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The field assessment of effects of dumping wastes at sea:
12 The disposal of sewage sludge, industrial wastes and dredged spoils in Liverpool Bay.

M. G. NORTON, A. FRANKLIN, S. M. ROWLATT,
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1. Introduction

The Ministry of Agriculture, Fisheries and Food (MAFF) is responsible for controlling the dumping of wastes from vessels loaded in English ports under the Dumping at Sea (DAS) Act 1974 (Gt Britain Parliament, 1974) and for the implementation of the Oslo and London Conventions on the Prevention of Marine Pollution by Dumping. These responsibilites include the licensing of all dumping operations to ensure that the marine environment and its living resources are protected, enforcing the conditions of licences, and monitoring the areas used for the dumping of wastes to establish the effects of disposal. This report discusses monitoring activities carried out by MAFF in Liverpool Bay to determine the effects of sewage sludge disposal.

Liverpool Bay is generally defined as that part of the Irish Sea delimited by the North Wales coast east of Great Ormes Head and by the Lancashire coast as far north as the Ribble Estuary (Figure 1a). It has a multiplicity of uses. The Bay itself and the Irish Sea in general provide an important commercial fishery for several species of fish and shellfish (Brander, 1980). Landings from Liverpool Bay in 1979 were valued at £1.5 million, the most important species being plaice, sole, cod, whiting and rays. It is also very important as a spawning and nursery area for several commercial species – particularly plaice and sole. Other uses of the Bay are recreation, navigation (particularly to and from the port of Liverpool) and sand and gravel extraction.

Liverpool Bay has been used for the dumping of sewage sludge since the turn of the century when the first sewage treatment plants were built near Manchester, and dumping has continued without interruption at the dumping site shown in Figure 1a. In the late 1960s industrial waste dumping also started in the same area. Both types of waste were subject to a voluntary control scheme from 1970 to 1974 and were brought under full statutory control by the DAS Act in 1974. Similarly the dumping of dredgings has been regulated under the DAS Act since 1974 and has been at two sites in the Bay (Sites Z and Y, see Figure 1); most of the dredged spoil (>95%) has been dumped at site Z. The Bay also receives many wastes derived from dumping and the discharge of effluents into the three major rivers (the Mersey, the Dee and the Ribble) and from coastal outfalls.

The dumping of sewage sludge in the Bay became a matter of particular interest in 1970 when solutions were sought to the problem of disposing of large quantities of sludge expected to arise from expansions in regional sewage treatment facilities. One suggestion was to increase substantially the quantities of sludge dumped in the Bay, but no data were then available to allow a prediction of the possible effects. Consequently, the Department of the Environment (DOE) set up a working party to study the proposal and a number of investigations were funded by that Department in order to describe in more detail the properties of the Bay and the behaviour of dumped sludge.

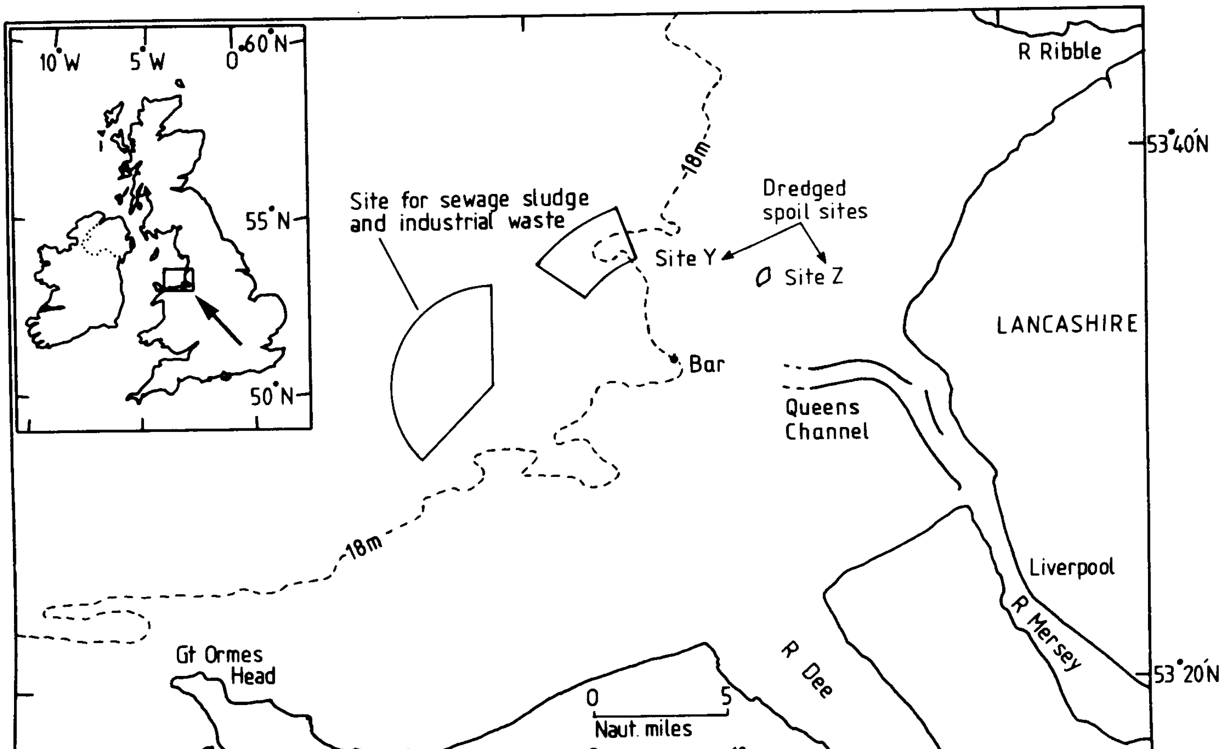
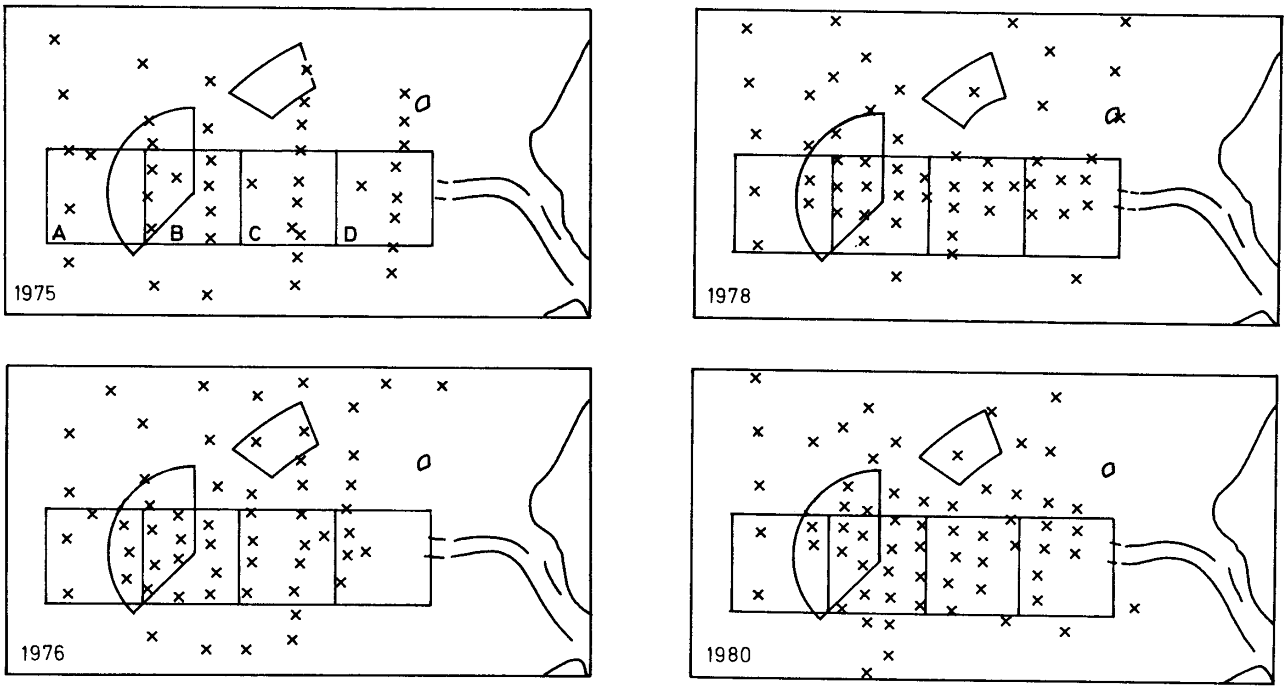


