample of these were taken as a representative example of a top predator. Numerous other fish species were caught downstream and included a mixture of the more common cyprinid species such as roach (*Rutilus rutilus*), dace (*Leuciscus (Leuciscus) leuciscus*, and bream (*Abramis brama*); these were not taken for residue analysis due to restraints on the analytical time available.

The fishery downstream of the discharge appeared more diverse than that upstream, but this probably related to the more 'natural' look of the downstream fishery. It is generally a mixture of riffles, deeper pools and glides, and as such provides a much wider variety of habitats for coarse fish species. The upstream fishery in contrast is semi-canalised and appears to be of much more consistent depth and flow. Since the banks of the upstream fishery are also much more exposed and level, and give good access to anglers and walkers, the fish were possibly more wary of disturbance. This lack of cover may also have meant that the sampling technique used (electrofishing from a punt, without a stop net) was not ideal, and that some of the fish were 'driven' by the electrofishing team compared to downstream where most fish were caught in pools and had little chance of escape.

11.2 Results

Analysis of the effluents, water samples and fish tissues was undertaken using previously published methodologies (Allchin and Law, 1988).

The results of the fish analyses are presented in Table 30. Concentrations of 2356-TCNB, 2356-TCA and 2356-TCTA (although generally low) were higher in the chub taken from downstream of the input, with the highest concentrations being found in the more lipid rich tissues.

The sample of pike taken from downstream had generally higher concentrations of 2356-TCNB and its metabolites than were found in the chub taken from either upstream or downstream sites. This was expected and is a reflection of the position of the pike in the aquatic food chain.

Tecnazene was found in all effluents sampled. The concentrations found were very variable, and dependent on the source and how recently the potatoes had been treated with tecnazene. The concentrations of tecnazene ranged from 0.5-720 µg l^{-1} but because of the brevity of this survey it is difficult to assess if this is a normal range. Certainly, higher concentrations of tecnazene (up to 3000 µg l^{-1}) have been recorded in the effluent of a food processing factory in Cornwall (D. Whitmarsh, personal communication).

Table 30. Concentrations (mg kg^{-1} wet weight) of tecnazene and its metabolites in chub and pike

	2356 TCNB	2356 TCA	2356 TCTA
Chub (upstream) gonad	0.001-0.005	<0.001	<0.001-0.29
Mean of 4	0.002		0.1
Chub (downstream) gonad	<0.001-0.25	<0.001-0.19	<0.001-0.1
Mean of 10	0.03	0.027	0 .002
Chub (upstream) muscle	<0.001	<0.001	0.006-0.017
Mean of 6			0.013
Chub (downstream) muscle	<0.001-0.023	<0.001-0.008	<0.001-0.008
Mean of 9	0.01	0.003	0.002
Pike (downstream) liver	0.002-0.051	0.017-0.13	0.015-0.18
Mean of 5	0.028	0.087	0.11
Pike (downstream) muscle	0.007-0.025	0.006-0.1	0.002-0.005
Mean of 5	0.012	0.007	0.004
Pike (downstream) gonad	0.009-0.19	0.008-0.024	0.006-0.035
Mean of 4	0.014	0.011	0.015
Pike (downstream) gonad (surface fat)	1.8-2	0.6-0.61	1.1-1.6
Mean of 2	1.9	0.6	1.4

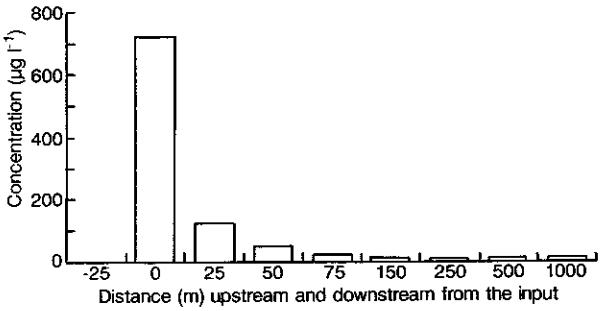


Figure 25. Concentrations of tecnazene in the River Widd in February 1988

Details of a sampling event on the River Widd are presented in Figure 25. It can be seen that the concentrations in the water column rapidly decrease downstream but are easily detected up to 1 km away from the discharge. Concentrations of tecnazene 25 m upstream of the input were below detectable limits.

11.3 Conclusions

Tecnazene and tecnazene-derived compounds are commonly found in the effluents of potato washing plants and food processors. These chemicals can find their way into the freshwater aquatic environment and, being lipophilic, are accumulated in fish tissues. Further work is being undertaken by the Department of the Environment to establish the occurrence and concentration of these chemicals in surface waters on a national basis. Tecnazine is currently being reviewed by the Advisory Committee on Pesticides and a number of data requirements have been set. To maintain their approval status, the suppliers of tecnazene will have to demonstrate *inter alia* that discharges of tecnazene are not causing deleterious effects to aquatic life.

12. TBT AND THE MARINE ENVIRONMENT : RECOVERY FOLLOWING LEGISLATION IN THE UK

12.1 Background

Following an extensive research and monitoring programme by MAFF into the effects of tributyltin (TBT)-based antifouling paints on marine organisms, an unequivocal link has been established between poor growth and reproduction in several molluscan species and low concentrations of TBT (< 100 ng l^{-1}), (Waldock and Thain, 1983; Thain and Waldock, 1986; Waldock, 1986). At higher concentrations of TBT, effects on a variety of other aquatic phyla have been demonstrated. In estuaries containing popular yachting centres, oyster culture has been adversely affected (Thain *et al.*, 1987) and common dogwhelks are no longer reproductively viable (Bryan *et al.*, 1986).

In response to the accumulated scientific evidence, the UK Government introduced a series of measures in 1986 and 1987 to control the sale and use of TBT. Restrictions on the sale of TBT-based products were phased in under the Control of Pollution Act (1974) (Great Britain — Parliament, 1974(a)), and the Food and Environment Protection Act (FEPA) (1985) (Great Britain — Parliament, 1985(a)). At present, under the provisions of the Control of Pesticides Regulations within FEPA, TBT can no longer be used in antifouling formulations on fish nets or on boats of less than 25m.

In order to monitor the efficacy of the control measures, the Department of the Environment and MAFF have sponsored further programmes of research and monitoring. Within these programmes, concentrations of TBT in water, oyster and mussel tissues and sediments have been measured in samples taken from several estuarine sites in the UK.

Other topics for research include studies on the rate of breakdown of TBT in contaminated estuarine sediments, the partitioning behaviour of sediment-bound and dissolved TBT and the potential availability of the reservoir of TBT in sediments to marine species.

12.2 Results

Preliminary results from the studies in 1988 and 1989 suggest that some recovery has taken place (Table 31). Data from the Crouch estuary are also shown in Figure 26(a). Concentrations of TBT in water at yachting centres dropped to at least one half of the corresponding 1987 figures. In some areas, and particularly within marinas, the reduction was even more marked, and in 1989 values of one third of the concentrations found in 1987 were measured.

Table 32 shows that concentrations of TBT in oysters held at the same sites showed a corresponding decrease, and there was a marked improvement in growth of young spat in 1988 and 1989, (Figure 26(b-c)). Such improvements in growth performance of Pacific oysters (*Crassostrea gigas*) led to the re-opening of an oyster cultivation business on the River Blackwater in Essex.

One indication of exposure to TBT is abnormal thickening of the shells of Pacific oysters. The length of the upper shell valve, divided by the thickness, provides an index which can be used to compare the growth performance of transplanted juvenile oysters. Normal oysters generally have thickness indices of >10; values of <5 indicate grossly malformed individuals. The growth of much thinner-shelled animals in 1988 and 1989 (Figure 26(d)) was consistent with the lower concentrations of TBT in the water.

Table 31. Mean summer (May to September) concentrations (ng l^{-1}) of tributyltin in the water of UK estuaries, marinas and harbours, 1986-1989

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

**Also a shellfish site*
[a] 263 ± 130 without June value; [b] 8 ± 2 without August value; [c] mean = 728 without May value
SD only given where n ≥ 4.

Table 32. Mean summer (June to August) concentrations (µg g^{-1} wet weight) of tributyltin in Crassostrea gigas, 1986-1989

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

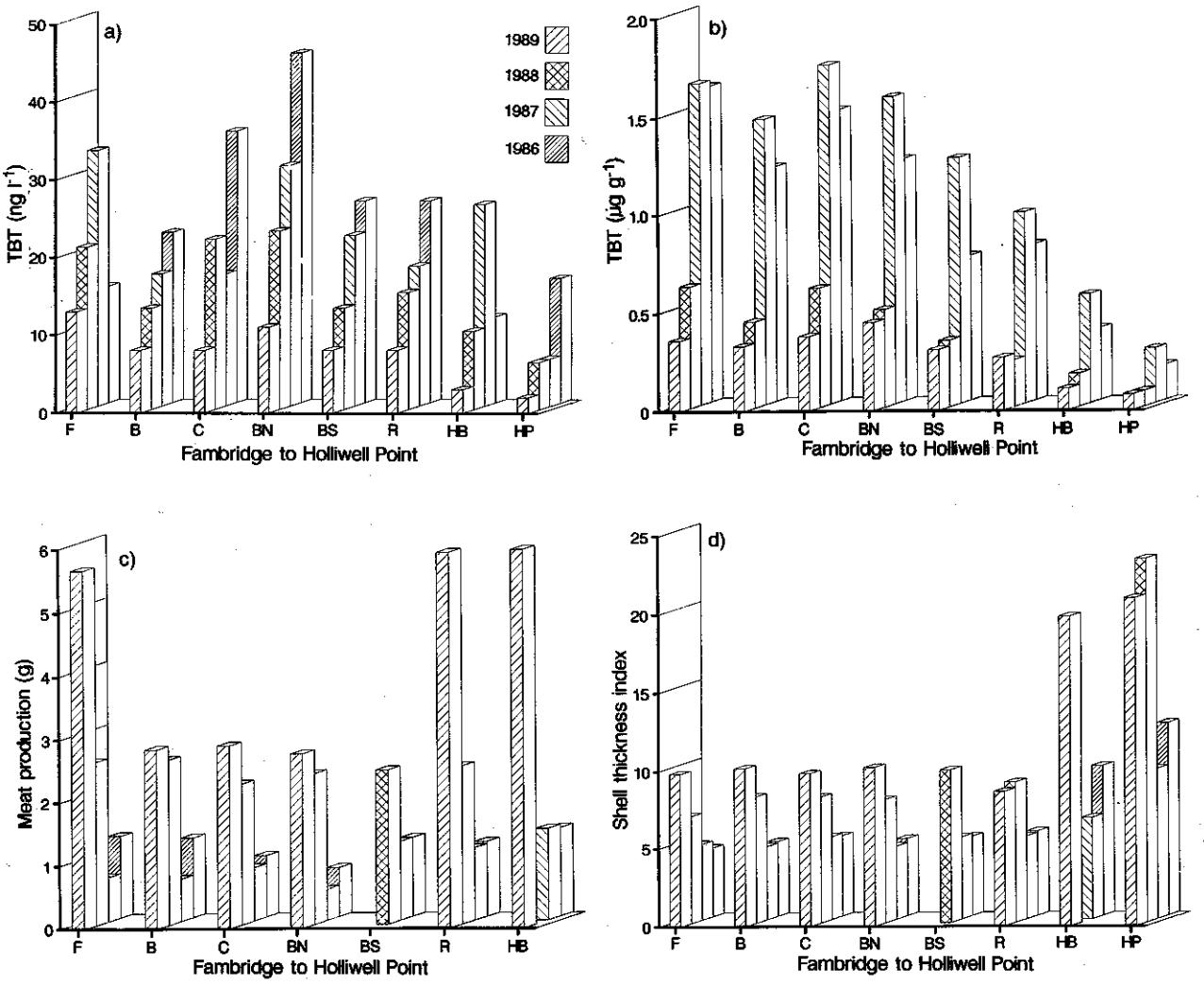


Figure 26. ***Concentrations of TBT in water and oysters, and performance of oyster growth in the Crouch estuary from 1986 to 1989: (a) mean concentrations (ng l^{-1}) of TBT in water, May to September; (b) mean concentrations (µg g^{-1}) of TBT in Crassostrea gigas tissue, June to August; (c) meat production (g) in young oysters, in August; and (d) shell thickness index of young oysters in August. (F = Fambridge; B = Bridgemarsh Island; C = Creeksea; Bn = Burnham-on-Crouch; BS = Bush Shore; R = Roach mouth; HB = Holliwell Buoy; HP =Holliwell Point. Fambridge is at the head of the estuary and Holliwell Point is at the mouth. Sites are approximately 3 km apart)***

Table 33 shows concentrations of TBT in surface sediments taken from the same sites. The inclusions of paint particles, in some samples taken close to marinas, caused difficulty in comparing changes in concentrations with time. However, laboratory-based studies on degradation of TBT in sediments showed that, in surface layers, it took less than a year for concentrations to reduce by half. In deeper anoxic layers, TBT degraded more slowly and a half-life value had not been achieved after 500 days. Bioavailability studies showed that some marine species, particularly clams, were able to ingest and accumulate TBT from contaminated muds, indicating that sediment-bound TBT cannot be regarded as being unavailable to many marine species.

12.3 Conclusions

Although the current signs are that concentrations of TBT in water and animal tissues are decreasing and a general lowering of concentrations in surface sediments is occurring, it is likely to be many years before the most sensitive species fully recover. More detailed reports of results from the research and monitoring programme have been published recently (Thain *et al.*, 1990; Waldock *et al.*, 1990; Waite *et al.*, in press).

Table 33. Summer concentrations (μg g $^{-1}$ dry weight) of tributyltin in sediments, 1986-1989

Estuary	Site	1986	1987	1988	1989
Crouch	Fambridge	0.08	0.03	0.02	0.10
	Bridgemarsh	0.15	0.04	0.08	0.05
	Creeksea	0.04	0.11[a]	0.07	0.04
	Burnham	0.36	0.15[b]	0.31	0.10
	Bush Shore	0.07	0.05	0.27	0.07
	Roach mouth	0.05	0.02	0.02	0.04
	Holliwell Buoy	<0.01	0.01	0.02	0.02
	Holliwell Point	0.01	0.01	0.05	<0.01
Blackwater	West Mersea	0.66	0.26	0.15	0.53[a]
Dart	Blackness Point	0.08	0.19	0.17	0.07
Kingsbridge	Frogmore	0.06[b]	0.03	0.07	0.04
Teign	Arch Brook	0.05[b]	0.02	0.03	0.02
Beaulieu	Bucklers Hard	4.6	10.8	1.1	0.23

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

DISPOSAL AT SEA (DAS) GROUP, FIELD ASSESSMENT STUDIES

13. BACKGROUND TO THE STUDIES

13.1 Introduction to the assessment

This assessment is divided into 6 sections : Section 13 gives the background and Sections 14-18 each relate to a particular type of disposal operation or other activity. The work carried out by the Disposal at Sea Group in 1988 and 1989 is summarised in Table 34. The right-hand column of Table 34 indicates the study from which the data are taken and the nature of the data collected in the course of the studies concerned.

13.2 Methods used in the assessment

Sediment samples were collected in accordance with the Co-ordinating Group on Monitoring of Sewage-Sludge Disposal Sites (CGMSD) guidelines (MAFF, 1989). Samples of surface sediment for microbiological or chemical analysis were collected using Day and Shipek grabs for soft and hard sediments respectively. Samples of sediment for analysis of infauna were collected using either Day grab, Tennant corer or Anchor dredge, depending on the substrate type and the purpose of the analysis.

Microbiological determinations were carried out on surface scrapes of sediment according to the methods described by West (1988).

The techniques used for determinations of trace metals, discussed in The Disposal at Sea Group's field assessment studies, are described in detail by Harper *et al.* (1989). Analyses of carbon and nitrogen were carried out using a carbon, hydrogen and nitrogen (CHN) analyser after pre-treatment of sediment samples with sulphurous acid to remove carbonates.

Samples of the epifauna at several disposal sites were collected using a 2 m beam trawl as described by Riley *et al.* (1986).

Table 34. Disposal ground surveys carried out during 1988 and 1989

Disposal Area	Date	Report sub-section	Studies described in this report
Thames	1988	14.7	Sediment quality and biological communities
Tyne	May 88	14.1	Sediment quality and biological communities
Tees	May 88	15.2	Sediment quality
Humber	May 88	14.2	Sediment quality
Liverpool Bay	Sep.88	15.1	Sediment quality
Plymouth	Dec.88	14.6	Sediment quality
South Falls	Dec.88	14.4	Sediment quality
Wearmouth	Apr.89	16.3	Survey of colliery waste disposal site
Tyne	May 89	14.1	Biological communities
Thames	1989	14.7	Sediment quality and biological communities
Seaham	Sep.89	16.2	Sediment quality
Liverpool Bay	Sep.89	15.1	Sediment quality
Roughs Tower	Dec.89	14.3	Bacterial distribution
Nab Tower	Dec.89	18.3	Sediment quality and biological communities
Exeter	Dec.89	14.5	Bacterial distribution
Aggregate extraction area			
Isle of Wight	Dec.88 Dec.89	18.3	Sediment and biological surveys of aggregate extraction sites
Southwold	Apr.89	18.2	Sonar survey of aggregate extraction site

14. SEWAGE SLUDGE

14.1 Tyne

There are several disposal areas off the Tyne designated to receive solid/liquid industrial wastes, sewage sludge and dredged material. The sewage-sludge disposal site receives approximately 500,000 wet tonnes of sludge annually. Monitoring studies carried out by MAFF included infrequent large-scale spatial surveys of sediment quality and more frequent small-scale surveys of sediment quality and biota. A survey on sediment quality was carried out in 1988 (Figure 27) and has been reported elsewhere (Rowlatt *et al.*, 1991). In summary, this work demonstrated that, with the possible exception of chromium, there was no evidence of any accumulation of metals or carbon at the disposal site. In the case of chromium, the data were inconclusive and further work is in hand. The survey, when compared with those from 1987 and 1989, indicated no year on year change in concentrations of metals in sediments. Also, there was no evidence of contamination by PCBs or pesticide residues associated with the sludge disposal site. The following sub-sections discuss results obtained in the course of the annual surveys conducted in 1988 and 1989.

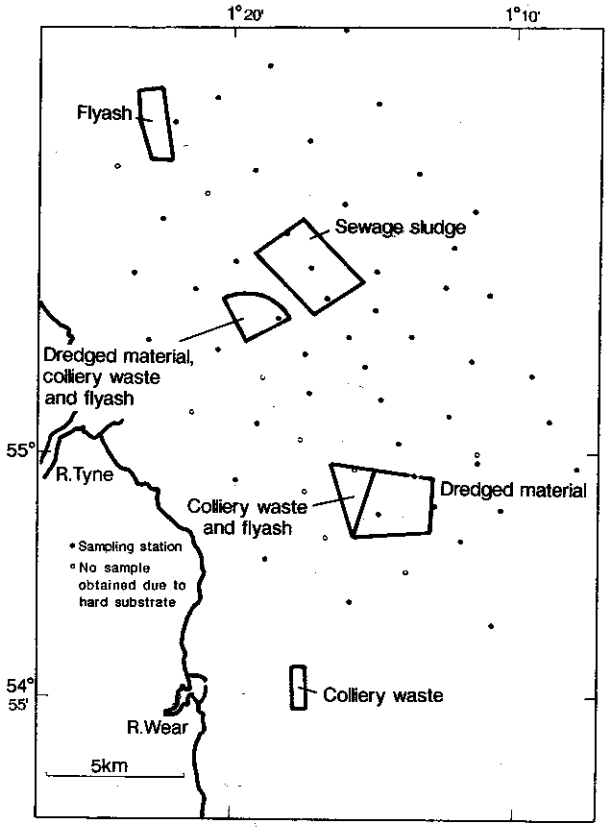


Figure 27. Sediment survey of the Tyne sewage-sludge disposal site in 1988

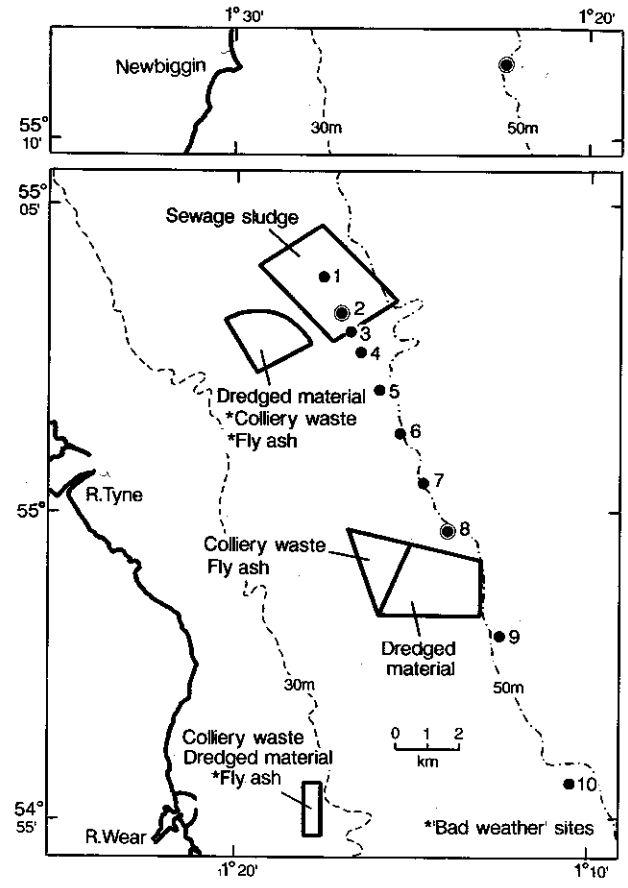


Figure 28. Benthic sampling stations in June 1986. Sites currently in use for sea disposal are specified (•); (double circles indicate stations sampled annually)

14.1.1. Trends in the macrofauna sampled by Day grab

Spatial changes in the benthic fauna sampled by Day grab, and in a range of physico-chemical measures of sediment quality, were examined on two occasions along a transect running south from the Tyne sewage-sludge disposal site (Figure 28). The results have been reported in Rees *et al.* (1985, 1990), and in a previous report in this series (MAFF, 1990).

