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Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1994

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Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1994

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FOREWORD

This is the seventh report in MAFF's series of publications on 'Monitoring and Surveillance of Non-Radioactive Contaminants in the Aquatic Environment'. The report describes work carried out, mainly by staff of the *Directorate of Fisheries Research's laboratory Burnham-on-Crouch, in 1994. The final two sections of the report deal with activities related to the control of disposal of material to the marine environment and these activities also involved staff of the Sea Fisheries Inspectorate and the Marine Environmental Protection Division of MAFF.

Following the pattern set by recent reports in the series, this issue gives information obtained from the main monitoring projects, together with a summary of the data and a brief interpretation of the significance of the findings. Until recently much of the monitoring work was concerned with establishing reliable baseline data on the concentrations of contaminants in water, sediments and marine organisms - particularly those likely to be consumed as food by Man. Although some studies of this nature continue - for example, those described in Sections 5 and 7, most of the work is now either directed at following trends in contaminant concentrations in areas where they are sufficiently high to cause concern, for example the work described in Sections 3 and 4 or in investigating possible new problems, for example the studies described in Sections 6 and 7.

The report also gives some examples of the developmental and investigative work that goes into ensuring the most effective monitoring techniques are used and that every attempt is made to explore possible reasons for unexpected monitoring results. Thus for example Section 1 of this report describes how sediment bioassay techniques are being introduced into routine monitoring programmes with a view to identifying problems which conventional chemical analysis, with its normally limited range of determinands, would miss. Similarly Section 9 describes work being undertaken to try and establish the source of the unexpectedly high concentrations of certain contaminants, notably chlorinated biphenyls, in the blubber of small cetaceans resident in Cardigan Bay.

As with other recent reports in this series, in addition to the general monitoring programmes, this report also includes information on licensing of deposits at sea, which are controlled under Part II of the Food and Environment Protection Act 1985 (FEPA II). These are presented in Sections 11 to 13 of the present report. Section 14 describes the related licensing activities and includes details of licences issued by the authorities for Scotland and Northem Ireland in order to provide statistics for the UK as a whole. The final section of the report relates to activities undertaken to protect marine fisheries interests from the potentially damaging effects of discharges from pipelines directly into the sea, e.g. from sewage works.

Thus the report covers a broad range of studies and reflects the diverse nature of the many contaminants and situations that merit investigation, as well as the monitoring which forms part of national and international programmes. The programme of work continues. It seeks to meet ongoing obligations and concerns but also to be dynamic. In this way MAFF is able to ensure that an accurate picture is maintained of the trends and patterns of non-radioactive contamination of UK waters, and that new issues and ones of topical concern are investigated promptly and effectively.

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* *The Directorate of Fisheries Research of the Ministry of Agriculture, Fisheries and Food will become an Executive Agency of MAFF on 1 April 1997. In preparation for this the name changed to the Centre for Environment, Fisheries and Aquaculture Science on 13 January 1997.*

BACKGROUND TO THE WORK

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

The most direct of these is a duty to license and thereby control the deposit of material in the sea. The statutory requirements for this are laid down in Part II of the Food and Environment Protection Act (1985) (FEPA) (Great Britain - Parliament, 1985) as amended by the Environmental Protection Act, 1990. Scientific and Technical aspects of licence applications are evaluated by inspectors at the Burnham Laboratory and regular visits are made to licence holders to discuss the terms of licences and to ensure that conditions stipulated in the licence are being met. An integral part of the scientific assessment process is the design and conduct of a marine disposal site monitoring programme. The environmental conditions at marine disposal sites are monitored by the Laboratory's Environmental Assessment team. Compared with the results of more general Environmental Quality Monitoring, the results of these monitoring programmes allows MAFF to take action if unexpected or unacceptable impacts do occur and also provide the basis for future licensing policy. Disposal at sea is regulated internationally by the Oslo and London Conventions and the monitoring programme allows the Government to demonstrate that it takes these obligations for the wider marine environment seriously. Some examples of monitoring investigations and licensing related data are included in this report.

Under the Water Resources Act (1991) (Great Britain - Parliament, 1991(a)), MAFF has the responsibility of statutory consultee for all discharges to controlled (tidal) waters. Each discharge must be consented and the consent will specify the substances the discharge may contain as well as limit concentrations and loads as appropriate. Fishery implications are assessed at Burnham-on-Crouch and take into consideration such factors as resources in the area, toxicity of the effluent, local hydrographic conditions and any standards set out in national policy or EC Directives.

Scientists at the laboratory also contribute to the control of pollution in other areas of industrial activity affecting the marine environment, including the offshore oil and gas industry and marine aggregate extraction.

To ensure that its advice on these specific activities covers the widest possible range of conflicts of interest, the Burnham-on-Crouch Laboratory conducts a more general programme of monitoring, which forms the basis of MAFF's contribution to the National Monitoring Programme as well as fulfilling international monitoring commitments. Samples of sea water, sediment and marine organisms are collected each year in which a wide range of chemical determinands is measured. These include, trace metals, organochlorine pesticides and polychlorinated biphenyls (PCBs). This programme provides valuable information on the quality of the marine environment, against which areas such as disposal sites can be assessed. It also aims to ensure that where potentially harmful substances occur, concentrations do not reach levels that present an unacceptable risk to either marine organisms or human consumers of marine produce. Most of this work is carried out in accordance with procedures agreed under the auspices of organisations such as the International Council for the Exploration of the Sea (ICES) and the Oslo and Paris Commissions.

All pesticides used in the UK have to be approved by the Ministry's Pesticide Safety Directorate and the Advisory Committee on Pesticides (ACP). The Burnham-on-Crouch Laboratory provides advice on risk assessment to the ACP and has conducted extensive research on pesticide leaching and run-off, the data from which are being used to validate improved computer models of leaching and run-off. In relation to particular measures of control, the Laboratory undertakes monitoring to demonstrate UK compliance with the European Union Directive on mercury discharges and similar requirements under the Paris Commission. It also undertakes work to assess the effectiveness of control measures taken, under Part II of FEPA, 1985, to regulate the use of antifouling paints containing tributyltin (TBT) oxide.

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

1. THE USE OF BIOASSAYS TO ASSESS THE QUALITY OF MARINE SEDIMENTS

1.1 Whole sediment bioassays

1.1.1 Introduction

Prior to 1992, the toxicity of sediments for monitoring purposes was assessed using the oyster embryo bioassay, which exposed the larvae to sediment elutriates. Because of the limitations of this technique, e.g. it did not mimic the exposure of sediment dwelling animals, whole sediment bioassays were developed using organisms that live in and feed directly on the sediment. In 1992, a bioassay using the polychaete lugworm *Arenicola marina* was used for the first time on board a MAFF research vessel; in 1993 the *Arenicola* bioassay was again used on board ship and an additional bioassay using the crustacean amphipod *Corophium volutator* was used to test the same sediments in the laboratory. In 1994, both bioassays were deployed on board a MAFF research vessel (*RV CIROLANA*, 10 June-1 July 1994) and the *Corophium* bioassay was also used on *RV CIROLANA*, 8-29 July 1994 to assess the toxicity of sediments from the Bremerhaven Transect in the North Sea (Stebbing and Dethlefsen, 1992).

In addition to monitoring the toxicity of sediments from estuaries and offshore sites around England and Wales, a programme to assess the usefulness of bioassays in the evaluation of dredged material was initiated in 1994. Twenty two samples of dredged material were obtained from 5 dock areas around southern England and Wales (see Table 3) and a battery of four tests was deployed to assess toxicity. The two infaunal species *A. marina* and *C. volutator* were used to test the whole-sediment phase and two pelagic species – the copepod *Tisbe battagliai* and the mollusc larvae *Littorina littorea* were used to test the elutriate phase, which mimics to some extent the conditions of the overlying water when dredge spoil is being extracted and deposited at sea.

1.1.2 Methods

Arenicola marina bioassay

Full details of the methods used in this bioassay have been described previously (MAFF, 1994(a)). In summary, worms are placed in plastic containers containing a layer of test sediment and overlying water. The worms burrow into the sediment and after a 10-day exposure are sieved out and the number of survivors recorded. The number of casts produced on the surface of the sediment is counted daily to obtain a measure of the feeding rate of the worms; a sublethal endpoint as opposed to an acute endpoint.

Corophium volutator bioassay

Full details of the methods used in this bioassay can be found in MAFF, 1995. Adult *C. volutator* are placed in glass beakers containing a layer of test sediment and overlying water. The amphipods burrow into the sediment and after a 10-day exposure period the sediment is sieved and the number of surviving animals counted.

Sediment elutriate bioassays

The elutriate bioassays recorded survival of *T. battagliai* copepodites and larvae of *L. littorea* after a 48-hour exposure. The sediment elutriates were prepared by adding 600 ml filtered (1 µm) natural sea water to 200 ml of test sediment and shaking for 3 hours on an orbital shaker. The mixtures were allowed to settle overnight and the overlying water filtered through a Whatman GF/C filter. Four replicates each containing five test organisms were exposed to the neat elutriate in cell wells (capacity 5 ml/well).

1.1.3 Results

RV CIROLANA, 10 June-1 July 1994

A total of 46 sediment samples were tested for toxicity using either the *C. volutator* or *A. marina* bioassay or both. Summary results are shown in Table 1 (full results are shown in Appendix 1). Tests 1 and 2 of the amphipod bioassay were deployed at sea and Test 3 was carried out in the laboratory. For the lugworm bioassay, Test 1 was deployed at sea while Test 2 was carried out in the laboratory.

Sediments from four stations showed acute toxicity to *C. volutator* – two in the North-East (Tees Redcar Jetty and Off Tyne/Tees, with 100 and 60 % mortality respectively) and two in the Mersey Estuary (Eastham Lock and Tranmere Oil Terminal, with 63 and 43% mortality). Only one sediment, Tyne South Ferry Landing, produced a significantly lethal response (compared to the control) in *A. marina*, but a total of 18 stations significantly reduced feeding rate compared to the control. The majority of these 18 stations were concentrated in the north-east estuaries of the Tyne, Wear and Tees.

Table 1. RV CIROLANA, 10 June-1 July 1994: 10-day sediment bioassays using the lugworm Arenicola marina and the amphipod Corophium volutator. (Only results significantly different from the control are shown - full results are given in Appendix 1)

Station	Actual position		Location	*Corophium*	*Arenicola*	
				% mortality	% mortality	Total casts in 10 days
				Test 1	Test 1	
			Shoebury Sands (Control)	10	13	123
13	54° 59.82' N	1° 26.47' W	Tyne (by South Ferry landing)	17	**73	*16
14	54° 59.17' N	1° 27.86' W	Tyne (buoy off Jarrow Slake)	20	27	*80
15	54° 59.09' N	1° 31.49' W	Tyne (Hebburn)	33	20	*84
16	54° 58.09' N	1° 36.25' W	Tyne (Tyne Bridge)	23	27	*80
23	54° 55.05' N	1° 21.43' W	Wear (Sandy Point)	40	27	*71
24	54° 54.58' N	1° 22.87' W	Wear (Wearmouth Bridge)	57	33	*29
25	54° 54.80' N	1° 24.24' W	Wear (Queen Alexandra Bridge)	33	7	*55
26	54° 44.03' N	0° 53.13' W	Off Tees	37	20	*79
27	54° 39.85' N	1° 04.56' W	Tees (anchor)	23	33	*88
				Test 2	Test 2	
			Shoebury Sands (control)	13	20	94
35	54° 37.79' N	1° 09.72' W	Tees (Phillips approach buoy)	30	27	*24
36	54° 37.40' N	1° 09.34' W	Tees (Redcar jetty)	**100**	27	*50
41	54° 49.95' N	1° 20.05' W	Off Tyne/Tees	**60**	7	93
44	51° 59.99' N	2° 19.90' E	Thames (Outer Gabbard)	17	0	*11
				Test 3		
			Shoebury Sands (control)	3		
57	50° 04.86' N	3° 00.12' W	Central Channel/Perintis 7	7	0	*45
67	50° 01.98' N	4° 21.99' W	Off Plymouth Sound/Perintis 8	7	20	*53
71	48° 30.04' N	7° 59.98' W	Western Approaches/Perintis 12	17	0	*33
85	53° 20.11' N	2° 57.22' W	Mersey (Bromborough E1 buoy)	10	33	*14
86	53° 19.40' N	2° 56.85' W	Mersey (Eastham Lock)	**63**	20	84
87	53° 23.70' N	2° 59.70' W	Mersey (Tranmere oil terminal)	**43**	13	107
88	53° 24.56' N	3° 00.48' W	Mersey (Seacombe Ferry)	0	0	*53

Note: Corophium mortality significantly different from control (ANOVA, p<0.05) indicated in ***bold***
*Arenicola mortality significantly different from control (ANOVA, p<0.05) indicated by '**'*
Arenicola casting significantly different from control (Dunnets t-test, p<0.05) indicated by ''*

RV CIROLANA, 8-29 July 1996

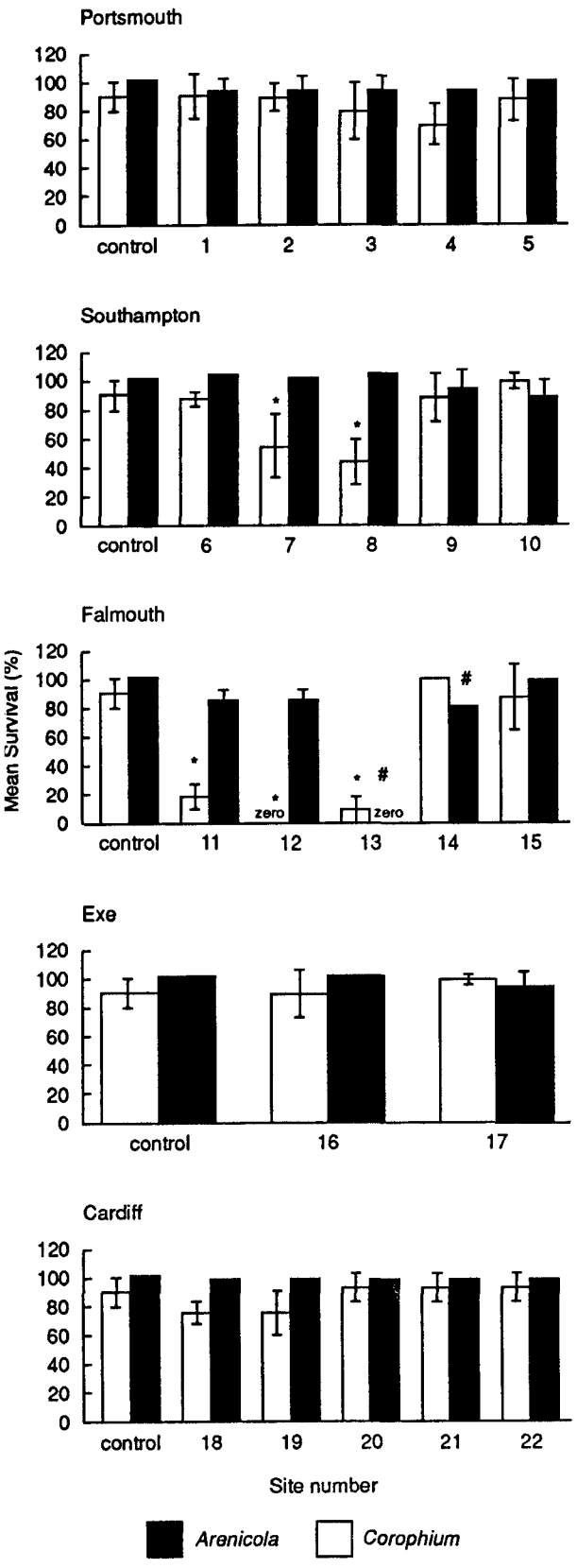
Nine stations along the Bremerhaven Transect were tested for toxicity using *C. volutator*. Results are shown in Table 2. None of the stations were found to be acutely toxic; the highest mortality was 7% at Station 4, within the German Bight.

Dredged material

Results of the whole sediment bioassays are shown in Figures 1 and 2. A significant reduction in the survival of *C. volutator* was found at a total of 5 sites; two in Southampton and three in Falmouth. Mortality was greatest at Sites 11 to 13 at Falmouth, ranging from 80%

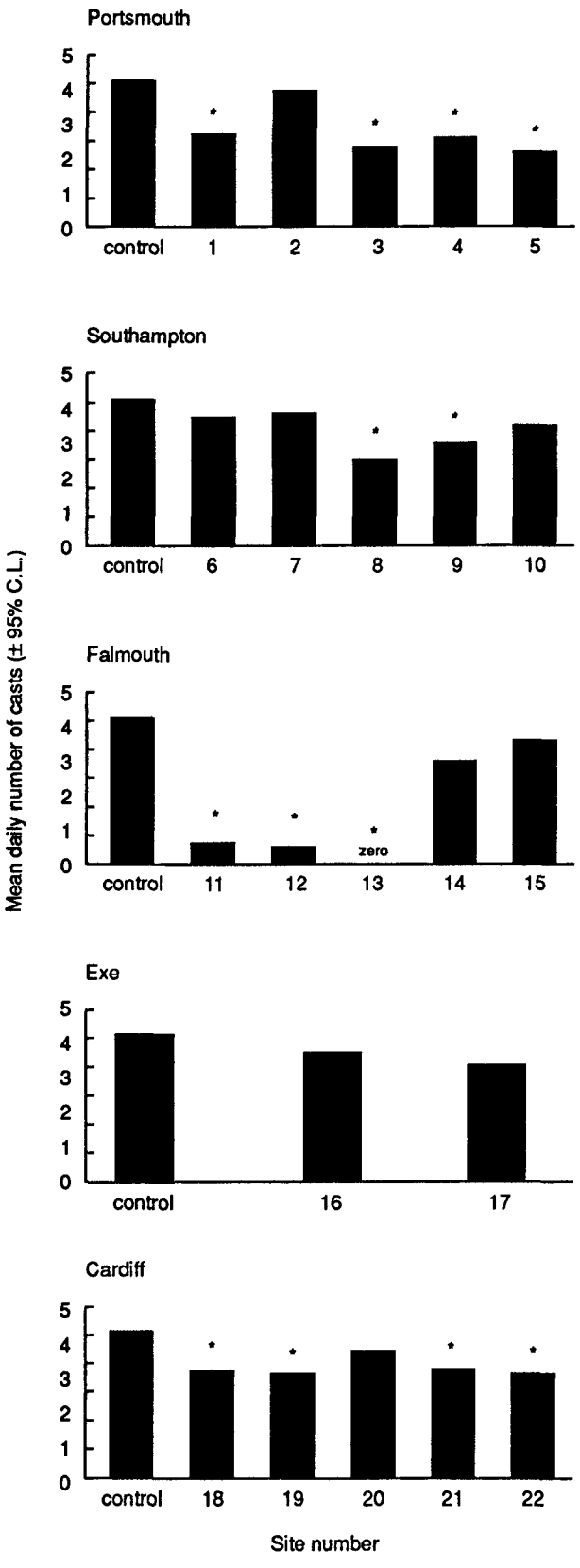
Table 2. RV CIROLANA, 8-29 July 1994: 10-day sediment bioassay using Corophium volutator

Station	Actual Position		Location	*Corophium* % mortality
			Shoebury Sands (control)	0
1	54° 03.98' N	8° 07.51' E	Bremerhaven Transect	0
2	54° 01.98' N	8° 02.85' E	Bremerhaven Transect	0
3	54° 00.08' N	8° 00.06' E	Bremerhaven Transect	0
4	54° 00.86' N	7° 48.85' E	Bremerhaven Transect	7
5	54° 06.53' N	7° 24.02' E	Bremerhaven Transect	3
6	54° 25.01' N	6° 14.90' E	Bremerhaven Transect	3
7	54° 50.11' N	5° 34.96' E	Bremerhaven Transect	0
8	55° 06.02' N	2° 00.07' E	Bremerhaven Transect	3
9	55° 29.96' N	4° 10.00' E	Bremerhaven Transect	0



Note: Significant differences from control denoted by * (Corophium) and # (Arenicola)

Figure 1. Survival of Corophium and Arenicola in samples of dredged material from five dock areas in southern England and Wales



Note: Comparisons significantly different from control casting (ANOVA, $p<0.05$) indicated by *

Figure 2. Effect of dredged material samples on the feeding rate of Arenicola marina

to 100%. No significant effects on survival were observed at Portsmouth or Cardiff.

Survival of *A. marina* was only significantly reduced at two sites – 13 and 14, both at Falmouth, with Site 13 having 100% mortality. Feeding rate of *A. marina*, expressed as the number of casts produced on a daily basis, was significantly reduced at a total of 13 sites (Figure 2), four of which were below 50% of the control.

Results of the sediment elutriate tests are shown in Table 3. A total of 6 sites elicited >40% mortality over 48 hours with *Tisbe*. These included one at Portsmouth (Site 1), three at Southampton (Sites 8, 9 and 10), and the two sites at Exe (Sites 16 and 17). Only three sites elicited a toxic response (>40% mortality) to *Littorina* larvae, and these were the same three sites at Southampton (8, 9 and 10) that elicited a response from *Tisbe*.

Table 3. Results of sediment elutriate bioassays used on dredged material samples

Area	Site No.	% mortality after 48h	
		Tisbe	*Littorina*
Shoebury	Control	0	0
Portsmouth	1	45	0
50°48.0'N 01°07.0'W	2	5	0
	3	0	0
	4	5	0
	5	10	0
Southampton	6	0	0
50°54.0'N 01°24.0'W	7	10	0
	8	75	100
	9	90	100
	10	70	75
Falmouth	11	0	0
50°03.0'N 05°03.0'W	12	5	0
	13	0	0
	14	10	15
	15	0	0
Exemouth	16	65	0
50°40.0'N 03°27.0'W	17	100	0
Cardiff	18	0	0
51°28.0'N 03°10.0'W	19	30	0
	20	15	0
	21	10	0
	22	0	0

1.1.4 Discussion

Differential sensitivity was observed between *C. volutator* and *A. marina* to the test sediments taken both on the research vessel and as part of the dredged material programme. This was also demonstrated in the 1993 survey (MAFF, 1995). *C.volutator* was more sensitive than *A. marina* in acute terms but feeding behaviour was, as expected, the most sensitive of the endpoints used. Of the *RV CIROLANA* 10 June-1 July 1994 samples, none of the four sediments which demonstrated significant toxicity to *C. volutator* had a significant effect on the mortality of *A. marina*, and only one (Tees Redcar Jetty) significantly reduced feeding behaviour. In the dredged material study, Site 13 at Falmouth was the only site that was acutely toxic to both species, and sublethal effects on the feeding of *A. marina* were found at three of the sites which caused significant mortality in *C. volutator*.

This indicates that the two test organisms are responding with different susceptibility to the concentrations of contaminants and/or physical properties of the sediments. It is therefore necessary to adopt a 'battery of tests' approach to sediment toxicity testing, particularly within a regulatory framework such as assessing dredged material for disposal at sea.

2. SURVEYS OF CONTAMINANTS IN MARINE MAMMALS

Resident or semi-resident populations of bottlenose dolphins occur in two main locations within the UK – in Cardigan Bay in west Wales and in the Moray Firth in north-eastern Scotland. Approximately one hundred and twenty individual animals have been identified in each area, with no evidence of movement between the two populations, which are approximately 1000 km apart (Evans, 1992; Lewis and Evans, 1993). Neither of the areas is either particularly heavily populated or industrialised, and both in fact are considered to be relatively clean (MAFF, 1993; Morris *et al.*, 1989; SOAFD, 1993). Table 4 lists concentrations of chlorobiphenyls (CB) (ΣICES7 = sum of ICES 7 congeners 28, 52, 101, 118, 138, 153 and 180) in the blubber of nine bottlenose dolphins and thirty-three porpoises, of which three dolphins and twelve porpoises were from Cardigan Bay and these were analysed by the MAFF Burnham Laboratory. The remainder were from the Moray Firth and were analysed at the Scottish Office Agriculture, Environment and Fisheries Department (SOAEFD) Marine Laboratory, Aberdeen. The analyses were conducted in two separate laboratories, but these have participated jointly in both investigative programmes and intercomparison exercises and the data are demonstrably directly comparable (Hall *et al.*, 1992). Further information on these can be found in the original publications (Law and Allchin, 1994; Morris *et al.*, 1989; MAFF, 1994(b); Wells *et al.*, 1994).