Figure 1a Liverpool Bay and the dumping areas for sewage sludge, industrial wastes and dredged spoil.



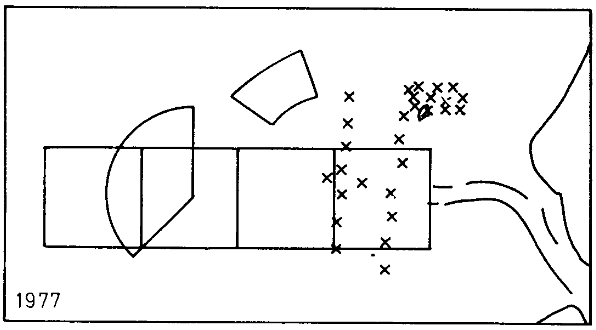


Figure 1b Sampling grids used in surveys:

X = sampling stations;

$\boxed{A}$ = delimits the grid squares referred to in Section 4.5.

The results of these studies and of further research on the area have been published (DOE, 1972a, 1972b, 1973, 1976; DOE/NWC, 1979a). They have provided a considerable amount of information on the hydrography of the area, water quality, the distribution of sludge in the water column after dumping, its subsequent settlement and movement and its impact on the sediments and the biota. Although MAFF has the statutory responsibility to monitor dumping grounds, because of the DOE studies, it has tended to concentrate its monitoring activities on the investigation of dumping-related effects which have not been fully covered in the DOE-funded programmes.

The results of MAFF surveys of sediments and benthos up to 1974 were reported by White *et al.* (1974). The investigations from 1975-80 have concentrated on the impact of dumping on the sediments and the identification of effects on the seabed biota (the benthos). This report presents the results of these detailed studies and attempts to (i) characterise the dumping area in physical, chemical and biological terms so as to provide a 'benchmark' against which the results of future studies can be compared, (ii) identify the dispersal paths and fate of the dumped waste, and (iii) identify the physical, chemical and biological effects of dumping. While the main area of investigation has been the sewage sludge dumping site, the results of more limited studies of the dredged spoil disposal sites are also included. (station positions for all surveys are shown in Figure 1 b).

A full description of the objectives and rationale of MAFF's monitoring programme for dumping grounds and details of the methods employed have already been published (Norton and Rolfe, 1978; Eagle *et al.*, 1978a). Results of studies on the effects of dumping in several other areas have been published in the MAFF Fisheries Research Technical Report series. Results of a study of the effects of dumping on water quality in Liverpool Bay in 1978/9 have been published elsewhere (Norton *et al.*, 1984) and a review of the effects of different inputs on the metal content of fish and shellfish in the Bay has also been completed (Norton and Murray, 1983).

The factors affecting sludge dispersal in the water column have been determined by earlier studies and these are summarised and reviewed in Section 4. Longer-term dispersal of sludge particles at the sea bed is linked with the movement of sediments; the hydrographic factors affecting this are also described in Section 4. The distribution of faecal bacteria at the sea bed allows areas of initial settlement of sludge particles to be identified; this is described in Section 5. Section 6 presents a detailed description of the distribution and dynamics of natural sediments and identifies the effect of dumping on their composition. The distribution and species composition of the benthos are discussed in Section 7 and, using the sediment data of Section 6, an analysis of the impact of dumping on the benthos of the survey area is made. The results of other studies carried out by MAFF (water quality investigations in 1978/79; fish and shellfish quality monitoring from 1970-81) have been published elsewhere. Nevertheless, in order to present a complete picture of the effects of waste disposal in Liverpool Bay, these results are summarised in Section 8. The conclusions reviewing the effects of dumping are summarised in Section 9.