Effects on the benthos were manifested by enhancement in numbers of individuals of a range of commonly- occurring species but there was no evidence for gross change in the composition of the community. This indicated an early stage in the process of organic enrichment of sediments.

From these data, representative stations were selected for the monitoring of temporal trends. These now form part of a regular MAFF sampling programme in this area: from 1987 onwards, 5 macrofaunal samples were taken annually in May from a station within the disposal site, and from 1 or 2 distant locations to the north and south (see Figure 28: the northern station is some 14 km from the disposal site).

This sampling regime allows a spatial comparison of 'treatment' and 'reference' sites. However, the continued validity of such a comparison requires checking by periodic grid surveys of the area. Presently, such surveys include those conducted under MAFF auspices by the licensee (Northumbrian Water plc) as part of a 'self-monitoring' scheme, whereby the licensee undertakes an agreed component of the monitoring programme at its sewage-sludge disposal site, usually with the emphasis on spatial studies (Rowlatt *et al.*, 1991).

A comparison of changes at the disposal site, relative to a reference site some 8 km to the south is shown in Figure 29. Summary statistics for the benthos show that the changes at both locations appear to be in synchrony, and are likely to reflect natural environmental variability. Such a synchronous relationship adds weight to the validity of the 'treatment' and 'reference' comparison, at least during the early stages of any impact, since it implies that similar environmental conditions prevail at both sites (see for example Skalski and McKenzie, 1982).

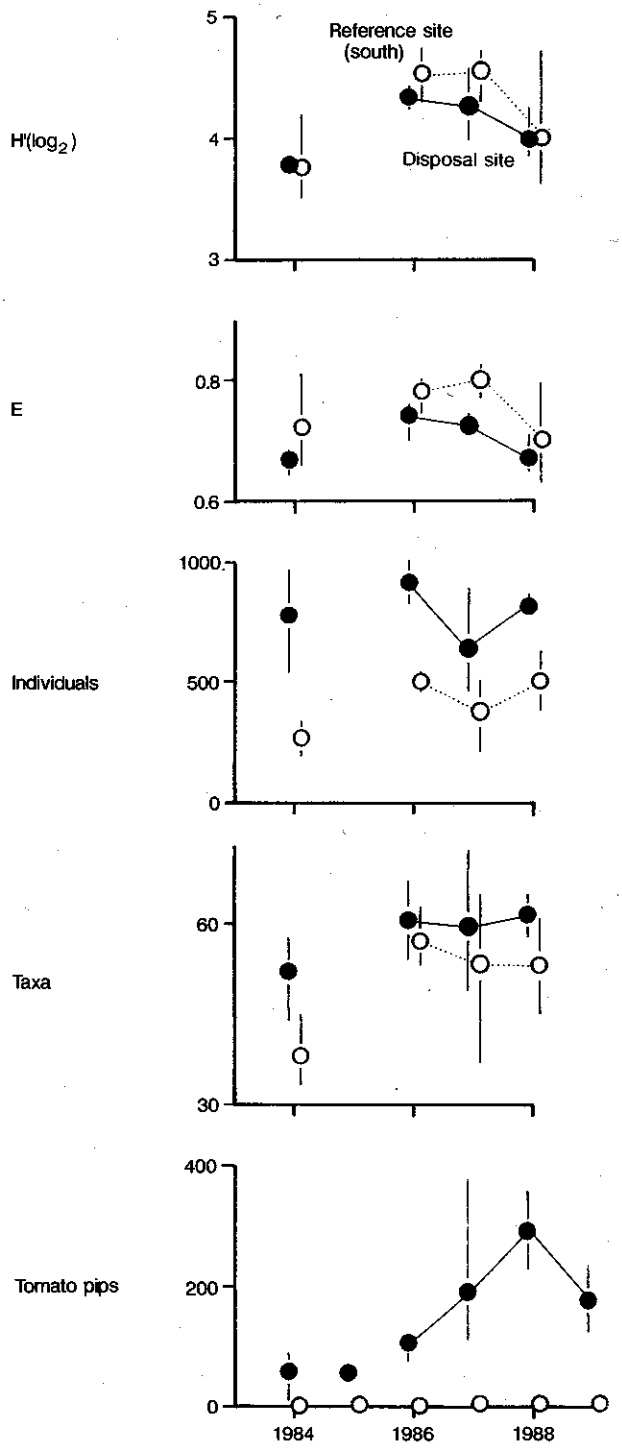


Figure 29. Temporal changes at two regularly sampled locations (x range). (Note: samples in 1984 and 1985 were taken in February/March; samples in 1986-1989 were taken in May/June)

Only in the case of total counts is there a marked difference in the benthos between the two sites, as might be expected from the results of the transect studies. It may be noted that this density difference is not clearly reflected in values of diversity (H') and evenness (E), demonstrating that it is important to employ a variety of measures of data structure when seeking to deduce subtle effects. Counts of tomato pips indicate that there can be appreciable variability between samples; nevertheless, there is a general upward trend with time at the disposal site.

Changes observed to date in the benthos near to the disposal site are judged to be relatively small and therefore acceptable. However, significant increases in the quantity of sewage sludge disposed of to this site are considered undesirable, as these may rapidly induce changes of a less benign nature which would be in breach of the proposed Environmental Quality Standard (EQS) (MAFF, 1989). Consequently, a tonnage limit was stipulated in 1985. Annual monitoring at this site is continuing.

14.1.2. Vertical distribution of the macrofauna from Tennant core sub-samples

In 1989, the vertical distribution of macrofauna was examined at the three locations referred to above, by sectioning 8 cm diameter sub-cores at 1 cm intervals. Each section was preserved for later laboratory analysis after extraction on a 0.5 mm mesh sieve. Prior to sectioning, Eh values were determined along one of the two sub-cores taken at each location, by inserting a 5 mm diameter combination redox electrode in a horizontal plane through a series of holes bored at 1 cm intervals. Readings were taken 60 s after electrode insertion and then corrected to the potential of a standard hydrogen electrode. Below 4 cm depth, there was a notable reduction in Eh values at the disposal site, compared with the other 2 locations (Figure 30 (a)). As might be expected, highest counts of macrofauna were found at or near the sediment surface (Figure 30 (b-c)). However, it was not unusual to encounter animals at depths of 10 cm or more, and there was no indication of reduced penetration of the macrofauna at the disposal site, compared with elsewhere. These results indicate that, despite the change in physico-chemical environment indicated by the Eh data, there is no evidence of any impairment in the suitability of sediments at the disposal site for colonisation by benthic organisms.

Higher counts of tomato pips were found at the disposal site (Figure 30 (d)). These were present even down to 15 cm, probably as a result of bioturbation of sediments through the feeding or burrowing activities of larger animals.

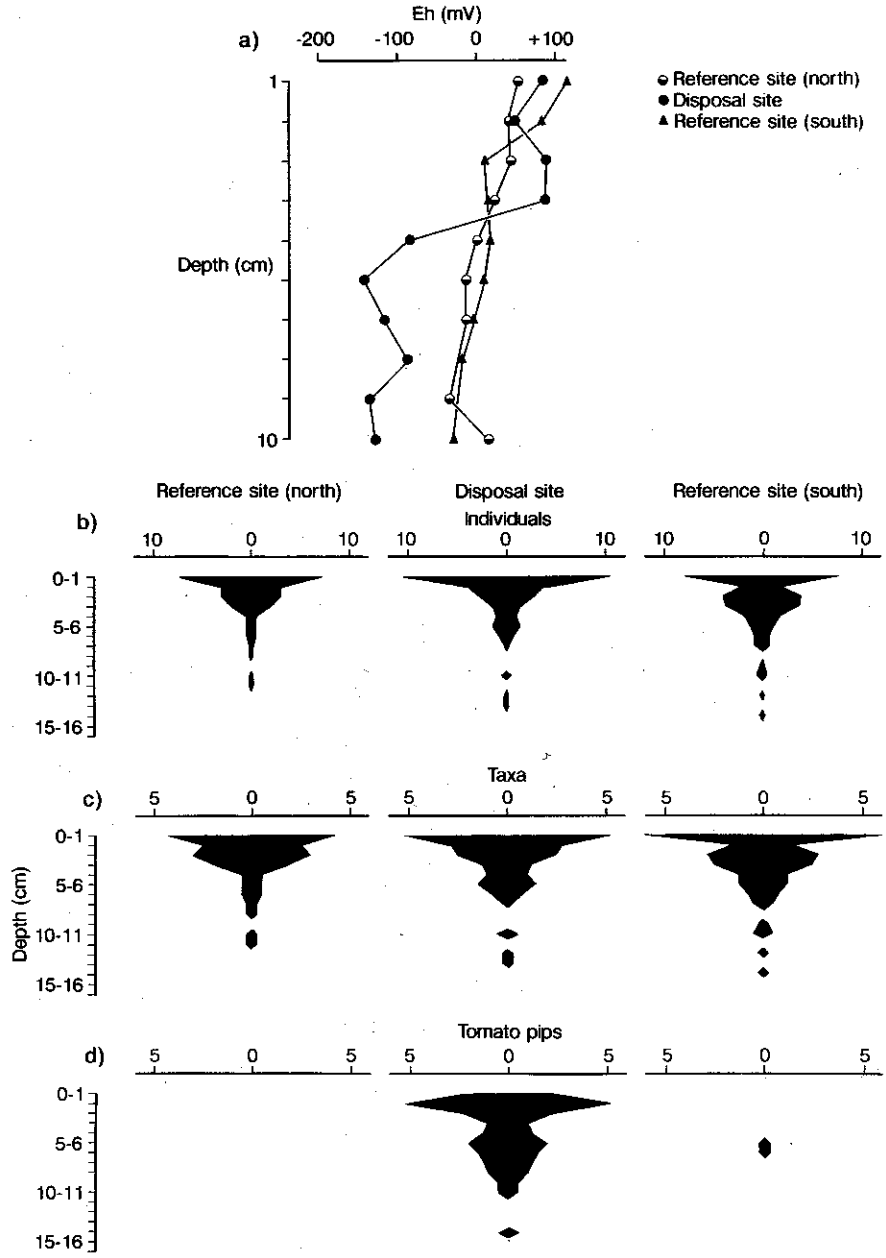


Figure 30. Results from analysis of Tennant core samples: (a) 'down-core' distribution of Eh values at regularly sampled locations; and (b-d) 'down-core' distribution of individuals, taxa and tomato pips (mean counts from two sub-cores)

14.2 Humber

Sediments around the Humber sewage-sludge disposal site, which receives about 100,000 wet tonnes of sewage sludge each year, were surveyed in 1988.

The organic carbon and metal contents of the <63 μm fraction of the sediments are shown in Figure 31 (a-g). This size fraction has been used in accordance with the guidelines specified by the CGMSD (MAFF, 1989) in order to detect 'sensitively' any variations in sediment quality. There is little evidence of any impact of sewage sludge at the sea bed, although a few stations do exhibit slightly elevated levels of metals. These elevations are judged to be within acceptable limits.

14.3 Roughs Tower

The Roughs Tower sewage-sludge disposal site receives about 130,000 wet tonnes of sewage sludge each year. Surveys of bacterial contamination at the sea bed were carried out in 1989 to assess the dispersion of the wastes. Figure 32 (a-b) shows the concentrations of *E. coli* and faecal streptococci in sea-bed sediment and indicates that sewage-sludge dispersion occurs along the tidal axis parallel to the coast and that under the survey conditions little accumulation of sludge- derived material occurred.

The survey was carried out under calm conditions and it is likely that, under more vigorous sea conditions, dilution will be greater and dispersion wider.

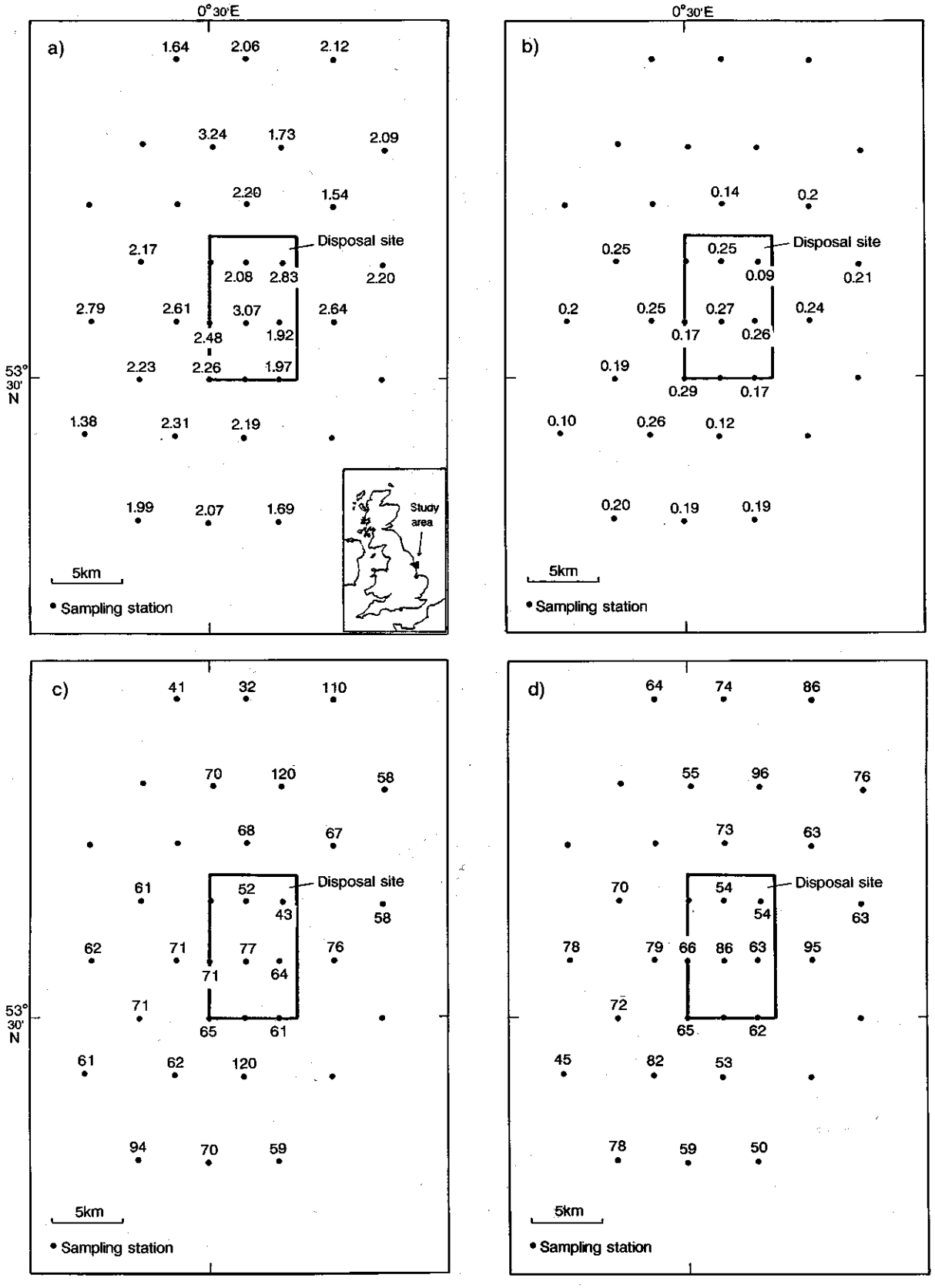


Figure 31. Sediment survey of the Humber sewage-sludge disposal site in 1988. Concentrations of metals (mg kg^{-1}) and carbon (%) in the <63 µm fraction of the sediment: (a) carbon; (b) mercury; (c) chromium; and(d) lead

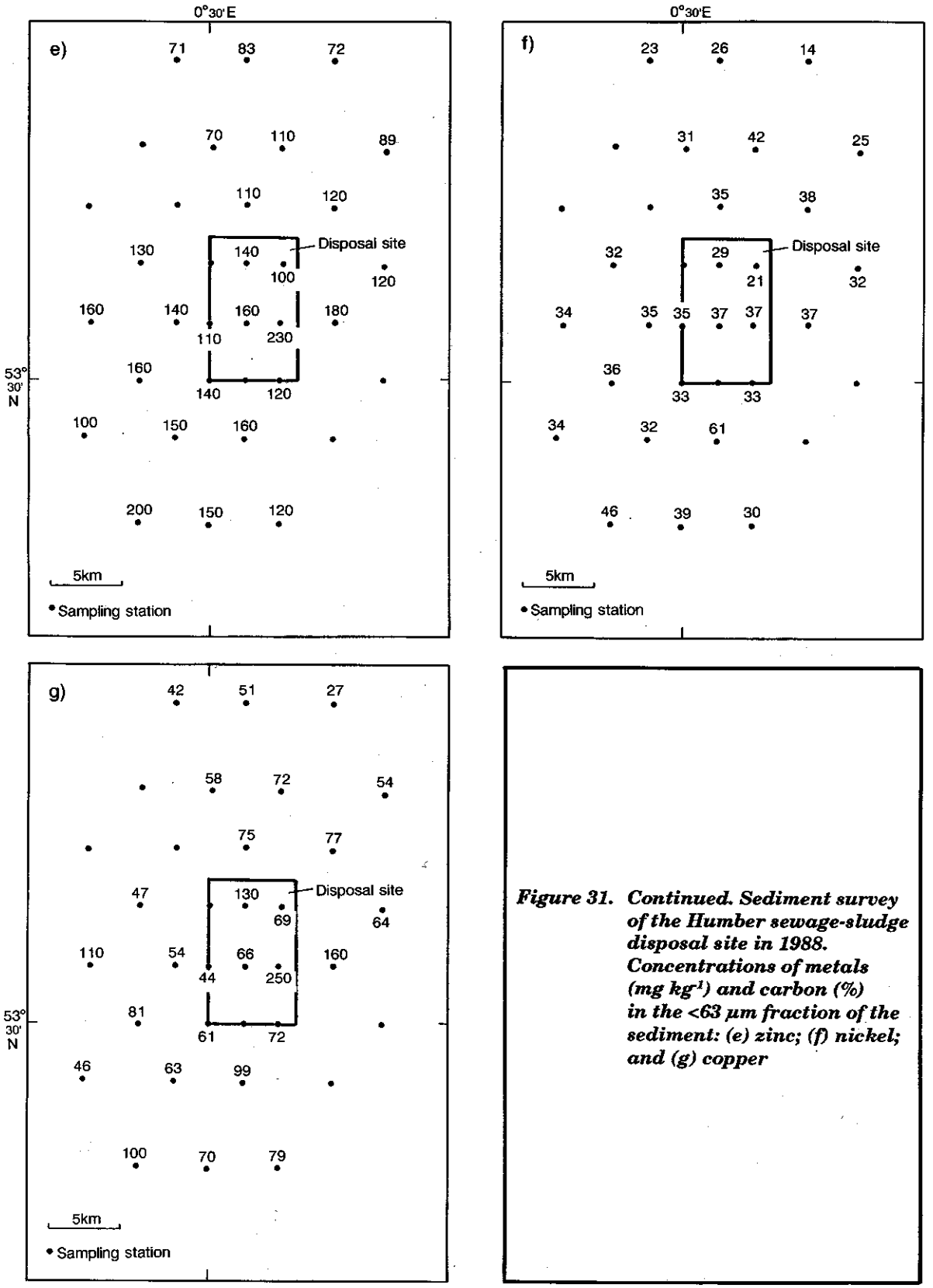


Figure 31. Continued. Sediment survey of the Humber sewage-sludge disposal site in 1988. Concentrations of metals (mg kg^{-1}) and carbon (%) in the <63 μm fraction of the sediment: (e) zinc; (f) nickel; and (g) copper

14.4 South Falls

A survey of sediments was carried out in 1988, at a time when the South Falls disposal site received sewage sludge from Tilbury, dredged material from, amongst other locations, the Thames Estuary and industrial waste (sludge from sugar purification).