From Table 4, it can be seen that some porpoises and all the dolphins from Cardigan Bay exhibit higher concentrations of CBs than those from the Moray Firth – in the case of the bottlenose dolphins, around an order of magnitude higher. A similar pattern was also apparent in data for the pesticide DDT and its metabolites (Law and Allchin, 1994). To explain this phenomenon an attempt was made to model the uptake of chloro-biphenyls by bottlenose dolphins using a bioenergetics-based approach, estimating changes in body burden during growth (Law *et al.*, 1995). These were then related to published estimates of food intake to yield

Table 4. Concentrations of chlorobiphenyls (ΣICES7) in the blubber of dolphins and porpoises from Cardigan Bay and the Moray Firth

Species/Area	ID no.	Age (years)	Length (cm)	Sex	Lipid %	ΣICES7 (mg kg^{-1} lipid)
Bottlenose dolphins						
Cardigan Bay	B-1	10 months	187	F	38	262.0
	B-2	ca. 4	277	M	78	115.0
	SW1991/42	23	342	M	68	107.0
Moray Firth	FA1TT	4	260	F	67	10.7
	FA2TT	9	330	F	49	16.9
	FA3TT	14	300	F	44	10.5
	FA4TT	17	310	F	57	1.4
	FA5TT	1	150	F	44	3.0
	MA1TT	4	240	M	57	12.3
Common (harbour) porpoises						
Cardigan Bay	P-1	-	95	-	83	8.8
	P-2	-	150	-	87	15.0
	P-3	-	103	-	81	21.4
	P-4	-	136	-	87	26.9
	SW1990/50A	<1	73	M	79	8.7
	SW1990/65	8	148	M	76	30.5
	SW1990/72	-	106	F	81	23.5
	SW1990/107	-	176	F	87	5.5
	SW1990/111	-	137	M	93	0.3
	SW1991/61	<1	72	F	52	3.4
	SW1991/136	-	144	M	82	64.8
	SW1992/7	-	131	F	88	11.8
Moray Firth	FA2PP	2	112	F	59	4.1
	FA3PP	-	117	F	78	1.0
	FA4PP	-	113	F	23	5.3
	FA5PP	-	113	F	67	1.6
	FA6PP	-	147	F	26	2.9
	FA10PP	-	120	F	83	2.2
	FA11PP	1	125	F	80	1.1
	FA12PP	-	-	F	88	1.0
	FA13PP	8	-	F	88	1.0
	FA14PP	-	120	F	88	1.9
	MA2PP	-	117	M	89	11.9
	MA3PP	1	117	M	93	5.5
	MA4PP	-	110	M	77	5.8
	MA7PP	-	147	M	79	3.8
	MA8PP	-	100	M	74	1.0
	MA13PP	7	139	M	88	7.0
	MA15PP	1	112	M	85	2.0
	MA16PP	7	141	M	86	5.3
	MA17PP	2	123	M	83	1.8
	MA19PP	24	148	M	87	16.7
	MA20PP	5	142	M	87	8.2

estimates of the extent of contamination of the dolphins' diet. Table 5 shows the range of estimates yielded by these calculations for three individual persistent chlorobiphenyl congeners (CBs 138, 153 and 180) compared with those determined in a sample of mackerel taken from Cardigan Bay in 1992 and analysed on a 'whole-body' basis (for detail see Law *et al.*, 1995).

From this, it is clear that the concentrations found in this single sample of mackerel are too low, in some cases by more than an order of magnitude, to supply the

Table 5. The range of concentrations of CBs predicted by the uptake model compared with those determined in mackerel (mg kg^{-1} wet weight)

	CB138	CB153	CB180
Predicted	0.042 to 0.11	0.058 to 0.16	0.022 to 0.06
Actual	0.007	0.005	0.004

chlorinated biphenyls that it is inferred are being consumed by dolphins in Cardigan Bay. Coastal bottlenose dolphins are reported to eat a wide variety of fish, squid and octopus, as well as shrimp and other crustaceans, although four or five locally abundant species usually form the majority of the diet (Evans, 1987; Martin, 1990). The specific diet of bottlenose dolphins in Cardigan Bay is not known in detail at present, nor are all of the locations from which food is taken. There is therefore, currently insufficient information to explain the source of the PCBs in dolphins from Cardigan Bay, and further studies must be conducted of the concentrations of contaminants in dolphins and fish from the area, and of the feeding locations and diet of the dolphins in order to try to explain their contaminant burdens.

3. MERCURY AND PCBS IN FISH AND CADMIUM IN MUSSELS

3.1 Introduction

In 1994, sampling was concentrated in those areas around England and Wales where previous monitoring (MAFF, 1991 and 1994) had indicated that the level of some contaminants could be above the range normally anticipated, as indicated by reference to guidelines issued by the Joint Monitoring Group (JMG) of the Oslo and Paris Commissions. This group developed guidelines for 'lower', 'medium' and 'upper' concentration values for contaminants, based on a statistical ranking of the results submitted by all of the countries which participated in the monitoring programmes 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 Annex 2. The presence of a contaminant in the 'upper' category does not necessarily imply any risk to either human health or to the environment, but countries are asked to provide data from affected areas on a biennial basis to ensure that up-to-date information is available. In 1994, data were obtained for this purpose on mercury and PCBs in fish from Liverpool and Morecambe Bays, and on cadmium in mussels from the Humber area and off Whitehaven. In the past, information has also been obtained on the cadmium levels in mussels taken from the Bristol Channel region where 'upper' level values have been recorded in some areas. In recent years problems have been encountered in obtaining the samples from the areas normally sampled, due to hydrographic changes. Information on contaminants in mussels will now be made available through monitoring carried out in 1995 to supply data for the EC Shellfish Hygiene Directive (91/492/EEC) (European Communities, 1991(a)).

3.2 Methods

3.2.1 Sampling

Samples of each of the main roundfish and flatfish species, cod (*Gadus morhua*), whiting (*Merlangius merlangus*), dab *(Limanda limanda*), flounder *(Platichthys flesus*), plaice *(Pleuronectes platessa*) and sole (*Solea solea*) were obtained, where available, and the muscle analysed in bulk for mercury. The livers of roundfish accumulate organochlorines to the greatest extent and this tissue was analysed for PCBs.

Samples of mussels (*Mytilus edulis*) were analysed in bulk for cadmium.

3.2.2 Analysis

Full details of the analytical methods employed by the Burnham-on-Crouch Laboratory are given in Jones and Laslett (1994) for metals and Allchin *et al.* (1989) for organochlorine pesticides and PCBs.

For consistency with earlier surveys, PCBs were quantified on a formulation basis using Aroclor 1254 as a standard. However, this will be the last year that this information is obtained as data are now normally provided on an individual chlorobiphenyl basis, initially on the seven (CBs 28, 52, 101, 118, 138, 153 and 180) on the priority list of the International Council for the Exploration of the Sea (ICES) (ICES, 1986). To enable comparisons to be continued to be made with earlier data, a provisional fish liver conversion factor has been calculated, using 1991-93 white fish liver data, plotting Aroclor 1254 concentrations against ICES 7 values (Figure 3). The data indicated that the ICES 7 value was approximately 30% of that calculated on the basis of Aroclor 1254 as a standard. Preliminary equivalent 'upper' categories for the concentration of ICES 7 CBs could therefore be suggested as >1.5 mg kg^{-1} wet weight for roundfish liver and >0.3 for flatfish liver.

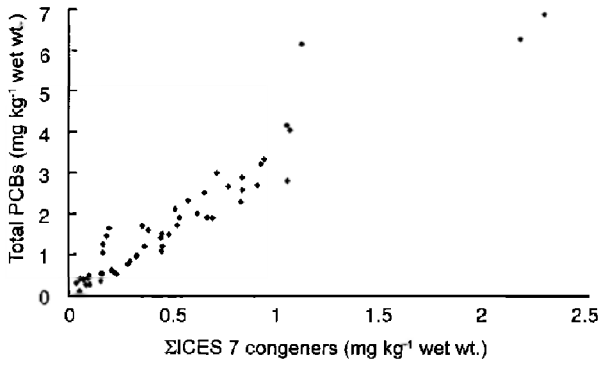


Figure 3. Total PCB and ICES 7 congeners in white-fish liver (1991-1993 data)

3.3 Results

The concentrations of mercury found in fish muscle are listed in Table 6, PCB residues in fish liver in Table 7 and cadmium in mussels in Table 8 .

3.3.1 Mercury in fish muscle

In 1994, no fish species from either Liverpool or Morecambe Bays contained mercury in the 'upper' (>0.30 mg kg^{-1} wet weight) JMP category (Table 6). Both areas are subject to discharges from chloralkali plants and thus to provisions of EC Council Directive 82/176/EEC (European Communities, 1982). Data are submitted regularly to indicate compliance with an Environmental Quality Standard of 0.30 mg mercury per kilogramme of representative wet fish flesh. For this purpose, the mercury concentration of each individual fish species is weighted according to the quantity of that particular species landed from the area in question. The weighted mean concentration of mercury calculated for 1994 for Liverpool Bay was 0.14 mg kg^{-1} and for Morecambe Bay 0.15 mg kg^{-1}; the standard was therefore met at both sites.

3.3.2 PCBs in roundfish liver

As in previous years, the concentration of PCBs in cod liver (Table 7) was only in the 'medium' JMP category (2.0-5.0 mg kg^{-1} wet weight) and this species will not be monitored as part of this particular programme in future years. Of the two whiting samples, that for Morecambe

Table 6. Concentrations of mercury in fish muscle in 1994 (JMP 'upper' level >0.30 mg kg^{-1} wet weight)

Area	Species	Number of fish analysed	Mean length (cm)	Concentration of mercury in fish muscle (mg kg^{-1} wet weight)
Liverpool Bay	Cod	23	30.9	0.10
	Whiting	25	30.8	0.27
	Dab	120*	22.7	0.20
	Flounder	45	32.5	0.17
	Plaice	25	26.8	0.13
	Sole	40	25.6	0.14
Morecambe Bay	Cod	-+	-	-
	Whiting	25	31.6	0.27
	Dab	25	26.1	0.15
	Flounder	25	32.0	0.23
	Plaice	25	27.9	0.09
	Sole	50	25.8	0.17

* *Large numbers of dab analysed for UK National Monitoring Programme*

\+ *No cod caught in Morecambe Bay area in 1994*

Table 7. Concentrations of PCB (Aroclor 1254) in fish liver in 1994 (JMP 'upper' level, for Aroclor 1254, >5.0 mg kg^{-1} wet weight for cod liver - value used for all round fish)

Area	Species	Number of fish analysed	Mean length (cm)	Concentration of PCB in fish liver (mg kg^{-1} wet weight)	
				Aroclor 1254	ICES 7 CBs
Liverpool Bay	Cod	23	30.9	2.6	0.74
	Whiting	25	30.8	5.9	1.7
Morecambe Bay	Whiting	25	31.6	4.7	1.4

Table 8. Concentrations of cadmium in Mytilus edulis in 1994 (JMP 'upper' level, >5.0 mg kg^{-1} dry weight)

Area	Number of mussels analysed	Mean length (cm)	Concentration of cadmium (mg kg^{-1} dry weight)
Humber (Bull Fort)	50	4.4	4.0
Humber (Cleethorpes)	50	4.6	5.2
Whitehaven (Parton)	50	5.3	18.7

Bay was also only in the medium JMP category. As this was also the case in 1992, further monitoring appears unnecessary. The concentrations of PCBs in whiting liver from Liverpool Bay remained in the 'upper' category (>5.0 mg kg^{-1}). The measured ICES 7 values of 28-30% that of the Aroclor concentration were as would be anticipated from the conversion factor calculated earlier. The Burnham laboratory now determines concentrations of 25 individual CBs (see Appendix 2). In the 1994 samples above, the total concentration of these was 44-46% of that calculated on the basis of Aroclor 1254 as a standard.

Analyses of the corresponding whiting muscle tissue was undertaken. As in previous years (MAFF 1993, 1995) concentrations were very much lower. The highest 1994 Aroclor value of 0.025 mg kg^{-1}, was only in the 'medium' JMP category.

3.3.3 Cadmium in mussels

Results are summarised in Table 8. The samples from Cleethorpes and off Whitehaven remain in the JMP 'upper' category (>5.0 mg kg^{-1} dry weight) and monitoring will continue at these sites as required. The results are not considered to have any significance from a human health point of view – the Whitehaven area does not in fact support a commercial fishery and the sample is simply used as an index of relative contamination.

3.4 Conclusions

Monitoring in 1994 in areas previously found to be most heavily contaminated by mercury, cadmium and PCBs indicated that fewer samples contain concentrations of these contaminants in JMG 'upper' categories. The Joint Monitoring Programme is now being revised and data requirements for the Oslo/Paris Commissions will differ in future. It is anticipated, however, that monitoring of heavily contaminated areas will continue under the UK's National Monitoring Programme (MPMMG, 1994).

4. HEAVY METALS IN OYSTERS FROM THE FAL ESTUARY

In January 1992, the Fal Estuary in Cornwall was subject to a major discharge of highly acidic, metal-rich mine water, as a result of the flooding of the Wheal Jane tin mine, following its closure in March 1991. The area supports a number of commercial shellfish beds and in view of the consequent threat to these beds and hence, human consumers, MAFF had taken samples of oysters before the anticipated overflow and since then, conducted regular monitoring of heavy metal levels in the main commercially produced shellfish, native oysters (*Ostrea edulis*).

Details of the programme and the results for the first two years of monitoring were included in earlier reports in this series (MAFF, 1993 and 1994). Monitoring was terminated at the end of March 1995 after a review of the three-year dataset. A summary of the results and the conclusions of the review, are given below.

4.1 Method

From January 1992 samples of oysters were collected, initially at fortnightly but subsequently at less frequent intervals, from two main stock areas, Parsons Bank and Messack Point and analysed for a suite of eight metals (Cd, Cu, Cr, Fe, Hg, Ni, Pb and Zn). The samples of oysters which were collected in October 1991, provided information on contaminant levels prior to the discharge event.

4.2 Results

Throughout the study period, concentrations of Cr, Hg, Ni and Pb were generally at, or below, the limits of detection of the method of analysis used. Figure 4 shows the time series of concentrations for the remaining metals. Copper and Fe showed some fluctuations, but there was no significant increase

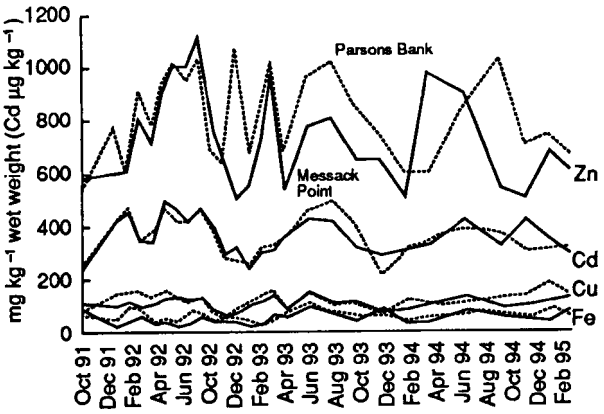


Figure 4. Time series of concentrations of metals in oysters from the Fal Estuary

compared with levels prior to the discharge despite the obvious contamination of water courses. The concentrations of Zn and Cd however, varied considerably throughout the period of monitoring, with levels ranging between 491-1098 mg kg^{-1} wet weight and 0.21-0.49 mg kg^{-1} wet weight, respectively.

4.3 Discussion

During the study period, water quality was monitored by the National Rivers Authority (NRA). Results from Devoran Bridge, down stream of the Wheal Jane mine, indicated that concentrations of Cd and Zn rose in January 1992, at the height of the discharge, to ~600 µg l^{-1} and 450,000 µg l^{-1} respectively, but then fell rapidly over the following two months. The concentrations of these metals in oysters also began to increase in January and this increase continued through until August despite reduction of concentrations in the water to pre-discharge levels by the end of March. Concentrations in the water remained at, or below, pre-discharge levels throughout the remainder of the monitoring period, yet concentrations in the oysters continued to fluctuate widely. The results therefore, did not indicate any direct correlation between the levels of metals in the oysters and those in the water and there did not appear to be any general increase in the levels of metals in the oysters due to the inputs from Wheal Jane.

Overall, the concentrations of Cd, Cu, Fe and Zn appeared to follow a seasonal pattern (Figure 4), rising in the spring/summer of each year and falling in the autumn/winter. It is likely that the increased metal uptake during the summer, is due to changes in biological or biochemical activity in the oysters during periods of growth and spawning.

Since the initial overflow event in 1992, water from the Wheal Jane mine has been pumped to a tailings dam in the Clemows valley, where lime and a flocculant are added, to precipitate the metals. This process removes around 97% of the metals present and when discharged into the Carnon River the water has a pH of 6-7 and a total metal concentration of ~10 ppm. This method of treatment is only short-term due to the limited life of the tailings dam and the high costs involved, but the NRA are currently investigating longer-term treatment methods which are expected to be in full scale use in the next few years. Concentrations of metals in the Carnon River are naturally high, due to inputs from other disused mine workings and the natural process of water percolation through the metalliferous geology of the catchment. However, with the treatment of the Wheal Jane mine water, levels are currently lower than before the overflow event and are expected to remain so in the future.

Throughout the monitoring period, even the highest concentrations of metals recorded in oyster tissue did not approach calculated Food Safety Advisory Action Limits (2 mg kg^{-1} wet weight for Cd and 2400 mg kg^{-1} wet weight for Zn) and MAFF was able to advise that there was no cause for concern with regard to the consumption of oysters. At the end of March 1995, it was therefore concluded that there was no longer a need to continue to monitor oysters while the levels of metals in the water remained at the current levels. Should the situation change in the future and inputs from Wheal Jane increase significantly, the need for monitoring will be re-assessed.

5. THE CONCENTRATIONS OF THALLIUM IN MARINE FISH/SHELLFISH

Some concern has recently been expressed regarding the concentrations of thallium in some shellfish from Canadian waters (pers. comm. J. M. Bewers). A review of the literature on concentrations of thallium in fish and shellfish around the coasts of England and Wales indicated that little information existed. A variety of fish and shellfish from around England and Wales were therefore analysed for thallium to provide some background information. Samples of fish livers, fish muscle, shellfish and crustacea were analysed using ICP-MS following digestion with nitric acid in closed microwave vessels.

5.1 Results and discussion

From Table 9 (overleaf), it can be seen that the concentrations found were very low, with almost all of the samples analysed below detection limits of 0.01-0.03 mg kg^{-1} wet weight. The results would indicate that the concentrations of thallium in fish and shellfish around the coast of England and Wales are low, arising only from natural sources and, therefore, do not give any cause for concern.

Table 9. Concentrations of thallium in marine fish/shellfish

Species	Tissue	Area	Number of samples	Thallium (mg kg^{-1})
Cod	MM	Cardigan Bay	2	<0.01
Cod	BL	Channel	2	<0.03
Cod	BL	Cardigan Bay	2	<0.02
Crab	Claw	Humber	2	0.01
Dab	BL	German Bight	2	<0.02
Dab	BL	Thames Wharf	1	<0.02
Dab	BL	Irish Sea	6	<0.02
Dab	BL	Tyne Tees	4	<0.02
Dab	BL	Cardigan Bay	1	<0.02
Dab	BL	Wash Area	3	<0.02
Dab	MM	Tyne Tees	4	<0.01
Dab	MM	Thames Wharf	9	<0.01
Dab	MM	Morecambe Bay	12	<0.01
Dab	MM	Morecambe Bay	6	<0.02
Dab	MM	Morecambe Bay	1	<0.03
Dab	MM	Irish Sea	12	<0.01
Dab	MM	Tyne Tees	8	<0.01
Dab	MM	Cardigan Bay	4	<0.01
Dab	MM	Wash Area	10	<0.01
Dab	MM	N E Irish Sea	2	<0.01
Flounder	MM	Morecambe Bay	6	<0.01
Herring	BL	Thames	1	<0.03
Herring	BL	Thames	1	<0.02
Herring	BL	N North Sea	2	<0.02
Herring	BL	N North Sea	3	<0.03
Herring	MM	N North Sea	6	<0.01
Herring	MM	SW of Beryl Oil Rig	4	<0.01
Modiolus	BT	Humber	18	<0.01
Mussels	BT	Menai Straits	8	<0.01
Oysters	BT	Morecambe Bay	1	<0.02
Oysters	BT	Messack Pt., R. Fal	2	<0.01
Oysters	BT	Parsons Bank, R. Fal	2	<0.01
Plaice	BL	Channel	1	<0.02
Plaice	BL	Liverpool Bay	2	<0.02
Plaice	IM	Liverpool Bay	2	<0.01
Queen Scallop	Bulk Viscera		1	0.02
Queen Scallop	Gonad		1	0.02
Queen Scallop	MM		1	0.01
Sole	MM	Liverpool Bay	4	<0.01
Whelk	BT	Nr Anglesey	8	<0.01

Note: BL = Bulked liver
BT = Bulked tissue
MM = Mixed muscle
IM = Individual muscle

6. OESTROGENIC EFFECTS PRODUCED BY SEWAGE EFFLUENTS – A PROGRESS REPORT

Recent research has shown that effluents from sewage works produce oestrogenic effects in male and female fish. The observed responses have included elevated serum levels of the lipophosphoprotein vitellogenin, a usually female-specific protein which during oocyte growth stages is translocated from the liver to the ovary for incorporation into the developing oocytes as yolk. In natural reproduction, vitellogenin is mediated by circulating oestrogens, usually 17b-estradiol. The vitellogenin levels in male fish stimulated by sewage effluent (caged trout held downstream of sewage treatment works in UK rivers) are of a similar magnitude to those in normal mature females undergoing the early stages of oocyte development (Harries *et al.*, 1995).

The causal factors for feminisation of male fish are currently under investigation. Ethynyl estradiol (the active ingredient in the contraceptive pill) is one candidate compound, as are a variety of non-ionic surfactants e.g. nonyl and octyl phenols derived from the breakdown of alkyl phenol polyethoxylates. The programme reported here was carried out in association with Brunel University and was jointly funded by the NRA. Its aim was to identify and quantify all oestrogenic compounds in sewage effluents.

An assay for determination of oestrogens in sewage effluents using a genetically modified strain of yeast has been refined at Brunel University and has been described recently (Routledge and Sumpter, 1996). Oestrogenic components in sewage have been isolated by testing the activity of whole sewage, which was then divided into coarse and successively finer fractions and re-tested.

Assays of coarse fractions of effluent samples from a local sewage treatment works (STW) showed considerable oestrogenic activity. This activity was not decreased by removal of the volatile components, nor by solvent extraction of the particulate phase. The result suggested that the oestrogenic factors present in the effluent were non-volatile compounds of medium to high polarity.

Non-polar solid phase extraction (SPE) was used to fractionate the effluent sample. Extraction removed most (at least 80%) of the oestrogenic activity present. On sequential removal of the components from the cartridge using a variety of solvents, those eluted using aqueous solutions of between 50 and 85% methanol were found to contain the oestrogenic activity, indicating that the oestrogenic factor/s were likely to be organic compounds of medium polarity.

Finer fractionation of the oestrogenic fractions by liquid chromatography (LC) showed that all of the oestrogenic activity could be isolated in a single fraction, eliminating most of the components present in effluent (Routledge *et al.*, 1995).

Similar results were obtained from effluent from 2 other STWs which are known to give a vitellogenic response in fish.

Alkylphenols and ethoxylates, although present in the effluent samples at low $\mu g\ l^{-1}$ levels do not coincide with the oestrogenic fraction and are therefore unlikely to be the main causative factors in these effluents. Identification of the compounds active in the fraction is currently in progress.

7. ALKYLPHENOLS IN RIVERS AND FISH TISSUES

7.1 Alkylphenols in rivers

Since the late 1980s, the Burnham Laboratory has carried out periodic investigations into the concentrations of alkylphenols in the aquatic environment generally and the possible toxicity of alkylphenols to marine life (MAFF, 1992). Work has been conducted at the Liverpool Bay and Outer Thames Estuary sewage-sludge disposal sites. Alkylphenols are a breakdown product of alkylphenolpolyethoxylate surfactants (APEOs). This work has recently received additional emphasis with the increasing concern over environmental concentrations of oestrogens or endocrine disruptors and the discovery that relatively low concentrations (approximately 20 $\mu g\ l^{-1}$) of nonylphenol, octylphenol and nonylphenol mono and diethoxylates can induce vitellogenesis in freshwater fish, affecting their fertility (Jobling and Sumpter, 1993).

In 1993, a study of the concentrations of alkylphenols in UK estuaries revealed generally low concentrations (<0.08-0.3 µg l^{-1}) of nonylphenol at most sites sampled, but relatively high concentrations in the Tees estuary (up to 27 µg l^{-1} nonylphenol adjacent to one outfall (MAFF, 1995)). In 1994, a survey of UK rivers was carried out to measure the concentrations of alkylphenols and their potential effect on fish.