2. Inputs of wastes and other substances to Liverpool Bay

The quantities of wastes dumped in each of the years 1976-80 are given in Tables 1-3 which feature sewage sludge, industrial wastes and dredged spoils respectively. In each table, estimates are also given of the quantity of contaminants (particularly metals) contained in the waste concerned.

The total amount of sewage sludge dumped each year varied between 0.5 and 1 million t a^{-1} from 1900 to 1972 since when it has increased to the present rate of 1.8 million t a^{-1} (Table 1). Present-day sewage sludges arise from treatment of sewage at several works adjacent to the River Mersey serving a population of nearly 2 million. The largest of these is Davyhulme at Manchester which produces 1.3 million t a^{-1} of anaerobically digested sludge. The other main works at Salford, Runcorn and Warrington produce mixed primary and secondary sludges which are undigested. The composition and quantity of sludge have varied a little during the study period and improvements in sludge quality have occurred as a result of controls imposed by the North West Water Authority (NWWA) on trade effluent discharges to sewage works.

The industrial wastes dumped in Liverpool Bay (Table 2) arise from a number of industries situated up to approximately 50 miles from the Bay. About 40 different wastes are licensed for disposal in the Bay, and include wastes from chemical, pharmaceutical, food, leather, textile and engineering processes.

Dredged spoils dumped in Liverpool Bay arise from the maintenance dredging of the channel approaches to the River Mersey, the Mersey itself, the docks and harbours along its length and the Manchester Ship Canal. The figures in Table 3 are based on the average composition of spoil from several different sources; details of the composition of spoils from specific sites have been given elsewhere (Murray and Norton, 1979).

At each of the dumping grounds the input of materials from dumping is likely to exceed that from other sources, but the effects of dumping on the Bay as a whole will be superimposed on those caused by wastes from other sources – particularly discharges from rivers, estuaries and coastal outfalls. It will be helpful in later sections of this report to be able to estimate the relative importance of different sources of each of the substances monitored; thus estimates of 1976 inputs to the Bay from sources other than dumping (DOE/NWC, 1979b) are presented in Table 4.

Table 1 Quantities of sewage sludge and its components dumped in Liverpool Bay 1976-80 (t a^{-1})

	1976*	1977	1978	1979	1980
Sewage sludge (wet weight)	1 586 888	1 688 805	1 664 605	1 650 472	1 805 969
" " (dry weight)	70 386	69 983	65 224	69 626	71 350
BOD	17 463	–	–	–	–
Nitrogen	3 490	3 880	3 660	3 590	3 700
Phosphorus	1 140	1 270	1 180	1 170	1 200
Mercury	2.7	2.5	0.9	0.7	0.8
Cadmium	2.1	2.05	1.95	1.7	1.4
Lead	55	66	58	55	60
Zinc	258	235	230	202	235
Copper	110	116	100	81	68
Chromium	93	123	110	88	78
Nickel	12	16	11	16	10

* For amounts dumped before 1976 see Head (1981)

– Not determined

Note: Lead analyses before 1979 have been adjusted by a factor of 2.2 to take account of different analytical procedures used

Table 2 Quantities of industrial wastes and their components dumped in Liverpool Bay 1976-80 (t a^{-1})

	1976	1977	1978	1979	1980
Industrial waste (wet weight)	212 146	310 738	102 234	81 754	52 449
" " (dry weight)	5 472	12 274	10 692	8 780	4 567
Nitrogen	928	2 304	1 630	1 835	1 137
Organic matter	9 362	7 822	2 856	4 375	2 811
Mercury	ND	ND	ND	ND	ND
Cadmium	ND	ND	ND	ND	ND
Lead	0.4	0.8	0.5	0.4	0.2
Zinc	0.9	1.4	1.6	1.5	0.5
Copper	0.7	0.6	1.3	2.1	1.2
Chromium	0.1	0.1	0.2	0.2	0.2
Nickel	0.1	0.2	0.5	0.3	0.3

ND – Not detected

Table 3 Quantities of dredged spoil and their components dumped* in Liverpool Bay 1976-80 (t a^{-1})

	1976	1977	1978	1979	1980
Dredged spoil (wet weight)	4 989 560	5 683 400	5 620 800	3 574 000	5 054 000
" " (dry weight)	2 066 349	2 773 626	2 434 864	1 370 420	2 094 820
Mercury	4.3	4.5	4.7	2.6	4.0
Cadmium	1.3	1.3	1.7	0.6	1.4
Lead	331	312	236	144	213
Zinc	872	752	853	486	741
Copper	109	123	142	76	124
Chromium	145	148	149	104	139
Nickel	105	136	108	71	96

* Over 95% of spoil dumped at site Z; remainder at site Y (Figure 1a)

Table 4 Discharges to Liverpool Bay other than dumping, 1976/7 (t a^{-1})

Source		BOD	N	P	Hg	Cd	Pb	Zn	Cu	Cr	Ni
Rivers	(Mersey (Dee (Ribble	20 244	29 371	2 078	0.35	34	–	155	75	80	98
Sewage and industrial discharge		102 400	11 515	2 836	2.9	16	47	1 492	53	152	74