Sewage-sludge disposal at this site ceased in February 1990.

Figure 33 (a-f) shows measurements of concentrations of metals in sediments collected during this survey. There is little evidence of elevated concentrations at the disposal site compared to the surrounding areas, indicating dispersion of the wastes. This is in contrast to the 1985 survey (MAFF, 1990) where areas of

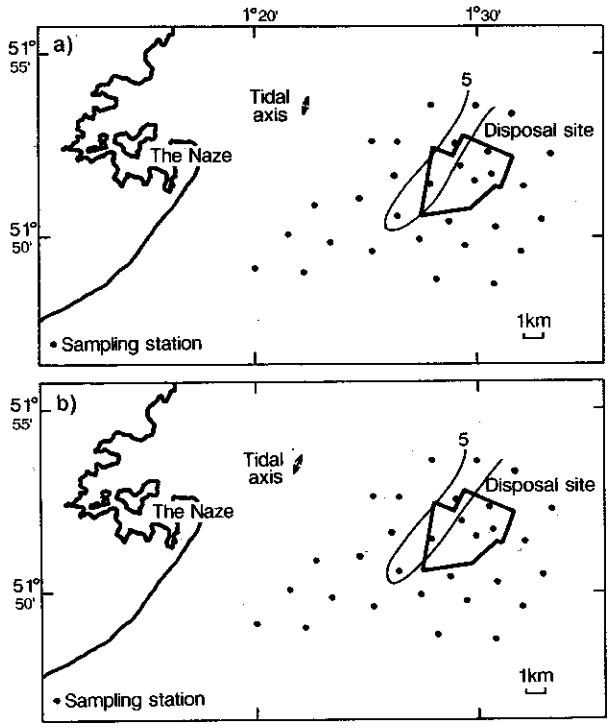


Figure 32. Distribution of faecal bacteria (numbers per ml sediment) around the Roughs Tower sewage-sludge disposal site in December 1989 : (a) E. coli; and (b) group D faecal streptococci

contaminated sediment around the disposal site were clearly demonstrated. Figure 34 shows a comparison of sedimentary concentrations of zinc between 1985 and 1988. Sediments to the east of the Falls Ridge have shown a marked decrease in concentrations of zinc during this period while those to the west have shown no significant change. The quantities of waste deposited changed little between these years so waste disposal practice cannot account for the observed difference in impact. As the area is dispersive, it must be concluded that the sampling programme in 1985 was conducted shortly after a disposal event and that the contamination observed then was a transitory feature.

14.5 Exeter

About 47,000 wet tonnes of sewage sludge from Exeter are deposited each year at a site in Lyme Bay. Due to tidal restrictions on the disposal vessels passing through the lock gates at Exeter, it is only possible for the vessels to reach the closest part of the designated area within one tidal cycle. Thus, only a small portion of the designated disposal site is actually used.

In 1989, a survey was carried out to assess dispersion of the sewage sludge by use of bacterial tracers (Figure 35 (a-b)). Concentrations of *E. coli* and faecal streptococci were highest below the main area of deposition and showed elevated concentrations to the north-east and south-west along the tidal axis. The data suggest that the south-west of the survey area may also be affected by coastal sources.

These results have been used to define more detailed monitoring work to be carried out by the licensee. This work will be reported by the CGMSD in its annual reports.

14.6 Plymouth

In 1988, a series of 15 sediment samples was collected from the vicinity of the sewage-sludge disposal site off Plymouth Sound. The samples were taken close to the disposal site where sewage-sludge settlement occurs (MAFF, 1990). Table 35 shows the mean concentrations of metals in the <63 μm fraction of the sediment. The data are the second in a series of biennial measurements designed to assess whether the quality of the surface sediments is changing. The earlier (1986) data are also shown in Table 35 for comparison, although further data will be required before firm conclusions can be drawn.

Table 35. Concentrations of metals (mg kg^{-1}) in the <63 μm fraction of sediment from the area of the Plymouth sewage-sludge disposal ground in 1986 and 1988

	Hg	Pb	Cu	Zn	Cr	Ni
1986 values						
Mean	0.23	79	32	100	38	22
SD	0.07	45.3	10.7	9.5	3.9	2.2
1988 values						
Mean	0.26	52	33	111	41	27
SD	0.05	7.1	7.7	7.8	2.8	1.1

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

14.7 Thames

Various measurements were made of sediment quality and benthic communities in the Barrow Deep area of the Thames Estuary. These form part of a time series given in an earlier report in this series (MAFF, 1990). As individual measurements cannot be viewed in isolation, the time series will be reported on only when sufficient new data points are available.

14.8 Liverpool Bay

Various measurements were made of sediment quality around the Liverpool Bay sewage-sludge disposal site. These form part of a time series and cannot be viewed in isolation. The time series will be considered in detail in MAFF's next annual report on monitoring and surveillance of non-radioactive contaminants, which will be issued in this series.

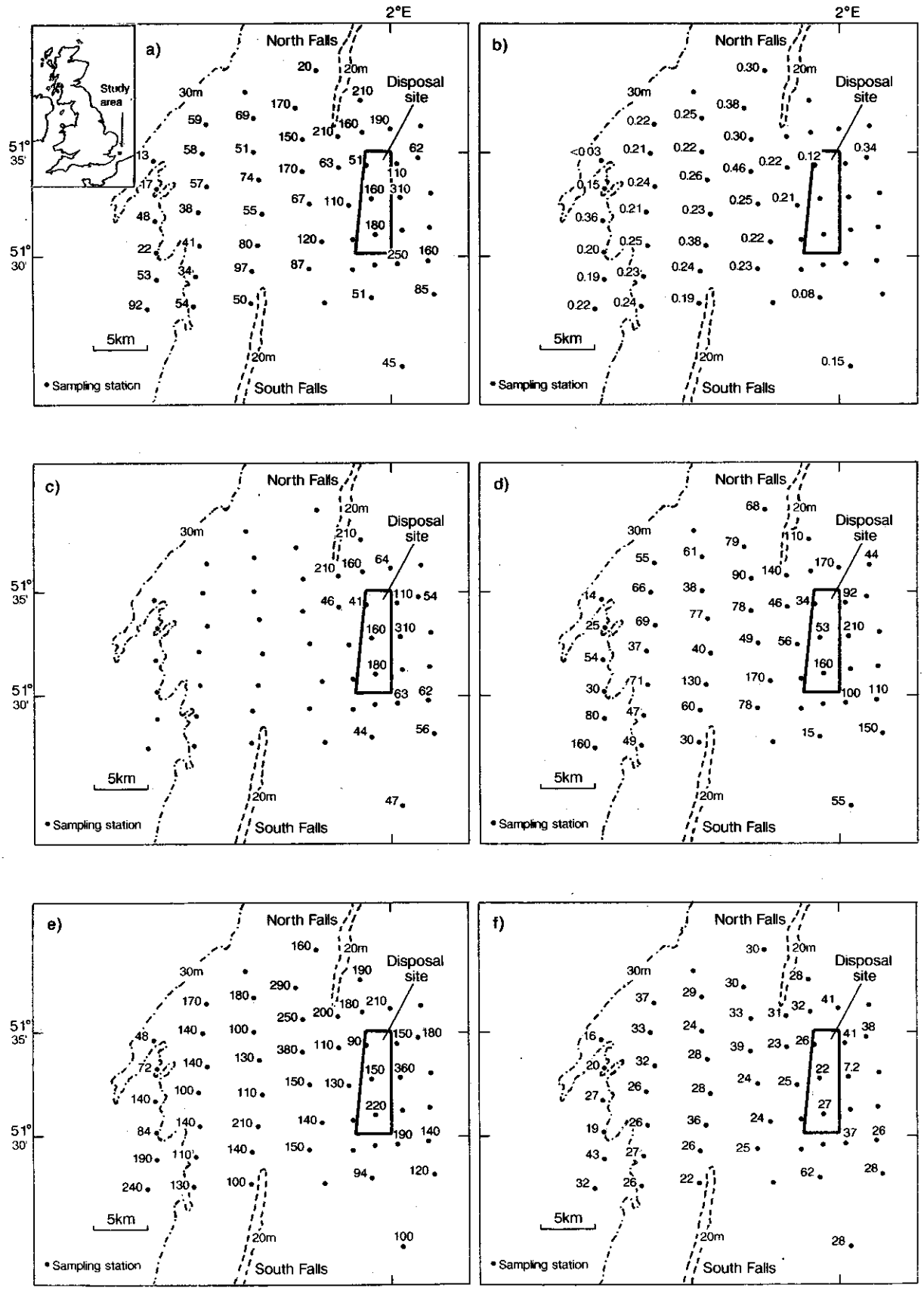


Figure 33. Sediment survey of the South Falls sewage-sludge disposal site in1988. Concentrations of metals (mg kg^{-1}) in the <63 μm fraction of the sediment: (a) copper; (b) mercury; (c) chromium; (d) lead; (e) zinc; and (f) nickel

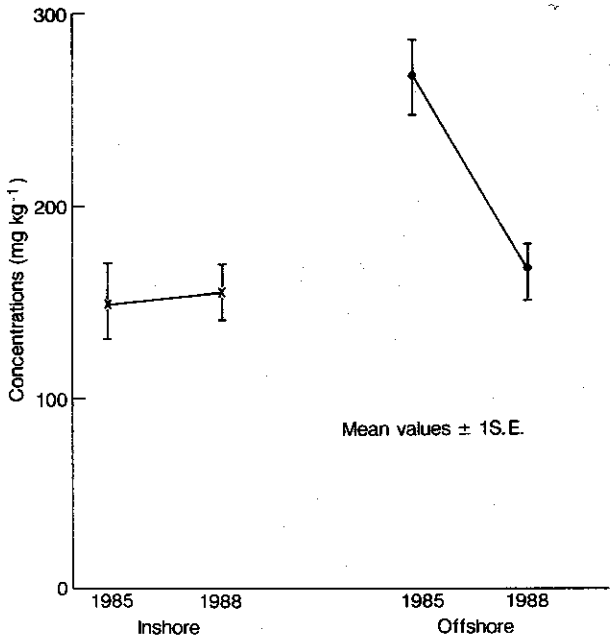


Figure 34. Variation in concentrations of zinc (mg kg^{-1}) in the < 63 μm fraction of the sediment in the South Falls area between 1985 and 1988

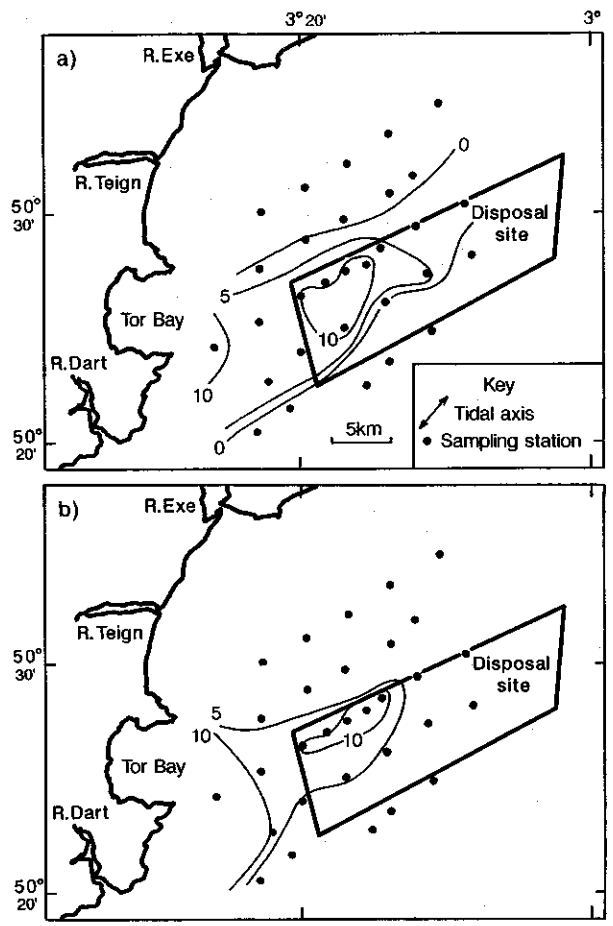


Figure 35. Distribution of faecal bacteria (numbers per ml sediment) around the Exeter sewage-sludge disposal site in December 1989: (a) E. coli; and (b) group D faecal streptococci

15. DREDGED MATERIAL

15.1 Liverpool Bay, Site Z

Each year, approximately 3 x 10^6 t of dredged material from the Mersey Estuary and its approaches are deposited at Site Z in Liverpool Bay. The disposal area is predominantly sand with muddy sands near the entrance to the Queen's Channel (Figure 36) and the deposited materials range between mud from the Liverpool Docks and sand from the approach channels.

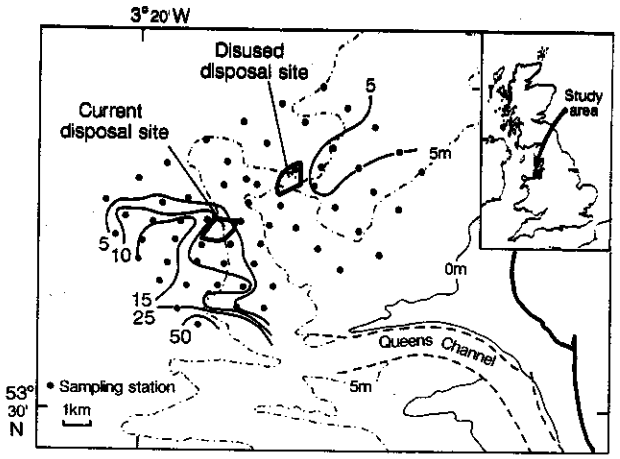


Figure 36. Distribution of mud(%) around Site Z, Liverpool Bay

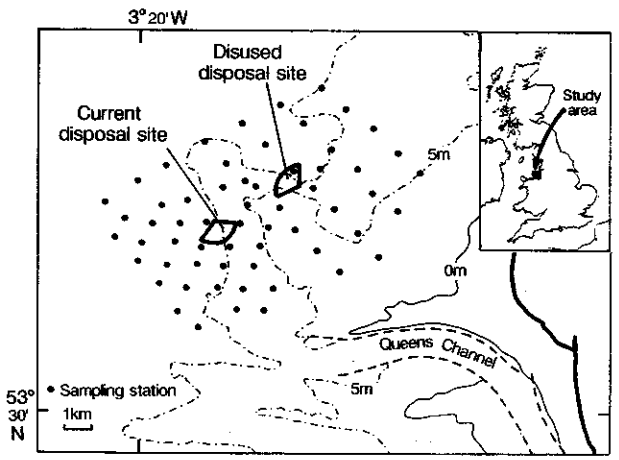


Figure 37. Sediment samples collected during the 1988 survey of Site Z, Liverpool Bay

In 1988 and 1989, sediment surveys were carried out to assess the impact of this disposal operation on the sediments around Site Z (Figures 37 and 38). There is a strong positive relationship between the concentrations of metals and the fines content of the sediments (Figure 39). This arises from the association of metals with the

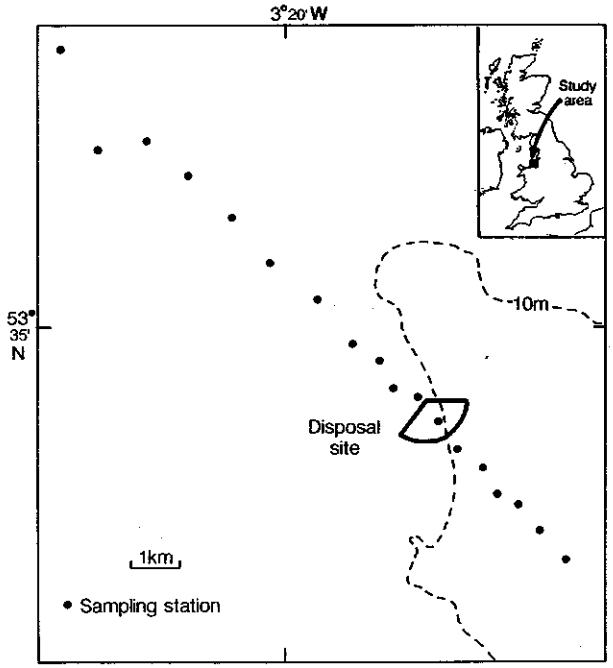


Figure 38. Sediment samples collected during the 1989 survey of Site Z, Liverpool Bay

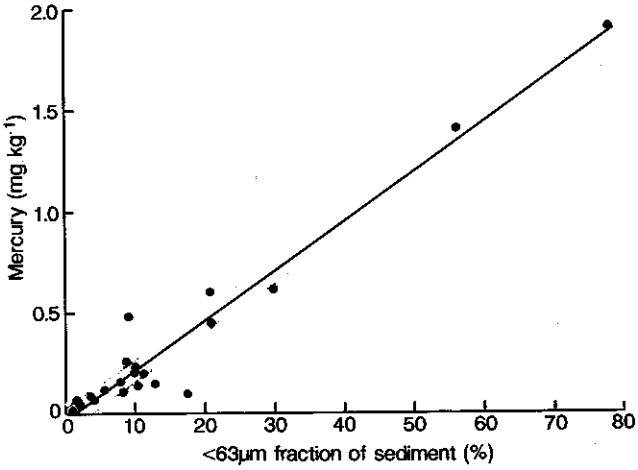


Figure 39. Mercury/fines relationship in sediments from the Site Z area

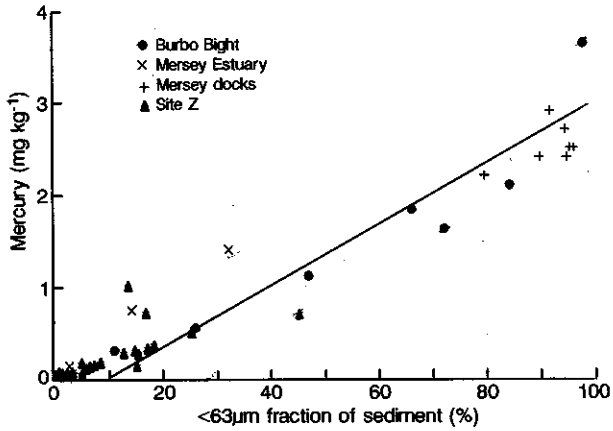


Figure 40. Mercury/fines relationship in sediments from the Mersey Estuary and the Site Z area

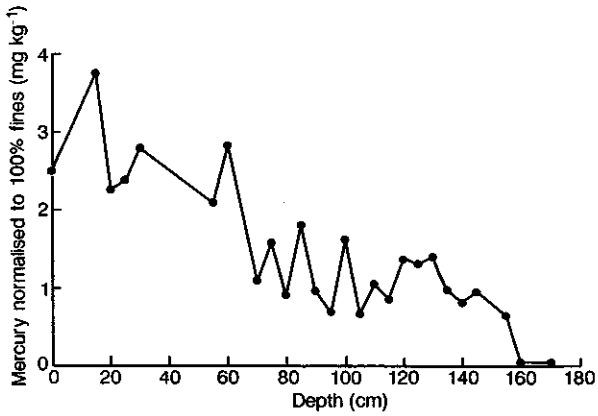


Figure 41. Mercury variation down a sediment core from the Burbo Bight, Liverpool Bay. Concentrations are expressed as mg kg⁻¹ normalised to 100% fines

clay minerals in the fines. The linear nature of the relationship indicates that the metal content of the fines is similar across the area and that the range of concentrations reflects dilution by coarse sand particles which are poor in metals. Such relationships between metals and fines have been observed previously for the Mersey Estuary by Taylor (1986) and Rowlatt (1988). Figure 40 shows the 1988 data for Site Z combined with the results of analyses of samples from the Mersey Estuary and docks and shows that the metals/fines relationship is the same in both areas and that sediments are all part of the same population.