Five rivers were sampled (Figure 5) during the summer of 1994, with samples collected at 5 or 6 sites on each river. In addition, one drinking water abstraction point, on the River Arun, was also sampled. The rivers were selected to provide a broad spectrum of potential alkylphenol inputs and concentrations, from the predominantly rural River Wye to the industrial River Aire (Figure 5). Three of the rivers, the Aire, the Arun and the Lea were the subjects of a biological field programme investigating the effect of sewage effluents on caged trout (Harries *et al.*, 1995). Samples were taken from mid-river where possible, and extracted and analysed by solid phase extraction followed by gas chromatography-mass spectrometry (SPE-GC-MS, Blackburn and Waldock, 1995).

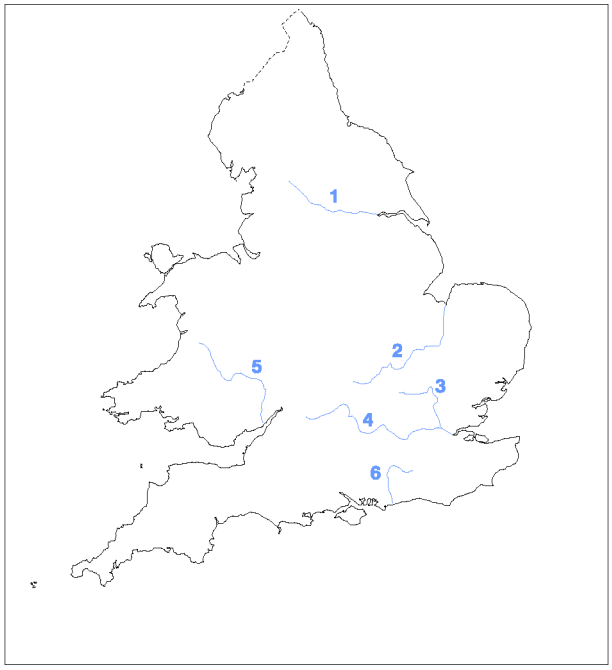


Figure 5. Map of rivers sampled: 1-Aire, 2-Great Ouse, 3-Lea, 4-Thames, 5-Wye, 6-Arun

Table 10. Concentrations of nonylphenol in rivers in England and Wales

River	Site	Dissolved nonylphenol (µg l^{-1})	Total extractable nonylphenol (µg l^{-1})
Aire	Kildwick Bridge	<1.6	<1.6
	Keighley	25	130
	Bingley	25	180
	Nab Wood	53	82
	Newlay Bridge	31	120
	A642 Bridge	24	140
Thames	Newbridge	<1.3	0.8
	Cookham	0.7	<1.3
	Staines	<1.3	<1.3
	Syon House	<1.3	2.3
	Battersea	0.6	2.2
Lea	Luton	0.5	1
	Nr Harpenden	1.1	<0.9
	Essendon	1.6	1.3
	Dobbs Weir	0.2	0.5
	Lower Hall	0.6	0.7
	Leyton (E. London)	9	12
Wye	Pant Mawr	<0.5	<1.2
	Builth Wells	<0.4	0.2
	Nr Hay on Wye	<0.2	<1.2
	Holme Lacey	0.9	<2.7
	Kearne Bridge	0.4	<1.3
	Llandogo	<0.4	<0.4
Ouse	Steane	<1.0	<1.0
	Olney	<1.9	0.6
	Great Barford	<0.9	<0.9
	Offord	0.9	1.1
	Ely	<0.5	1.6
	Kings Lynn	1.3	5.3
Arun*	Holdwick Farm	<0.2	<0.2

Octylphenol <1 µg l^{-1} at all sites
* *Drinking water abstraction point*

The results are presented in Table 10 . Most of the rivers sampled contained relatively low levels of nonylphenol, with concentrations >5 µg l^{-1} only at specific sites affected by sewage discharges or industrial effluents. Concentrations at the River Arun water abstraction point were below instrumental limits of detection and substantially below analytical limits of detection. Concentrations were, however, very much greater in the River Aire, due to inputs resulting from the extensive use of APEO surfactants during wool cleaning operations in local textiles plants, which reach the river via local sewage treatment works. Concentrations of dissolved nonylphenols exceeded the threshold limit for vitellogenesis and total extractable concentrations approached acute toxic levels. High levels of nonylphenol monoethoxylate and nonylphenol diethoxylate were also recorded at several sites on this river.

In general, the data show that concentrations of alkylphenols are well below toxic levels in most areas and that the oestrogenic effects observed in fish near many domestic sewage works (Purdom *et al.*, 1994; Harries *et al.*, 1995) may be due to compounds other than alkylphenols (as also indicated in the previous section of this report). In the case of the River Aire, where concentrations of nonylphenol exceed the threshold for induction of vitellogenesis the NRA has taken action to control the inputs which cause the problem.

7.2 Alkylphenols in fish

Several alkylphenolic compounds are used as production chemicals at offshore installations and are released into the open sea as part of the production water discharge. To ensure that these chemicals are not reaching the food supply a number of herring, haddock and dab, taken from around North Sea offshore installations, were analysed for nonylphenol and octylphenol as part of a preliminary Food Quality Assurance monitoring programme. Samples were extracted by Soxhlet extraction with dichloromethane followed by an alumina clean up and quantitative analysis by GC-MS. This preliminary study showed that concentrations of each compound in liver and muscle tissue were below the limits of detection (Table 11), suggesting that food stocks from the North Sea are not at risk from this source. This survey was repeated with a greater number of fish in 1995 and the results will be included in a future report in this series.

Table 11. Alkylphenols in North Sea fish tissue

Species and tissue	Area	Nonylphenol isomers (mg kg^{-1})	Octylphenol isomers (mg kg^{-1})
Herring muscle	Beryl	<0.02	<0.004
Herring muscle	Forties	<0.02	<0.004
Haddock liver	Beryl	<0.1	<0.04
Haddock liver	Forties	<0.1	<0.04
Haddock liver	Ninian	<0.1	<0.04
Haddock muscle	Beryl	<0.02	<0.1
Dab muscle	Ravenspurn	<0.02	<0.004

All concentrations are on a 'per isomer' basis and are the mean of 5 samples

SEDIMENTS

8. ESTUARINE SEDIMENTS AROUND ENGLAND AND WALES

8.1 Introduction

Estuaries, although a recognised sink for a wide range of trace elements, are often also a significant source of metals inputs to the marine environment. However, the behaviour and distribution of these elements in estuaries is not always well-known. Consequently, a collaborative project between MAFF and the NRA was carried out to examine metals in a range of estuaries.

8.2 Methods

Samples were collected from selected sites within various estuaries using either a hand-held or winch-deployed stainless steel grab. A sample of approximately 1 kg of the surface 1 cm of sediment, was taken from the grab sample using a plastic spoon. The

samples were stored in self-sealing plastic bags held within plastic boxes and frozen as soon as possible after collection.

Depending on the primary purpose of the survey, either a single sample was collected or, where sampling was required to be broadly in line with the National Monitoring Programme (see MPMMG, 1994), a grid of 5 samples, spaced at 250-m intervals, was taken.

The estuaries sampled were the Tyne, Tees, Humber, Thames, Severn, Mersey, Medway, Adur, Exe, Fowey, Fal, Dart, Taw/Torridge, Bristol Avon, Wye, Usk, Cleddau, Conwy, Derwent (NW) and Solway Firth.

8.2.1 Analytical methods

On return to the laboratory, the samples were defrosted, sub-sampled wet and approximately 150 g freeze dried; the remainder was returned to an archive freezer. The freeze dried sample was then split into 2 equal fractions, one for particle size analysis and one for metals and carbon analyses. The metals/carbon sub-sample was sieved at 2 mm to remove gravel particles and any large detritus. This sub-sample was then split to yield a further sub-sample of 30 g which was ground to a powder using a mechanical ball mill. The ground sample was then split into three portions, one for metal analysis, one for carbon analysis by a wet chemical technique and one for storage. The collection and analytical methods used in this study followed ICES advice (ICES, 1989).

8.2.2 Metal analysis

Five hundred milligrammes of the ground sediment were completely digested in a microwave furnace using a mixture of hydrofluoric acid and nitric acid (Jones and Laslett, 1994). The resulting solution was treated to minimise matrix effects before being analysed using ICP/MS for a range of elements (Al, Cd, Cr, Cu, Pb, Ni, As, Fe, V and Zn). Mercury was also measured, using an atomic fluorescence technique.

8.3 Interpretation of the observed concentrations

Generally, the interpretation of the concentrations of metals in sediments is facilitated by the removal of complicating factors such as the sediment type (whether mud or sand). This is a process called normalisation which in the present context, requires the use of a non-contaminant element associated with clay minerals to account for mineralogical variations. Aluminium and lithium are two frequently used normalising elements. Both of these elements are present in clay minerals and each may act as a surrogate for the clay mineral fraction.

Loring (1991) recommended the use of lithium in areas with sediments derived by glacial processes where aluminium may be present in feldspars in the sand-sized sediment fraction. Rowlatt and Lovell (1994) showed that aluminium normalised against lithium indicates an excess of aluminium in the northern North Sea and suggested that the effect described by Loring could affect the present dataset. However, examination of the sediment metal concentrations showed little difference in the final results obtained using lithium and aluminium as normalisers, indicating that any such effect was only minor. Since aluminium is more widely accepted as a normaliser, aluminium was preferred to lithium in this survey.

8.4 Discussion

8.4.1 The application of normalisation to sediments using the Mersey as an example

The mean metal contents of the sediments from the Mersey Estuary are shown in Table 12. The concentrations of metals were found to decrease in the order docks > lock entrances > sea channels > Eastham Channel. This correlates with the sediment types in these areas which, due to differences in the hydrodynamic regime, range from mud in the docks to sand in Eastham Channel.

Sands contain lower concentrations of trace metals than the muds because quartz particles generally contain only very low concentrations of trace metals, quartz in the mud/sand mixtures acts as a diluent, lowering sediment metal burdens.

The dock sediments which may be dredged are contaminated with, amongst other metals, mercury derived from industry in the Mersey catchment area and contain a range of concentrations of mercury from 84 to 3020 $\mu g\ kg^{-1}$. Sediments from the outer estuary have mercury contents ranging from 9 to 1709 $\mu g\ kg^{-1}$. However, direct comparison of the dock dredgings with the estuarine sediments is made difficult by the fact that the latter sediments contain a large amount of quartz relative to those in the docks.

The position can be clarified by use of the conservative element normalisation technique, as follows. Figure 6, shows a strong correlation between the mercury and aluminium contents of the sediments from the Mersey Estuary and its associated docks. As aluminium is primarily located in the structure of the clay minerals it is effectively a marker for them. These data, therefore, indicate that mercury is strongly associated with the clay minerals, either directly or by adsorption onto organic coatings. Whilst it is true that some mercury will be present in the lattice structure of the clay minerals, as the mercury content of average shale is 400 $\mu g\ kg^{-1}$

Table 12. General statistics of samples collected from the Mersey

Variable	N	Mean	Std. Deviation	Minimum	Maximum
Al (%)	45	3	1.83	0.93	6.36
Cd (µg kg^{-1})	45	639	528	11	2555
Cr (mg kg^{-1})	45	71	47.08	11	152
Cu (mg kg^{-1})	45	38	33.23	1.36	107
Fe (%)	45	2	1.11	0.51	3.8
Hg (µg kg^{-1})	44	1823	3656	8.8	19609
Li (mg kg^{-1})	45	32	20.75	1	70
Ni (mg kg^{-1})	45	22	15.32	3.98	48.3
Pb (mg kg^{-1})	45	104	150	2.1	812
Zn (mg kg^{-1})	45	211	153	2.66	651
As (mg kg^{-1})	45	16	8.41	0.2	38.5
V (mg kg^{-1})	45	50	32.31	1.61	107
OC (%)	45	1	0.98	0.03	3.78
Median (µm)	22	139	106	16	360

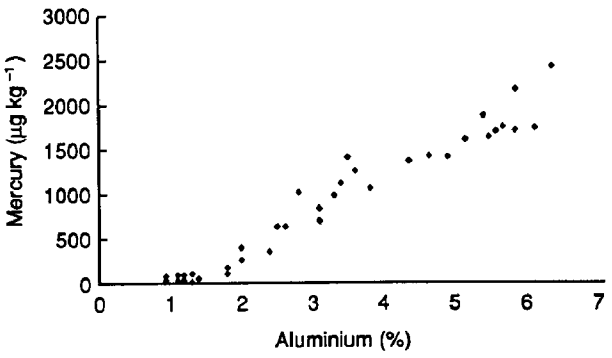


Figure 6. The relationship between mercury and aluminium in sediments from the Mersey Estuary

(Krauskopf, 1967) compared with about 4000 µg kg^{-1} in the muddiest samples from Liverpool docks, it is unlikely that mercury in clay mineral lattices contributes significantly to the total mercury burden of the sediments.

The relationship shown in Figure 6, indicates that the sediments all contain the same degree of contamination in the metal-bearing clay fraction and that the sediments are mixtures of two particle populations, a mercury- and aluminium-bearing population (the clays) and a low-mercury population (the sands). Similar relationships were observed by Taylor (1986) using granulometric data and by Rowlatt (1988) using iron as a normaliser.

These sediments taken throughout the Mersey Estuary and the dock areas of Liverpool are part of a continuum from sand to mud and the total metal burden of any dredgings produced is merely a reflection of the sand/mud ratio of the material. Thus, the normalisation technique has identified the fine sediments throughout the sample domain as constituting one population.

It is worth noting that when the dredgings are deposited at sea the currents tend to separate the mud and sand components of the sediments. Thus, there is little difference between depositing a given amount of contaminant either in a large volume of muddy sand (low contaminant concentration) or a small volume of mud (high contaminant concentration). For environmental control purposes, the total loading of contaminant must be considered as well as the contaminant concentration in the dredgings.

Any exceptionally contaminated material (i.e. not from the same fines population) would have shown up as outliers to the main trend line. This effect is illustrated in Figure 7 where samples from the Manchester Ship Canal (MSC) can be seen to deviate markedly from the mercury/aluminium relationship.

In conclusion, the concentration of contaminants in sediments collected from the Mersey/Liverpool docks system depends largely on the fines content of the sediments and normalisation against aluminium suggests that there is only one population of fines in the area.

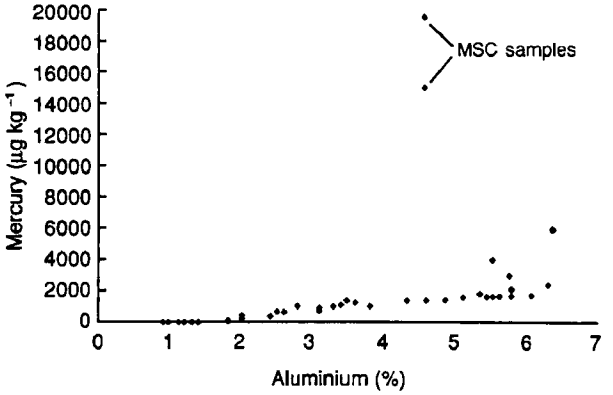


Figure 7. The relationship between mercury and aluminium in sediments from the Mersey Estuary and Manchester Ship Canal (MSC)

8.4.2 The use of sediment metal data to assess the quality of the Mersey Estuary

The ranges of concentrations of metals in the Mersey Estuary are quite large. Table 12 shows the concentrations of metals in sediments and shows for example that concentrations of cadmium range between 11 and 2555 µg kg^{-1} and concentrations of mercury range between 8.8 and 19,609 µg kg^{-1}. Any use of raw concentrations will therefore be subject to enormous variability depending on where the samples are collected.

The previous discussion of the Mersey data showed that all the sediment samples taken in the Mersey Estuary area are part of the same general population, with the exception of those from the MSC. Thus, a parameter which describes the whole population (except those from the MSC) will be insensitive to variations in sampling location and can be used to describe sediments in the whole estuary. A measure which should be considered as a candidate for this role might be the slope of the relationship between metal and normaliser.

This normalisation technique has been applied to the concentrations of lead in samples from a number of estuaries. Figure 8 shows an example of the ranges of lead burdens. The size of the slopes of the regression lines can be used to rank the estuaries according to their lead content; the steepest slopes indicating the estuaries or parts of estuaries with a higher sediment metal status. Sediment metal status is only indicative of the relative metal concentrations and provides no indication as to the origin (natural or anthropogenic) of high metal concentrations. Further work is needed to assess background concentrations before any assessment of the origin of the metal can be carried out.

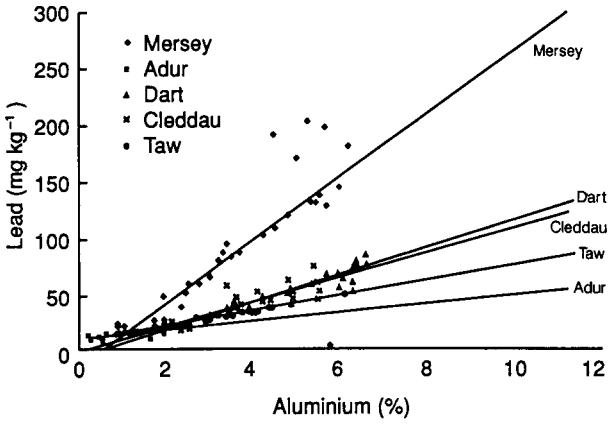


Figure 8. The relationship between lead and aluminium in sediments from various estuaries

9. CHLOROBIPHENYLS IN FINE SURFICIAL SEDIMENTS FROM CARDIGAN BAY

Concentrations of organochlorine compounds (chlorobiphenyls (CBs) and 4,4'-DDT and its metabolites) in the blubber of small cetaceans from Cardigan Bay, are higher than would be expected (see Section 2) in an area which is apparently relatively uncontaminated (Law and Allchin, 1994; Morris *et al.*, 1989). The source of the PCB is unknown at present. This study presents the results of a spatial survey of CBs in fine surficial sediments from Cardigan Bay, conducted as an adjunct to the marine mammal studies in Section 2.

Samples were collected from both estuarine and offshore locations, where predominantly fine-grained sediments were located. Fifteen locations (shown in Figure 9) were sampled between 13-21 July 1994. Offshore sites (9 samples) were sampled by means of a 0.1m^2 modified Day grab during a MAFF research cruise, and estuarine sites (6 samples: five in estuaries adjoining Cardigan Bay, and a sixth at Neyland within Milford Haven) were taken intertidally by means of a PTFE spoon. In both cases, the upper 2 cm of the sediment were taken and the samples placed in pre-cleaned glass jars and stored at -20°C prior to analysis. Analyses were conducted using

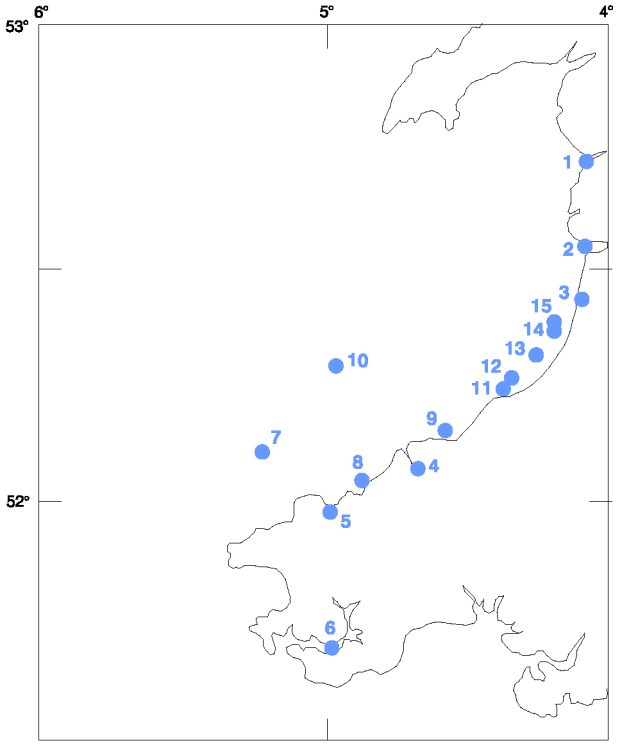


Figure 9. Cardigan Bay sediment sampling sites

standard methodology (Allchin *et al.*, 1989), modified in the light of the recommendations from the stepwise intercomparison programme conducted under the auspices of the International Council for the Exploration of the Sea (de Boer *et al.*, 1992; 1994), and were conducted under an analytical quality control protocol involving the analysis of blanks and reference samples within each batch. A list of the 25 CB congeners determined is given in Appendix 2.

The results of analyses for organic carbon, nitrogen and chlorobiphenyls (sums of the 25 CB congeners analysed and of the seven congeners of the ICES primary list (ΣICES7)) are given in Table 13. Following the Piper Alpha incident in 1988, Wells *et al.* (1989) defined a series of (arbitrary) concentration guidelines (on a dry mass basis) for categorisation of concentrations of CBs in sediments:

<0.2 μg kg^{-1}	contamination not detectable
0.20 to 20 μg kg^{-1}	slightly contaminated
21 to 100 μg kg^{-1}	contaminated
>100 μg kg^{-1}	heavily contaminated

Applying these criteria to the Σ25CB concentrations (Table 13), all but one of the sites sampled were either uncontaminated or slightly contaminated. The exception was Site 3, located in a small basin just inside the river entrance at Aberystwyth, which was highly contaminated at almost 400 μg kg^{-1} dry mass. Additional studies undertaken subsequently by the Welsh region of the National Rivers Authority in the Rivers Rheidol and Ystwyth in the vicinity of Aberystwyth (9 sites) have yielded concentrations of ΣICES7 in the range 0.2 to 20 μg kg^{-1} dry mass, with the majority of values in the lower part of this range (unpublished data; R. Milne, pers. comm.), the highest value being found close to the above mentioned Site 3. It is likely that the sediments sampled at Site 3 were disturbed by water flows resulting from excavation works in the harbour area just prior to sampling, but the presence of an historical PCB source in the catchment cannot be ruled out. This possibility will be the subject of further studies by the Environment Agency.

The Cardigan Bay data can be compared with results obtained from the more industrialised/urbanised Thames area. A set of predominantly fine sediment samples were collected along a transect of the River Thames and its estuary in August 1991, between Richmond Bridge (close to the tidal limit at Teddington Lock) and Shoeburyness (see Figure 10). The concentrations of CBs in these samples are given in Table 14. Essentially all samples were in the moderately contaminated category, except for two, from Grays and Wapping, which were in the heavily contaminated range.

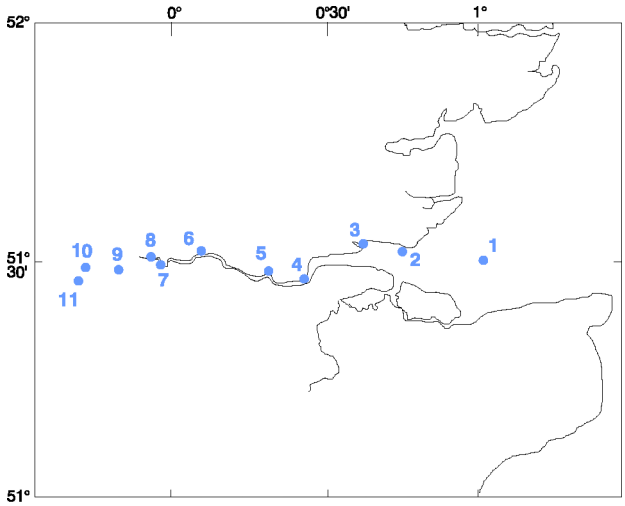


Figure 10. Thames transect

This study has indicated that surficial fine sediments in Cardigan Bay and the adjoining estuaries are not generally subject to substantial contamination by

Table 13. Concentrations of chlorobiphenyls (μg kg^{-1} dry mass), organic carbon and nitrogen in fine surficial sediments from Cardigan Bay

Site no.	Location	organic C (%)	organic N (%)	ΣICES7	Σ25CBs	Contamination
1	Barmouth	1.06	0.11	<0.2	<0.2	uncontaminated
2	Aberdyfi	0.45	0.03	<0.2	<0.2	uncontaminated
3	Aberystwyth	2.27	0.2	191	396	heavy
4	Cardigan	5.23	0.48	<0.2	1.9	slight
5	Fishguard	1.99	0.12	7.0	11.7	slight
6	Neyland	1.38	0.14	0.22	1.08	slight
7	Off Strumble Head	4.1	0.31	<0.2	<0.2	uncontaminated
8	Off Newport	1.48	0.08	1.7	3.4	slight
9	Off Aberporth	1.86	0.16	2.0	4.2	slight
10	Offshore Cardigan Bay	0.26	<0.01	1.1	2.2	slight
11	Off New Quay	0.57	0.07	1.6	2.1	slight
12	Off Aberaeron	1.17	0.09	<0.2	<0.2	uncontaminated
13	Off Llansantffraid	0.73	0.07	<0.2	<0.2	uncontaminated
14	Off Llanddeiniol	0.83	0.08	<0.2	<0.2	uncontaminated
15	Off Llanfarian	0.1	<0.01	<0.2	<0.2	uncontaminated

Table 14. Concentrations of chlorobiphenyls (µg kg^{-1} dry mass) in predominantly fine sediments from a transect along the River Thames

Site	Location	ΣICES7	Σ25CBs	Contamination
1	Shoeburyness	1.2	1.4	slight
2	Old Leigh	8.3	23	moderate
3	Tilbury	29	65	moderate
4	Grays	132	280	heavy
5	Creekmouth	16	36	moderate
6	Isle of Dogs	17	46	moderate
7	Wapping	52	125	heavy
8	Battersea	24	72	moderate
9	Kew Bridge	20	60	moderate
10	Richmond Bridge	10	24	moderate

chlorobiphenyls, and confirm the earlier view that Cardigan Bay appears to be a relatively clean environment. It has provided no clues, therefore, to the source of the high concentrations observed in dolphins from this area.