Source: DOE/NWC (1979b) and Preston and Portmann (1981)
– Not determined

3. Surveys and sampling

Surveys of the seabed sediments and their biota were carried out in October 1975, October 1976, September 1977, October 1978 and November 1980. The 1975, 1976, 1978 and 1980 surveys included between 41 and 68 sampling stations and concentrated on the sewage sludge dumping area. The 1977 survey included 30 stations and was directed to the dredged spoil disposal area at Site Z. The location of sampling stations in each survey is shown in Figure 1b. At each station sediment was sampled, using a 0.1 m^2 or 0.14 m^2 Day grab. Where the substrate permitted and the smaller grab was used, three samples were taken: the first of these was used for sedimentological purposes, a sample of the surface 1.2 cm of sediment or a short core being retained for particle-size, organic carbon, metal and (in 1976, 1978 and 1980) bacteriological analysis, and the other two were sieved to collect the macrobenthos, using a 1 mm sieve. Use of the larger (0.14 m^2) grab allowed samples of both sediment and benthos to be taken from one sample. The biota were preserved in 5% formalin and returned to the laboratory for identification. Details of the methods employed in the analysis of sediments and the identification of fauna are given by Eagle *et al.* (1978 a).

4. Factors affecting the dispersal of dumped waste

4.1 The general characteristics of the receiving waters

Liverpool Bay is a semi-enclosed water body occupying the south-eastern extremity of the northern Irish Sea. The sewage sludge dumping site (Figure 1a) is located in waters of 20-30 m depth. The salinity of Liverpool Bay waters is lowered by the inflow of fresh water from the Rivers Ribble, Mersey and Dee. In the region of the sludge dumping ground, however, surface water salinities are 'not usually directly controlled by the discharge from rivers but more importantly controlled by variations in the patterns of dispersal of fresh water and hence in changes in the general circulation of water within the Bay' (Spencer, in press). Near surface salinity distributions are therefore complex and subject to considerable variability around a mean value of approximately 33^{0}/oo. Bottom water salinity distributions appear to be simpler and more stable, increasing steadily with water depth (Ramster, 1972a). Vertical variations in salinity have been regularly detected at the sludge dumping ground (Ramster, 1972b ; Bowen *et al.*, 1973) with maximum surface/bottom salinity differences of approximately 0.76°/oo (Spencer, in press).

Water temperatures at the dumping ground typically vary between 6°C in January and 16°C in August (Lee and Ramster, 1976). Thermoclines have been observed to form in the vicinity of the dumping ground, particularly during summer months (Ramster, 1972b; DOE/NWC 1979a, Driver *et al.*, 1977), but there is little information concerning their persistence or spatial distribution.

Czitrom-Baus (1982) made a detailed study of stratification and the associated formation of fronts in Liverpool Bay and found that the main factors causing stratification are density-driven advection and surface heat flux. In the advective process, fresh water enters the Bay from the east and moves westward at the surface while dense saline water moves eastward at depth. Heat flux at the sea surface causes the upper layers of water to be warmed, resulting in stratification. The main factor causing a weakening or breakdown of stratification is mixing of the water column by wind or tidal bottom friction. These factors, which cause the formation or breakdown of stratification, are precariously balanced in Liverpool Bay and, given the appropriate circumstances, any one of them may become dominant.

Czitrom-Baus (1982) gave an indication of the likely impact of these processes at various times of the year. In winter, large river discharge may result in an estuarine-type circulation and stratification. In spring, this type of circulation may result from the effects of both river discharge and heat flux warming the surface waters. Stratification in the summer may develop at neap tides during periods of warm, calm weather (i.e. when tidal and wind stirring are small and heat flux great). In autumn, the water column remains mixed for most of the time due to the dominance of wind and tidal mixing over the low heat flux at this time of year.

Patchy patterns of stratification may also occur, due to lateral variations in density and therefore advective transport. Also, near the coast the water column remains mixed for most of the time, due to strong tidal mixing (Czitrom-Baus, 1982).

4.2 Tidal currents

Tidal currents were studied intensively in 1970 and 1971 using two current meters (5 and 21 m above the sea bed in 33 m water depth) at the sludge dumping ground (Ramster, 1972c). The strongest flood and ebb flows were aligned along a 105-285^{0} axis, attaining peak values of 91 cm s^{-1} (surface) and 71 cm s^{-1} (bottom). Surface tidal ellipses were extremely elongated, but those near the bed became almost rotary during neap periods, with slack water velocities rarely dropping below 5 cm s^{-1}. Moored current meters were deployed at a further five sites (both inshore and offshore of the dumping ground) during November 1970 (Talbot, 1972) and showed the ranges of tidal speeds to be similar throughout the Bay.

4.3 Residual water movements

Several studies have been conducted on the long-term circulation of the northern Irish Sea but there is as yet no clear consensus on the sense of the residual circulation. Movement of bottom waters is believed to be generally south-eastward and landward between Anglesey and north of the Isle of Man (Ramster and Hill, 1969; Figure 2). Surface water movements are more variable and there are at least two schools of thought as to the general pattern that exists (Ramster, 1973).

The main division of opinion has been whether the circulation in Liverpool Bay moves clockwise or anticlockwise. Evidence for a clockwise circulation has come mainly from studies of plankton distribution (Williamson, 1952, 1956; Jones and Haq, 1963; Khan and Williamson, 1970) while current meter studies (Ramster and Hill, 1969 and Ramster, 1972c) have demonstrated an anticlockwise circulation in the Bay. This latter model was supported by the trace metal studies of Abdullah, *et al.* (1972). Ramster (1972c) also suggested that the anticlockwise circulation is characteristic of the winter months, but that a surface clockwise gyre might occur in the summer.