Figure 41 shows that sediment samples collected at depth from a core collected in the Mersey entrance contained a lower level of mercury than those recently deposited (at the top of the core). These sediments are interpreted as representing the sediments deposited in pre-industrial or early industrial times when environmental concentrations of mercury were lower than presently. Thus it may be concluded that at least 90% of the inventory of mercury in the sediments in the Mersey and at Site Z arises from anthropogenic sources.

15.2 Tees Bay

There are two disposal sites for dredged material in Tees Bay used for maintenance and for capital dredgings (MAFF, 1990). A survey of the chemical quality of sediments around these sites was carried out in 1988. Figure 42 (a-g) shows the results for several metals and for carbon. Concentrations of metals and carbon around the sites show only limited evidence of an increase that could be attributed to the disposal operation.

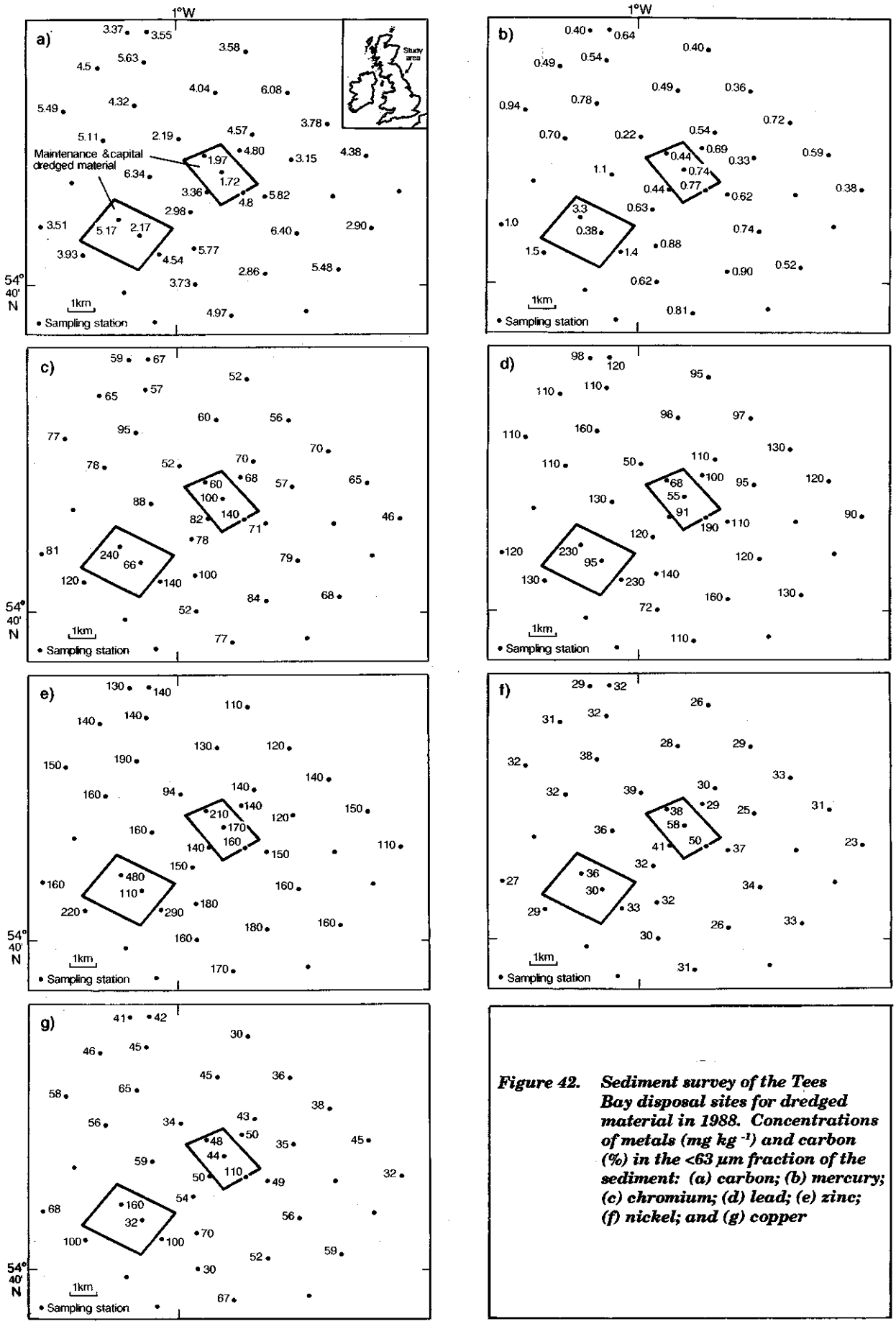


Figure 42. ***Sediment survey of the Tees Bay disposal sites for dredged material in 1988. Concentrations of metals (mg kg $^{-1}$) and carbon (%) in the <63 µm fraction of the sediment: (a) carbon; (b) mercury; (c) chromium; (d) lead; (e) zinc; (f) nickel; and (g) copper***

16. COLLIERY WASTE

16.1 Background

During coal mining operations, a quantity of rock is brought to the surface along with the coal. The rock is separated from the coal in washeries, giving rise to two waste materials - minestone and tailings. Minestone is a gravelly material produced during the initial stages of processing. Most of the particles are in the size range 1 mm-30 cm. Tailings are produced during a subsequent froth flotation process to separate the fine coal and fragments of rock and are composed of particles finer than 2 mm. The washery procedures do not extract all of the coal and can leave the waste containing up to 20% coal, largely as inclusions in shale fragments.

Due to the historical development of some collieries in towns in the north-east of England, (e.g. Wearmouth colliery in Sunderland and Westoe in South Shields), there is limited opportunity for the disposal of waste on land near to the colliery and there can be problems with transportation of the waste inland. However, in cases where collieries are situated close to rivers or on the coast, it is possible to transport the rock waste by ship and to deposit it in the sea or to deposit it along the shore. These options have been used for a proportion of the waste for many years and, at least until recently, have been considered as an acceptable consequence of the industry and not dissimilar to input of rock, etc. by cliff erosion slightly further south on the same coast.

16.2 Seaham

Waste materials from both coal mining and harbour dredging are disposed to the marine environment off Seaham, County Durham. Minestone and tailings (the latter being consented by the National Rivers Authority (NRA)) are discharged directly onto the beach where they are subjected to weathering and dispersal by wave action. Dredged material is disposed of at a disposal site 1 km east of the colliery waste beach disposal site.

A diver survey of the sea bed was carried out in 1989, to assess the extent to which these disposal operations had affected the physical and chemical nature of the sediments. The survey was based on visual inspections of the sea bed (including photography and video) and sampling of sediment for physical and chemical analysis.

Despite the disposal of 1 x 10^6 t of minestone on the beach at Dawdon each year, there was no evidence of significant accumulations of waste in the nearshore zone and in particular no evidence of smothering of rocky habitats locally important for crustacea.

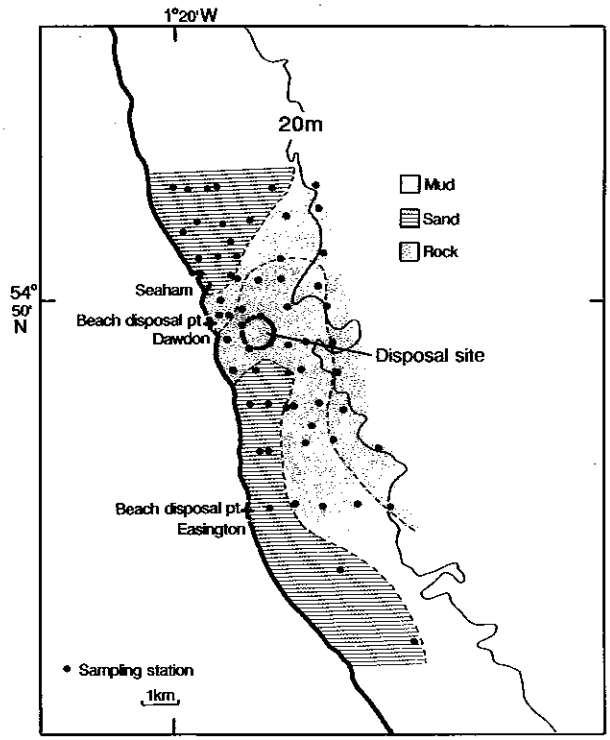


Figure 43. Sediment distribution off Seaham (from diver observations)

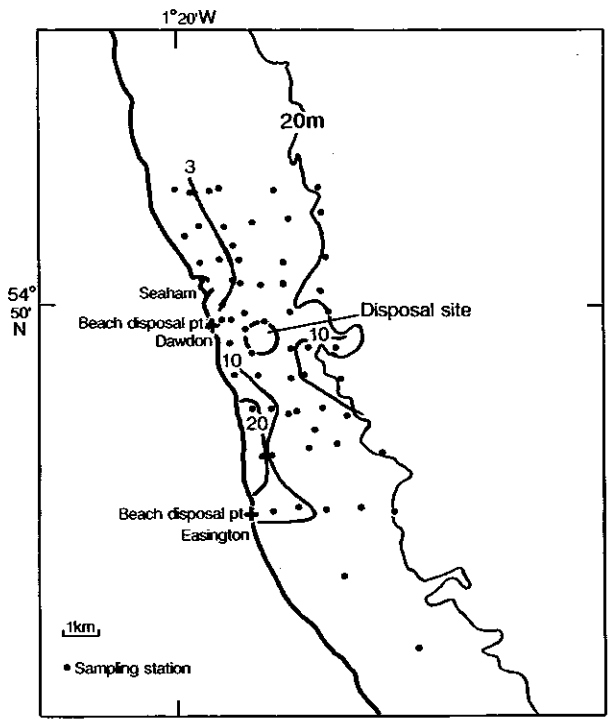


Figure 44. Organic carbon (%) in sediments off Seaham in 1989

Sediments over the survey area range from thick deposits of fluid mud, to sand, to bed rock (Figure 43). The mud was found covering predominantly sandy substrates but occasionally encroaching into rocky areas, although not to an extent sufficient to fill in crevices likely to provide habitat for crabs and lobsters.

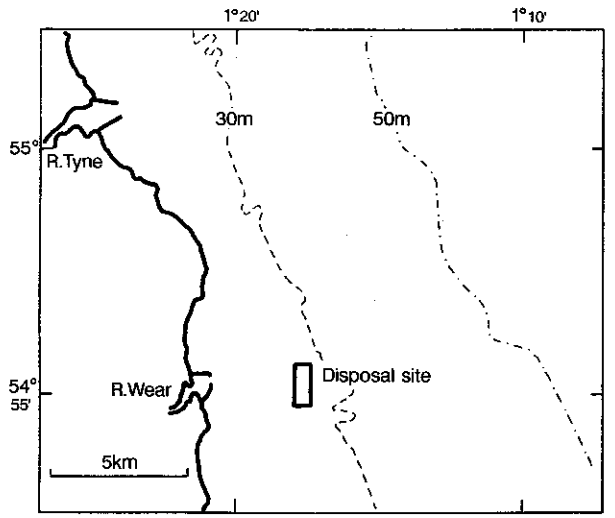


Figure 45. Location of the Wearmouth colliery waste disposal site

There was no significant chemical contamination of sediments by either colliery waste or dredgings, although carbon analyses indicated an enhancement of sediment coal over the grid, particularly south of Seaham (Figure 44). The main conclusion to be drawn from this survey is that the disposal of colliery waste and dredgings has little effect on the natural sediment over the area studied.

16.3 Wearmouth

A detailed survey of the Wearmouth disposal site (Figure 45) was carried out in 1989 as a follow-up to that reported previously in this series (MAFF, 1990). The mound of rock waste observed at the sea bed in the earlier survey was defined using echo sounder and side-scan sonar techniques (Rowlatt *et al.*, 1990).

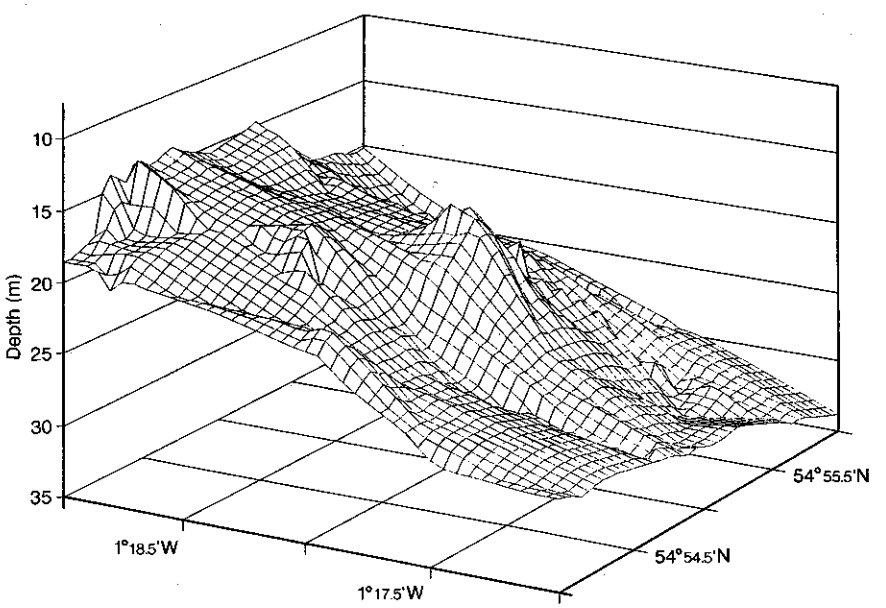


Figure 46. Three-dimensional representation of sea-bed topography around the area of the Wearmouth colliery waste disposal site

Given the local input of large amounts of fine material via colliery tailings and dredged material, and the similarity of these to natural muddy sediments present throughout the grid, it seems likely that some of the mud may originate from one or both of these disposal operations. As only limited quantities of mud were observed within the survey area, it appears that a large proportion of the disposed material is dispersed away from the locality by tides and storms.

A 3-D visualisation of the rock mound is shown in Figure 46 with a chart of the quality of the sea bed in 1989 in Figure 47. The volume of waste at the sea bed in the mound has been calculated as 4.2×10^6 t, equivalent to about 5 years' disposal. As disposal has taken place for more than 20 years, it may be concluded that most of the minestone has dispersed from the area, after weathering to sand sized particles (Rowlatt *et al.*, 1990).

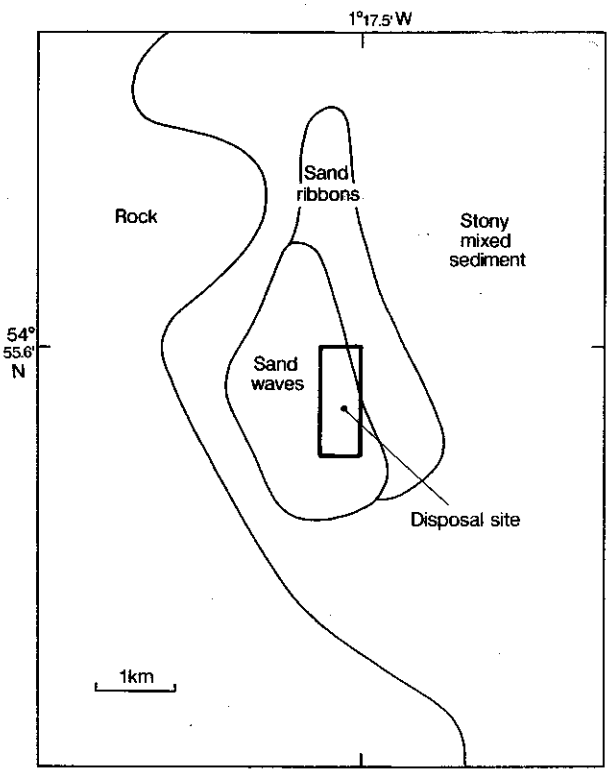


Figure 47. Distribution of sediments around the Wearmouth colliery waste disposal site

17. BEAM TRAWL SURVEYS AT DISPOSAL SITES OFF THE ENGLISH AND WELSH COASTS

17.1 Methods

Along with traditional methods for quantitative sampling of the soft-sediment macrofauna using grabs, several disposal sites have also been sampled for the epifauna using a standard Lowestoft 2m beam trawl (see Riley *et al.*, 1986 for specifications). Sampling and analytical procedures were similar to those reported in MAFF,1990.

Although it is recognised that an appreciable amount of the variability between sites will be accounted for by differences in tow lengths and month and year of sampling, the data are nevertheless considered adequate to identify any major differences in the structure of epifaunal assemblages that might arise from sea disposal activity, or geographical location. A full account of the work, along with additional data, is intended for publication as a separate MAFF report.

17.2 Results and discussion

17.2.1 Description of the study areas

Figure 48(a-b) shows the locations of the eight study areas, and the nature and quantities of material deposited at these sites in 1989. Station positions, along with depth contours and boundaries of disposal sites, are shown in Figure 49 (a-h). In summary, the Tyne and Plymouth sites occupy deeper water (of about 50 m) than elsewhere. Off the Tyne, the substrates along this depth contour range from muddy sand to sandy mud. Sediments at the Plymouth ground typically range from clean shelly sands to gravel and stones.

Muddy sands, interspersed to a variable extent with cleaner sands, are found at South Falls, Exeter and Liverpool Bay (Site Z). The sediments at the Thames and Roughs Tower locations are more mixed, and frequently contain significant quantities of shell debris and stones. At the latter location, larger stones and boulders also occur, with a resultant risk of trawl damage. This is also true for parts of Swansea Bay, where pockets of very muddy sediments are encountered, some of which may be derived from the disposal of dredgings.

The shallower waters at the south-eastern locations and within Swansea and Liverpool Bays, coupled with relatively strong tidal currents, might be expected to exert an influence on the fauna inhabiting these substrates.

17.2.2 Epifauna

Averages of numbers of taxa, individuals, diversity and evenness are shown in Figure 48 (c-f). Highest numbers of taxa occur off the Tyne, and at Exeter and Plymouth (Figure 48 (c)). A comparison of the numbers of taxa found at stations within the disposal sites, with the average for other stations, is shown in Figure 48 (d). There are no marked differences between the pairs of values. This evidence strongly conflicts with a popular misconception that sites used for sea disposal of wastes are, by the nature of the activity, devoid of animal life. This is not to imply that effects do not arise as a result of waste disposal, but rather that such effects as are encountered are usually more subtle and not necessarily found within the confines of the disposal site. The studies undertaken by the Burnham-on-Crouch Laboratory are designed to pick up such subtle effects both in and outside the immediate confines of the area.

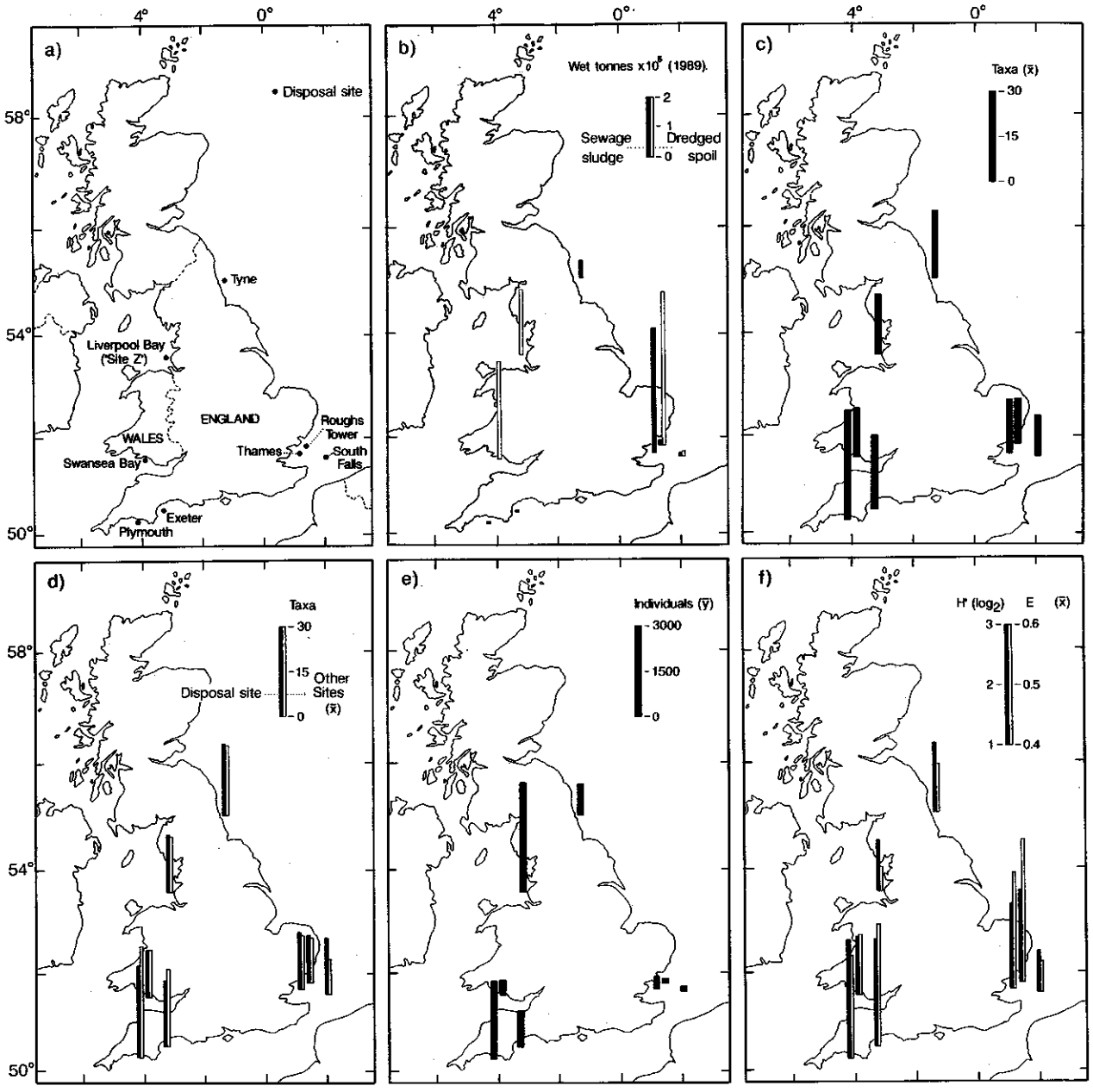


Figure 48 (a-f). Site locations, quantities of wastes disposed of and summary statistics for the benthos from beam trawl samples. Data are expressed as bars extending upwards from the location of each disposal site

Abundances (Figure 48 (e)) show relative impoverishment at the south-eastern locations, and at Swansea Bay. This may be due in part to greater exposure to wave and tidal current action, at least compared with the deeper water locations off the Tyne, Plymouth and Exeter. Very high counts were encountered at some of the sites in Liverpool Bay.