10. LONG-TERM MONITORING OF SEDIMENT QUALITY IN LIVERPOOL BAY

Liverpool Bay has been used for the disposal of sewage sludge since the 19th Century, but during the 1970s the quantity deposited increased from 40,000 to 70,000 tonnes per annum (dry solids). Monitoring has shown that the sedimentary concentrations of several trace metals in the vicinity of the sludge disposal site are elevated relative to the natural background levels and the benchmark concentrations, e.g. those concentrations found in surface sediments unaffected by the disposal operation (see e.g. Norton *et al.*, 1984). This sediment metal enrichment is a permanent feature of the disposal area and monitoring is carried out regularly to assess the quality of the sediments.

Each year, approximately 15 samples of sediment are collected from the disposal site. Figures 11(a-d) show the maximum, minimum and median values for the concentrations of Hg, Pb, Cu and carbon in these samples over the period 1975-93. The maximum values

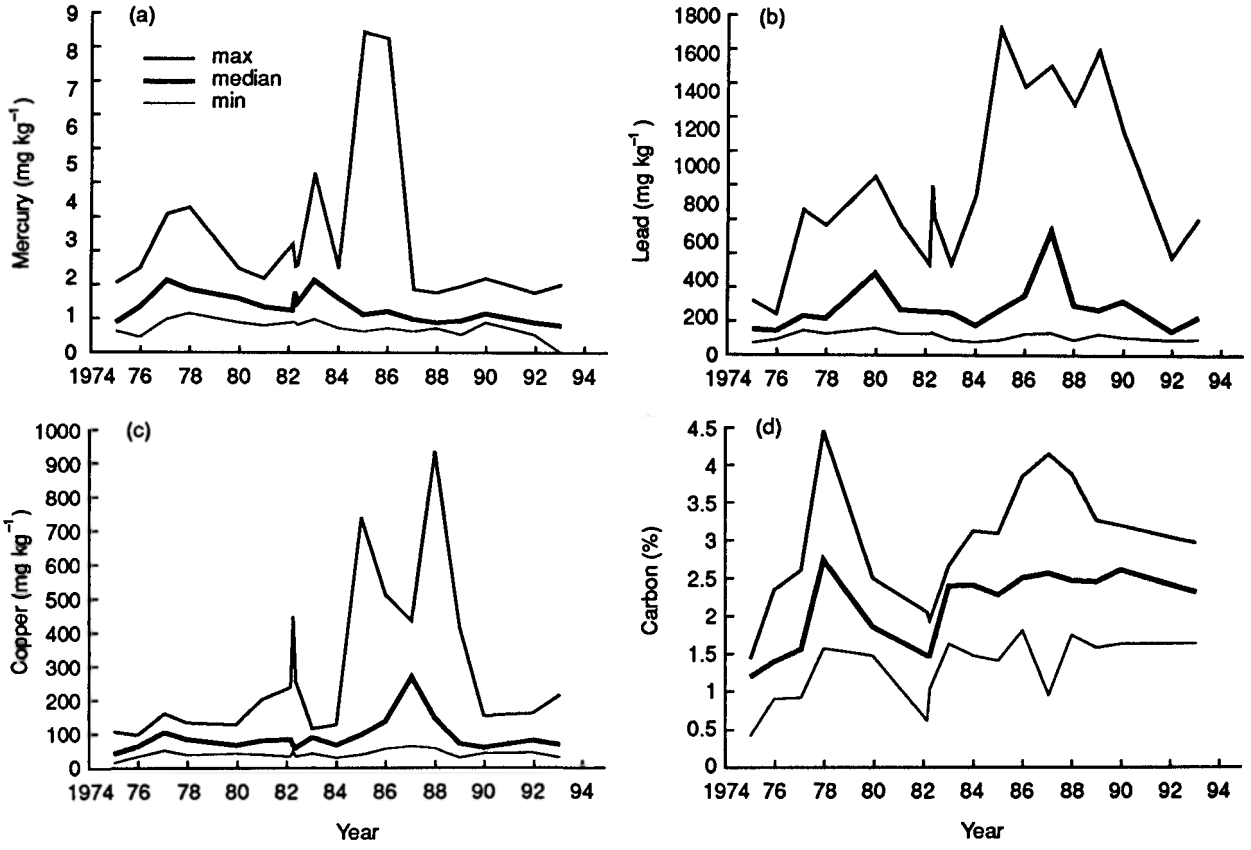


Figure 11. Concentrations of trace elements in sediments from the Liverpool Bay sewage-sludge disposal site: (a) mercury, (b) lead, (c) copper and (d) carbon (%)

vary considerably between years, suggesting patchiness of metal concentration. The metal values are not normally distributed and the median value therefore gives a more stable measure of central tendency and may be taken as a good indicator of the general quality of the area.

The median concentrations of mercury, lead, copper and carbon have shown little variation since the mid to late 1980s and levels have certainly not shown any signs of increasing over the last few years. Given that the disposal of sewage sludge will cease in 1998, it has been decided to reduce the frequency of monitoring.

DISPOSAL AT SEA: ENVIRONMENTAL ASSESSMENT STUDIES

11. BENTHIC COMMUNITY STUDIES AT THE OUTER THAMES SEWAGE-SLUDGE DISPOSAL SITE

11.1 Introduction

Sewage sludge from the London area has been deposited at two sites in the outer Thames Estuary for about a hundred years. Since 1967, a location in the Barrow Deep has been used (see Figure 12), and this received about 4.4 million wet tonnes in 1993, an amount comparable to the annual average disposed of since commissioning of the site.

MAFF has conducted a number of spatially extensive studies of the environment around the disposal site, as part of its responsibilities as the licensing authority for sea disposal of wastes under the Dumping at Sea Act (1974) (Great Britain - Parliament, 1974) and its successor the Food and Environment Protection Act (1985) (Great Britain - Parliament, 1985) as amended by the Environmental Protection Act 1990 (Great Britain - Parliament, 1990). Additionally, the licensee (Thames Water) has in recent years commissioned its own studies of the disposal operation under an agreed 'self-monitoring' scheme. Examples of these studies include Shelton (1971), Norton *et al.* (1981), Talbot *et al.* (1982), Whitelaw and Andrews (1988(a), (b)) and ap Rheinallt *et al.* (1990). Recent work by MAFF has concentrated on the evaluation of seasonal and annual trends in the concentrations of trace metals and benthic organisms at

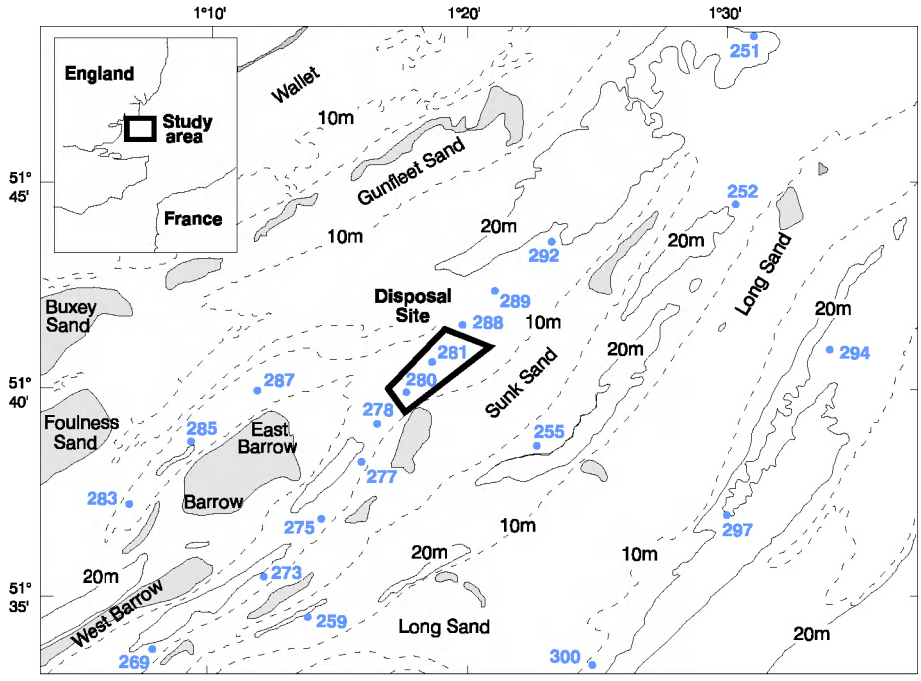


Figure 12. Map of the outer Thames Estuary showing the location of the disposal site and sampling stations (May 1992)

selected sites (see MAFF, 1990, 1993) as a complement to the more infrequent spatial surveys that have been conducted. All of these studies concluded that the dispersive properties of the receiving environment were sufficient to ensure that effects on the benthic fauna were limited in both intensity and spatial extent. For example, high densities of species indicative of organic enrichment, such as the polychaete worm *Capitella*, did not occur, in contrast to findings at a more quiescent location off the west coast of Scotland (Pearson, 1987). The main influences on the distribution of the benthic fauna of the outer Thames area were substrate type *per se*, and associated physical perturbations arising from wave and tidal current action at the seabed. A subsidiary influence of sewage-sludge disposal was, however, identified at some locations.

In 1992, MAFF collected a series of samples at stations on a wider geographical scale than previous studies, and included transects running along the main channel systems, a distinctive feature of the outer Thames environment. The results from analyses of concentrations of trace metals were reported in MAFF, 1994. A number of these stations were also sampled for the benthic macrofauna, and the results from analyses of these samples are the subject of this account.

The objectives of the survey were to examine relationships between the spatial distribution of the benthic fauna and other environmental variables including the concentrations of trace metals and in so doing to check the validity of earlier judgements concerning limitations of effects arising from sewage-sludge disposal.

11.2 Methods

Field sampling

Single samples of the benthic macrofauna were collected in May 1992 by Day grab at twenty stations in the outer Thames area, using the MAFF research vessel *RV CORYSTES*. The depth of material retained in each grab was measured, and those with a depth of less than 5 cm were discarded.

After removal of a sub-sample of sediment for later particle size analysis using a 3-cm diameter perspex corer, the fauna sample was washed over a 1-mm mesh sieve, and the contents preserved in 5% formalin in sea water with added Rose Bengal (a vital stain).

Laboratory analysis

In the laboratory, the sediment sub-samples for particle size analysis were initially separated into fine and coarse fractions by wet sieving at 63 µm. Particle size distributions for the coarse fraction (>63 µm) were determined by dry sieving at 0.5 phi intervals. In cases where the fine fraction (<63 µm) exceeded 10% by weight of the sample, size distributions were determined instrumentally using a Sedigraph.

After removal of the preservative, samples of the benthic macrofauna and any residual sediment were transferred to white trays, where specimens were sorted by eye and then counted and identified as far as possible to species level, using a range of standard taxonomic keys. Wet blotted weights of individual species were determined, and then converted to ash-free dry weights (AFDW) using standard conversion factors, notably as per Rumohr *et al.* (1987).

Data analysis

The PRIMER software package (see Clarke and Warwick, 1994) was employed for analysis of species abundance data, and for exploring relationships between the distribution of the benthos and environmental variables. A fourth-root transformation was used in order to reduce the influence on the outcome of multivariate analyses, of the presence of one or two numerically dominant species.

11.3 Results

Three station groups were identified from cluster analysis of the benthos data (Figure 13). These are described as follows:

(i). Group 1. This group appears as a chain of linked stations with no distinct sub-groups, and their distribution is shown in Figure 14. It includes stations sampled in the Barrow Deep (except those in the immediate vicinity of the disposal site) as well as some from the Middle, Black and Knock Deeps. The fauna at these stations was relatively impoverished, with means and ranges of densities and numbers of taxa per grab of 30 (6-46) and 9.6 (4-16), respectively. There was no marked numerical dominance by one or more species. Of the five species which were most influential in the formation of this station group, two were typical of muddy sands (*Abra alba* and *Nephtys hombergi*) and two typical of clean sands (*Bathyporeia elegans* and *Nephtys cirrosa*). The average AFDW biomass per grab amounted to 200 mg. However, if the weights of 4 adult cockles at a single station are excluded, the value drops to 50 mg.

(ii). Group 2. This includes stations within and in close proximity to the disposal site (Figure 14), which are characterised by relatively high densities and numbers of taxa, with means and ranges per grab of 230 (126-405) and 26 (22-31), respectively. The five species which were most influential in establishing this group included three infaunal species (*Abra alba*, *Lanice conchilega* and *Cirratulid* sp) and two attached epifaunal species (*Sagartia* sp and *Pomatoceros* spp). Occurrence of the latter indicates a significant coarse component to superficial sediments (see below). The serpulid worm *Pomatoceros* spp occurred in relatively high densities only at the three stations

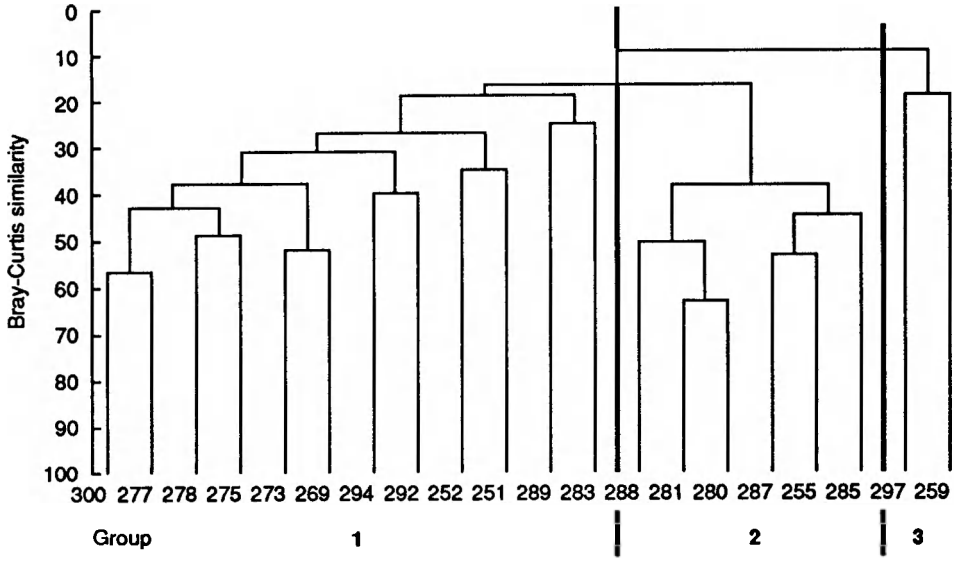


Figure 13. ***Output from cluster analysis of the macrofauna data, employing the Bray-Curtis similarity index and 'group-average' sorting***

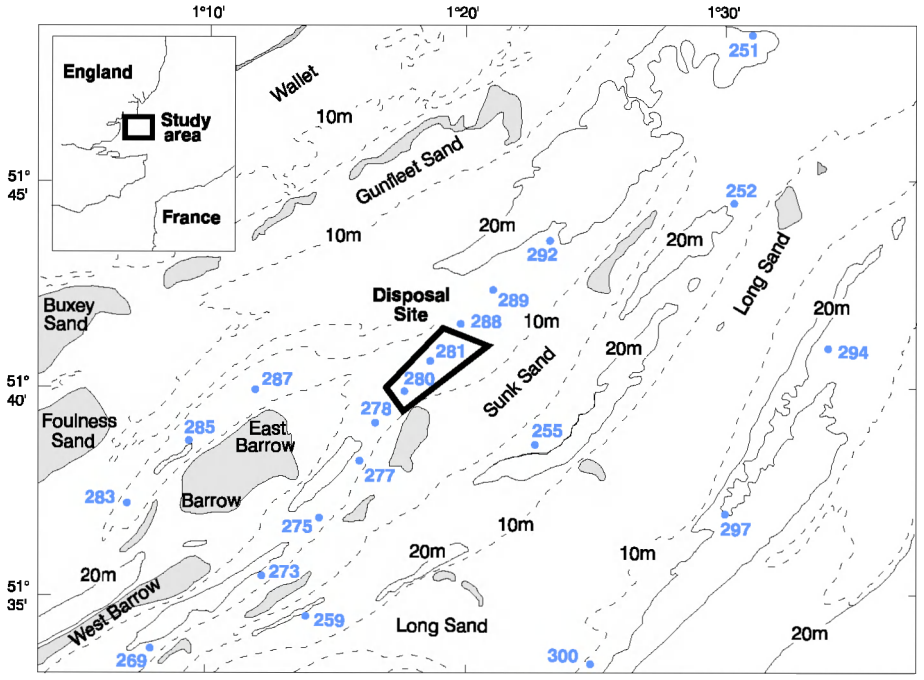


Figure 14. ***Map of the outer Thames Estuary showing the disposition of station groups identified from the cluster analysis of Figure 13***

close to the disposal site, while the sea anemone *Sagartia* sp was present at all stations within this group. The average AFDW biomass per grab amounted to 550 mg, i.e. substantially greater than for Group 1 stations.

(iii). Group 3. This consists of two stations to the south of the survey area, linked at a low level of similarity (Figure 14). Densities and numbers of taxa per grab were very low at 11 (3-18) and 4.5 (3-6), respectively. Moderate numbers of the polychaete *Ophelia* at Station 297 are suggestive of naturally unstable sandy sediments. The other station (259) was located at the now disused Black Deep disposal site (see Discussion). The average AFDW biomass per grab amounted to only 7 mg.

The distribution of median particle sizes of sediments (Figure 15) demonstrates that the 'enriched' benthos samples from Group 2 stations, which include the disposal site, are associated with generally coarser sediments than elsewhere. The main channel systems characteristically comprise well-sorted fine sands or muddy sands, and support a relatively impoverished fauna (Group 3, above).

The concentrations of lead, zinc, copper and mercury at stations at which benthos samples were collected, and at several additional locations, were reported in MAFF (1994). The results showed that, in general, values did not vary greatly over the sampling area, which at its seaward end extended some 50 km beyond the disposal site into the southern North Sea. However, elevated concentrations along the Barrow and Middle Deeps may be linked with the dispersion of sewage-sludge particulates, and this would accord with the findings of earlier sediment transport studies by Talbot *et al.*, 1982. Higher concentrations were also observed in the immediate vicinity of the Black Deep disposal site, which may be linked with an earlier history of dredgings and sewage-sludge disposal here.

An examination of relationships between the concentrations of trace metals and the configuration of stations on a 2-dimensional plot of the outcome of multivariate analysis of the benthos data (Figure 16(a-d)) provided no evidence of any correlation.

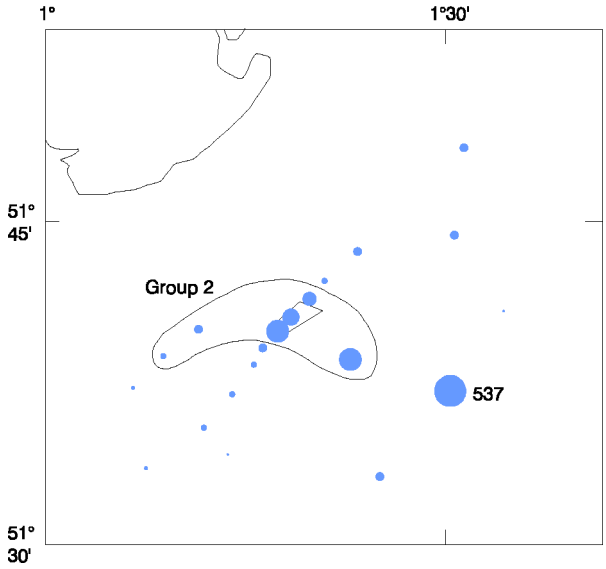


Figure 15. ***Distribution of the median particle diameter (mm) of sediments within the survey area. Circles are scaled relative to the highest value encountered, which is specified on the plot. Stations within Group 2 of the output from cluster analysis (Figure 13) are highlighted***

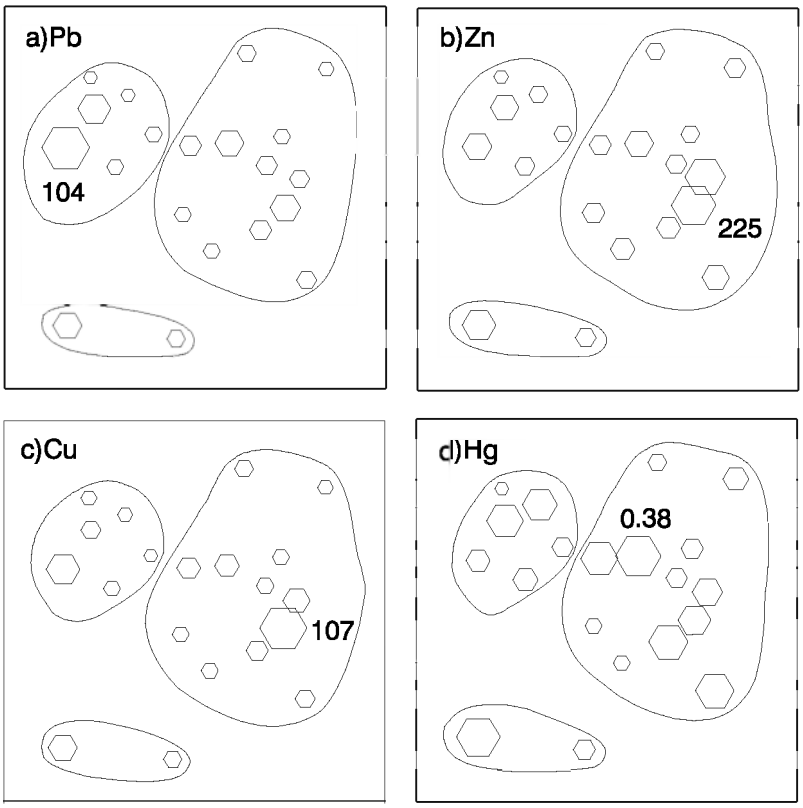


Figure 16. ***Multi-Dimensional Scaling ordination of the benthos data with Pb, Zn, Cu and Hg concentrations (mg kg^{-1}) superimposed. In each plot, values are scaled relative to the highest concentration encountered, which is specified. Additionally, station groups identified from cluster analysis (Figure 13) are circled***

11.4 Discussion

Norton *et al.* (1981) and Talbot *et al.* (1982) concluded that the distribution of the benthic fauna was largely governed by sediment type and the associated degree of sediment mobility under wave and tidal current action, with sewage sludge having only a secondary influence. The results from the present study are in agreement with these findings. The substrates along much of the channel systems consisted of well-sorted sands or muddy sands which supported a relatively impoverished fauna, typically including *Bathyporeia elegans*, *Nephtys* spp, *Abra alba* and *Scoloplos armiger*.

At some localities, the occurrence of coarser, mixed sediments provided greater stability and, as a result, a richer benthic fauna was encountered. Coincidentally, such substrates are encountered at the disposal site and, as in 1977 when Norton *et al.* (1981) conducted their survey, the dominant taxa were serpulid worms (*Pomatoceros* spp) and attached sea anemones. Thus there is a natural explanation for elevated densities and numbers of species here, although the consistent occurrence (albeit in low numbers) of taxa such as Cirratulid worms associated with the fine particulate component of the sediment provided a suggestion of a limited additional effect of organic enrichment. Cirratulids have been cited as indicators of organic enrichment by, among others, Pearson and Rosenberg (1978). As *Pomatoceros* spp and sea anemones are suspension-feeders, their presence in relatively large numbers indicates that the seabed environment at the disposal site is not adversely affected by settling particulates since, in excess, these would be expected to clog ciliary or siphonal mechanisms or, in extreme cases, smother the organisms (Pearson and Rosenberg, 1978).

Both stations within Group 3 of the output from cluster analysis (Figure 13), possessed very low numbers of species and individuals. One was located at a site in the Black Deep which, until 1967, had been used for the disposal of both dredged material and sewage sludge, and the presence of plant/wood debris may be indicative of the former activity. The silt/clay content of the sediment was high (20%), but was clearly unattractive to benthic organisms, i.e. it did not appear to be organically enriched; it is presumed to have originated from the erosion of clay outcrops which are known to occur at a number of locations within the estuary (Norton *et al.*, 1981). The other station was characterised by a well-sorted coarse sand with a very low silt/clay content (<1%). The only prominent species (the polychaete *Ophelia*) is typically found in association with naturally unstable sandy locations.