Heaps and Jones (1977) reconciled the various views using a numerical model which showed that a density driven circulation results in a clockwise gyre in Liverpool Bay while an anticlockwise circulation is the result of wind stress, particularly from the south-west quadrant. Which of the two circulation patterns dominates is dependent on the

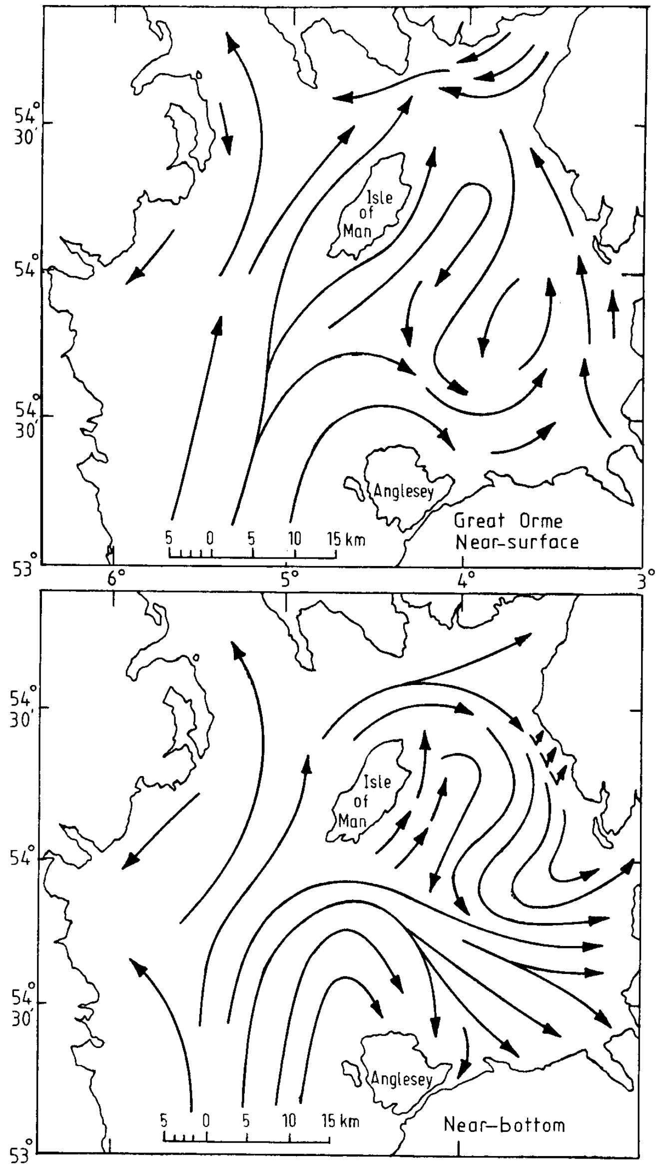


Figure 2 Residual water movements in Liverpool Bay (Ramster and Hill, 1969): (a) surface; (b) bottom.

relative effectiveness of wind and density forcing agents. In general, wind becomes a significant factor at speeds greater than 5 m s^{-1} ($\simeq$10 knots) and the dominant factor at speeds greater than 10 m s^{-1} ($\simeq$20 knots).

Ramster (1972c) showed that the bottom water residual flows at the dumping ground were generally south-east (± 45°), with speeds of ~4 cm s^{-1}. Surface water residual flows were more variable but tended to be northwards.

Drifter releases have been carried out to determine the direction of residual water movements over a wider area (Ramster, 1965; Halliwell, 1972, 1976). Surface drifters released at the dumping ground were returned from areas to the north, while seabed drifters from the dumping ground were washed ashore along the North Wales coast, in the Mersey and Dee estuaries, and along the south-west Lancashire coasts. Most seabed drifters were returned from the North Wales/Dee zone but during north-westerly gales more drifters were recovered from the Mersey. The proportion of seabed drifters returned was high (>50% in a year at some stations) even when releases were made seaward of the dumping ground.

Modelling work has shown that the residual current at the sea bed due to density variations is persistently landward with a corresponding offshore flow at the surface, but superimposed on this residual are significant effects due to changes in wind direction (Heaps and Jones, 1977). Proctor (1981) used a numerical model of the Irish Sea the results from which reinforced the conclusions of Heaps and Jones and explained some of the wind driven variability in the near-bed currents. Of particular note was his finding that strong westerly winds cause a westwards bottom current. Booth (1978) conducted a series of experiments using drogues and found that in areas sheltered from winds blowing along Saint Georges Channel residual currents at depth frequently opposed wind direction. This result confirmed those of Bowen *et al.* (1973) who drew a similar conclusion using current meters and of Ramster and Hill (1969) and Ramster (1973) who found that north-westerly gales may reverse the direction of the frequent south-easterly residual drift at depth. Several workers have suggested that the data which they collected point to a two-layered circulation (e.g. Ramster and Hill, 1969; Ramster, 1973), particularly in the east of the Bay (Booth, 1978).

Variations in river discharge may alter the density field within the Bay and result in changes in residual currents. This effect is complicated because the residual currents themselves have a pronounced effect on density distributions (Bowen *et al.*, 1973). It should be noted that as well as short-term changes in residual circulation in the Irish Sea there may also be variations on the scale of years caused by, for example, variations in the Atlantic inflow to the sea (Bowen *et al.*, 1973).

Jefferies *et al.* (1982) showed that the flow through the North Channel during the 1970s varied from a minimum of 1.2 km^3 d^{-1} for July 1973 – July 1974 to a maximum of 9.3 km^3 d^{-1} for September 1977 – May 1978. These authors suggested that one possible reason for the variability is that there have been changes in the wind regime, but the meteorological data for the area do not reveal a simple relationship between wind and flow through the North Channel.

Taking these factors into consideration it is clear that temporal variations in residual currents in Liverpool Bay may be caused by the motion of the Irish Sea as a whole (either wind or density driven) and direct forcing by wind stress and density currents on a local scale.

4.4 Tidal and wave-induced currents at the sea bed

In order to quantify the conditions under which sludge particles may accumulate at the sea bed and to provide a basis for the discussion of sediment dynamics in Section 6.2, it is necessary to estimate the strength of tidal and wave-induced currents at the sea bed.