Figure 48 (f) shows that, as expected, the lower the value of evenness (i.e. the greater the numerical dominance by one or two species) the lower the diversity. Highest values of diversity occur off Plymouth and Exeter, which accords with observations on the mean numbers of taxa recorded at these locations (Figure 48 (c)).

The proportional numerical contribution of the major taxonomic groups is shown for each station in Figure 49 (a-h). Crustaceans, notably hermit crabs and shrimps, are more prominent at the east-coast sites, and at certain stations in Swansea Bay and Plymouth. Echinoderms, especially the brittle-star (*Ophiura*) and the common starfish (*Asterias*), are generally more prominent on the west coast. This is especially true for Liverpool Bay, where exceptionally high densities of *Ophiura* were encountered near to the mouth of the Mersey.

High numbers of the turret shell (*Turritella*), characteristically found near the surface of muddy sediments, occurred off the Tyne, and also at Exeter, but not elsewhere. Though intermittently present at several locations, the predatory mollusc (*Philine*) was notably abundant at, and to the south of, Site Z in Liverpool Bay.

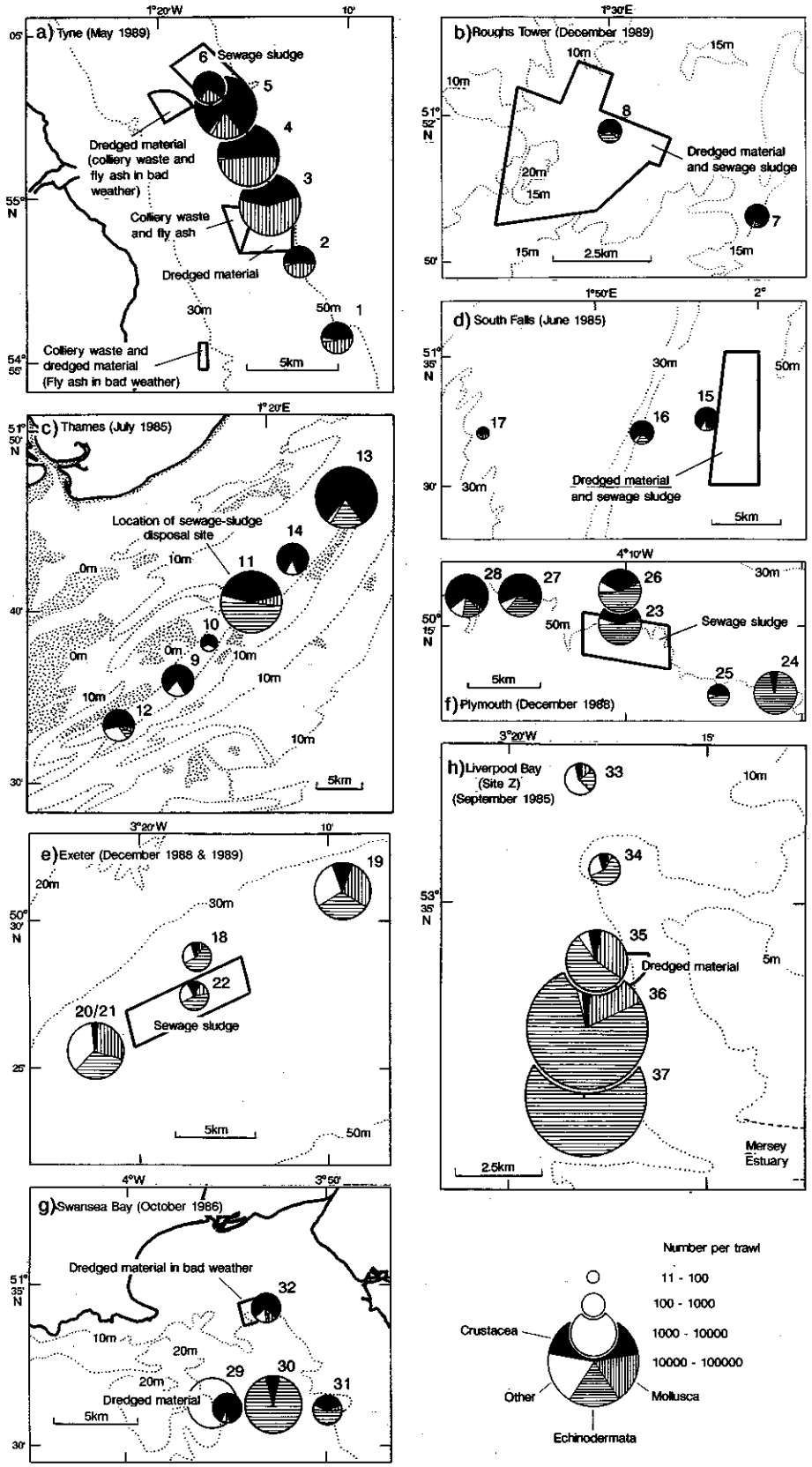


Figure 49 (a-h). Faunal abundance in beam trawl samples: contributions of the major taxonomic groups are superimposed. Boundaries of the major disposal sites are outlined (for explanation of material disposed of, see text)

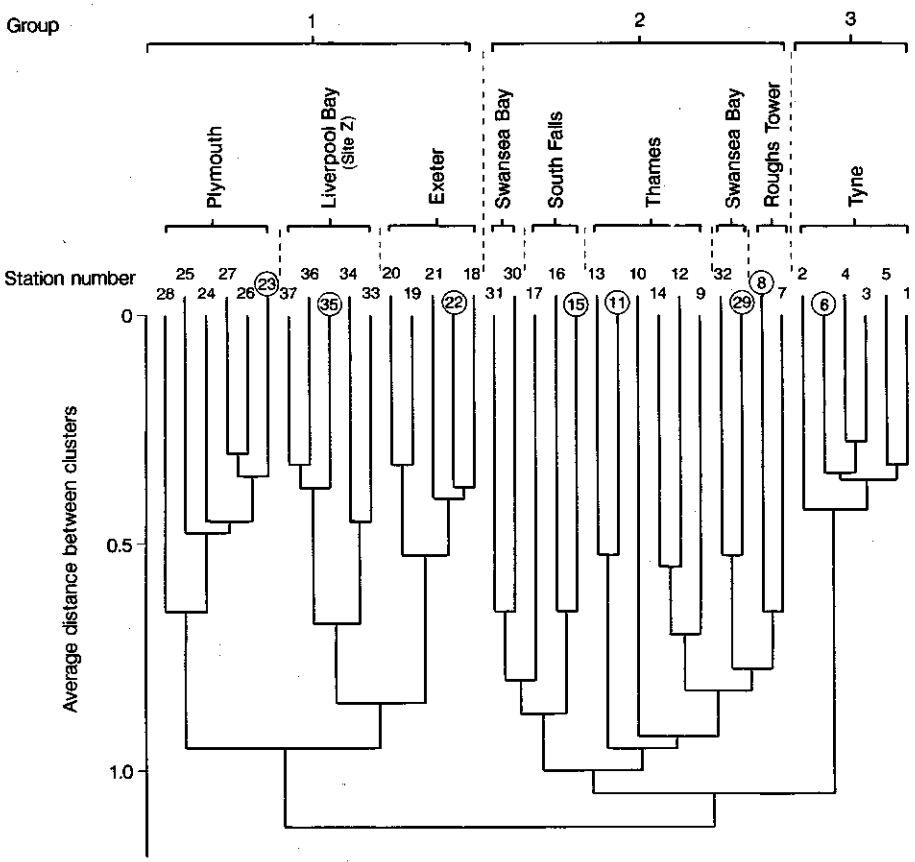


Figure 50. Dendrogram output from cluster analysis of the benthos from beam trawl samples. (Transformation: $\log_e(x+1)$; similarity measure: Bray-Curtis; sorting: average-linkage). Stations within or nearest to the major disposal sites are encircled

Although the spatial coverage within each location is limited, there is little evidence to suggest a marked deterioration in conditions at the disposal sites investigated. The high proportional dominance of hermit crabs just to the south of the Tyne disposal site might suggest an enhanced supply of food for these scavengers (see also Rees *et al.*, 1990). Possible explanations for the low counts just inshore of the Thames disposal site, namely an inhibitory effect of dispersing sludge particles, and for the high counts south of Site Z in Liverpool Bay, namely a combination of dredgings dispersal and organic efflux from the Mersey estuary, were given in MAFF, 1990.

Cluster analysis of the epifaunal data (Figure 50) identified three major station groups - the first comprising Plymouth, Exeter and Liverpool Bay, a second, somewhat ill-defined group, comprising east-coast sites and Swansea Bay, and a third consisting of stations off the Tyne. The linkage is broadly as expected from a consideration of geographical location and prevailing environmental conditions. As can be seen, stations within the major disposal sites are not systematically isolated from others. However, rather unexpectedly, in Liverpool Bay, the inshore stations linked with Exeter within Group 1. Among other attributes shared by these two sites are the 'co-occurrence' in significant numbers of the brittle-star (*Ophiura*) and the solenette (*Buglossidium*).

Apart from the 'co-occurrence' of the predatory mollusc (*Natica catena*) the only notable similarity between two of the four stations in Swansea Bay and those at South Falls appears to be the reduced numbers of taxa compared with elsewhere. The other two stations in Swansea Bay link with those of Roughs Tower and it is perhaps significant that both sites also receive relatively large quantities of dredged material.

17.2.3 Artefacts

Figure 51 gives the combined results of cluster analyses by location and by attribute. Artefacts, predominantly of human origin, were recorded on a presence/absence basis, and the data from each location were pooled. The site grouping does not closely match the nature or the quantity of the material disposed of. Of the three distinct groups of artefacts, the first two are largely inert in nature and, while indicative of human activity, a number may have arrived many years ago and can be just as legitimately connected with the

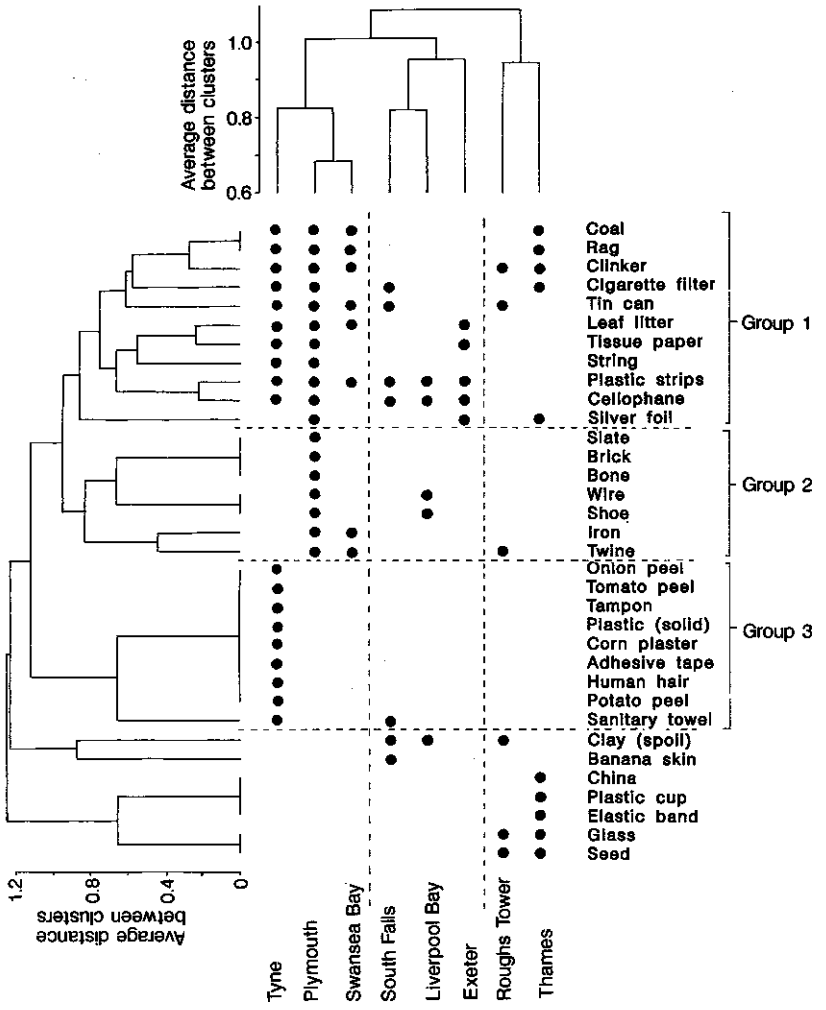


Figure 51. Dendrogram output from cluster analyses of artefacts from beam trawl samples.(Similarity measure: Jaccard; sorting: average-linkage)

proximity to major centres of population and shipping activity as with sewage-sludge disposal. In the case of sites for dredged material, the artefacts which are present will probably have originated from estuarine sediments during dredging.

The third group of artefacts is (with one exception) unique to the Tyne site, and though not present in large quantities, indicates an unacceptable degree of contamination of the sea bed arising from the disposal of primary settled sewage sludge. Similar findings have been reported elsewhere (Rowlatt *et al.*, 1991). Since these surveys were carried out and the problem identified, the Regional Water Company has improved the effectiveness of screening prior to disposal. Further monitoring will establish whether this has solved the problem.

18. AGGREGATE EXTRACTION

18.1 Background

During the period covered by the report, surveys were conducted of six areas of marine aggregate extraction in the Southern North Sea and the English Channel. These areas were studied partly to ensure compliance with the conditions of the licence regarding the location of dredging activity and also as part of an ongoing programme of research to determine the effects of such operations upon the benthic environment and fisheries.

18.2 Southwold Area, Southern North Sea

In May 1989, areas of aggregate extraction (221, 229 and 305/3) (Figure 52) were surveyed using side-scan sonar. This enabled the general relief and composition of the sea floor to be ascertained. Additionally, the extent of dredged tracks was determined, thereby providing an indication of the focus of dredging activity. The side-scan track followed during this survey is shown in Figure 53.

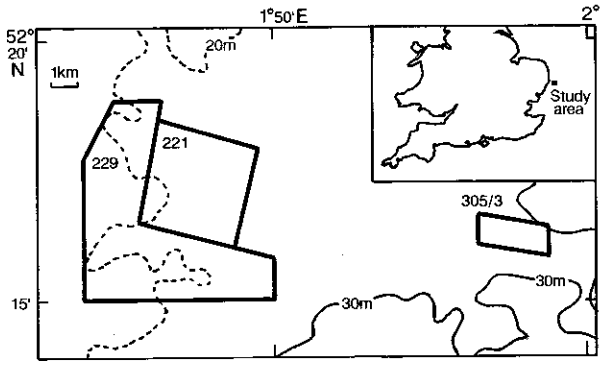


Figure 52. Location of study area and licensed extraction areas 221, 229 and 305/3 east of Southwold

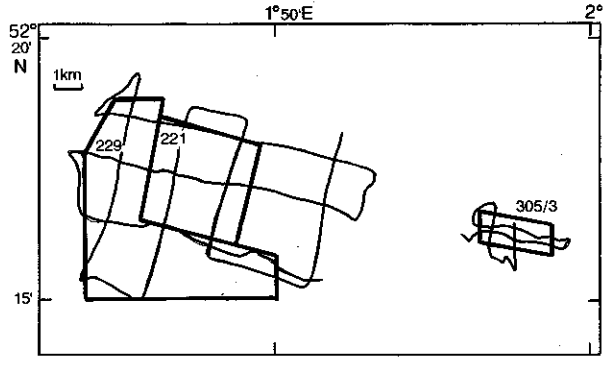


Figure 53. Side-scan track followed in the 1989 survey of areas of aggregate extraction off Southwold

The results of this survey are presented in Figure 54.

The sea floor is covered by a variety of bedforms indicative of strong tidal activity upon sandy and gravelly bottoms, such as sand waves, sand ribbons and gravel lineations. These features all indicate bed transport parallel to an axis aligned NNE-SSW, equivalent to the directions of peak tidal flow in this region. Two general bed types may be distinguished, dominated by sandy sediments and gravelly sediments respectively. These tend to merge into one another; thus sand ribbons are common upon the gravelly floor while patches of gravel occur in otherwise sandy areas. Dredged tracks are confined to the gravelly substrates. They are most frequently encountered in the south-west of licence areas 229 and 221 where, at most, they

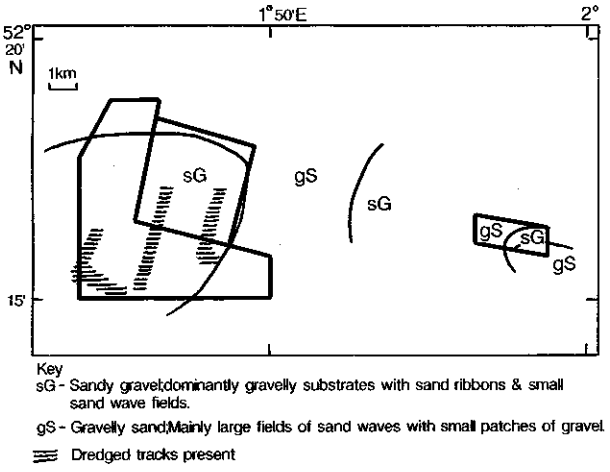


Figure 54. Generalised sediment distribution and location of dredged tracks as determined using side-scan sonar

cover approximately 10% of the sea floor. Apart from a few tracks recorded immediately beyond the western boundary of area 229, no evidence of dredging was found outside the licensed areas or in area 305/3.

18.3 Isle of Wight Area, English Channel

In December 1988, a preliminary side-scan sonar survey was conducted of aggregate extraction areas 213 and 122/3 (Figure 55). This indicated that intensive dredging had been taking place in the north-east corner of area 213. MAFF's diving team subsequently examined part of this impacted area during the following August and found clear evidence of the effects of dredging upon the substrate in the form of numerous intersecting dredged tracks and pits.

On the basis of these observations, a more intensive survey of the region was conducted in December 1989. The survey area was expanded to incorporate the Nab Tower Disposal Ground and also two newly licensed extraction areas (340 and 351). Firstly, the presence of dredged tracks, and hence the location of dredging activity, was ascertained using side-scan sonar (Figure 56). Secondly, a grid of stations was sampled by anchor dredge to allow differences in sediment type and benthic associations across the region to be documented (Figure 57).

Analysis of the side-scan sonar record reveals that sporadic dredging activity has taken place in licenced areas 340 and 351. The most heavily dredged area is 213. In particular, dredging appears to have been intensive at, and immediately beyond, the northern boundary of the licence area. Complete coverage by dredged tracks of a 1 km section of the sea floor (A-A' on Figure 56) is apparent here. Figure 58, which is an echo-sounder trace of this section, indicates the presence of a furrowed 3 m deep depression coincident with this.