11.5 Conclusions

The present status of the benthic fauna in the vicinity of the sewage-sludge disposal site is very similar to that encountered some 15 years previously, and spatial differences within the survey area can, at any one time, be largely explained by natural influences, especially substrate type, tidal current and wave action. Coarser substrates which coincidentally occur in the immediate vicinity of waste disposal provide a natural explanation for elevated numbers of individuals, taxa and biomass here. However, there is an indication of an additional 'enrichment' effect associated with sewage-sludge disposal but the effect is marginal, since there is no evidence for the elimination of suspension-feeders, or for the proliferation of classical 'indicator' species. These findings point to the continued acceptability of this disposal practice at current levels of input.

12. DEVELOPMENTS IN THE USE OF MEIOBENTHOS AS A MONITORING TOOL

Traditionally, most studies investigating the manifestation of pollution in the marine benthos have concentrated on monitoring changes in the macrofauna (Pearson and Rosenberg, 1978 and Gray, 1979). However, the meiofauna (animals passing through a 500 µm sieve and retained on a 63 µm sieve) are being increasingly used as a means of detecting environmental perturbations (Lambshead, 1986; Austen *et al.*, 1989; for review see Coull and Chandler, 1992).

Significant gaps remain in our understanding of the role of the meiobenthos in marine ecosystems, and their utility in monitoring is still relatively unexplored. This is partly due to difficulties in the practical application of working with such small taxa compounded by the incomplete and fragmented state of the meiofaunal taxonomic literature. However, recent publications have brought meiofaunal taxonomy and assessment more within the scope of the non-specialist worker.

In view of this, a project, funded by MAFF, has been initiated to investigate the practical implications for the in-house use of marine meiofauna as a means of assessing disturbances resulting from man's activities. This project aims to evaluate the usefulness of meiofaunal studies alongside conventional macrofaunal surveys. Accordingly, samples have been collected in collaboration with both MAFF and the NRA during routine assessments, at a range of sites impacted by licensed disposal activities.

A prerequisite in quantitative meiofaunal studies is undisturbed sediment samples. Therefore, in attempting to employ the meiobenthos in monitoring programmes, the choice of quantitative sampler is a primary concern. A multiple corer designed to take four undisturbed sediment cores simultaneously has been used with some success in the collection of meiobenthic samples from subtidal soft sediments. Underwater video images have confirmed that the operation of the corer creates a minimal amount of disturbance during sample collection.

This device was used successfully during a survey of the Liverpool Bay 'New Site Z' dredged material disposal ground. The purpose of this study was to examine the responses of meiobenthic taxa, particularly nematodes, to dredged material disposal and identify any depth related faunal trends that may be attributable to such disposal practices. To this end, replicate sediment cores were collected at three stations within the disposal ground and at reference stations to the north and south. Cores were sectioned in order to examine the vertical distribution of the fauna and sub-samples from complimentary cores were also retained for trace metal and particle size analyses.

Preliminary data indicate that there is spatial separation of different species of nematodes from both the reference and disposal site communities, although the depth separation of nematode species at the disposal site was less well-defined. Peak densities of certain nematodes species (e.g. *Richtersia inaequalis*) were also found to be distributed lower in the sediment at the disposal site (see Figure 17(a)) compared to the reference station (Figure 17(b)). This may be a direct consequence of burial from the disposal of dredged material or the result of reduced competition allowing animals to extend their depth horizons. Further work is being conducted to determine whether nematode vertical distribution is a useful measure of physical stress in marine sediments.

In conjunction with the NRA, a survey of the Howden sewage outfall in the Tyne was also successfully completed using a multiple corer. The aims of the survey were two-fold: firstly, to examine the sensitivity of the meiofaunal community to the discharge of primary treated sewage; and secondly, to examine the vertical distribution of the faunal assemblages in relation to the proximity of the outfall. Replicate core samples for meiofaunal investigations were sliced and preserved for later faunal examination from eight stations. Sub-samples of sediment for particle size, trace metals and bacterial analyses were also taken.

The results from these and additional studies will be reported at a later date.

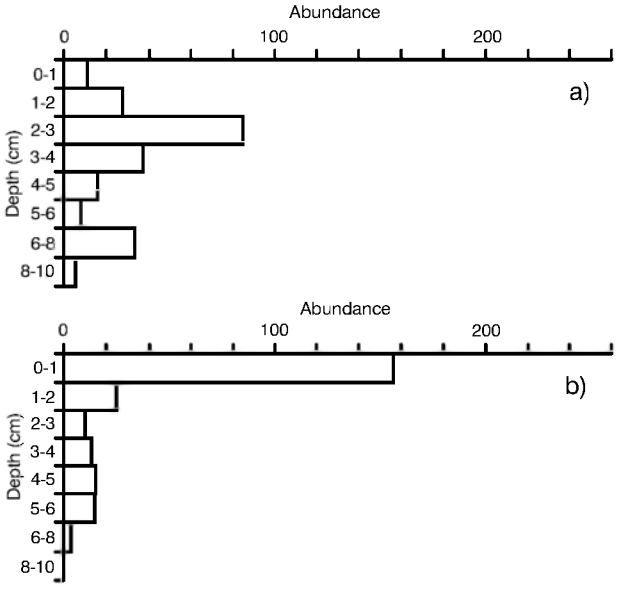


Figure 17. The vertical distribution of Richtersia inaequalis at (a) the disposal site and (b) the reference site

DEPOSITS IN THE SEA: LICENSING AND RELATED ACTIVITIES

13. CHLOROBIPHENYL CONTAMINANTS IN DREDGED SEDIMENT

13.1 Introduction

Sea disposal of dredged sediment remains, to a large extent, the most cost-effective means of disposal. All dredged-material disposal operations are licensed under the Food and Environment Protection Act, part II (1985) (Great Britain - Parliament, 1985) with MAFF as the licensing authority; details of licensing activities under the act are presented in Section 14 of this report.

The assessment of licence applications to dispose of dredged material at sea, includes the need to determine the concentrations of certain chemicals in the dredged sediment, to ensure they will not represent an unacceptable hazard to the marine environment and its resources after disposal has taken place. When considering which chemical compounds need to be determined in dredged material, MAFF follows the guidelines set out by the Oslo Commission for the management of dredged sediment (Oslo Commission, 1993). These guidelines recommend the methodology for sampling and the determinands to be measured in the sediment prior to dredging and sea disposal and where appropriate, the guidelines recommend analysing for chlorobiphenyls. These compounds remain significant

and widely dispersed contaminants. They can be rapidly assimilated into the food chain and they have recently been reliably linked with the inability of certain marine mammals to reproduce (Morris *et al.*, 1989).

Over the last few years, MAFF has established a useful and informative set of data on chlorobiphenyl concentrations in dredged sediment. This summary paper aims to review the data collected since 1990 and discusses the approach the licensing authority has taken in monitoring chlorobiphenyl contamination of dredged sediment.

13.2 Methods

Oslo Commission Guidelines suggest that each dredged area is sampled every 3 to 5 years and using this time scale, samples are requested by MAFF in response to the appropriate licence renewal application. Samples are sometimes requested from across the whole sediment arising, to ensure they are representative of the area to be dredged, but more frequently, sample locations are selected by MAFF to reflect areas where sediments are fine enough to retain contaminants.

The selection of contaminants for analysis is based on a number of factors, including; the type of sediment, any previously collected data from the area and any known commercial or industrial activities in the area that may contribute to the contamination of the sediment.

If chlorobiphenyl analysis is considered to be necessary, hexane washed glass jars are forwarded to the applicant with instructions on how to obtain and store the sediment. The labelled jars are returned, usually within 48 hours of the sampling operation and processed using published methodologies with minor modifications (Allchin *et al.*, 1989). The detection limit for individual chlorobiphenyl congeners is 0.2 µg kg^{-1} (dry weight) and values of <0.20 µg kg^{-1} are treated as zero.

The Oslo Commission Guidelines recommend measurement of the 7 congeners in the ICES primary list (ICES, 1986), with a secondary list of 8 congeners which should be measured where there are suspected local sources of contamination or known historic inputs.

MAFF actually undertakes measurement of the ICES 7 congeners, the Oslo Commission Guidelines secondary 8 congeners and a further 10 individual congeners, so that 25 separate chlorobiphenyl congeners are routinely determined. The full suite of IUPAC congener numbers measured by MAFF is shown in Appendix 2.

13.3 Results

A summary of the collected data for 1990 to 1994 is shown in Table 15 and Figure 18. The full dataset is given in Appendix 3.

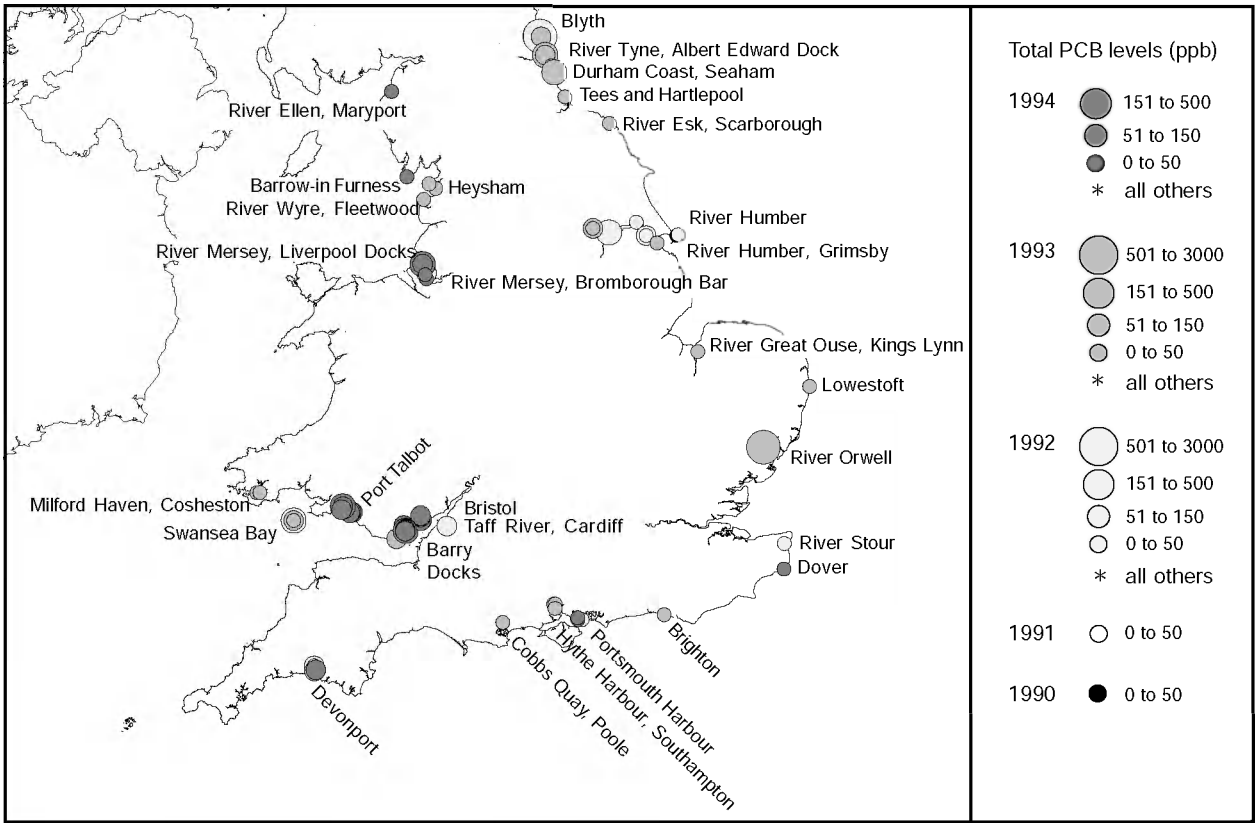


Figure 18. Total PCB levels in dredged material licensed for sea disposal in the UK, 1990-1994

Table 15. Mean concentrations of chlorobiphenyls measured in sediment samples from specific areas since 1990

Year	Location		Mean ΣICES7 (μg kg^{-1})	Mean Σ25CBs (μg kg^{-1})
1990	River Humber (n = 2)		4.00	6.00
1991	Durham Coast (n = 1)		26.80	39.80
1992	River Mersey, Liverpool Docks (n = 9)		22.53	52.50
	Tyne (n = 1)		10.80	21.80
	River Test, Southampton (n = 1)		6.40	12.40
	River Mersey, Garston (n = 7)		21.20	46.94
	River Stour (n = 1)		1.00	3.50
	River Humber, Hull (n = 1)		11.90	28.30
	River Humber, Goole (n = 1)	*	71.80	298.70
	Blyth (n = 28)		36.72	67.11
	Blyth Excluded Area (n = 6)	*	691.40	2138.55
	Devonport (n = 2)		18.20	38.80
	Swansea Bay (n = 2)		27.80	62.30
	River Usk, Newport (n = 1)		56.80	137.70
	River Taff, Cardiff (n = 1)		62.70	151.10
	Bristol (n = 1)		21.50	54.20
	Portsmouth Harbour (n = 6)		6.70	14.23
	River Tyne (n = 10)		13.51	28.84
1993	Swansea Bay (n = 2)		34.85	76.00
	Blyth (n = 5)		54.14	85.30
	Milford Haven (n = 1)		6.20	10.60
	River Wyre, Fleetwood (n = 3)		1.77	3.50
	Southampton (n = 1)		14.80	30.10
	River Wear, Port of Sunderland (n = 3)		9.67	16.80
	Devonport (n = 23)		13.73	28.83
	River Humber (n = 4)		19.63	53.68
	Brighton (n = 2)		0.10	0.30
	River Great Ouse, Kings Lynn (n = 3)		9.37	15.27
	Heysham (n = 5)		1.66	3.26
	Hythe Harbour, Southampton (n = 2)		14.25	28.20
	Tees and Hartlepool (n = 4)		5.68	12.88
	Barry Docks (n = 2)		30.00	69.80
	Milford Haven, Cosheston (n = 2)		1.40	3.35
	Tyne Dock, South Shields (n = 2)		3.20	11.50
	River Tyne, Albert Edward Dock (n = 9)	*	93.59	226.82
	River Orwell Marina (n = 16)		9.23	23.57
	River Orwell Marina Excluded Area (n = 3)	*	320.17	710.47
	River Wear, Port of Sunderland (Capital dredge) (n = 3)		6.57	17.40
	River Humber, Grimsby (n = 9)		7.21	16.84
	Cobbs Quay, Poole (n = 1)		5.10	11.40
	Lowestoft (n = 3)		6.43	11.53
	River Test (n = 2)		2.80	4.75
	River Esk, Scarborough (n = 2)		5.55	10.00
	Durham Coast, Seaham (n = 3)		36.87	85.07
1994	Portsmouth Harbour, Gosport (n = 4)		10.73	21.58
	Cardiff Bay Barrage Project (n = 3)		53.90	123.50
	River Ellen, Maryport (n = 6)		3.48	6.87
	Barrow-in-Furness (n= 6)		1.53	3.33
	River Mersey, Liverpool Docks (n = 10)		41.98	99.04
	Manchester Ship Canal (n = 1)		10.60	24.20
	River Mersey, Bromborough Bar (n = 2)		<0.20	0.70
	Portsmouth Harbour (n = 3)		4.17	6.33
	Devonport (n = 31)		15.96	32.82
	Dover (n = 1)		1.80	3.00
	Cardiff Docks (n = 9)		71.74	171.88
	River Usk, Newport (n = 5)		48.44	120.10
	River Taff, Cardiff (n = 5)		50.08	121.22
	Port Talbot (n = 5)		23.90	55.56
	Swansea (n = 5)		49.40	110.58

** = sea disposal licence refused*

n = number of individual samples tested for PCBs

Table 15 lists the areas sampled each year since 1990, together with the sum of the ICES 7 congeners (in µg kg^{-1}) and the sum of the full suite of 25 congeners (in µg kg^{-1}) that were measured. The data shown in this table are the mean values of the results of several samples taken at a different number of sites for each area.

Figure 18 is a geographical representation of the data showing the areas sampled for each year from 1990 to 1994.

It can be seen from Table 15, that 36 out of a total of 60 areas measured since 1990, contained sediments with concentrations of chlorobiphenyls greater than 50 µg kg^{-1}. Seventeen sites (28%) fell into the 51-150 µg kg^{-1} range and 5 sites (8%) fell into the 151-500 µg kg^{-1} range. Two sites contained sediments with concentrations of chlorobiphenyls above 500 µg kg^{-1}. Those sites where a sea disposal licence has been refused on the basis of the chlorobiphenyl contamination are marked in Table 15.

13.4 Discussion

The impact of chlorobiphenyls on the marine environment is difficult to quantify although it is possible to assess the potential impact using ecotoxicological information. In the study following the Piper-Alpha incident in the North Sea in 1988, Wells *et al.* (1989) defined a series of concentration guidelines (see Section 8), based on an Aroclor 1254 formulation and these have been used previously by MAFF to assess contamination levels in sediment samples (MAFF, 1993).

Using these guidelines, it is clear that the majority of dredged areas chosen for PCB analysis, shown in Table 15, fall into the slightly contaminated and contaminated categories; albeit that these data are based on the sum of 25 individual congeners and not on the Aroclor 1254 formulation used by Wells *et al.* This is not altogether surprising as these areas were specifically chosen for analysis because of suspected contamination; this in itself is encouraging in that the correct areas appear to have been targeted for sampling.

The concentrations of individual congeners are combined to assess the overall degree of contamination by PCBs. The arithmetic sum of the 25 individual congeners routinely determined (Σ25CBs) is used to express the total content. If (Σ25CBs) is found to be close to, or below, low level concentrations that are now found in the environment, the sediment may be considered to be acceptable for sea disposal. As the (Σ25CBs) increases there is increasing cause for concern, and high concentrations can be indicative of local inputs.

In those cases where the (Σ25CBs) is high enough to give cause for concern, further information is required from the applicant regarding potential sources of contamination and whether or not the contamination is a result of an historic input or of continuing practices. The relative concentrations of individual congeners can be a useful guide as to the type and timing of the inputs. The outcome of these further enquiries can lead to one of a number of options being suggested to the applicant in their further handling of the sediment.

High concentrations of chlorobiphenyls in all samples from a particular area can lead to a sea disposal licence being refused (Table 15). In such cases an alternative, non-dispersive disposal arrangement needs to be found, such as landfill. However, this does not preclude dredged sediment from the same area being permitted for sea disposal in subsequent years, if the source of the contamination can be identified and removed and concentrations of chlorobiphenyls can be shown to be reducing with time and be below an arbitrarily defined limit value.

It is more often the case that only part of the area to be dredged contains high concentrations of chloro-biphenyls. The practical solution in such cases, is to exclude the contaminated area or areas from the sea disposal licence whilst alternative disposal arrangements are found. The rest of the site can then be dredged for sea disposal. Sampling at Blyth in 1992 and at a marina on the River Orwell in 1993, found some very heavily contaminated areas; these are defined as 'Excluded Area' in each case in Table 15. In both cases further sampling was undertaken to determine the spatial extent of the contamination and once this had been achieved, the contaminated areas were removed to landfill in advance of the uncontaminated areas being dredged for sea disposal.

13.5 Conclusions

The analysis of samples taken within areas of dredging activity has been useful in identifying areas where high concentrations of chlorobiphenyls are found, so allowing decisions to be made on what sediment is acceptable for sea disposal. As was expected, the data show that CBs are ubiquitous environmental contaminants. In particular, sediments in or adjacent to industrial areas can contain high concentrations of chlorobiphenyls as a result of local and prolonged discharges. These areas require careful assessment when considering sea disposal of dredged sediment.

The data can also provide a useful record of broad changes in concentrations of chlorobiphenyls in dredged areas over time. Where significant reduction in concentrations occur with time, sediments from previously excluded areas may become acceptable for sea disposal.

Although the analysis procedure for chlorobiphenyls is a time-consuming and a costly process, there is a clear need for continued assessment of chlorobiphenyl

contamination in dredged sediment. The supply and new use of chlorobiphenyls has been prohibited in the UK since 1986 and the Government is committed to phasing out all identifiable uses of chlorobiphenyls by the end of 1999 (Department of the Environment, 1994). Continuing attention will be paid to the presence of chlorobiphenyls in dredged sediment by MAFF to prevent unacceptable levels of these compounds from entering the marine environment through this route.

The dataset will continue to expand as additional samples from selected sites are analysed and several of the sites listed in Table 15, will be re-analysed over the next few years. More detailed reports of the dataset will be published in due course.

14. LICENSING OF DEPOSITS IN THE SEA

14.1 Introduction

This section gives information about the licensing of deposits in the sea during 1994 under Part II of the Food and Environment Protection Act (1985) (FEPA) (Great Britain - Parliament, 1985(a)). It fulfils an undertaking given to Parliament by the Government to report on the licensing, enforcement and monitoring of activities related to the disposal of wastes at sea.

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

This report also includes statistics of other deposits in the sea which are principally for construction purposes.

14.2 Legislation and licensing authorities

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

Section 147 of the Environmental Protection Act (1990) (Great Britain-Parliament, 1990) provides for further additional material about sea disposal activities to be added to the public registers, established under FEPA. A Statutory Instrument relating to licences issued in England and Wales came into force on 1 July 1996, extending the information to be held on the public register about licence applications, breaches of the legislation and enforcement action. Similar Orders are in preparation in respect of licensing activity in Scotland and Northern Ireland.

14.3 Enforcement

Scientists from the MAFF Burnham-on-Crouch Laboratory have powers to enforce licence provisions. Visits are made to construction sites and treatment works, storage facilities and disposal vessels. Samples are taken and records, including logbooks, are checked. Scientific staff carried out 5 inspections in 1994. The Sea Fisheries Inspectorate, with staff based on the coast, detects unlicensed activities and enforces licence conditions relating to construction and the disposal of wastes in designated disposal areas. They made 423 inspections in 1994.

In Scotland, similar enforcement powers are held by staff of the SOAEFD Marine Laboratory, Aberdeen and by the Scottish Fisheries Protection Agency (SFPA). The Marine Laboratory made 3 inspections in 1994 and a further 29 enforcement visits were made by the SFPA. In Northern Ireland, 8 inspections were made in 1994.

MAFF took enforcement action following the investigation of a number of alleged instances of licence infringement during 1994. Legal proceedings were initiated against a dredging company over dumping short of the licensed site and log book falsification. Two licences were revoked due to breaches of terms and conditions of the consent. The first was a licence held by consultants acting for a south coast marina, where the total tonnage of dredged material for disposal was exceeded by a significant margin; the other was a licence held by a processing company, where there was concern about the effects of depositing shellfish waste in a river estuary.

Several instances of unlicensed works were identified during the course of monitoring studies of a designated Special Protection Area in North Wales. In those cases where works were continuing, or had only recently been completed, licences applications were required to be submitted.

A robust warning was issued and similar enforcement action taken in respect of a programme of sewerage improvements being carried out by a water company in south west England for which licences had not been secured.

A warning letter was issued to a marina on the east coast where a vessel deposited dredged material at the wrong location having used incorrect co-ordinates for the disposal site. A similar warning letter was sent to a hotel owner in Devon who carried out unlicensed rock armouring works to protect his sea frontage.

Enforcement action in Scotland resulted in six investigations of alleged illegal deposits and in four cases retrospective applications were submitted. Two investigations were pursued in Northern Ireland.

14.4 Report on licensing activities

Tables 16 to 20 give details, over the period 1990-1994 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)).

14.5 Licensing of minestone disposal

Only two licences were issued for disposal of minestone in 1994, for Ellington Colliery, (County Durham) and Point of Ayr (North Wales). The quantity actually deposited was less than a quarter of that disposed of during 1993. At Ellington, production was suspended in February 1994; the mine being sold to RJB Mining. No further disposal onto the beach took place that year. At Point of Ayr, a seawall was constructed to contain the minestone and this was completed in October 1994. Thereafter, since all minestone disposal took place behind this wall, no licence renewal was required.