4.4.1 *Tidally-induced currents*

A boundary layer is developed when a steady unidirectional current flows over the sea bed, usually extending for 1-2 m above the bed. Unfortunately the lower current meter used during the 1970/71 dumping ground survey was situated 5 m above the bed and was thus above the boundary layer. However, Bowen *et al.* (1973) compared tidal velocities 1 m and 4 m above the bed, which indicated that the tidal velocity at 1 m ($\bar{U}_{100}$) was approximately 0.8 of the velocity at 4 m ($\bar{U}_{400}$). This relationship can be used to calculate approximate values of $\bar{U}_{100}$ from the results of the lower meter at the dumping ground. Crickmore (1972a) has analysed the records over a 2-month period and derived percentage exceedence curves for one-hourly mean velocities. These curves, corrected to $\bar{U}_{100}$ velocities, are shown in Figure 3.

The bottom shear stress over a rough bed of uniformly sized particles such as naturally sorted sands and gravels can be estimated from values of $\bar{U}_{100}$ using the quadratic stress law (Sternberg, 1968) where

$$\tau = C_{100} \times \rho (\bar{U}_{100})^2 \qquad (1)$$

where τ = bottom shear stress

C_{100} = drag coefficient

ρ = density of seawater.

Sternberg observed values of C_{100} to vary from between 2×10^{-3} and 4×10^{-3} while Heathershaw and Simpson (1978) measured a mean value of 1.89×10^{-3} at a station 18 km north-west of the dumping ground; furthermore, Bowden and Fairbairn (1956) and Charnock (1959) recorded values from 2.0×10^{-3} to 11.6×10^{-3} in shallow waters off the north-east coast of Anglesey.

The different values of C_{100} are largely attributable to the effects of bed forms which cause 'form drag' over and above the 'skin friction' between the tidal flow and the sediment surface (Dyer, 1980). Only skin friction is responsible for sediment transport and this must be estimated if the forces governing the entrainment and dispersion of sludge in the sediment are to be determined. Since bed features are widely developed in Liverpool Bay (Section 6), it is assumed that the higher values of C_{100} calculated above reflect the presence of form drag and that the low value of 1.89×10^{-3} (Heathershaw and Simpson, 1978) is the best approximation of the skin friction component of C_{100}. Using this value of C_{100}, the range of shear velocities ($U_* = \sqrt{\tau/\rho}$) expected over sand and gravel substrates in the vicinity of the dumping ground has been calculated and is shown in Figure 3.

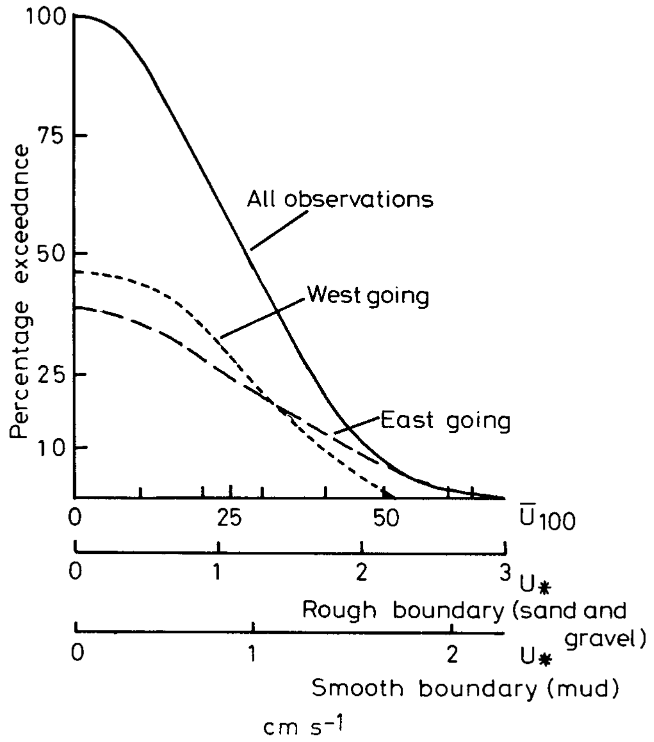


Figure 3 Tidal velocities 1 m above the sea bed and estimated friction velocities at the dumping ground (after Crickmore, 1972a):
$\bar{U}_{100}$ = velocity 1 m above the sea bed;
U_* = friction velocity.

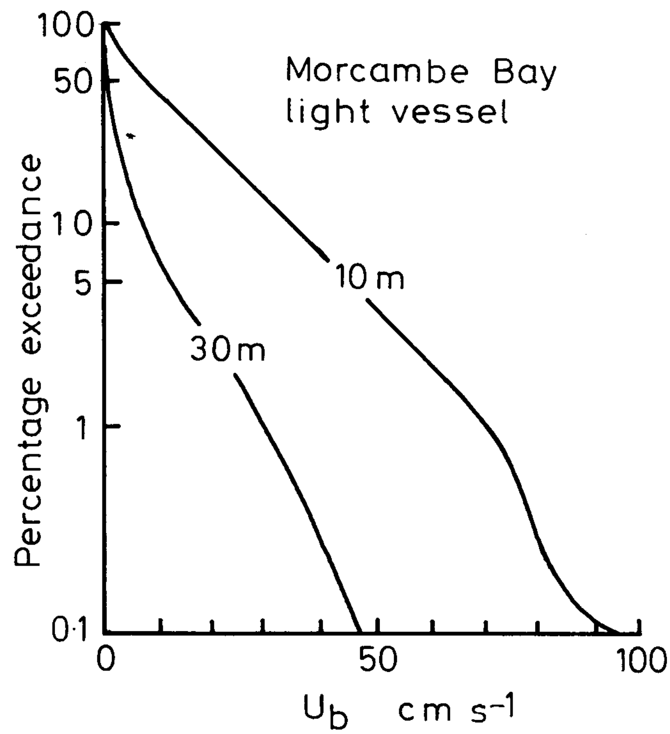


Figure 4 Wave-induced near-bed velocities: U_b = maximum velocity attained during the passage of the wave.