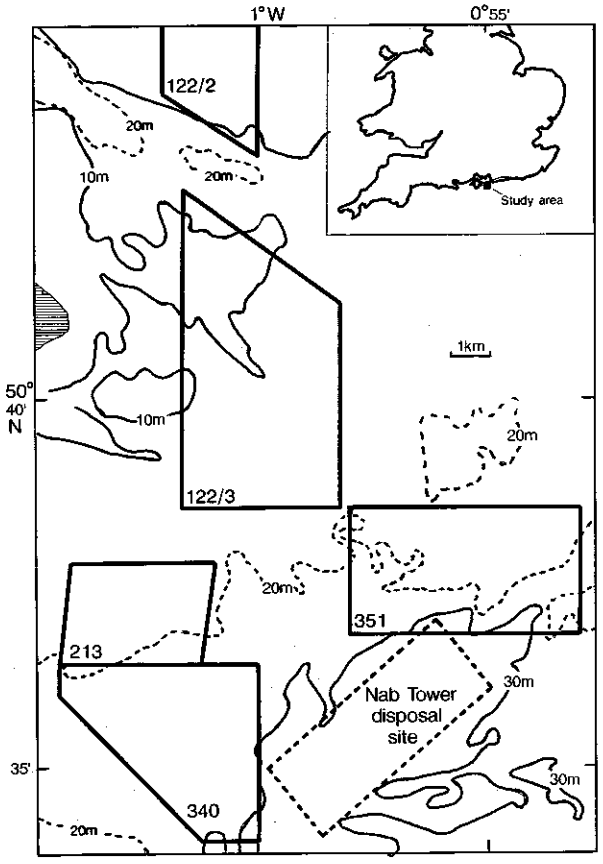


Figure 55. Location of the study area east of the Isle of Wight, including licensed extraction sites and the Nab Tower disposal site

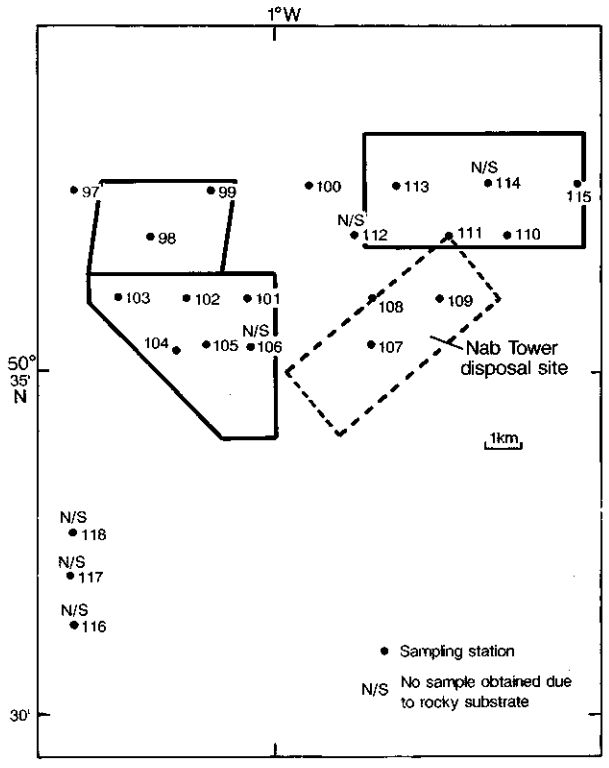


Figure 57. Location of stations sampled by anchor dredge during the 1989 survey of areas of aggregate extraction around the Isle of Wight

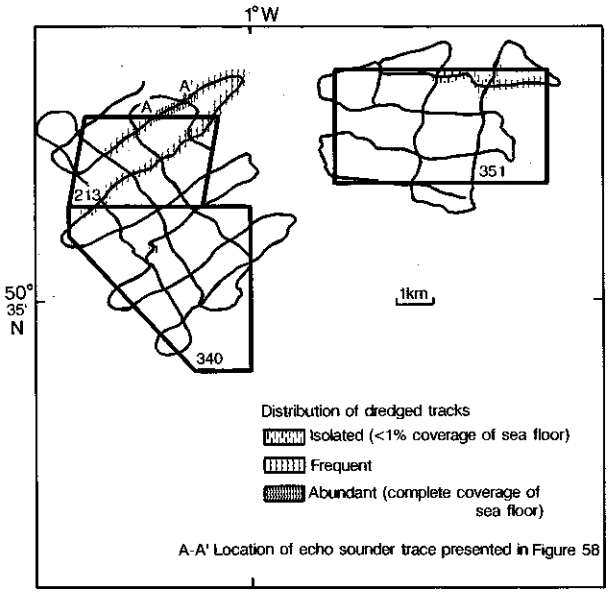
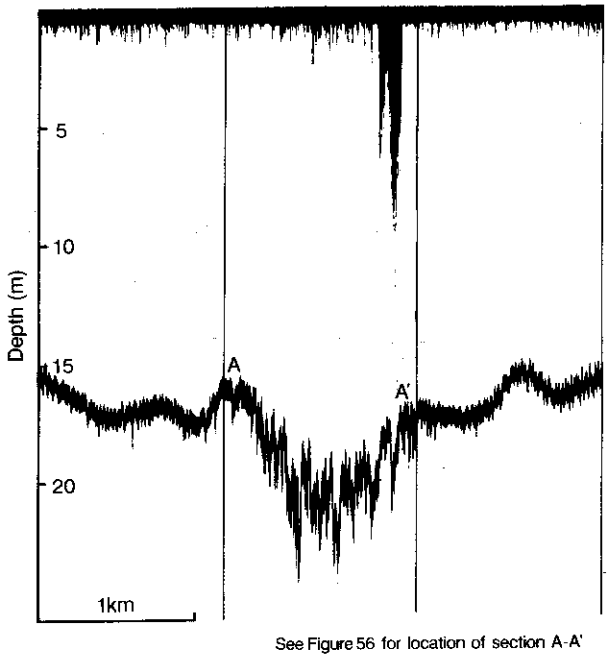


Figure 56. Side-scan track followed during the 1989 survey of areas of aggregate extraction around the Isle of Wight and the location of dredged tracks recorded on the survey



See Figure 56 for location of section A-A'

Figure 58. Echo-sounder record from the northern boundary of extraction area 213, in the Isle of Wight area in 1989

As a result of these findings, the Crown Estate Commissioners, who licence the extraction activities at these sites, were notified. A subsequent side-scan sonar survey of the area, commissioned by them, confirmed that unauthorised dredging had taken place outside area 213. In consequence, dredging was suspended while the matter was investigated further.

Of the twenty-two stations originally selected for study by anchor dredge, only sixteen provided samples; at the remainder, the dredge hit the rocky floor. Apart from these, the sediments are broadly similar across the survey area consisting of admixtures of sand and gravel with less than 10% mud.

Preliminary analysis of the benthic fauna shows the associations to be dominated by sessile epifaunal species, principally sponges, hydroids, bryozoans and ascidians, accompanied by the gastropod *Crepidula fornicata*, the bivalve *Nucula* sp. and occasionally other molluscs, polychaetes and crustaceans. Variations in total numbers of individuals (non-colonial taxa only) and total taxa retained on a 5 mm sieve are considerable but do not appear to be related to either dredging activity, at the areas of extraction, or disposal operations at the Nab Tower disposal ground (Figures 59 and 60). By contrast, the number of non-colonial ascidians sampled, though nowhere great, is preferentially distributed away from the areas of impact (Figure 61). The significance of this is yet to be ascertained but may be due to outwash fines, at the extraction sites, or mud, at the disposal ground, impeding the ascidians' filter feeding mechanism.

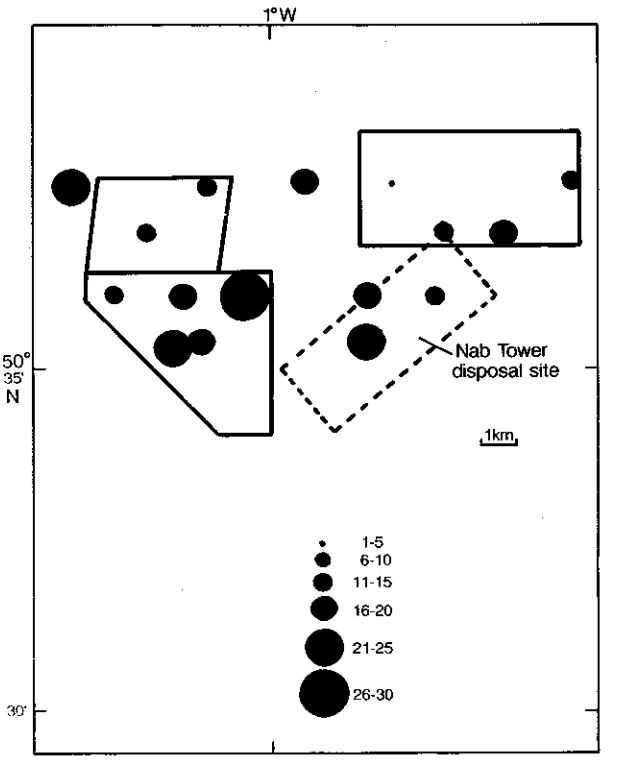


Figure 60. Total taxa in the > 5 mm fraction of samples taken by anchor dredge in the Isle of Wight area in 1989

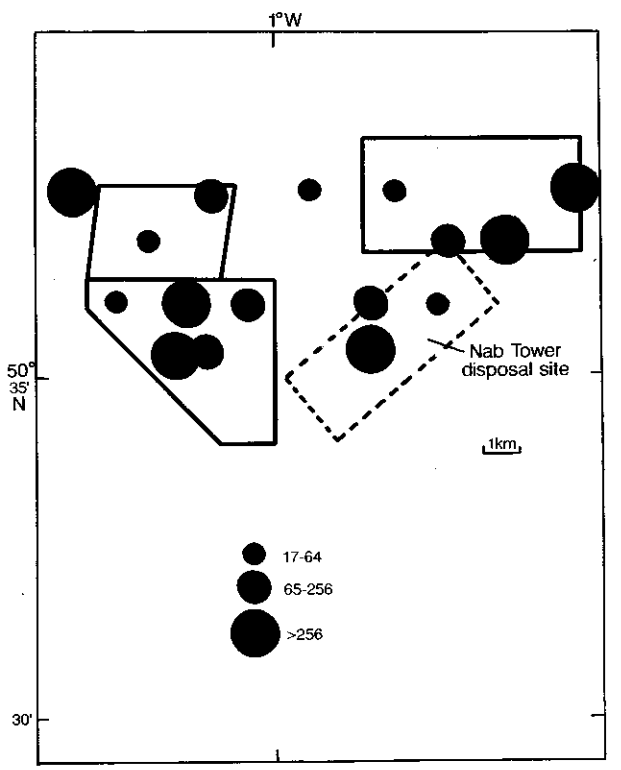


Figure 59. Total individuals (non-colonial taxa only) (expressed per 50 l of sediment) in samples taken by anchor dredge in the Isle of Wight area in 1989

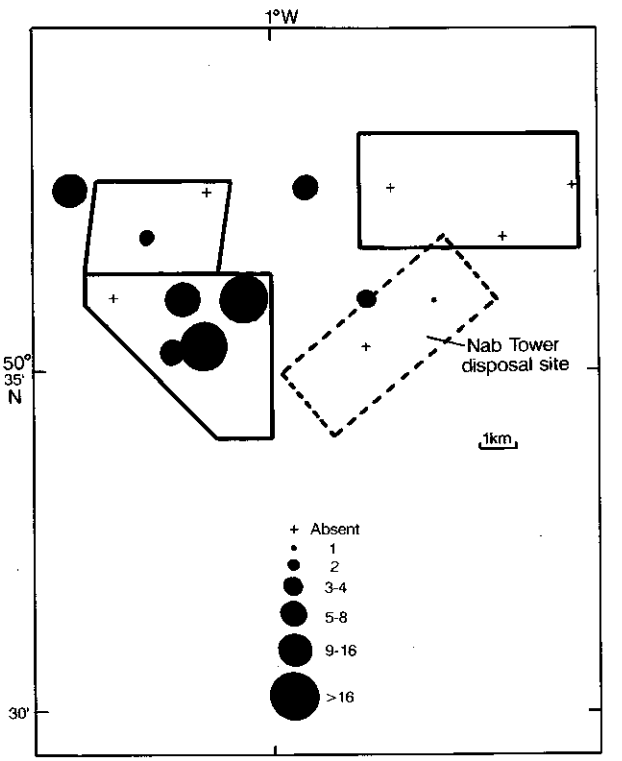


Figure 61. Total ascidians (non-colonial taxa only) (expressed as numbers per 50 l of sediment) in samples taken by anchor dredge in the Isle of Wight area in 1989

19. LICENSING OF DISPOSAL OF WASTES AT SEA

19.1 Introduction

This section gives information about the licensing of disposal of wastes at sea in the UK during 1988 and 1989 under Part II of the Food and Environment Protection Act 1985, (Great Britain — Parliament, 1985(a)). It fulfils an undertaking by the Government to report on the licensing, enforcement and monitoring of sea disposals.

An initial report, describing the licensing system, and giving information on wastes licensed in 1986 and 1987, was published in 1989 (MAFF/DAFS, 1989). For this and future reports, the information on licensing and enforcement for England and Wales, is brought together with the related monitoring activity in a single publication. For convenience, licensing statistics for Scotland and Northern Ireland are included in this Section to provide statistics for the UK as a whole.

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 the Food and Environment Protection Act 1985 (Great Britain — Parliament, 1985(a)). 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 Department of Agriculture and Fisheries for Scotland (DAFS). In Northern Ireland, the issuing of licences for disposal at sea is the responsibility of the Department of the Environment for Northern Ireland (DOE (NI)).

19.3 Enforcement

Scientists from MAFF's Directorate of Fisheries Research at Burnham-on-Crouch 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 81 inspections in 1988 and 42 in 1989, the reduction reflecting a temporary staff vacancy. 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 87 inspections in each of 1988 and 1989.

In Scotland, similar enforcement powers are held by staff of DAFS Marine Laboratory and by the Scottish Sea Fisheries Inspectorate (SSFI). The Laboratory made five enforcement visits in 1988 and eight in 1989 and a further three visits were made each year by the SSFI. In Northern Ireland, enforcement duties are carried out by officers of the Department of the Environment's Marine Environmental Protection Division. No inspectional visits to vessels engaged in sea disposal were considered necessary.

MAFF investigated one report of unlicensed disposal during 1988 and six in 1989. In three instances, responsibility for the disposal was established and warning letters were issued. In the other cases, it proved difficult to substantiate some reports and impossible to establish who had disposed of the waste. In Scotland, DAFS investigated two cases of alleged illegal disposal operations in 1988 and five in 1989 (some of these involved apparent breaches of licence conditions). Warning letters were issued in four cases. No reports of unlicensed disposals were received in Northern Ireland in either year.

During the course of 1989, most vessels disposing of sewage sludge and industrial waste were fitted with a tamper proof marine position recorder in order to provide independent evidence of their activities.

19.4 Report on licensing activities

Tables in this Section give details, over the period 1986-89, 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)).

During the period of this report, a number of enquiries were received by the licensing authorities about the possibility of sea disposal for a wide range of wastes.

Table 36(a). Liquid industrial wastes licensed for disposal at sea [1]

Licensed quantity (t) [2]		Company	Description of waste	Disposal site	Quantity deposited (t)	
1988/89	1989/90				1988	1989
20000	15000	Allied Colloids Ltd	Ammoniacal liquor	Tees	10091	4397
1999	-	Angus Fire Armour	Caustic sulphide	Liverpool Bay/ Tyne/Spurn Head	2801	0
8000	-	British Petroleum Development Ltd	Drilling mud	Nab Tower	2100	0
-	10000	British Petroleum Development Ltd	Drilling mud	Swanage Bay	-	9686
5000	1000	Chlor Chem Ltd	Acidic waste	Tees	4101	928
400	100	Chorley Bleaching	Oxygen demanding waste	Liverpool Bay/ Tyne	169	0
-	-	Dunlop Textiles	Industrial slurry	Liverpool Bay	50	-
-	-	Esso Petroleum Co Ltd	Caustic sulphide	South of Isle of Wight	4045	-
10000	8000	Fine Organics	Oxygen demanding waste	Tees	4889	5287
6000	6000	Fisons Plc	Oxygen demanding waste	Liverpool Bay/ Tyne	300	42
1000	1000	Fisons Plc	Oxygen demanding waste	Liverpool Bay/ Tyne	84	0
1800	800	Formica Ltd	Phenolic liquor	Tyne/Spurn Head	1268	960
6000	6000	Imperial Chemical Industries Ltd	Oxygen demanding waste	Liverpool Bay/Tyne/ Spurn Head	2474	826
165000	165000	Imperial Chemical Industries Ltd	Ammoniacal liquor	Tees	146227	161808
999	999	A H Marks and Co Ltd	Oxygen demanding waste	Tyne	629	102
3000	3000	Orsynetics Ltd	Caustic alkali	Tyne/Spurn Head	1591	1638
13	-	Pilkington Brothers Ltd	Oxygen demanding waste	Liverpool Bay	13	-
250	70	Robinson Brothers Ltd	Caustic sulphide	Tyne	78	146
42000	42000	Sterling Organics Ltd	Phenolic liquor	Tyne/Spurn Head	38501	33079
33000	33000	Tate and Lyle Refineries Ltd	Industrial slurry	South Falls/ Roughs Tower	28765	29402
950	999	Woolcombers Ltd	Oxygen demanding waste	Liverpool Bay/Tyne	622	153
6000	-	Yorkshire Water	Oxygen demanding waste	Spurn Head	946	-

Notes:
(1) Table excludes three 1987 licences valid into 1988, but on which no waste was disposed of in 1988.
(2) Licensed amount is for licences issued in 1988 and 1989 respectively.

Table 36(b). Summary of liquid industrial wastes licensed for disposal at sea [1]

Country	Licences issued		Licensed quantity (t)	Quantity deposited (wet weight) (t)	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England	1986	28	365995	283066	0.03	0.55	1.84	0.01	0.80	1.78	2.06
and Wales	1987	21	338871	264131	0.02	0.35	0.66	0.01	0.57	0.76	0.83
	1988	19	311411	249744	0.03	0.29	0.88	0.01	0.41	0.63	0.76
	1989	16	292968	248454	0.02	0.19	0.78	0.00	0.27	0.81	0.60
Northern Ireland	1986	0	0	650[2]	0.00	0.00	0.59	0.00	0.00	0.01	0.03
UK Total	1986	28	365995	283716	0.03	0.55	2.43	0.01	0.80	1.79	2.09
	1987	21	338871	264131	0.02	0.35	0.66	0.01	0.57	0.76	0.83
	1988	19	311411	249744	0.03	0.29	0.88	0.01	0.41	0.63	0.76
	1989	16	292968	248454	0.02	0.19	0.78	0.00	0.27	0.81	0.60

Notes
[1] No liquid industrial wastes were licensed or disposed of in Scotland in the period covered by this report, nor in Northern Ireland after 1986.
[2] Disposed of under licence issued in 1985.
Figures for 1986 and 1987 have been revised from those previously published in The Annual Report to Parliament 1986/87 (MAFF/ DAFS, 1989).
Licensed quantities : licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages deposited : relate to quantities deposited in a calendar year, which may be covered by two licences, including one from the previous calendar year.

In many cases, these enquiries were not followed by submission of applications. Two applications were refused in England and Wales in 1988 and five in 1989. No applications were refused in Scotland or Northern Ireland.

19.5 Licensing of liquid industrial wastes

Table 36(a) lists the liquid industrial wastes licensed for disposal at sea in the period covered by this report, together with the producer, the disposal sites used, the quantity licensed, and the quantity deposited. Table 36(b) summarises contaminants in the wastes. Locations of the disposal sites are given in Figure 62.

Following the Second International Conference on the Protection of the North Sea held in London on 24-25 November 1987 (Department of the Environment, 1987), the Authorising Departments required licence holders to take urgent action to identify and implement alternative means of dealing with their wastes, in order that sea disposal of these materials could be terminated as soon as practicable. Whereas there were 20 licences valid at the beginning of 1988, only 10 remained at the end of 1989.

19.6 Licensing of solid industrial wastes

Table 37(a) gives details of licences issued for the disposal of solid industrial wastes, and Table 37(b) details the contaminants in the material deposited. Figure 63 shows the locations of the relevant disposal sites.