Table 16(a). Solid industrial wastes licensed for disposal at sea in 1994[1]

Licensed quantity (t)	Company and source of waste	Description of waste	Disposal sites	Quantity Deposited (t)
	British Coal Collieries			
	Seaham	Minestone	Bankside at Seaham	0 [2]
	Easington	Minestone	Foreshore at Easington	0 [2]
600,000	Ellington	Minestone	Foreshore at Ellington	71,849
100,000	Point of Ayr	Minestone	Foreshore at Point of Ayr	91,935

Notes: [1] *No solid industrial wastes were licensed or disposed of in Scotland or Northern Ireland during the period covered by this report*
[2] *Disposed of under licences issued in 1993*
***Licensed quantities**: licences were issued throughout the calendar year 1993 and were generally valid for twelve months*
***Tonnages deposited**: relate to quantities deposited in the calendar year 1994, which may be covered by two licences, including one issued in 1993*

Table 16(b). Summary of solid industrial waste licensed and disposed of at sea in 1994[1]

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1990	9	6,575,000	4,919,654	0.29	24	182	0.21	64	223	441
	1991	8	5,750,000	4,530,860	0.28	23	172	0.21	60	212	414
	1992	8	5,080,000	3,649,727	0.22	18	133	0.16	46	163	319
	1993	4	1,800,000	2,205,670	0.14	12	98	0.09	36	117	241
	1994	2	700,000	163,784	0.00	0	0	0.00	0	0	0

Notes: [1] *No solid industrial wastes were licensed or disposed of in Scotland or Northern Ireland in the period covered by this report*
For information on licensed quantities and tonnages deposited see footnote to Table 16(a)

14.6 Licensing of sewage sludge disposal

Table 17(a) gives details of licences issued for the disposal of sewage sludge at sea. Total quantities of key metallic contaminants in sewage sludge actually disposed of at sea are shown in Table 17(b). Figure 19 shows the location of the disposal sites for sewage sludge and the quantities deposited at each site.

At the 1987 Second International Conference on the Protection of the North Sea, (Department of Environment, 1987), the Government indicated it was taking urgent action to reduce the contamination by persistent, toxic or bioaccumulable materials present in sewage sludge deposited in the North Sea and to ensure that the quantities of such contaminants deposited in the North Sea did not increase above 1987 levels. Earlier reports explained that to apply this control, limits were set for a series of key contaminants deposited under each licence. Table 18 shows how the aggregate figures authorised for disposal in the North Sea have been reduced, in most cases substantially in percentage terms, with the estimated quantity at 1987 licensed levels. Most of these contaminants come from general domestic sources rather than from industry.

14.7 Licensing of dredged material disposal

The bulk of the dredged material licensed for disposal at sea is silt and sand, but coarse sand and shingle can occur in 'maintenance' dredging and shingle, cobbles, rocks and heavy clay can be present in 'capital' material arising, for example, from channel development and deepening. Table 19 shows the numbers of licences issued, the quantity licensed, and the quantity deposited, together with figures for the quantity of a

Table 17(a). Sewage sludge licensed for disposal at sea in 1994

Country	Licensed quantity (t) [1]	Company and source of waste	Disposal sites	Quantity Deposited (t) [1]
England	80,000	Anglian Water (Cliff Quay STW, Ipswich)	Roughs Tower	76,386
and Wales	65,000	Anglian Water (Colchester STW)	Roughs Tower	9,486
	150,000	Anglian Water (Tilbury STW)	Roughs Tower	139,720
	554,000	Northumbrian Water (Howdon, Chester-le-Street,Cramlington,Washington STWs)	Tyne/Spurn Head	484,231
	105,000	Northumbrian Water (Portrack,Billingham, Guisborough,Ayton STWs)	Tyne/Spurn Head	70,346
	1,965,000	North West Water (Davyhulme,Liverpool, Warrington STWs)	Liverpool Bay	1,951,806
	4,970	North West Water (Walney Island)	Liverpool Bay	3,745
	300,000	Southern Water (Woolston,Portswood, Millbrook,Slowhill Copse STWs)	Nab Tower	264,405
	58,000	South West Water (Countess Wear STW)	Lyme Bay	38,012
	55,000	South West Water (Plympton,Radford, Camel's Head,Ernesettle,Ivybridge, Saltash,Newton Ferrers STWs)	Plymouth	44,341
	1,400,000	Thames Water (Crossness STW)	Barrow Deep	1,331,445
	3,100,000	Thames Water (Beckton,Riverside and Deephams STWs)	Barrow Deep	2,922,324
	140,000	Yorkshire Water (Knostrop STW)	Spurn Head	138,602
Scotland	500,000	Lothian Regional Council	St Abb's Head/ Bell Rock	327,510
	2,500,000	Strathclyde Regional Council	Garroch Head	1,603,000
Northern Ireland	251,860	Dept. Environment (Northern Ireland)	Belfast Sludge	251,860 [2]

Notes: [1] *All figures are for tonnage in wet weight*
[2] *Disposed of by DOE (NI) Water Services under an administrative authorisation*
STW = Sewage Treatment Works
For information on licensed quantities and tonnages deposited see footnote to Table 16(a)

Table 17(b). Summary of sewage sludge licensed and disposed of at sea in 1994

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England and Wales	1990	14	8,553,579	7,098,944	1.76	68	113	0.88	18	109	238
	1991	14	8,373,569	7,449,084	1.88	67	121	0.92	17	103	269
	1992	12	7,985,000	7,739,369	1.70	61	123	0.85	17	93	273
	1993	13	7,884,000	7,733,656	1.09	49	112	0.65	15	74	216
	1994	12	7,911,970	7,474,849	0.94	45	123	0.69	13	81	217
Scotland	1990	2	3,000,000	1,946,430	0.27	34	30	0.16	3	18	39
	1991	2	3,000,000	1,984,035	0.24	35	41	0.10	3	20	44
	1992	2	3,000,000	1,984,525	0.21	30	36	0.10	2	18	40
	1993	2	3,000,000	1,946,340	0.19	18	26	0.07	3	18	35
	1994	2	3,000,000	1,930,510	0.14	27	29	0.10	3	16	42
Northern Ireland	1990	1	80,000	290,030 *	0.03	2	3	0.04	0	2	11
	1991	1	80,000	302,370 *	0.04	2	3	0.04	0	3	11
	1992	0	0	261,000 *	0.04	2	3	0.02	0	3	10
	1993	0	0	243,200 $	0.04	2	3	0.02	0	3	9
	1994	0	0	251,860 @	0.03	4	4	0.03	1	2	12
UK Total	1990	18	10,701,305	9,642,847 *	2.06	104.00	146.59	1.07	20.67	129.21	288.34
	1991	17	11,633,579	9,335,404 *	2.15	103.56	164.97	1.06	20.05	125.07	325.17
	1992	17	11,453,569	9,735,489 *	1.95	93.37	163.07	0.97	19.46	114.01	322.73
	1993	14	10,985,000	9,984,894 $	1.31	69.35	140.74	0.75	18.18	94.65	259.77
	1994	15	10,884,000	9,923,196 @	1.10	77.15	156.09	0.83	16.95	99.44	270.43

*Notes: * Includes 200,000 t disposed of by DOE(NI) Water Services under an administrative authorisation*
$ Includes 261,000 t disposed of by DOE(NI) Water Services under an administrative authorisation
@ Includes 243,200 t disposed of by DOE(NI) Water Services under an administrative authorisation
For information on licensed quantities and tonnages deposited see footnote to Table 16(a)

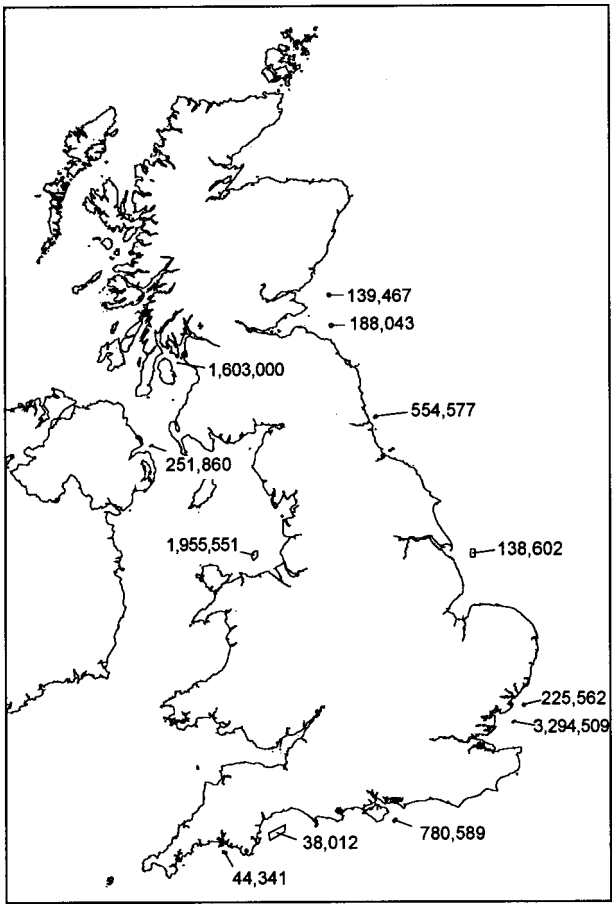


Figure 19. UK sewage-sludge disposal sites and amounts deposited in tonnes for 1994

range of trace metals which enter the sea in the dredged materials. As noted in previous reports, a proportion of the trace metals in dredged material is natural and occurs within the mineral structure or is otherwise tightly bound, such that it will not be available to marine organisms. Figure 20 shows the main disposal sites used in 1994 and the quantities deposited at each site. However all applicants for licences are now required to show evidence that they have considered alternative disposal options including beneficial uses for dredged material and why such alternatives are not considered practical.

14.8 Other materials deposited at sea

Under Part II of FEPA, licences are also required for certain activities other than for the disposal of wastes, but nevertheless involving the deliberate deposit of articles or substances in the sea. The majority of such cases relate to construction works below mean high water. Each licence application is carefully considered, in particular, to assess the impact upon tidal and inter-tidal habitat, the hydrological effects and potential interference to other users of the sea.

Further activities involved the use of tracers, the application of biocides and burials at sea. Generally, the anticipated environmental impact from deposit of these substances is minimal and little or no monitoring is required. Table 20 shows the numbers of such licences issued in 1994.

Table 18. Contaminants in sewage sludge authorised for disposal in the North Sea in 1994 compared against estimated quantities in 1987

Year	Cd	Cr	Cu	Hg	Ni	Pb	Zn
1987	3.7	56.2	133.6	1.2	19.4	146.4	468.2
1989	3.7	55.4	133	1.2	19.2	143.9	460
1990	3.4	49.7	132.5	1.2	18.6	136	435.6
1991	2.4	43.5	128.4	1.1	15.9	126.2	340
1992	2.3	43.8	127.1	1.1	15.8	125.3	339.4
1993	2	39.3	123	1	14.9	114.8	295.5
1994	1.2	28.7	120.3	0.8	14	91.8	212.8

Table 19. Summary of dredged material licensed and disposed of at sea in 1994

Country	Year	Licences issued	Licensed quantity (t)	Wet tonnage deposited	Quantities of metal contaminants in wastes deposited (t)						
					Cd	Cr	Cu	Hg	Ni	Pb	Zn
England	1990	135	63,983,920	33,728,978	12.2	1,023	834	6.8	484	1,426	3,724
and Wales	1991	108	57,782,520	39,886,812	7.4	1,189	773	7.0	518	1,263	3,394
	1992	123	55,741,813	24,243,998	6.0	812	512	4.2	291	876	2,271
	1993	110	66,074,966	26,086,503	7.3	875	606	5.2	458	1,004	2,461
	1994	106	53,187,009	34,049,468	8.0	1,295	734	5.9	587	1,375	3,375
Scotland	1990	21	3,031,960	2,109,114	0.8	67	64	0.8	30	106	207
	1991	26	5,147,245	2,788,611	0.6	70	53	0.6	22	79	167
	1992	35	5,920,005	3,841,296	0.9	108	82	1.7	39	111	245
	1993	26	3,174,050	2,025,525	2.4	50	44	0.8	21	63	132 $
	1994	23	3,643,250	1,822,053	0.9	42	36	0.5	20	56	122
Northern	1990	6	261,700	317,082	0.1	3	3	0.0	3	5	7
Ireland	1991	10	807,400	519,049	0.1	2	2	0.1	3	2	6
	1992	7	2,956,601	891,087	0.3	2	3	0.2	2	3	10
	1993	7	996,500	3,392,994 @	1.8	11	26	1.1	13	23	70
	1994	5	113,200	91,314	0.0	0	0	0.0	0	0	1
UK total	1990	162	67,277,580	36,155,174	13.2	1,093	900	7.6	518	1,537	3,938
	1991	144	63,737,165	43,194,472	8.1	1,260	828	7.8	543	1,344	3,566
	1992	165	64,618,419	28,976,381	7.2	923	597	6.1	332	990	2,527
	1993	143	70,245,516	31,505,022 @	11.5	937	676	7.1	491	1,090	2,663
	1994	134	56,943,459	35,962,835	8.9	1,338	770	6.4	608	1,432	3,498

Notes: For information on licensed quantities and tonnages deposited see footnote to Table 16(a)
$ Amended contaminant data since last report due to updated analyses being received from SOAEFD
@ Amended quantity since last report due to additional data received from DOE (NI)

Table 20. Other categories of licences issued in 1994

Licence category	England and Wales	Scotland	Northern Ireland	Total
Construction - new and renewal	222	70	5	297
Tracers, biocides etc.	16	0	0	16
Burial at Sea	11	0	0	11

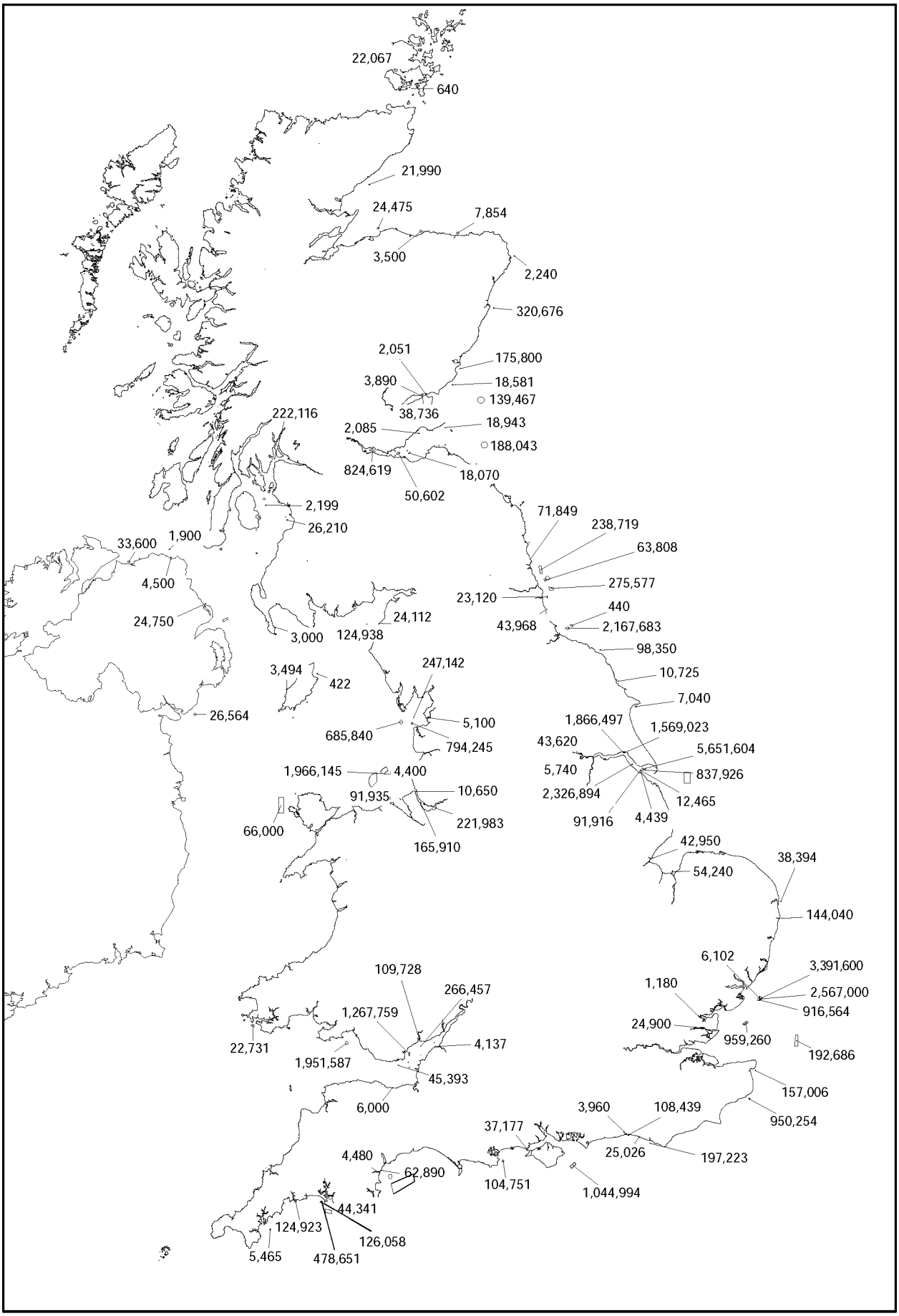


Figure 20. UK dredged material disposal sites and amounts deposited in tonnes for 1994

15. ADVICE ON FISHERY IMPLICATIONS OF PIPELINE DISCHARGES

This section gives a brief summary of activities carried out during 1994 in connection with the provision of advice on fishery implications of pipeline discharges. The background to this work in relation to MAFF's responsibilities as a statutory consultee under the Water Resources Act, 1991 (Great Britain - Parliament, 1991(a)), was described in previous reports in this series (MAFF, 1991, 1992, 1993, 1994 and 1995).

A total of 206 applications was sent to MAFF, Burnham for comment during 1994. As in previous years, the majority of these (about 80%) was for sewage discharges. Discharges of domestic sewage to coastal waters pose few problems for finfish or crustacea unless they contain persistent plastics or other material which could foul fishing gear. The discharge of such material is gradually being reduced through upgrading to comply with the requirements of EC Directive 91/271 concerning urban waste water treatment (European Communities, 1991(b)).

The main fishery implication 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 which may have entered the water from discharges of sewage or treated sewage effluent. Although these micro-organisms do not harm the shellfish, they may accumulate to levels which put public health at risk. In England and Wales, public health controls are currently based on EC Directive 91/492 laying down the health conditions for the production and the placing on the market of live bivalve molluscs (European Communities, 1991(a)). This requires all commercially harvested beds of bivalve molluscs to be classified according to levels of microbiological contamination in the shellfish flesh. The degree of contamination determines whether the shellfish may be marketed without treatment (Category A), or must be purified (Category B), or relaid in cleaner water then purified or heat treated by an approved process (Category C) before sale. In cases of extreme contamination harvesting is prohibited altogether.

Because of practical difficulties with relaying of shellfish in clean areas, MAFF's aim in advising on sewage discharges near shellfish areas is to press for schemes which will achieve a shellfish classification of at least Category B as soon as possible. However, Directive 91/492 does not place a statutory obligation on Member States to maintain a particular quality of shellfish harvesting area, merely to ensure that shellfish undergo the level of treatment before sale which is appropriate to the classification of the bed. Water Companies are therefore under no obligation to design or modify a scheme for the sole purpose of reducing contamination of shellfish by sewage micro-organisms. Many shellfish beds are in areas where re-routing or upgrading of sewage discharges is required in order to meet statutory obligations, particularly EC Directive 76/160 concerning the quality of bathing water (European Communities, 1976) and EC Directive 91/271 concerning urban waste water treatment (European Communities, 1991(b)). In 1994, MAFF used its role as statutory consultee to try

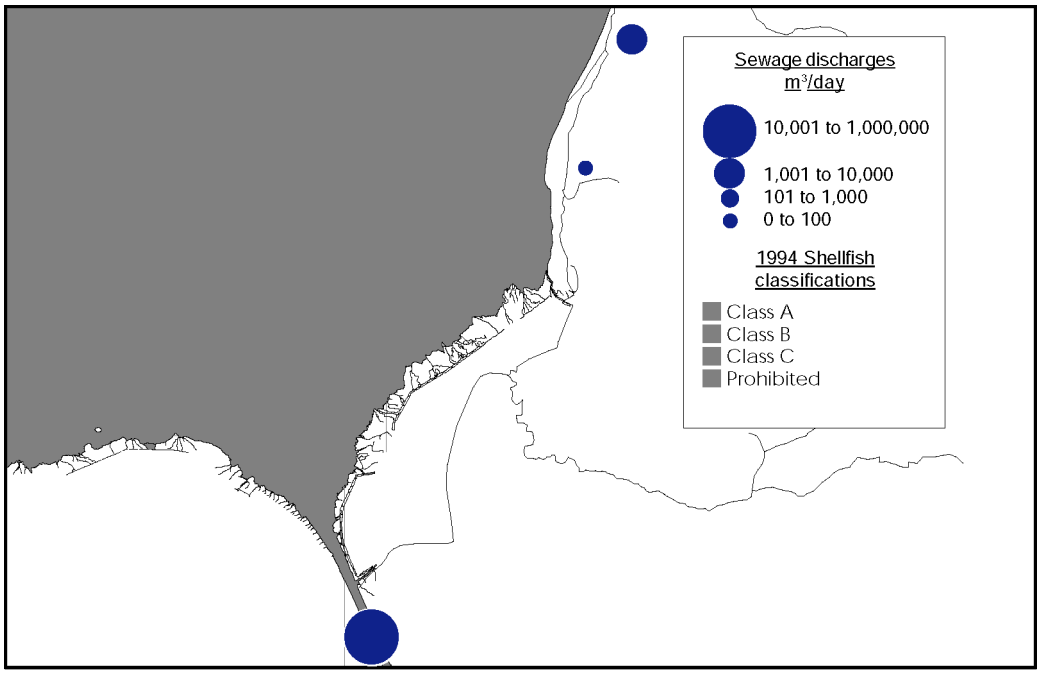


Figure 21. Location of sewage discharges in relation to hygiene classification of bivalve mollusc production areas

and negotiate the best possible advantage for shellfisheries (e.g. increased level of treatment, better outfall location or lower storm overflow frequency) in the design of such schemes (about 18 in all).

The classification of shellfish harvesting areas is determined by MAFF's Fish Diseases Laboratory (FDL) at Weymouth, based on an assessment of microbiological data collected by local food authorities. To assist in the classification, the Burnham and Weymouth laboratories have collaborated to produce maps showing the location of sewage discharges in relation to shellfish beds (see Figure 21). By helping to identify existing discharges which contribute to unsatisfactory shellfish classifications, the maps will also aid in the assessment of discharge applications. Future plans include integrating the shellfish maps and discharge data with other information on activities in the coastal zone, to produce a computerised geographic information system (GIS). This will be used to help provide advice in response to statutory consultations, licensing of deposits at sea under the Food and Environment Protection Act and other coastal zone management issues, as well as responding to emergencies.

During 1994, about 20% of the discharge applications sent to MAFF for comment under the Water Resources Act were for trade effluent (industrial) discharges. Most of these were for non-hazardous discharges (e.g. surface water), or for upgrading of existing discharges, and required little input from MAFF on fishery implications. Since 1 April 1991, discharges from the most potentially polluting or technologically complex industrial processes in England and Wales have no longer been controlled by the NRA under the Water Resources Act, but have been subject to authorisation by Her Majesty's Inspectorate of Pollution (HMIP) under Part I of the Environmental Protection Act, 1990 (Great Britain - Parliament, 1990). This Act has replaced separate controls for releases of polluting substances from industrial processes to air, water and land by a system of Integrated Pollution Control (IPC). IPC requires that all prescribed processes (as listed in The Environmental Protection (Prescribed Processes and Substances) Regulations, 1991 (Great Britain - Parliament, 1991(b))) operate according to Best Available Techniques (not entailing excessive costs). Wastes from these processes are disposed of after an assessment of the Best Practicable Environmental Option (BPEO). New processes came under immediate control, with controls on existing processes being phased in according to a rolling programme. During 1994, procedures were put in place to ensure that all applications for new processes which include a discharge to tidal water will be assessed for fishery implications in the same way as those covered under the Water Resources Act.

Acknowledgement

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

BIOTA

The use of bioassays to assess the quality of marine sediments	Y Allen J E Thain
Contaminants in marine mammals	R J Law
Mercury and PCBs in fish and cadmium in mussels	A Franklin
Heavy metals in oysters from the Fal Estuary	J Jones
Thallium in marine fish/shellfish	B Jones

WATER

Oestrogenic effects produced by sewage effluent	M J Waldock
Alkylphenols in rivers and fish tissue	M Blackburn J Osborne

SEDIMENTS

Estuarine sediments around England and Wales	S Rowlatt
Chlorobiphenyls in fine surficial sediments from Cardigan Bay	R J Law
Long-term monitoring of sediment quality in Liverpool Bay	S Rowlatt

DEPOSITS AT SEA: ENVIRONMENTAL ASSESSMENT STUDIES

Benthic studies at the Thames Estuary sewage-sludge disposal site	H Rees E Wootton*
Developments in the use of meiobenthos as a monitoring tool	S Boyd

DEPOSITS IN THE SEA: LICENSING AND RELATED ACTIVITIES

Chlorobiphenyl contaminants in dredged sediment	N Gilbert J Reed L Murray
Licensing of Deposits in the Sea	G Boyes C M G Vivian
Advice on fishery implications of pipeline discharges	F L Franklin

* *University College Swansea*

ANNEX 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).