Over smooth muddy substrates, as opposed to sands and gravels, hydraulically smooth boundaries persist with the velocity distribution taking the form

$$\frac{\bar{U}_{100}}{U_*} = 2.5 \ln \left(\frac{100 U_*}{\nu}\right) + 5.5 \qquad (2)$$

where ν = kinematic viscosity of seawater ($cm^2\ s^{-1}$).

Shear velocities (U_*) expected over mud deposits have been calculated from this equation and also added to Figure 3.

4.4.2 *Wave-induced water movements*

Wind-generated waves produce at the sea bed an oscillatory water movement whose periods are too short (<20 s) for a boundary layer to develop more than a few millimeters above the bed. The strength and frequency of these wave-induced oscillating motions at the sea bed can be computed if the prevailing wave climate is known.

The wave climate in Liverpool Bay can be assumed to be similar to that recorded at the Morecambe Bay Light Vessel (Draper, 1967) which, although situated some 33 km to the north of the sludge dumping ground, has similar water depths and exposure. The frequency of occurrence of significant peak water-particle velocities (U_b) at the sea bed for 10 m and 30 m depths is shown in Figure 4. Bottom shear stress may be calculated from these velocities by

$$\tau = 0.5 f_w \rho U_b^2 \qquad (3)$$

where f_w = friction factor (Jonsson, 1966) which for the upper range of oscillatory velocities encountered at the dumping ground is approximately 10^{-2}.

Using equations 1 and 3, and the frequency data of Figures 3 and 4, shear velocities (U_*) expected under the strongest tidal and wave-induced currents, and under conditions which occur for 10% and 50% of the time have been calculated for both the dumping ground and inshore (Table 5).

The range of shear velocities demonstrates that peak tidal flows exceed wave-induced flows in all but the most severe storms. Thus, at the dumping ground, tides are the dominant cause of sediment transport, although it must be noted that wave shear stress may be an important factor in sediment disturbance when it adds to the prevailing tidal stress. Further inshore in shallower waters, wave-induced flows will become more effective in causing sediment movement, for instance in depths of 10 m over smooth substrates, wave-induced flows exceed peak tidal flows for 10-20% of the time.

4.5 The behaviour of sewage sludge after discharge

Approximately 7000-8000 t of sewage sludge are dumped each weekday at the dumping ground at any state of the tide. The dumping vessels range in capacity from 500 to 3000 t and up to four vessels may dump during any one tide. Discharge takes place at 4-6 knots, over at least 15 min through valves in the ship's bottom.

The dilution of the sludge in the wake of the dumping vessels has been observed normally to be in the order of 200:1 within a minute of discharge (Crickmore, 1972b). This dilution is sufficient to prevent widespread flocculation of the dispersed sludge particles (Crickmore, 1972b; Eagle *et al.*, 1978b). Settling rates of the coarsest sludge particles approach 10 m h^{-1}, and the median settling velocity lies in the range 0.2-0.5m h^{-1} (Crickmore, 1972b), although it should be noted that the finest particles may never settle out.

Movement of the sludge through the water column also results from eddy diffusion processes which vary with the strength of the tidal and wave-induced currents and the degree of density stratification. Talbot (1972) estimated that under non-stratified conditions eddy diffusion would result in the dispersion of sludge as far as the sea bed within 2 h of discharge. This was confirmed by Barrett *et al.* (1972a) who found radioactively labelled sludge to be distributed to below 20 m depth within 2.5 h of discharge when salinity gradients were not present. These authors also showed that stratification which occurred during the ebb delayed dispersion of sludge to the sea bed. The exact conditions following dumping will thus determine the time taken for sludge particles to reach the sea bed and the amount and sense of tidal advective movement which occurs. Sludge remaining in suspension in surface waters over a tidal cycle could be dispersed by as much as 9 km (neaps) and 16 km (springs) east or west of the discharge site and several km to the north or south. On the other

Table 5 Estimated shear velocities (cm^{-1}) at the dumping ground and inshore

	Dumping ground		Inshore	
	Tidal flow (rough boundary)	Wave-induced flow (30 m)	Tidal flow (smooth boundary)	Wave-induced flow (10 m)
Maximum	2.8	3.2	2.2	6.3
Value exceeded for 10% of time	2.05	0.6	1.6	2.5
Value exceeded for 50% of time	1.2	0.2	1.0	0.6

hand, material reaching bottom waters within a few hours of dumping, particularly around slack water, would be expected to move very little.

The good mixing conditions which prevail much of the time have been shown in surveys by the North West Water Authority (NWWA) and the Lancashire and Western Sea Fisheries Joint Committee (LWSFJC) to be effective in avoiding detectable reductions in the oxygen concentration of the receiving waters. In conditions where the waters are stratified, however, a reduction in the level of oxygen saturation may occur in bottom waters as a result of biological activity and a failure of oxygen replenishment from surface waters. Differences between surface and bottom waters under such conditions are generally around 10% of the saturation value but during an extended period of warm and calm weather in July 1976 they reached 20% (DOE/NWC, 1979a), and Driver *et al.* (1977) recorded one measurement of 60% saturation in bottom waters when the difference between surface and bottom was 25%. Sludge is not the only source of degrading organic material in these bottom waters; during the summer months decaying phytoplankton is also likely to be a major source of oxygen demand.

Once sludge particles have reached the bottom waters, they settle and accumulate on the sea bed when shear velocities fall below ~1.5 cm s^{-1} over sand and gravel substrates (McCave, 1971) and ~0.7 cm s^{-1} over muds (Partheniades, 1965). Table 5 and Figures 3 and 4 show that such conditions prevail in the vicinity of the dumping ground for around 50% of the time; thus deposition frequently occurs over neap periods and in calm weather. The area of initial settlement of an individual vessel load depends on the state of tide when dumped, but it is likely that the integrated effect of several vessel loads will lead to the area of initial settlement being centred near the dumping site with extensions along the tidal ellipse (major axis 105°–285°). This was confirmed by the radioactive tracer experiment (Crickmore, 1972a) which showed sludge to settle after one tidal cycle over an elliptical area extending 10 km from east to west and 6 km from north to south which encompassed the dumping ground.