The bulk of the material licensed was stone extracted from mines with the coal. The material is accepted internationally as inert in the marine environment. During the period of this report, British Coal sought a new licence for disposal at sea of 'tailings', a slurry of finely ground stone arising from the process of washing coal. After tests, MAFF was satisfied that the material would cause no toxic effect in the sea, and an assessment provided evidence that no alternative methods of disposal of the tailings were practically available. British Coal was licensed to dispose of tailings at sea from 1 October 1988.

As a condition of its licences in 1988 and 1989, British Coal was required to undertake further studies of the practicality of disposing of the colliery waste on land, in order that disposal on the beaches in Durham could be terminated as soon as possible.

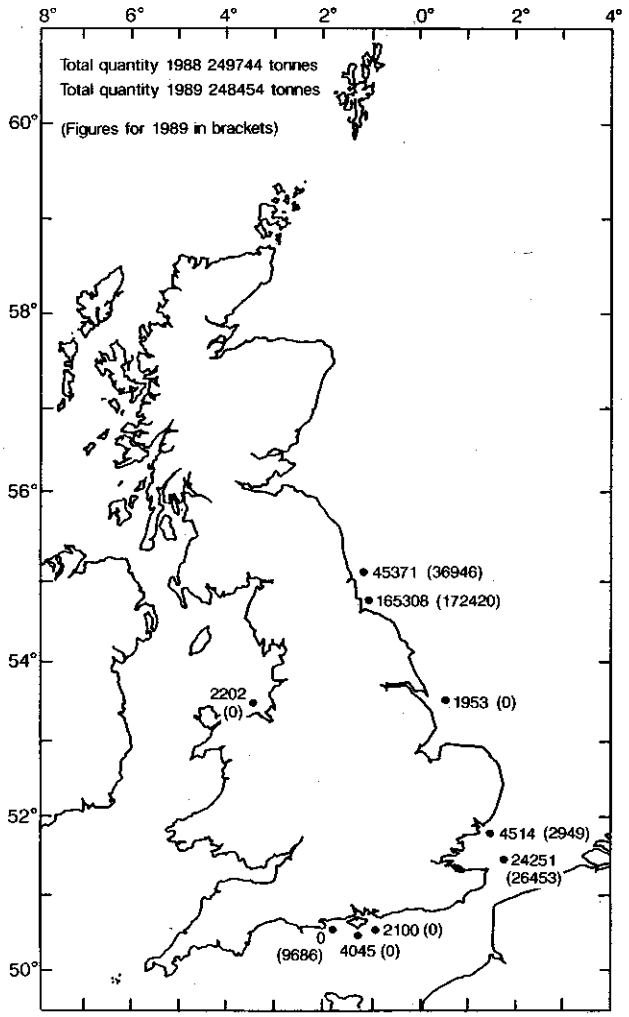


Figure 62. Liquid industrial wastes disposed of in 1988 and 1989. Figures for 1989 are in parentheses

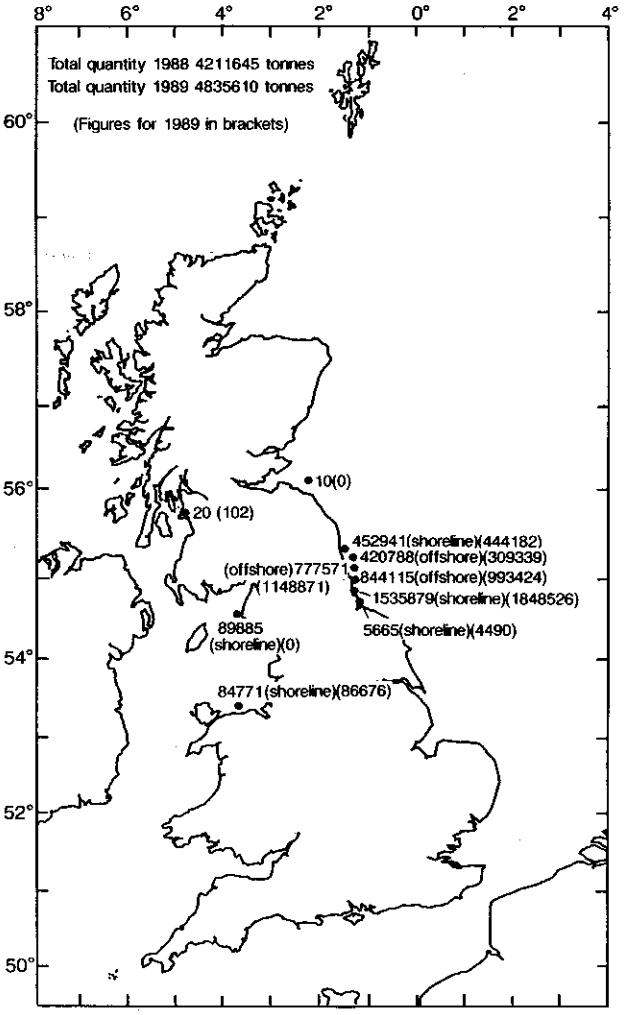


Figure 63. Solid industrial wastes disposed of in 1988 and 1989. Figures for 1989 are in parentheses

The Central Electricity Generating Board was required to take the necessary steps to terminate the disposal at sea of flyash as soon as practicable, in line with the declaration from the Second International Conference on the Protection of the North Sea (Department of the Environment, 1987).

In Scotland, the only licences for disposal of solid industrial wastes at sea were terminated in 1989. The materials involved were explosive waste which was disposed of in the Firth of Clyde and *Nephrops* offal disposed of off Dunbar.

19.7 Licensing of sewage sludge

Table 38(a) shows the details of licences issued for disposal of sewage sludge at sea. At the 1987 Second International Conference on the Protection of the North Sea, (Department of the Environment, 1987) the Government agreed to take urgent action to reduce the contamination by persistent, toxic or bioaccumulatable materials of sewage sludge deposited in the North Sea and to ensure that the quantities of such contaminants disposed of in the North Sea did not increase above 1987 levels. To apply this control, limits were set for a series of key contaminants deposited under each licence. The following table compares the aggregated figures (in tonnes) authorised for disposal in the North Sea in 1989 with the estimated quantity (in tonnes) at 1987 licensed levels:

Year	Hg	Cd	Cr	Ni	Cu	Zn	Pb
1987	1.2	3.7	56.2	19.4	133.6	468.2	146.4
1989	1.2	3.7	55.4	19.2	133.0	460.0	143.9

The licensed levels for 1989 did not exceed those for 1987. The scope for reducing the licensed levels during 1988/9 was small. This reflects the fact that contamination in UK sewage sludge generally is at relatively low levels. Moreover, since the bulk of these contaminants does not arise from industrial sources but comes from more general domestic sources, the timescale will be slow for achieving further reductions.

Total quantities of key metallic contaminants in sewage sludge licensed for disposal at sea are shown in Table 38(b). Figure 64 shows the location of the disposal sites for sewage sludge.

Table 37(a). Solid industrial wastes licensed for disposal at sea

Country	Licensed quantity ('000 t)		Company and source of waste	Description of waste	Disposal site	Quantity deposited ('000 t)	
	1988/89	1989/90				1988	1989
England and Wales			**British Coal Collieries**				
	700	725	Westoe	Minestone	Souter Point/ North Tyne	658	699
	1200	1400	Dawdon/Seaham	Minestone	Bankside, Seaham	1279	1233
	1150	1125	Wearmouth	Minestone	Sunderland	843	828
	150	150	Point of Ayr	Minestone	Foreshore at Point of Ayr	85	87
	650	525	Ellington	Minestone	Foreshore at Ellington	453	444
	600	525	Easington	Minestone	Foreshore at Easington	256	615
	-	-	Haig, Workington	-	Saltom Bay	90	-
	645	587	Westoe and Wearmouth	Mine tailings	Souter Point/ Sunderland	95	590
			National Power				
	4*	86+	Stella Power Stations	Power station pulverised flyash	Souter Point/ North Tyne	26	26
	650	790	Blyth Power Station	Power station pulverised flyash	Blyth A	421	309
	7.5	16	**Jacksons Fuels**	Washed sand	Foreshore at Hartlepool	5.7	4.5
Scotland	0.16	-	**HOMAC Seafoods**	*Nephrops* offal	Dunbar	0.010	0.00
	0.16	-	**Border Lairds**	*Nephrops* offal	Eyemouth	0.00	0.00
	0.25	-	**Imperial Chemical Industries Ltd.**	Explosives	Birch Point	0.020	0.102

Notes:
**Valid only for 10 weeks.*
+2 licences covering 19 months.
Licensed quantities : licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages deposited : relate to quantities deposited in a calendar year, which may be covered by two licences, including one from the previous calendar year.

Table 37(b). Summary of solid industrial wastes licensed for disposal at sea+

Country	Licences issued		Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1986	5	302800	4209783*	0.22	19	195	0.22	63	249	490
	1987	7	4290100	4275382*	0.23	20	197	0.23	63	251	486
	1988	10	5756200	4211615	0.26	21	193	0.25	63	244	470
	1989	10	5928917	4835508	0.28	24	200	0.24	68	245	481
Scotland	1986	2	410	19	0	0	0	0	0	0	0
	1987	1	250	117	0	0	0	0	0	0	0
	1988	3	570	30	0	0	0	0	0	0	0
	1989	0	0	102	0	0	0	0	0	0	0
UK Total	1986	7*	303210	4209802	0.22	19	195	0.22	63	249	490
	1987	8*	4290350	4275499	0.23	20	197	0.23	63	251	486
	1988	13	5756770	4211645	0.26	21	193	0.25	63	244	470
	1989	10	5928917	4835610	0.28	24	200	0.24	68	245	481

Notes:
+No Northern Ireland solid industrial wastes were licensed or disposed of in the period covered by this table.
**Tonnage deposited is covered by 5-year licences issued in 1983.*
Figures for 1986 and 1987 have been revised from those previously published in The Annual Report to Parliament 1986/87 (MAFF/ DAFS, 1989).
Licensed quantities: licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages deposited : relate to quantities deposited in a calendar year, which may be covered by two licences, including one from the previous calendar year.

Table 38(a). Sewage sludge licensed for disposal at sea

Country	Licensed quantity ('000 t) [1]		Company	Disposal site	Quantity deposited ('000 t) [1]	
	1988/89	1989/90			1988	1989
England and Wales	80	80	Anglian Water (Cliff Quay, Ipswich STW) [3]	Roughs Tower	81	74
	84	84	Anglian Water (Colchester STW)	Roughs Tower	58	57
	15	-	Anglian Water (Canvey and Benfleet STW)	South Falls	17	-
	210	150	Anglian Water (Tilbury STW)	Roughs Tower/South Falls	144	131
	564	554	Northumbrian Water (Howdon/Cramlington/ Washington, Chester le Street STWs)	Tyne/Spurn Head	440	466
	90	105	Northumbrian Water (Portrack/Guisborough/ Ayton/ Billingham STWs)	Tyne/Spurn Head	71	72
	1965	1965	North-West Water (Davyhulme and Warrington STWs)	Liverpool Bay	1625	1643
	5	5	North-West Water (Walney Island)	Liverpool Bay	3	3
	0.4	0.9	Severn-Trent Water (Harworth STW)	Spurn Head/Liverpool Bay	3	0.9
	263	307	Southern Water (Woolston/Portswood Millbrook/Slowhill Copse STWs)	Nab Tower	235	268
			Southern Water (Peel Common STW)	Nab Tower	1[4]	-
	58	58	South-West Water (Countess Wear STW)	Lyme Bay	47	46
	94	77	South-West Water (Camels Head, Ernsettle, Plympton, Ivybridge, Newton Ferrers, Radford, Saltash STWs)	Plymouth	78	78
	3600	4450[2]	Thames Water	Barrow Deep	4083	4135
	55	65	Welsh Water	Bristol Channel	55	62
	300	300	Wessex Water	Bristol Channel	218	236
	120	120	Yorkshire Water	Spurn Head	109	101
Scotland	500	500	Lothian Regional Council	Bell Rock/St Abbs Head	249	248
	750	1800	Strathclyde Regional Council	Garroch Head	1756	1693
N.Ireland	90	80	Dept of Environment (NI)	Belfast Lough	292[5]	329[5]

Notes:
1. All figures are for tonnage in wet weight.
2. The increased tonnage in 1989/90 consisted of additional water. Dry weight and contaminant loads were unchanged.
3. STW = Sewage treatment works.
4. Sludge was dumped from Peel Common under licence issued in 1987.
5. Includes 200,000 t yr^{-1} disposed of by DOE (NI) Water Services under an administrative authorisation.
For information on licensed quantities and tonnages deposited see footnote to Table 38(b).

Table 38b. Summary of sewage sludge licensed for disposal at sea

Country	Licences issued		Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in sludge (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1986	17	8702925	7532480	3.26	91	140	1.08	23	161	473
	1987	17	8682510	6692654	3.24	104	136	1.08	21	156	439
	1988	16	7503580	7267935	2.45	92	130	0.95	18	144	355
	1989	15	8321305	7373212	2.17	85	128	0.94	18	157	297
Scotland	1986	2	2300000	2002817	0.49	30	28	0.20	2	30	57
	1987	2	1900000	1978041	0.41	34	29	0.19	2	28	54
	1988	2	1250000	2004963	0.42	45	33	0.19	3	26	56
	1989	2	2300000	1940575	0.46	53	32	0.17	3	24	58
Northern Ireland	1986	0	0	285436	0.09	3	3	0.08	1	5	12
	1987	1	100000	309489	0.09	3	3	0.08	1	4	15
	1988	1	90000	291904	0.06	3	3	0.03	1	2	18
	1989	1	80000	329060	0.05	3	4	0.03	1	3	12
UK Total	1986	19	11002925	9820733	3.84	124	170	1.37	26	196	541
	1987	20	10682510	8980184	3.73	140	167	1.35	24	188	508
	1988	19	8843580	9564802	2.92	140	166	1.17	22	172	429
	1989	18	10701305	9642847	2.68	141	164	1.14	22	184	367

Notes : Northern Ireland and UK figures for tonnage deposited include 200,000 t yr^{-1} disposed of by DOE(NI) Water Services under an administrative authorisation.
Figures for 1986 and 1987 have been revised from those previously published in the Annual Report to Parliament 1986/87 (MAFF/DAFS, 1989).
Licensed quantities: licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages deposited: relate to quantities deposited in a calendar year, which may be covered by two licences, including one from the previous calendar year.

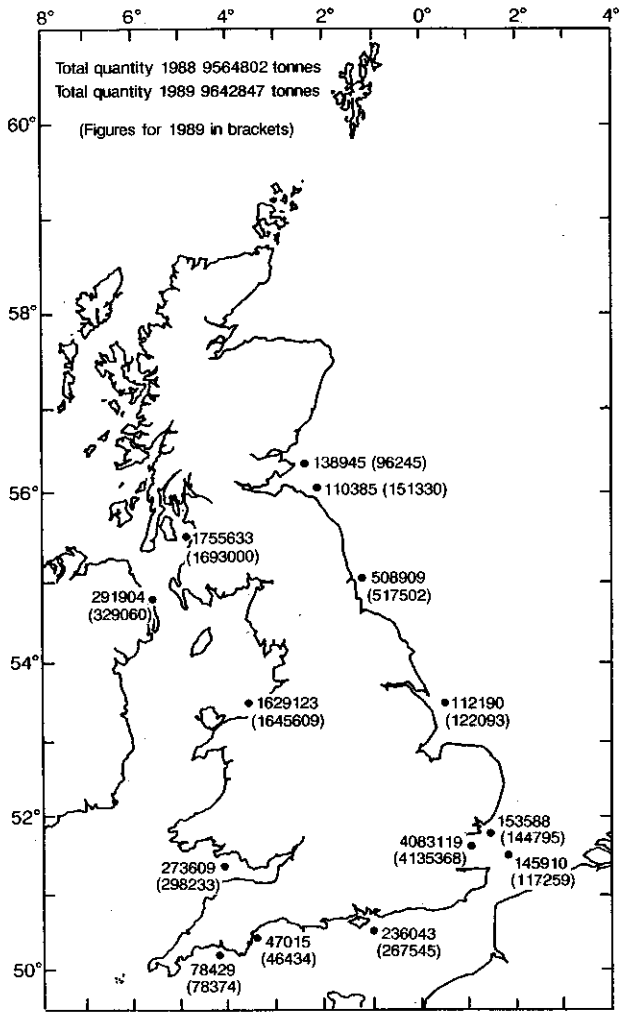


Figure 64. Sewage sludge disposed of in 1988 and 1989. Figures for 1989 are in parentheses

19.8 Licensing of dredged materials

The bulk of the dredged material licensed for disposal at sea is silt, but coarse sand and shingle can occur in 'maintenance' dredging, and shingle, cobbles, small rocks and heavy clay can be present in 'capital' spoils arising from development. Table 39 shows the numbers of licences issued, the quantity licensed, and the quantity deposited, together with figures for the quantity of a range of trace metals which enter the sea in the dredged materials. A proportion of the trace metals associated with dredged material occurs within the mineral structure or is tightly bound, such that generally it will not be available to marine organisms. Figure 65 (a - b) shows the main disposal sites used in 1988 and 1989 respectively.

19.9 Licensing of marine incineration

Table 40 gives details of licences issued and material incinerated. The waste itself does not enter the marine environment, but the table gives figures for the constituents of the waste which were expected to be released to the environment in exhaust gases.

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

Table 39. Dredged material licensed for disposal at sea

Country	Licences issued		Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in spoils (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1986	96	27917115	31698959*	29.7	1211	1230	13.5	848	2405	5338
	1987	107	28689146	38692856*	35.0	1410	1438	14.6	997	2876	6661
	1988	131	61645223	34691093	23.2	1165	1091	11	753	2199	5191
	1989	138	66408100	40810718	18.6	1234	1037	9.3	638	1877	4938
Scotland	1986	23	3460850	3516224	1.7	132	86	1.9	43	113	251
	1987	35	8813850	3927264	1.3	120	67	1.6	43	101	206
	1988	25	4148690	3506685	1.2	114	89	1.5	43	123	259
	1989	27	4252950	3154756	1.1	106	106	1.3	40	141	313
Northern Ireland	1986	2	1544000	1927575	3.3	41	48	0.4	40	68	498
	1987	5	338400	547052	1.2	6	4	0.0	6	4	15
	1988	9	1534200	1077023	0.1	10	7	0.1	8	11	26
	1989	6	383300	338521	0.1	2	2	0.0	2	2	7
UK Total	1986	121	32921965	37142758	34.8	1384	1363	15.8	931	2585	6086
	1987	147	37841396	43167172	37.5	1537	1509	16.2	1046	2981	6881
	1988	165	67328113	39274801	24.6	1289	1187	12.5	805	2333	5476
	1989	171	71044350	44303995	19.8	1343	1145	10.6	679	2021	5258

**Some deposits were covered by 5-year licences.*
Figures for 1986 and 1987 have been revised from those previously published in the Annual Report to Parliament 1986/87 (MAFF/DAFS, 1989).
Licensed quantities: licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages deposited: relate to quantities deposited in a calendar year, which may be covered by two licences, including one from the previous calendar year.