Community Decision 93/351 EEC (European Communities, 1993) applies to samples of fishery products. This states that the mean total mercury content of the edible parts of fishery products must not exceed 0.5 mg kg^{-1} fresh weight, increased to 1.0 mg kg^{-1} fresh weight for some species listed in an annex.

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

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

(b) Cadmium

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

The JMP guidelines for cadmium in mussels are as follows:

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

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

(c) Lead

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

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

(d) Copper

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

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

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

(e) Zinc

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

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

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

A2.2 Pesticides/PCBs

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

(a) HCB

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

(b) HCH

Codex Alimentarius Commission's maximum residue limit (MRL) (FAO/WHO, 1987) is 2 mg kg^{-1} in meat fat for 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*

A2.3 References

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APPENDICES

These appendices contain the raw data associated with the work carried out by the Burnham-on-Crouch Laboratory in 1994. Although not all of the data are referred to in the report, they are included here for completeness.

Appendix 1. RV CIROLANA 10 June-1 July 1994: 10 day sediment bioassays using the lugworm Arenicola marina and the amphipod Corophium volutator

Station	Actual Position		Location	*Corophium*	*Arenicola*	
				% mortality	% mortality	Total casts in 10 days
				Test 1	Test 1	
			Shoebury Sands (Control)	10	13	123
1	55° 45.75' N	1° 57.61' W	Tweed (anchor)	23	13	91
2	55° 45.83' N	1° 59.58' W	Tweed (buoy off Spittal Point)	33	20	121
3	55° 46.12' N	2° 00.35' W	Tweed (Berwick / Tweed Bridges)	30	13	94
4	55° 07.26' N	1° 28.21' W	Blyth (anchor)	23		
7	55° 01.48' N	1° 23.24' W	Tyne (anchor)	47	33	111
13	54° 59.82' N	1° 26.47' W	Tyne (by South Ferry landing)	17	**73	*16
14	54° 59.17' N	1° 27.86' W	Tyne (buoy off Jarrow Slake)	20	27	*80
15	54° 59.09' N	1° 31.49' W	Tyne (Hebburn)	33	20	*84
16	54° 58.09' N	1° 36.25' W	Tyne (Tyne Bridge)	23	27	*80
17	55° 00.45' N	1° 07.45' W	off Tyne		33	93
22	54° 56.32' N	1° 19.76' W	Wear (anchor)	53	33	97
23	54° 55.05' N	1° 21.43' W	Wear (Sandy Point)	40	27	*71
24	54° 54.58' N	1° 22.87' W	Wear (Wearmouth Bridge)	57	33	*29
25	54° 54.80' N	1° 24.24' W	Wear (Queen Alexandra Bridge)	33	7	*55
26	54° 44.03' N	0° 53.13' W	Off Tees	37	20	*79
27	54° 39.85' N	1° 04.56' W	Tees (anchor)	23	33	*88
				Test 2	Test 2	
			Shoebury Sands (control)	13	20	94
35	54° 37.79' N	1° 09.72' W	Tees (Phillips approach buoy)	30	27	*24
36	54° 37.40' N	1° 09.34' W	Tees (Redcar jetty)	**100**	27	*50
40	54° 35.53' N	1° 15.03' W	Tees (Bamlett's Bight)	47		
41	54° 49.95' N	1° 20.05' W	Off Tyne/Tees	**60**	7	93
42	54° 00.09' N	1° 59.99' E	Off Humber/Wash	10		
43	52° 49.93' N	2° 49.82' E	Southern Bight (Smith's Knoll)	10		
44	51° 59.99' N	2° 19.90' E	Thames (Outer Gabbard)	17	0	*11
45	51° 30.70' N	0° 58.01' E	Thames (Warp)		7	87
51	50° 39.29' N	1° 54.15' W	Poole Harbour (anchor)	23		
52	50° 41.15' N	1° 57.35' W	Poole Harbour (Brownsea buoy : no. 42)	13		
54	50° 43.30' N	1° 59.93' W	Poole Harbour (Holes Bay : Cobb's Quay marina)	13		
				Test 3		
			Shoebury Sands (control)	3		
57	50° 04.86' N	3° 00.12' W	Central Channel / Perintis 7	7	0	*45
58	50° 20.81' N	4° 08.95' W	Plymouth Sound (anchor)	0		
59	50° 22.93' N	4° 11.60' W	Tamar (Hamoaze)	3	20	83
60	50° 24.37' N	4° 12.05' W	Tamar (Royal Albert Bridge)	0		
61	50° 25.45' N	4° 11.99' W	Tamar (E. Tamar north buoy)	0	13	100
66	50° 17.71' N	4° 09.73' W	Off Tamar	3		
67	50° 01.98' N	4° 21.99' W	Off Plymouth Sound / Perintis 8	7	20	*53
71	48° 30.04' N	7° 59.98' W	Western Approaches / Perintis 12	17	0	*33
74	53° 30.06' N	3° 41.76' W	Liverpool Bay	7		
75	54° 00.05' N	3° 50.12' W	SE Isle of Man	0		
76	53° 58.02' N	3° 19.88' W	Off Morecambe Bay	3	20	96
77	53° 28.76' N	3° 16.03' W	Burbo Bight / Mersey (anchor)	3		
85	53° 20.11' N	2° 57.22' W	Mersey (Bromborough E1 buoy)	10	33	*14
86	53° 19.40' N	2° 56.85' W	Mersey (Eastham Lock)	**63**	20	84
87	53° 23.70' N	2° 59.70' W	Mersey (Tranmere oil terminal)	**43**	13	107
88	53° 24.56' N	3° 00.48' W	Mersey (Seacombe Ferry)	0	0	*53
89	53° 28.00' N	3° 02.90' W	Mersey (C20 buoy)	3	13	158
93	53° 19.98' N	3° 12.08' W	Dee (no.2 buoy)	0		
94	51° 14.98' N	5° 59.88' W	Celtic Deep	10		

Note: Corophium mortality significantly different from control (ANOVA, p<0.05) indicated in ***bold***
*Arenicola mortality significantly different from control (ANOVA, p<0.05) indicated by '**'*
Arenicola casting significantly different from control (Dunnets t-test, p<0.05) indicated by ''*

Appendix 2. Suite of 25 chlorobiphenyl congeners routinely measured at the Burnham-on-Crouch Laboratory

Congener number	Structure
CB18	2,2',-trichlorobiphenyl
CB28 *	2,4,4'-trichlorobiphenyl
CB31	2,4',5-trichlorobiphenyl
CB44	2,2',3,5'-tetrachlorobiphenyl
CB47	2,2',4,4'-tetrachlorobiphenyl
CB49	2,2',4,5'-tetrachlorobiphenyl
CB52 *	2,2',5,5'-tetrachlorobiphenyl
CB66	2,3',4,4'-tetrachlorobiphenyl
CB101 *	2,2',4,5,5'-pentachlorobiphenyl
CB105	2,3,3',4,4'-pentachlorobiphenyl
CB110	2,3,3',4',6-pentachlorobiphenyl
CB118 *	2,3',4,4',5-pentachlorobiphenyl
CB128	2,2',3,3',4,4'-hexachlorobiphenyl
CB138 *	2,2',3,4,4',5'-hexachlorobiphenyl
CB141	2,2',3,4,5,5'-hexachlorobiphenyl
CB149	2,2',3,4',5',6-hexachlorobiphenyl
CB151	2,2',3,5,5',6-hexachlorobiphenyl
CB153 *	2,2',4,4',5,5'-hexachlorobiphenyl
CB156	2,3,3',4,4',5-hexachlorobiphenyl
CB158	2,3,3',4,4',6-hexachlorobiphenyl
CB170	2,2',3,3',4,4',5-heptachlorobiphenyl
CB180 *	2,2',3,4,4',5,5'-heptachlorobiphenyl
CB183	2,2',3,4,4',5',6-heptachlorobiphenyl
CB187	2,2',3,4',5,5',6-heptachlorobiphenyl
CB194	2,2',3,3',4,4',5,5'-octachlorobiphenyl

Note: The suite of CBs comprises 3 trichlorobiphenyls, 5 tetrachlorobiphenyls, 4 pentachlorobiphenyls, 8 hexachlorobiphenyls, 4 heptachlorobiphenyls and 1 octachlorobiphenyl. The 7 congeners comprising the ICES primary list are indicated by asterisks

Appendix 3. Summary of PCB data collected between 1990 and 1994

Year	Location	Concentration (µg kg^{-1})								
		CB#18	CB#28	CB#31	CB#44	CB#47	CB#49	CB#52	CB#66	CB#101
1990										
	River Humber (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
1991										
	Durham Coast (n = 1)	<0.20	5.70	<0.20	<0.20	<0.20	<0.20	6.30	2.60	10.00
1992										
	River Mersey, Liverpool Docks (n = 9)	1.82	4.17	2.91	2.03	0.66	1.58	3.07	2.98	3.39
	Tyne (n = 1)	<0.20	1.60	1.60	0.60	<0.20	0.50	1.90	0.90	1.50
	River Test, Southampton (n = 1)	<0.20	1.40	0.90	0.70	<0.20	0.50	0.60	0.80	0.80
	River Mersey, Garston (n = 7)	2.49	4.36	2.17	2.04	0.37	2.90	3.37	3.10	2.76
	River Stour (n = 1)	<0.20	0.50	0.50	0.40	<0.20	0.30	<0.20	0.70	<0.20
	River Humber, Hull (n = 1)	<0.20	<0.20	<0.20	2.20	<0.20	0.30	<0.20	2.70	3.30
	River Humber, Goole (n = 1)	1.20	<0.20	3.30	0.60	<0.20	50.00	0.20	4.80	1.30
	Blyth (n = 28)	1.86	14.56	0.01	0.42	0.18	0.34	0.61	1.83	2.83
	Blyth Excluded Area (n = 6)	4.68	19.33	<0.20	5.90	9.13	15.40	19.58	69.00	95.17
	Devonport (n = 2)	<0.20	2.45	<0.20	0.20	1.20	0.35	0.30	0.40	2.65
	Swansea Bay (n = 2)	5.50	5.40	0.65	2.15	0.65	1.50	2.75	4.75	3.55
	River Usk, Newport (n = 1)	8.10	17.00	11.00	6.40	2.10	4.40	5.90	9.90	5.10
	River Taff, Cardiff (n = 1)	9.50	19.00	12.00	7.00	2.30	4.90	6.60	11.00	5.50
	Bristol (n = 1)	4.90	6.30	3.60	2.40	0.70	1.60	2.10	3.80	1.60
	Portsmouth Harbour (n = 6)	1.30	0.63	0.17	0.43	0.08	0.35	0.63	1.03	1.02
	River Tyne (n = 10)	1.95	2.37	1.03	0.63	0.09	0.54	1.41	1.12	2.15
1993										
	Swansea Bay (n = 2)	<0.20	4.90	3.10	1.85	0.65	1.35	1.90	3.70	4.15
	Blyth (n = 5)	1.66	25.00	0.06	1.04	0.08	0.58	7.74	3.16	3.08
	Milford Haven (n = 1)	0.30	2.80	0.60	0.30	<0.20	0.20	<0.20	1.00	0.40
	River Wyre, Fleetwood (n = 3)	<0.20	0.27	0.13	0.07	0.07	0.07	0.23	0.20	0.27
	Southampton (n = 1)	0.80	1.70	1.10	1.10	0.30	0.80	1.70	1.50	2.20
	River Wear, Port of Sunderland (n = 3)	1.00	1.73	0.70	0.37	<0.20	0.17	2.90	0.77	0.93
	Devonport (n = 23)	0.50	0.80	0.19	0.38	0.09	0.64	0.74	1.33	2.19
	River Humber (n = 4)	1.58	2.18	4.10	1.40	0.60	1.40	3.05	3.08	2.03
	Brighton (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	0.20	<0.20
	River Great Ouse, Kings Lynn (n = 3)	0.40	1.43	0.70	0.27	<0.20	0.20	4.03	0.93	0.77
	Heysham (n = 5)	<0.20	0.28	0.10	<0.20	<0.20	<0.20	0.48	0.46	0.14
	Hythe Harbour, Southampton (n = 2)	0.35	1.60	0.65	0.65	0.30	0.90	1.60	1.35	2.30
	Tees and Hartlepool (n = 4)	<0.20	1.38	0.78	<0.20	<0.20	1.50	0.08	1.80	0.70
	Barry Docks (n = 2)	3.00	8.45	5.05	2.65	0.70	2.20	2.75	5.80	2.50
	Milford Haven, Cosheston (n = 2)	<0.20	0.55	0.30	<0.20	<0.20	<0.20	<0.20	0.50	<0.20
	Tyne Dock, South Shields (n = 2)	<0.20	0.40	0.45	<0.20	<0.20	3.00	<0.20	0.95	0.85
	River Tyne, Albert Edward Dock (n = 9)	5.39	10.96	10.37	9.00	2.29	15.91	13.54	19.11	15.69
	River Orwell Marina (n = 16)	0.08	1.11	0.59	0.31	<0.20	5.81	0.66	1.48	1.11
	River Orwell Marina Excluded Area (n = 3)	4.07	2.70	2.20	14.00	1.40	8.67	37.00	49.00	68.33
	River Wear, Port of Sunderland (Capital dredge) (n = 3)	0.33	1.10	0.77	0.30	<0.20	3.63	0.43	1.53	0.87
	River Humber, Grimsby (n = 9)	0.36	1.69	0.76	0.34	<0.20	1.87	0.42	1.38	0.76
	Cobbs Quay, Poole (n = 1)	<0.20	0.90	0.40	<0.20	<0.20	0.60	<0.20	1.40	0.70
	Lowestoft (n = 3)	<0.20	0.60	<0.20	0.23	<0.20	<0.20	0.43	0.67	0.73
	River Test (n = 2)	0.20	0.35	<0.20	<0.20	<0.20	<0.20	0.10	0.55	0.20
	River Esk, Scarborough (n = 2)	<0.20	0.40	<0.20	<0.20	<0.20	<0.20	0.25	1.00	1.00
	Durham Coast, Seaham (n = 3)	7.10	12.33	9.20	4.40	1.57	2.83	6.13	6.07	4.20
1994										
	Portsmouth Harbour, Gosport (n = 4)	<0.20	<0.20	<0.20	0.45	<0.20	<0.20	0.78	1.98	1.78
	Cardiff Bay Barrage Project (n = 3)	6.77	15.33	9.33	5.07	2.13	<0.20	5.60	9.60	4.50
	River Ellen, Maryport (n = 6)	<0.20	0.08	<0.20	0.07	<0.20	<0.20	0.23	0.50	0.57
	Barrow-in-Furness (n = 6)	<0.20	0.28	0.10	0.05	0.28	0.20	0.07	0.30	0.13
	River Mersey, Liverpool Docks (n = 10)	4.59	12.42	7.08	4.48	2.02	5.73	6.08	8.22	4.58
	Manchester Ship Canal (n = 1)	1.30	2.10	1.20	0.90	0.40	0.60	1.60	1.70	1.30
	River Mersey, Bromborough Bar (n = 2)	0.70	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
	Portsouth Harbour (n = 3)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	0.13	0.83
	Devonport (n = 31)	0.98	3.62	0.80	1.20	0.56	1.71	2.17	2.68	1.91
	Dover (n = 1)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
	Cardiff Docks (n = 9)	8.67	18.33	10.47	6.91	2.58	6.19	7.78	11.11	6.46
	River Usk, Newport (n = 5)	7.60	13.60	7.60	4.88	1.80	3.78	5.04	8.58	4.18
	River Taff, Cardiff (n = 5)	7.00	15.40	7.28	5.42	2.00	4.32	5.44	8.94	4.36
	Port Talbot (n = 5)	2.64	6.82	3.70	2.06	0.74	1.80	2.32	4.20	2.12
	Swansea (n = 5)	2.74	7.18	3.86	2.84	0.98	2.68	4.60	7.16	7.14

Appendix 3. continued

Year	Location	Concentration (µg kg^{-1})								
		CB#105	CB#110	CB#118	CB#128	CB#138	CB#141	CB#149	CB#151	CB#153
1990										
	River Humber (n = 2)	<0.20	<0.20	<0.20	<0.20	2.00	<0.20	2.00	<0.20	2.00
1991										
	Durham Coast (n = 1)	<0.20	8.20	<0.20	<0.20	2.00	<0.20	<0.20	<0.20	2.80
1992										
	River Mersey, Liverpool Docks (n = 9)	2.38	6.48	3.37	0.81	3.50	0.76	2.86	0.68	3.29
	Tyne (n = 1)	0.70	3.00	1.60	0.20	1.80	0.30	1.60	0.30	1.70
	River Test, Southampton (n = 1)	<0.20	2.10	0.80	0.20	1.20	0.20	0.20	<0.20	1.10
	River Mersey, Garston (n = 7)	0.79	5.70	3.13	0.47	3.10	0.49	2.21	0.47	2.84
	River Stour (n = 1)	<0.20	0.60	<0.20	<0.20	0.30	<0.20	<0.20	<0.20	0.20
	River Humber, Hull (n = 1)	0.20	5.20	1.60	0.20	3.60	0.70	2.40	0.70	3.40
	River Humber, Goole (n = 1)	75.00	17.00	52.00	27.00	1.60	0.40	9.50	<0.20	15.00
	Blyth (n = 28)	1.54	3.87	0.95	0.89	5.58	2.13	5.83	2.27	7.58
	Blyth Excluded Area (n = 6)	374.32	117.70	29.33	56.95	163.62	59.75	183.67	74.50	205.50
	Devonport (n = 2)	2.65	5.60	3.55	1.00	4.35	1.05	3.70	0.55	3.45
	Swansea Bay (n = 2)	<0.20	9.70	3.40	0.85	5.20	1.00	3.35	0.85	5.05
	River Usk, Newport (n = 1)	<0.20	17.00	5.30	1.60	9.10	1.80	6.50	1.80	7.80
	River Taff, Cardiff (n = 1)	<0.20	18.00	5.80	1.60	10.00	2.10	6.50	2.10	8.70
	Bristol (n = 1)	<0.20	7.10	2.10	0.50	3.70	0.60	2.60	0.70	3.10
	Portsmouth Harbour (n = 6)	<0.20	2.32	1.18	0.15	1.32	0.15	0.87	0.12	1.38
	River Tyne (n = 10)	0.49	5.11	1.99	0.32	2.35	0.40	1.54	0.33	2.01
1993										
	Swansea Bay (n = 2)	<0.20	11.50	4.75	1.40	7.95	1.80	5.55	1.70	7.35
	Blyth (n = 5)	0.04	5.52	0.24	0.54	5.96	2.16	5.10	1.66	6.86
	Milford Haven (n = 1)	0.20	1.30	0.70	<0.20	0.80	<0.20	<0.20	<0.20	0.90
	River Wyre, Fleetwood (n = 3)	0.20	0.60	0.27	<0.20	0.30	<0.20	0.20	<0.20	0.30
	Southampton (n = 1)	<0.20	4.90	2.60	0.60	2.70	0.40	1.60	0.30	2.80
	River Wear, Port of Sunderland (n = 3)	0.63	2.43	1.03	0.07	1.20	0.17	0.20	<0.20	1.10
	Devonport (n = 23)	1.11	5.23	3.04	0.92	3.31	0.44	1.59	0.29	2.54
	River Humber (n = 4)	0.70	6.75	0.78	0.45	2.88	1.20	3.80	1.08	6.68
	Brighton (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
	River Great Ouse, Kings Lynn (n = 3)	0.37	1.67	0.60	0.27	0.90	0.27	0.20	<0.20	0.83
	Heysham (n = 5)	0.16	0.38	0.14	0.16	0.16	0.18	<0.20	<0.20	0.14
	Hythe Harbour, Southampton (n = 2)	0.40	4.80	2.40	0.35	2.65	0.35	1.55	0.35	2.45
	Tees and Hartlepool (n = 4)	0.13	2.80	0.68	<0.20	1.08	0.03	0.03	<0.20	0.95
	Barry Docks (n = 2)	0.95	8.75	3.30	0.60	4.85	0.65	3.25	0.65	4.40
	Milford Haven, Cosheston (n = 2)	0.10	0.70	0.30	<0.20	0.30	<0.20	0.20	0.15	0.25
	Tyne Dock, South Shields (n = 2)	0.30	2.45	0.75	0.15	0.75	0.20	0.45	0.35	0.45
	River Tyne, Albert Edward Dock (n = 9)	7.74	36.33	17.53	2.11	16.73	2.09	9.26	1.93	12.34
	River Orwell Marina (n = 16)	0.40	2.87	1.91	0.37	1.93	0.24	0.94	0.14	1.59
	River Orwell Marina Excluded Area (n = 3)	25.20	138.33	74.00	16.40	71.67	13.90	46.67	10.50	51.67
	River Wear, Port of Sunderland (Capital dredge) (n = 3)	0.43	2.37	1.20	0.23	1.33	0.07	0.53	<0.20	0.90
	River Humber, Grimsby (n = 9)	0.60	2.23	0.80	0.12	1.42	0.20	0.26	0.02	1.16
	Cobbs Quay, Poole (n = 1)	0.70	2.60	1.30	0.20	1.20	<0.20	<0.20	<0.20	0.60
	Lowestoft (n = 3)	<0.20	2.33	1.47	0.23	1.50	0.20	0.57	0.10	1.13
	River Test (n = 2)	<0.20	1.00	0.55	<0.20	0.70	<0.20	0.10	<0.20	0.55
	River Esk, Scarborough (n = 2)	<0.20	2.00	1.25	<0.20	1.20	0.15	0.45	<0.20	0.95
	Durham Coast, Seaham (n = 3)	<0.20	8.60	4.70	0.90	4.27	0.83	2.73	0.57	3.50
1994										
	Portsmouth Harbour, Gosport (n = 4)	<0.20	4.05	2.80	0.65	2.43	0.35	1.35	0.18	1.98
	Cardiff Bay Barrage Project (n = 3)	<0.20	14.33	6.13	1.53	8.63	1.77	6.17	1.87	7.37
	River Ellen, Maryport (n = 6)	0.38	1.62	0.92	0.15	0.92	0.10	0.32	0.05	0.67
	Barrow-in-Furness (n = 6)	0.17	0.43	0.32	0.10	0.28	<0.20	0.07	<0.20	0.32
	River Mersey, Liverpool Docks (n = 10)	2.01	10.67	5.78	1.06	5.38	1.02	3.74	0.90	4.90
	Manchester Ship Canal (n = 1)	1.00	3.40	1.50	0.30	1.60	0.30	1.10	0.20	1.70
	River Mersey, Bromborough Bar (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20
	Portsmouth Harbour (n = 3)	<0.20	1.93	0.87	<0.20	0.73	<0.20	0.10	<0.20	1.63
	Devonport (n = 31)	0.74	4.12	2.61	0.40	2.63	0.25	1.40	0.65	2.22
	Dover (n = 1)	<0.20	0.40	<0.20	<0.20	0.50	<0.20	0.20	<0.20	0.60
	Cardiff Docks (n = 9)	3.32	20.78	7.49	2.03	11.63	2.42	7.98	2.61	10.37
	River Usk, Newport (n = 5)	2.22	14.40	5.12	1.38	7.76	1.68	5.80	1.82	6.80
	River Taff, Cardiff (n = 5)	2.20	14.00	5.20	1.40	7.60	1.58	5.74	1.76	6.62
	Port Talbot (n = 5)	1.08	6.40	2.78	0.62	3.80	0.70	2.52	0.66	3.40
	Swansea (n = 5)	3.06	16.46	7.80	1.96	10.10	2.04	6.60	1.74	8.08