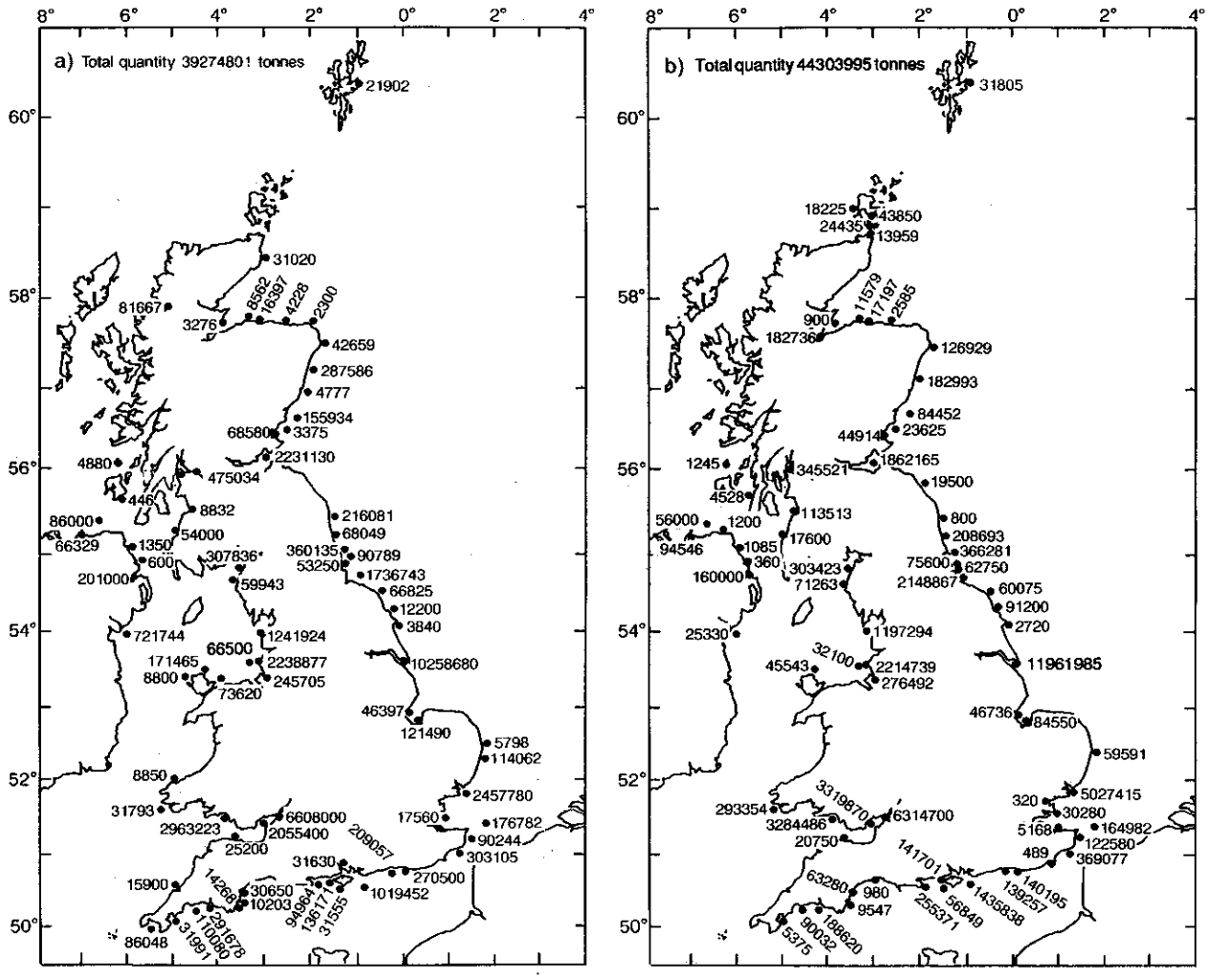


Figure 65. Dredged material disposed of: (a) in 1988 ; and (b) in 1989

Table 40. Licensing of marine incineration, quantities of waste burned and contaminants emitted

Licences issued		Licensed quantity (t)	Amount burned (t)	Estimated amount of contaminants released (kg)							Organo-halogens*
				Cd	Cr	Cu	Hg	Ni	Pb	Zn	
1986	20	7530	3754	<0.30	<1.9	<3.2	<0.20	<1.9	<2.3	<1.9	38.0
1987	13	10681	4551	0.90	14.1	65.0	0.40	92.0	15.2	13.8	45.0
1988	12	7497	5407	0.54	12.3	59.2	0.31	84.2	8.2	10.3	30.1
1989	8	6900	4152	0.65	19.6	8.0	0.13	8.7	4.4	8.3	8.3

**Based on an average destruction efficiency of 99.995%.*
Figures for 1986 and 1987 have been revised from those previously published in The Annual Report to Parliament 1986/87 (MAFF/DAFS, 1989).
Licensed quantities: licences are issued throughout a calendar year and are generally valid for twelve months.
Tonnages burned: relate to quantities burned in a calendar year, which may be covered by two licences, including one from the previous calendar year.

In response to the agreement at the Second International Conference on the Protection of the North Sea (Department of the Environment, 1987) that the quantities of waste incinerated should be reduced by not less than 65% by the beginning of 1991 and that marine incineration should be terminated by the end of 1994, licensees were required to look at alternative methods of dealing with their wastes.

19.10 Other materials deposited at sea

Under Part II of the Food and Environment Protection Act 1985 (Great Britain — Parliament, 1985), licences are also required for certain activities which do not involve the disposal of bulk 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 are involved with construction work below mean high water, the use of tracers, the application of biocides and burials at sea.

20. ADVICE ON FISHERY IMPLICATIONS OF PIPELINE DISCHARGES

20.1 Legislation

Prior to 1984, there was no statutory requirement for MAFF to be consulted regarding discharges of non-radioactive wastes to sea via pipelines. Discharges to estuaries were controlled by river boards under the Clean Rivers (Estuaries and Tidal Waters) Act 1960 (Great Britain — Parliament, 1960), while the Sea Fisheries Regulation Act 1966 (Great Britain — Parliament, 1966) allowed Sea Fisheries Committees to make bylaws regulating the discharge of certain categories of effluent to coastal water. However, on 26 July 1984, these earlier controls were re-enacted and extended by the implementation of Part II of the Control of Pollution Act 1974 (COPA) (Great Britain — Parliament, 1974(a)). Under Section 32 of COPA, the discharge of sewage, trade effluent and surface water, via a pipeline to any waters in or around the country, was only permitted in accordance with the terms of a consent. Consents were issued by the relevant regional water authority or, in the case of water authorities' own discharges, by the Department of the Environment. Where the discharge was to tidal water, the consenting authority was required to copy the consent application to MAFF (or for discharges in Wales, to the Welsh Office Agriculture Department (WOAD)) for comment on marine fishery implications. Salmon, trout, eel and freshwater fisheries remained the responsibility of the consenting authority and were not therefore normally subject to comment from MAFF or WOAD.

Part II of COPA was implemented in phases and initially only new discharges, or existing discharges containing certain hazardous substances, required consent. The exemption order was lifted in October 1986 and each of the previously exempt discharges was covered by an unconditional 'deemed' consent, most of which simply reflected the *status quo*. There are about 3000 'deemed' consents, all of which are to be replaced by full positively determined consents reflecting the needs of the receiving water. MAFF will thus for the first time have a statutory right to comment on the marine fishery implications of all discharges to tidal waters.

In September 1989, Part III of the Water Act 1989 (Great Britain — Parliament, 1989) replaced Part II of COPA, as the legislation for controlling discharges, and transferred consenting responsibility to the newly-formed National Rivers Authority (NRA). Under Schedule 12 of the Water Act, the NRA are obliged to send MAFF copies of all applications for consent to discharge effluent to estuaries (referred to in the Act as 'coastal waters') and territorial waters. This includes all waters from the fresh water limit up to three nautical miles offshore as well as discharges from land in England and Wales through an outfall ending outside the three-mile limit.

In addition to its obligations under COPA/the Water Act, MAFF also has certain responsibilities for outfall construction under Part II of the Food and Environment Protection Act 1985 (FEPA) (Great Britain — Parliament, 1985(a)). Because any construction extending into the sea causes permanent deposition of material below the high water mark, construction of a new sea outfall has to be licensed by MAFF. Conditions may be included in such licences to ensure that the marine environment is protected from unnecessary harm as a consequence of the pipe laying operations and/or the final presence of the outfall on the sea bed.

20.2 Consultation and assessment

Comments on fishery implications of discharges are passed to the consenting authority by the Marine Environmental Protection Division of MAFF (or for discharges in Wales by WOAD), based on advice provided by the Aquatic Environment Protection Division 2 (AEP2) of the Directorate of Fisheries Research, the local MAFF District Inspector of Fisheries and Sea Fisheries Committee. The consenting authority is required to consider any comments which are made to it within six weeks of the application being advertised.

MAFF's primary role as a statutory consultee under COPA/the Water Act is protection of fishery interests. This is taken in the widest sense to include protection of the life cycle and food chain of commercial species (which accordingly reflects the general well-being of the marine environment) and a consideration of the catchability and marketability of the product, including avoidance of adverse effects on the consumer.

Factors which are taken into account in carrying out an assessment vary depending on the type of effluent and receiving water, but may include the following :

(a) *biological parameters*

- resources in the area (existing and potential fisheries, spawning and nursery grounds, food organisms);
- toxicity, mutagenicity and bioaccumulation potential of the effluent or of its components;
- microbiological characteristics of the effluent and receiving water (bacteria, viruses, etc);

(b) *chemical parameters*

- oxygen demand of the effluent and receiving water;
- chemical characteristics of the effluent (persistence, solubility, volatility, etc);

(c) *physical parameters*

- solids content of the effluent and receiving water;
- temperature of the effluent and receiving water;
- degree of effluent treatment proposed;

(d) *hydrographic parameters*

- dilution and dispersion of the effluent;
- water movements, etc;

(e) *standards and guidelines*

- national policy;
- international agreements (EC Directives, North Sea Conference Agreement, Paris Convention, etc.)

The number of applications sent to MAFF for comment has increased steadily since COPA was implemented. Table 41 shows the number of each type of application assessed in 1988 and 1989.

Table 41. Applications for consent to discharge assessed by MAFF in 1988-89

Year	Sewage (including storm and emergency overflows)	Trade effluent	Surface water	Total
1988	132	53	60	245
1989	214	32	53	299

Two of these applications were recommended for refusal: one was a peat-based material resulting from construction of a power station which, following tests in the laboratory, was considered likely to result in significant fouling of fishing gear; the other was leachate from a waste disposal site which was contaminated with a number of materials liable to cause carcinogenic effects, taint fish flesh, or bioaccumulate.

Additional treatment, tighter consent limits, or inclusion of a toxicity condition were requested for about 10% of applications. A further 15% were only aproved subject to certain provisos, such as adequate precautions being taken to prevent contamination by hazardous materials. Of the remaining applications, about 15% were for continuation of existing, previously unconsented, discharges, which were thought to be contributing to poor water quality in fishery areas. In such cases, the consent was normally approved subject to a strict time limit (usually two years) to give the regulating authority time to determine the improvements which should be carried out. Care was taken to ensure that consent limits were set as low as possible in relation to existing effluent quality in order to prevent any further deterioration during this period. Most of the remaining applications were recommended for unconditional approval on the grounds that they were for existing discharges which were not causing any problems, or new discharges which were unlikely to adversely affect fishery interests. Many of these were for uncontaminated surface water, small sewage discharges to areas with little or no fishery interests, or improvements to existing discharges.

The final consent contains details of the grid reference of discharge point, the nature, composition and volume of discharge, plus any additional conditions considered necessary by the consenting authority or additionally recommended by MAFF. All consents are copied to MAFF and details entered onto a computer database linked to that used to store details of wastes licensed for disposal at sea under FEPA.

20.3 Types of effluent

20.3.1 Sewage

As Table 41 shows, the majority of discharge applications sent to MAFF for comment are for sewage effluent. These can range from the septic tank effluent from a single household (typically less than 1 $m^3 d^{-1}$) to discharges from urban areas which may have dry weather flows in excess of 100 000 $m^3 d^{-1}$, together with associated storm and emergency overflows.

There are still many short outfalls around the coast which have been discharging untreated sewage since they were constructed in the last century. These are now considered to be unsatisfactory and are gradually being replaced by inland treatment works or properly designed long sea outfalls, often of several kilometres in length. Modern outfalls are fitted with diffusers which provide rapid initial dilution of at least 1:100. Computer predictions, validated by field studies, are used by the design engineers to ensure that the diffusers are sited in areas where water movements will promote subsequent dispersion and prevent the effluent plume from returning to the shore. When discharged in this way, sewage and sewage effluents are rapidly decomposed by natural processes (salinity, sunlight, microbial breakdown, etc.). The impending EC Directive on Municipal Waste Water Treatment (European Communities, 1990) is likely to require at least primary treatment of all significant sewage flows before discharge.

Domestic sewage and treated sewage effluent are of low acute toxicity to aquatic organisms and, when discharged via a properly designed modern long sea outfall, do not pose a risk to mobile species such as fin fish or crustacea. The fine screens (typically 6 mm), which must now be fitted to all new outfalls, coupled with rapid dilution and dispersion, minimise the risk of fouling of nets and pots by recognisable sewage solids. Provided that the discharge point is carefully chosen to give good dilution and avoid the discharge plume contaminating any sensitive beds of bivalve molluscan shellfish (see below), this type of discharge does not pose a risk to fisheries.

One of the main fishery implications of domestic sewage discharges is the risk of contamination of bivalve shellfish such as oysters, mussels, cockles and clams. These feed by filtering large volumes of water, retaining and accumulating solid particles, including any bacteria and viruses from sewage. The microorganisms do not harm the shellfish themselves but may accumulate to such levels that the shellfish consumer is at risk of contracting gastro-intestinal ailments. Where the degree of contamination is not excessive, purification (e.g. relaying for 48 h in water sterilised using ultra-violet light), or cooking the shellfish, can reduce the risk of public health problems to acceptable levels. However, when levels of contamination are very high, such procedures are inadequate and harvesting of the shellfish may have to be prohibited completely, or preceded by additional cleansing by relaying in clean water for a period of several weeks prior to the usual purification procedures. The sale of shellfish from contaminated waters is regulated by imposition by the local health authority of a 'closure order' under the 1934 or 1948 Public Health (Shellfish) Regulations (Great Britain — Parliament, 1934 and 1948) which specifies the degree of treatment required to render the product acceptable for human consumption. MAFF advice, regarding sewage discharges to waters which support bivalve shellfisheries, has been aimed at ensuring that water quality is not allowed to deteriorate to such an extent that the degree of treatment has to be increased. The long-term aim is for all major shellfish waters to be clean enough for the shellfish to be harvested without purification. The pending EC Directive on Municipal Waste Water Treatment requires resources at risk to be taken into account when deciding whether primary or secondary treatment is required. The measure should assist in achieving this long-term aim.

Although domestic sewage may cause adverse effects associated with biochemical oxygen demand (BOD), solids and ammonia, these tend to occur only in relatively enclosed areas such as estuaries. For this reason, sewage discharges to estuaries are usually given primary or secondary treatment at a sewage treatment works. Primary treatment is a physical process involving settlement to reduce suspended organic solids and BOD. Microbial contaminants, which are associated with the solids, will also be reduced by primary treatment, usually by about an order of magnitude. This may be followed by secondary treatment in which the effluent is subjected to biological processes to further reduce its BOD. Although this reduces microbial levels by about another order of magnitude, the effluent from secondary treatment works still contains significant concentrations of bacteria and viruses. When such effluents are discharged to estuaries where dilution is limited, there may still be a significant risk of contamination of bivalve shellfish, such that discharge of sewage effluent via a sea outfall will be required.

One way of reducing the microbial content of sewage is by disinfection. However, most of the currently available disinfectants are limited in their effectiveness against viruses, carry the risk of formation of hazardous compounds by reacting with organic matter in the sewage, or are too expensive to be used on a routine basis. In May 1989, a 3-4 year programme of

field trials to investigate the efficiency and environmental hazards of all potential sewage disinfection methods commenced in south-west England. MAFF (including staff from the Burnham-on-Crouch Laboratory) is represented on the working group which is planning these trials, and has therefore been able to advise on appropriate biological and chemical monitoring of effects and also to ensure that the studies are carried out in areas where they will not pose a risk to fisheries.

Many sewers receive trade effluents and care needs to be taken to ensure that these do not result in the sewage discharge containing significant concentrations of hazardous materials. The factors which are considered when assessing direct discharges of trade effluent to sea (see below) also apply to sewage containing significant trade effluent inputs.

20.3.2 Trade effluent

Trade, or industrial, effluents vary considerably in their composition and potential effect on fisheries. Although a detailed assessment of fishery implications of each new industrial discharge to sea is carried out by MAFF, this is normally based on data provided by the consenting authority, rather than studies carried out at MAFF's laboratories.

Acute toxicity is one of the main factors to be taken into account when considering whether or not an industrial effluent is likely to adversely affect fisheries. Where the effluent contains only one or two chemicals, it is often possible to predict its toxicity from published data. However, if the effluent is composed of a complex mixture of materials, it may be necessary to request that toxicity tests be carried out on appropriate marine species. The toxicity information is then considered in conjunction with data on the dilution available, both at the outfall and at the point of discharge. A waste is only considered acceptable for discharge to sea if there is a large margin of safety between its acute toxicity to the most sensitive species in the area and the concentration discharged to the environment. MAFF may recommend that the consent should contain limits, not only for concentrations of potentially hazardous substances, but also for the rate and the period of discharge (e.g. ebb tide only). Monitoring of effluent quality to check for compliance with consent is carried out by the consenting authority. In some cases (e.g. where it is not possible to define the composition of the waste because of its variability), the consent may be expressed in terms of acute toxicity to a representative species and monitoring to check compliance will be demonstrated by a regularly executed toxicity test.

A waste which is not in itself acutely toxic may nevertheless pose a hazard because it contains substances which are particularly persistent, bioaccumulative or mutagenic. Many of these substances have been identified under national or international agreements or regulations such as the UK 'Red list' (Anon., 1990), Annex A of the Paris Convention (Great Britain — Parliament, 1974(b)), or Lists I and II of the EC Directive on Dangerous Substances (European Communities, 1976). Fishery interests are protected, in part, by ensuring that such substances are rapidly diluted to levels well below their Environmental Quality Standard (EQS), but also by ensuring a long-term aim of reducing inputs of these materials by taking advantage of technical advances in effluent treatment methods. In addition to monitoring the effluent to determine compliance with consent conditions, the consenting authority (the NRA in England and Wales) is required to monitor concentrations of many of these materials in the receiving water.

Wastes which have a high solids content (e.g. mine tailings) may have a low toxicity, but risk damaging fisheries by blanketing areas of the sea bed and thus making them unable to support commercial species or the organisms on which they feed. Such wastes are only considered acceptable for discharge to areas where there are commercial fisheries, if detailed studies on environmental impact have shown that there is no significant risk. The consent for such a waste would normally be conditional on the pre- and post-discharge monitoring of physical and biological effects being carried out, with a requirement for remedial action to be taken if unacceptable adverse effects occur.

20.3.3 Surface water

Surface water usually consists of rain water drained from roofs and impervious surfaces. Where this comes from domestic areas, there is little risk to fisheries, provided that the water passes through an interceptor to remove any oil which is picked up from roads. However, surface water from industrial premises may be contaminated by spilt chemicals, in which case, controls similar to those imposed on discharges of trade effluents may be required.

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

Report Section	Project	Staff
1.	Contaminants in marine fish and shellfish	A Franklin
2.	Contaminants in marine mammals	R J Law C R Allchin
3.	Diseases in fish	D Bucke S W Feist B D Rackham S Swaine
4.	PSP toxin in mussels	D J Alderman
5.	Oyster embryo bioassay	P Matthiessen J E Thain
6.	Mercury in sea water	M J Waldock C F Fileman
7.	Specific organic compounds in sea water:	
	(i) Xylene, styrene, chlorobenzene and phthalate esters	R J Law T W Fileman
	(ii) Alpha and gamma HCH	C R Allchin C A Kelly
8.	Organochlorine contaminants in sediments	C R Allchin C A Kelly
9.	Oil and gas exploration:	
	(i) Ravenspurn gas field studies	R J Law J A Osborne
	(ii) Southern gas field studies	R J Law J A Osborne
10.	Loss of the *MV PERINTIS*	R J Law J E Portmann C R Allchin
11.	Tecnazene studies	C R Allchin
12.	TBT studies	M J Waldock M E Waite
13.	DAS* : Background to field assessment studies	S M Rowlatt
14.	DAS : Sewage sludge	S M Rowlatt H L Rees
15.	DAS : Dredgings	S M Rowlatt
16.	DAS : Colliery waste	S M Rowlatt D S Limpenny
17.	DAS : Beam trawl surveys	H L Rees
18.	DAS : Aggregate extraction	R G Lees
19.	DAS : Licensing and enforcement	C M G Vivian G Boyes
20.	DAS : Pipeline discharges	F L Franklin

* *DAS = Disposal at Sea*

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

A2.1 Metals

(a) Mercury

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

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

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

(b) Cadmium

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

The JMP guidelines for cadmium in mussels are as follows:

Level	Mussel tissue	Approximate equivalent
Lower	< 2 mg kg^{-1} dry weight	(≡ < 0.4 wet weight)
Medium	2-5 mg kg^{-1} dry weight	(≡ 0.4-1.0 wet weight)
Upper	> 5 mg kg^{-1} dry weight	(≡ > 1.0 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 net exceed 2.0 mg kg^{-1} wet weight, and lead in shellfish 10.0 mg kg^{-1} wet weight.

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

(d) Copper

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

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

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

(e) Zinc

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

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

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

A2.2 Pesticide 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 γHCH. The 'expected' values are up to 0.05 mg kg^{-1} wet weight for each of α and γHCH in fish liver.

(c) Dieldrin

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

(d) Total DDT

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

(e) PCBs

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

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

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

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