Appendix 3. continued

Year	Location	Concentration (µg kg^{-1})								
		CB#156	CB#158	CB#170	CB#180	CB#183	CB#187	CB#194	ΣICES7	Σ25CBs
1990										
	River Humber (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	4.00	6.00
1991										
	Durham Coast (n = 1)	<0.20	<0.20	<0.20	<0.20	<0.20	2.20	<0.20	26.80	39.80
1992										
	River Mersey, Liverpool Docks (n = 9)	0.42	0.49	1.17	1.76	0.57	1.13	0.27	22.53	52.50
	Tyne (n = 1)	<0.20	0.20	0.30	0.70	0.20	0.60	<0.20	10.80	21.80
	River Test, Southampton (n = 1)	0.20	<0.20	0.20	0.50	<0.20	<0.20	<0.20	6.40	12.40
	River Mersey, Garston (n = 7)	0.09	0.04	1.10	1.64	0.17	0.93	0.21	21.20	46.94
	River Stour (n = 1)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	1.00	3.50
	River Humber, Hull (n = 1)	<0.20	<0.20	<0.20	<0.20	0.50	1.30	<0.20	11.90	28.30
	River Humber, Goole (n = 1)	23.00	4.30	0.60	1.70	2.40	7.60	0.20	71.80	298.70
	Blyth (n = 28)	0.49	0.66	2.76	4.61	1.84	2.83	0.62	36.72	67.11
	Blyth Excluded Area (n = 6)	16.87	122.95	89.67	158.87	128.00	97.33	21.33	691.40	2138.55
	Devonport (n = 2)	0.70	0.65	1.15	1.45	0.60	0.55	0.25	18.20	38.80
	Swansea Bay (n = 2)	0.55	<0.20	1.35	2.45	0.60	0.70	0.35	27.80	62.30
	River Usk, Newport (n = 1)	0.70	0.90	2.90	6.60	1.60	3.30	0.90	56.80	137.70
	River Taff, Cardiff (n = 1)	0.80	1.00	3.20	7.10	1.80	3.70	0.90	62.70	151.10
	Bristol (n = 1)	0.40	0.40	1.20	2.60	0.60	1.30	0.30	21.50	54.20
	Portsmouth Harbour (n = 6)	0.10	<0.20	0.23	0.53	0.05	0.18	<0.20	6.70	14.23
	River Tyne (n = 10)	0.21	0.19	0.48	1.23	0.44	0.36	0.10	13.51	28.84
1993										
	Swansea Bay (n = 2)	0.75	0.85	3.40	3.85	1.00	2.00	0.55	34.85	76.00
	Blyth (n = 5)	0.96	0.46	3.18	5.26	1.52	2.68	0.76	54.14	85.30
	Milford Haven (n = 1)	<0.20	<0.20	0.30	0.60	<0.20	<0.20	0.20	6.20	5.60
	River Wyre, Fleetwood (n = 3)	<0.20	<0.20	0.07	0.13	0.07	0.07	<0.20	1.77	3.50
	Southampton (n = 1)	0.40	0.30	0.50	1.10	0.30	0.40	<0.20	14.80	11.20
	River Wear, Port of Sunderland (n = 3)	<0.20	<0.20	0.23	0.77	0.17	0.23	<0.20	9.67	16.80
	Devonport (n = 23)	0.50	0.41	0.60	1.10	0.26	0.48	0.13	13.73	28.83
	River Humber (n = 4)	0.40	0.35	2.20	2.05	1.13	2.98	0.88	19.63	53.68
	Brighton (n = 2)	<0.20	<0.20	<0.20	0.10	<0.20	<0.20	<0.20	0.10	0.30
	River Great Ouse, Kings Lynn (n = 3)	0.20	0.10	0.13	0.80	0.07	0.13	<0.20	9.37	15.27
	Heysham (n = 5)	0.08	<0.20	0.08	0.32	<0.20	<0.20	<0.20	1.66	3.26
	Hythe Harbour, Southampton (n = 2)	0.30	0.25	0.50	1.25	0.30	0.60	<0.20	14.25	28.20
	Tees and Hartlepool (n = 4)	<0.20	<0.20	<0.20	0.83	<0.20	0.13	0.03	5.67	12.88
	Barry Docks (n = 2)	0.40	0.35	1.35	3.75	0.75	2.20	0.50	30.00	69.80
	Milford Haven, Cosheston (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	1.40	3.35
	Tyne Dock, South Shields (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	3.20	11.50
	River Tyne, Albert Edward Dock (n = 9)	1.98	1.89	2.78	6.79	1.27	2.92	0.87	93.59	226.82
	River Orwell Marina (n = 16)	0.09	0.06	0.38	0.94	0.09	0.48	0.01	9.23	23.57
	River Orwell Marina Excluded Area (n = 3)	11.30	9.80	26.43	14.80	4.33	6.77	1.33	320.17	710.47
	River Wear, Port of Sunderland (Capital dredge) (n = 3)	0.07	<0.20	0.27	0.73	<0.20	0.30	<0.20	6.57	17.40
	River Humber, Grimsby (n = 9)	0.26	0.40	0.44	0.97	0.03	0.24	0.12	7.21	16.84
	Cobbs Quay, Poole (n = 1)	<0.20	<0.20	0.40	0.40	<0.20	<0.20	<0.20	5.10	4.00
	Lowestoft (n = 3)	0.13	0.13	0.20	0.57	0.07	0.17	0.07	6.43	11.53
	River Test (n = 2)	<0.20	<0.20	0.10	0.35	<0.20	<0.20	<0.20	2.80	4.75
	River Esk, Scarborough (n = 2)	<0.20	<0.20	0.15	0.50	0.35	0.35	<0.20	5.55	10.00
	Durham Coast, Seaham (n = 3)	0.57	0.57	0.83	1.73	0.37	0.87	0.20	36.87	85.07
1994										
	Portsmouth Harbour, Gosport (n = 4)	0.38	0.23	0.55	0.97	0.18	0.43	0.10	10.73	21.58
	Cardiff Bay Barrage Project (n = 3)	0.87	0.93	2.70	6.33	1.73	3.87	0.93	53.90	123.50
	River Ellen, Maryport (n = 6)	0.08	0.07	0.05	0.10	<0.20	<0.20	<0.20	3.48	6.87
	Barrow-in-Furness (n = 6)	<0.20	<0.20	0.10	0.13	<0.20	<0.20	<0.20	1.53	3.33
	River Mersey, Liverpool Docks (n = 10)	0.79	0.34	1.66	2.84	0.67	1.65	0.43	41.98	99.04
	Manchester Ship Canal (n = 1)	<0.20	<0.20	0.50	0.80	0.20	0.50	<0.20	10.60	11.10
	River Mersey, Bromborough Bar (n = 2)	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	<0.20	0.70
	Portsmouth Harbour (n = 3)	<0.20	<0.20	<0.20	0.10	<0.20	<0.20	<0.20	4.17	6.33
	Devonport (n = 31)	0.25	0.17	0.37	0.81	0.14	0.44	0.02	15.96	32.82
	Dover (n = 1)	<0.20	<0.20	0.20	0.70	<0.20	0.40	<0.20	1.80	0.00
	Cardiff Docks (n = 9)	1.28	1.32	3.70	9.69	2.23	4.94	1.59	71.74	171.88
	River Usk, Newport (n = 5)	0.76	0.86	2.42	5.94	1.64	3.48	0.96	48.44	120.10
	River Taff, Cardiff (n = 5)	0.72	0.82	2.30	5.46	1.50	3.28	0.88	50.08	121.22
	Port Talbot (n = 5)	0.38	0.38	1.08	2.66	0.66	1.66	0.38	23.90	55.56
	Swansea (n = 5)	1.28	1.20	2.14	4.50	1.18	2.62	0.64	49.40	110.58

Appendix 4. Stations sampled during RV CIROLANA Cruise 7, 10-26 June 1994

Station	Date	Position		Location	NMP
1	12-Jun	55° 45.75' N	1° 57.61' W	Tweed (anchor)	
2	12-Jun	55° 45.83' N	1° 59.58' W	Tweed (buoy off Spittal Point)	
3	12-Jun	55° 46.12' N	2° 00.35' W	Tweed (Berwick/Tweed Bridges)	
4	12-Jun	55° 07.26' N	1° 28.21' W	Blyth (anchor)	
5	12-Jun	55° 07.30' N	1° 29.70' W	Blyth (South Harbour entrance)	
6	12-Jun	55° 08.10' N	1° 30.75' W	Blyth (North Blyth)	
7	13-Jun	55° 01.48' N	1° 23.24' W	Tyne (anchor)	
8	13-Jun	54° 59.82' N	1° 26.47' W	Tyne (by South Ferry landing)	TY3
9	13-Jun	54° 59.17' N	1° 27.86' W	Tyne (buoy off Jarrow Slake)	
10	13-Jun	54° 59.09' N	1° 31.49' W	Tyne (Hebburn)	225
11	13-Jun	54° 58.09' N	1° 36.25' W	Tyne (Tyne Bridge)	
12	13-Jun	55° 01.57' N	1° 23.32' W	Tyne (anchor)	
13	13-Jun	54° 59.82' N	1° 26.47' W	Tyne (by South Ferry landing)	TY3
14	13-Jun	54° 59.17' N	1° 27.86' W	Tyne (buoy off Jarrow Slake)	
15	13-Jun	54° 59.09' N	1° 31.49' W	Tyne (Hebburn)	225
16	13-Jun	54° 58.09' N	1° 36.25' W	Tyne (Tyne Bridge)	
17	13-Jun	55° 00.45' N	1° 07.85' W	Off Tyne	245
18	14-Jun	54° 56.24' N	1° 19.70' W	Wear (anchor)	
19	14-Jun	54° 55.05' N	1° 21.43' W	Wear (Sandy Point)	275
20	14-Jun	54° 54.58' N	1° 22.87' W	Wear (Wearmouth Bridge)	
21	14-Jun	54° 54.80' N	1° 24.24' W	Wear (Queen Alexandra Bridge)	265
22	14-Jun	54° 56.32' N	1° 19.76' W	Wear (anchor)	
23	14-Jun	54° 55.05' N	1° 21.43' W	Wear (Sandy Point)	275
24	14-Jun	54° 54.58' N	1° 22.87' W	Wear (Wearmouth Bridge)	
25	14-Jun	54° 54.80' N	1° 24.24' W	Wear (Queen Alexandra Bridge)	265
26	14-Jun	54° 44.03' N	0° 53.13' W	Off Tees	295
27	15-Jun	54° 39.85' N	1° 04.56' W	Tees (anchor)	
28	15-Jun	54° 37.79' N	1° 09.72' W	Tees (Phillips approach buoy)	TS3
29	15-Jun	54° 37.40' N	1° 09.34' W	Tees (Redcar jetty)	
30	15-Jun	54° 36.22' N	1° 09.90' W	Tees (ICI no. 4 buoy)	
31	15-Jun	54° 35.66' N	1° 10.84' W	Tees (no. 23 buoy)	TS2
32	15-Jun	54° 35.07' N	1° 13.57' W	Tees (Transporter bridge)	
33	15-Jun	54° 35.53' N	1° 15.03' W	Tees (Bamlett's Bight)	TS1
34	15-Jun	54° 39.83' N	1° 04.58' W	Tees (anchor)	
35	15-Jun	54° 37.79' N	1° 09.72' W	Tees (Phillips approach buoy)	TS3
36	15-Jun	54° 37.40' N	1° 09.34' W	Tees (Redcar jetty)	
37	15-Jun	54° 36.22' N	1° 09.90' W	Tees (ICI no. 4 buoy)	
38	15-Jun	54° 35.66' N	1° 10.84' W	Tees (no. 23 buoy)	TS2
39	15-Jun	54° 35.07' N	1° 13.57' W	Tees (Transporter bridge)	
40	15-Jun	54° 35.53' N	1° 15.03' W	Tees (Bamlett's Bight)	TS1
41	15-Jun	54° 49.95' N	1° 20.05' E	Off Tyne/Tees	285
42	16-Jun	54° 00.09' N	1° 59.99' E	Off Humber/Wash	345
43	16-Jun	52° 49.93' N	2° 49.82' E	Southern Bight (Smith's Knoll)	395
44	16-Jun	51° 59.99' N	2° 19.90' E	Thames (Outer Gabbard)	475
45	17-Jun	51° 30.70' N	0° 58.01' E	Thames (Warp)	465
46	17-Jun	51° 55.98' N	1° 16.64' E	South Varne	485
47	17-Jun	50° 44.97' N	0° 10.01' W	Perintis 1	

Appendix 4. continued

Station	Date	Position		Location	NMP
48	17-Jun	50° 40.00' N	0° 29.92' W	Perintis 2	
49	17-Jun	50° 39.01' N	0° 50.06' W	Selsey Bill/Perintis 3	495
50	18-Jun	50° 29.96' N	1° 29.85' W	Perintis 4	
51	18-Jun	50° 39.29' N	1° 54.15' W	Poole Harbour (anchor)	
52	18-Jun	50° 41.15' N	1° 57.35' W	Poole Harbour (Brownsea buoy : no. 42)	
53	18-Jun	50° 42.40' N	1° 58.95' W	Poole Harbour (Stakes buoy : no. 55)	
54	18-Jun	50° 43.30' N	1° 59.93' W	Poole Harbour (Holes Bay : Cobb's Quay marina)	
55	18-Jun	50° 29.96' N	1° 59.97' W	Perintis 5	
56	18-Jun	50° 29.99' N	2° 39.94' W	Perintis 6	
57	18-Jun	50° 04.86' N	3° 00.12' W	Central Channel/Perintis 7	535
58	19-Jun	50° 20.81' N	4° 08.95' W	Plymouth Sound (anchor)	
59	19-Jun	50° 22.93' N	4° 11.60' W	Tamar (Hamoaze)	565
60	19-Jun	50° 24.37' N	4° 12.05' W	Tamar (Royal Albert Bridge)	
61	19-Jun	50° 25.45' N	4° 11.99' W	Tamar (E. Tamar north buoy)	555
62	19-Jun	50° 20.83' N	4° 08.97' W	Plymouth Sound (anchor)	
63	19-Jun	50° 22.93' N	4° 11.60' W	Tamar (Hamoaze)	565
64	19-Jun	50° 24.37' N	4° 12.05' W	Tamar (Royal Albert Bridge)	
65	19-Jun	50° 25.45' N	4° 11.99' W	Tamar (E. Tamar north buoy)	555
66	19-Jun	50° 17.71' N	4° 09.73' W	Off Tamar	575
67	19-Jun	50° 01.98' N	4° 21.99' W	Off Plymouth Sound/Perintis 8	585
68	19-Jun	49° 35.00' N	5° 14.99' W	Perintis 9	
69	20-Jun	49° 10.01' N	6° 09.92' W	Perintis 10	
70	20-Jun	48° 49.87' N	7° 04.95' W	Perintis 11	
71	20-Jun	48° 30.04' N	7° 59.98' W	Western Approaches/Perintis 12	595
72	21-Jun	51° 18.06' N	3° 32.95' W	Severn (Nash Point)	615
73	22-Jun	53° 37.66' N	4° 30.32' W	Irish Sea	775
74	22-Jun	53° 30.06' N	3° 41.76' W	Liverpool Bay	715
75	22-Jun	54° 00.05' N	3° 50.12' W	SE Isle of Man	805
76	22-Jun	53° 58.02' N	3° 19.88' W	Off Morecambe Bay	795
77	23-Jun	53° 28.76' N	3° 16.03' W	Burbo Bight/Mersey (anchor)	705
78	23-Jun	53° 31.83' N	3° 08.80' W	Mersey (C1 buoy)	765
79	23-Jun	53° 28.00' N	3° 02.90' W	Mersey (C20 buoy)	
80	23-Jun	53° 24.56' N	3° 00.48' W	Mersey (Seacombe Ferry)	755
81	23-Jun	53° 23.70' N	2° 59.70' W	Mersey (Tranmere oil terminal)	
82	23-Jun	53° 20.11' N	2° 57.22' W	Mersey (Bromborough E1 buoy)	745
83	23-Jun	53° 19.40' N	2° 56.85' W	Mersey (Eastham Lock)	
84	23-Jun	53° 28.72' N	3° 16.24' W	Burbo Bight/Mersey (anchor)	705
85	23-Jun	53° 20.11' N	2° 57.22' W	Mersey (Bromborough E1 buoy)	745
86	23-Jun	53° 19.40' N	2° 56.85' W	Mersey (Eastham Lock)	
87	23-Jun	53° 23.70' N	2° 59.70' W	Mersey (Tranmere oil terminal)	
88	23-Jun	53° 24.56' N	3° 00.48' W	Mersey (Seacombe Ferry)	755
89	23-Jun	53° 28.00' N	3° 02.90' W	Mersey (C20 buoy)	
90	23-Jun	53° 31.83' N	3° 08.80' W	Mersey (C1 buoy)	765
91	24-Jun	53° 28.68' N	3° 16.09' W	Burbo Bight/Mersey (anchor)	
92	24-Jun	53° 23.35' N	3° 14.25' W	Dee (buoy HE3/Hilbre Island)	
93	24-Jun	53° 19.98' N	3° 12.08' W	Dee (no.2 buoy)	695
94	25-Jun	51° 14.98' N	5° 59.88' W	Celtic Deep	605

Appendix 5. Water column data for stations sampled during RV CIROLANA Cruise 7, 10-26 June 1994

Station	Depth (m)	Salinity	Temperature (°C)	Concentrations of nutrients (µmol l^{-1})					Sediment type
				Phosphate	Silicate	Nitrate	Nitrite	Ammonia	
1	19	33.850	10.31	0.91	2.0	1.9	0.10	3.2	sand and shell
2		26.312	14.51	0.72	3.7	21.1	0.70	8.1	fine sand
3		7.962	12.75	1.56	6.8	74.9	1.51	2.9	sand and stones
4	18	34.151	11.34	0.48	2.1	1.7	0.29	1.9	shell
5				0.75	2.3	1.3	0.30	3.0	anoxic mud
6		33.600		1.29	3.2	2.4	0.51	5.8	anoxic mud
7	26	34.120	9.98	0.51	2.3	0.9	0.21	2.0	sand
8		33.860	10.50	0.54	2.2	1.9	0.35	4.4	NS
9		31.152	11.37	6.65	7.1	8.6	1.23	77.0	NS
10		29.482	12.08	5.10	7.9	16.4	2.45	43.6	NS
11		21.775	13.39	2.26	13.0	31.9	4.01	33.9	NS
12	23	34.073	10.57	0.54	1.9	1.6	0.31	3.2	NS
13		29.491	12.98	3.71	6.8	16.4	2.32	39.5	mud and sand
14		23.958	12.58	19.70	21.0	25.6	3.52	220.0	anoxic mud
15		22.301	13.68	2.23	12.3	39.8	3.79	36.9	anoxic mud
16		15.014	15.49	1.66	15.7	50.0	4.10	32.2	anoxic mud
17	74	34.126	11.77	0.24	2.6	1.5	0.21	1.9	mud
18	24	34.108	10.22	0.47	2.4	3.0	0.23	2.3	sand over mud
19		30.732		3.83	2.0	32.0	2.36	9.7	NS
20		29.839	11.30	4.67	2.1	38.3	3.08	11.7	NS
21		29.002	11.63	5.56	2.2	45.8	3.57	13.6	NS
22	21	33.769	11.10	1.58	2.3	4.6	0.67	3.9	sand over mud
23		32.187	11.89	2.80	2.2	17.3	1.40	7.3	sand
24		23.027	14.27	11.80	2.0	90.7	7.40	25.0	mud
25		13.536	14.86	21.20	1.7	158.8	14.10	41.7	mud
26	53	34.244	11.82	0.13	1.6	1.8	0.09	2.2	mud
27	27	34.166	11.19	0.16	1.4	1.9	0.35	3.6	sand
28		33.234	12.78	1.26	2.1	13.4	2.18	71.0	NS
29		31.958	12.49	3.89	4.1	22.5	3.56	215.0	NS
30		32.293	12.33	44.80	4.7	18.9	4.01	950.0	NS
31		32.711	12.06	3.98	2.5	15.8	2.68	167.0	NS
32		27.563		7.82	6.8	47.3	7.00	530.0	NS
33		29.252	12.20	6.96	5.3	45.8	11.30	450.0	NS
34	24	34.173	11.12	1.54	4.6	6.7	0.54	5.8	NS
35	6	32.411	12.75	2.44	1.7	23.3	3.88	167.2	sand
36	6.5	31.042	12.87	0.27	2.7	58.0	15.80	152.4	anoxic mud
37	8.6	31.551	12.69	22.30	4.7	26.4	4.06	657.2	NS
38	7.5	30.976	12.99	6.08	3.0	26.4	4.68	345.0	NS
39	7.6	25.794	13.81	8.87	6.7	56.0	6.69	659.1	NS
40	5.1	30.792	14.81	0.33	3.3	65.5	17.50	180.5	anoxic mud
41	29	34.789	11.47	0.28	1.2	0.5	0.29	0.5	fine sand and shell
42	76	34.777	11.77	0.16	0.5	0.5	0.14	1.3	sand and mud
43	34	34.218	11.94	0.13	2.1	0.8	0.14	2.1	fine sand
44	44	34.670	12.79	0.11	1.1	0.6	0.17	1.3	fine sand
45	18	34.010	14.96	0.39	1.2	0.8	0.18	1.8	sand (anoxic > 2cm)
46	31	34.786	13.00	0.25	2.7	1.0	0.21	1.2	NS - hard ground
47	21	34.605		0.18	1.4	1.0	0.16	1.3	NS

Appendix 5. continued

Station	Depth (m)	Salinity	Temperature (°C)	Concentrations of nutrients (µmol l^{-1})					Sediment type
				Phosphate	Silicate	Nitrate	Nitrite	Ammonia	
48	20	34.665		0.13	1.3	0.8	0.16	1.1	NS
49	18	34.679	14.36	0.08	0.8	0.5	0.21	1.3	NS - hard ground
50	37	34.364		0.13	1.2	2.3	0.29	1.8	NS
51	51	34.017	15.49	0.06	2.0	0.5	0.14	0.9	sand, stones and shell
52	6.8	32.542	16.84	0.10	0.7	0.3	0.14	1.3	sand (anoxic patches > 2cm)
53	2.4	30.959	17.75	0.16	0.9	0.5	0.14	1.3	anoxic mud
54	1.3	29.393	18.96	26.50	14.4	20.2	2.56	13.8	anoxic mud
55	37	34.644		0.20	2.0	0.6	0.12	1.3	NS
56	43	34.880		0.23	2.3	0.5	0.11	1.2	NS
57	66	35.011	12.83	0.13	1.3	0.5	0.11	1.0	sand and shells
58	10	34.596	13.75	0.23	0.8	0.6	0.13	1.2	sand, stones and shell
59	12.8	32.017	16.28	0.46	2.2	2.0	0.32	2.7	mud
60	5.1	25.447	17.09	0.54	4.8	13.8	0.59	4.1	mud
61	2.2	22.878	17.55	0.22	1.5	25.4	0.80	6.3	mud
62	12	34.709	13.83	0.41	1.3	1.6	0.19	2.7	NS
63	17.5	33.704	16.15	0.41	1.1	1.4	0.38	3.8	NS
64		32.149		0.24	0.9	1.3	0.29	3.2	NS
65	4.7	31.929	15.60	0.25	0.6	1.3	0.24	2.8	NS
66	42	35.090	13.13	0.46	0.5	0.8	0.23	2.4	mud
67	74	35.038	14.99	0.13	1.4	1.0	0.15	1.9	sand and shell
68	93	35.300		0.03	0.6	0.8	0.14	1.9	NS
69	117	35.265		0.66	1.0	1.3	0.32	3.1	NS
70	140	35.150		0.31	1.3	1.3	0.23	2.8	NS
71	169	35.273	14.62	0.15	1.1	1.3	0.16	1.8	sand and shell
72	26	29.109	14.49	2.05	15.5	57.8	0.95	1.3	NS - hard ground
73	68	34.179	11.00	0.56	2.6	2.6	0.28	3.1	shell, stones, sand and mud
74	38	33.811	12.43	0.41	2.1	1.2	0.15	1.7	sand
75	35	33.604	11.93	0.43	2.6	1.0	0.18	3.0	sand and shell
76	21	33.117	13.45	0.43	0.8	0.8	0.16	1.4	mud
77	18	32.539	13.85	0.79	2.0	2.5	0.53	4.1	sand over mud
78	14.2	31.303	15.30	2.38	3.0	20.6	5.88	10.1	NS
79		31.385		2.16	3.0	18.4	5.16	8.8	NS
80	22.6	31.108	14.99	2.48	3.3	24.6	7.06	11.8	NS
81		30.296		3.25	6.1	41.6	12.4	19.3	NS
82	4.7	28.529	15.39	4.67	11.0	74.5	23.7	37.1	NS
83		21.985		7.75	41.5	154.2	43.3	141.0	NS
84	13	32.092	14.88	1.38	3.3	5.4	1.3	4.0	NS
85	3.1	20.167	16.65	7.75	47.4	184.0	45.1	162.0	sand
86	4.8	21.319	16.46	7.75	43.2	182.5	46.2	141.0	mud
87	12.3	24.132	16.02	6.19	30.0	148.4	39.4	103.0	mud
88	5.3	25.100	15.91	6.31	25.5	131.7	35.7	90.0	sand
89	11.8	28.604	15.54	4.41	10.5	72.3	22.4	43.0	sand
90	20.8	31.293	15.32	2.38	2.7	20.1	4.9	9.6	NS
91	19	32.327	14.22	1.11	2.2	3.1	0.7	3.9	NS
92	11.2	32.377	15.35	0.89	0.9	0.8	0.2	1.4	NS
93	11.5	31.969	15.75	1.51	1.4	0.6	0.1	2.6	sand
94	96	34.792	14.44	0.13	1.1	0.9	0.2	1.9	mud

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