The movement of sludge particles remaining in suspension over the longer term will be determined by the residual water flows described in Section 4.3. These were shown to

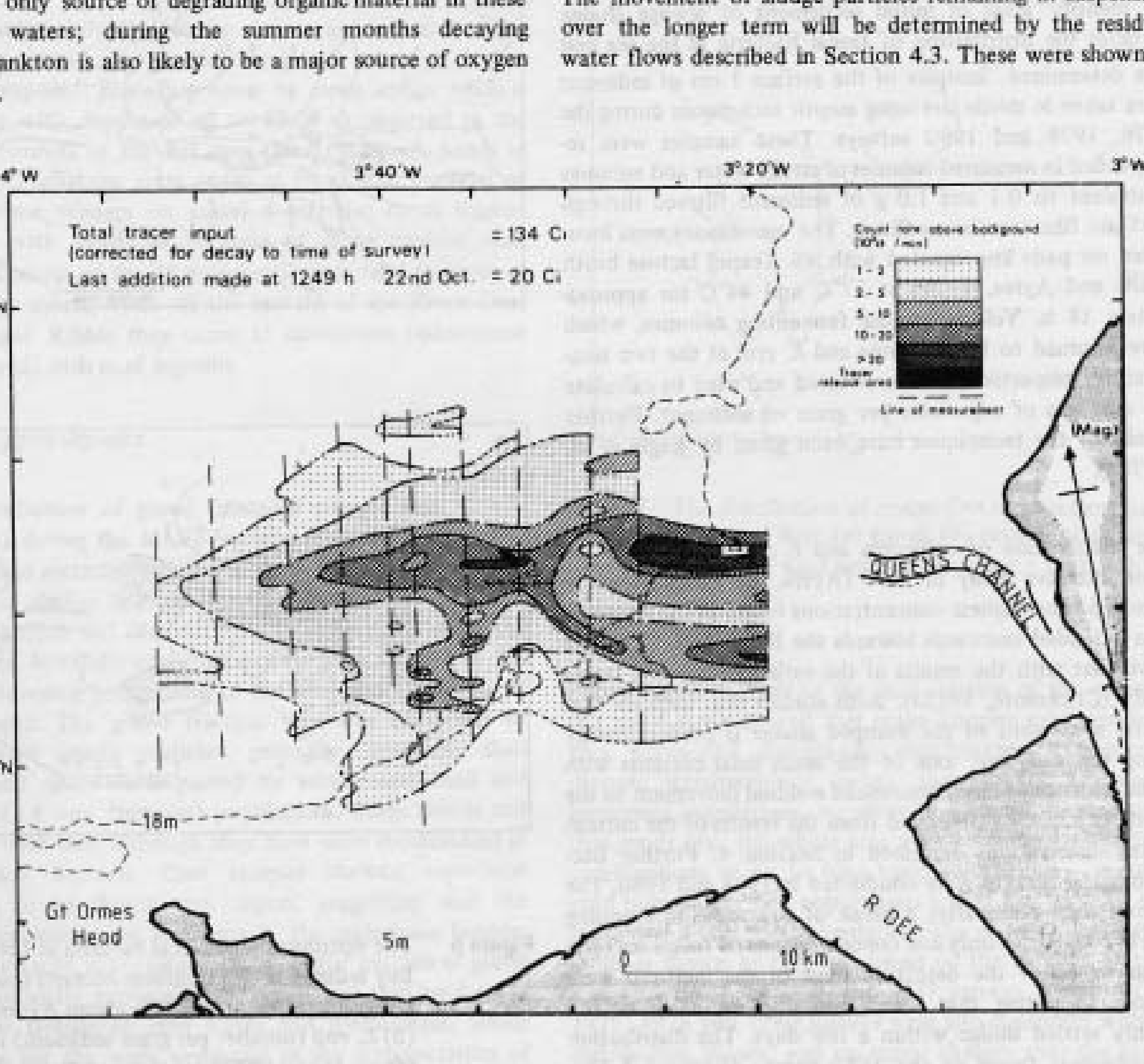


Figure 5 Distribution of radiotracer at the sea bed after dumping of tracer amended sludge (Crickmore, 1972a): date of survey – 22-24 October 1970; total tracer input (corrected for decay to time of survey) = 134 Ci; last addition made 7h before start of survey = 20 Ci.

be largely inshore in the bottom waters and suggested movement within the quadrant between south and east of the dumping ground. During periods of peak tidal flows or high wave activity the finer sludge particles may well move in the middle or near-surface waters for at least part of the tidal period, leading to wider dispersion in other directions. Much of the sludge may therefore become widely dispersed before settlement occurs. The integrated effect of these processes was determined in October 1970 following the addition of five batches of labelled sludge over a 4-week period (Crickmore, 1972a). The distribution of radioactivity at the sea bed (Figure 5) showed that considerable dispersion north and west occurred as well as the expected southerly and inshore movement towards the Mersey.

5. Distribution of faecal bacteria at the sea bed

To identify areas of initial settlement of sewage sludge particles the distribution of faecal bacteria at the sea bed was determined. Samples of the surface 1 cm of sediment were taken in sterile jars using aseptic techniques during the 1976, 1978 and 1980 surveys. These samples were resuspended in measured volumes of sterile water and volumes equivalent to 0.1 and 1.0 g of sediment filtered through 0.45 μm film membrane filters. The membranes were incubated on pads impregnated with 4% Teepol lactose broth (Halls and Ayres, 1974) at 37°C and 44°C for approximately 18 h. Yellow, lactose fermenting colonies, which were assumed to be coliforms and *E. coli* at the two temperatures respectively, were counted and used to calculate the numbers of organisms per gram of sediment. Further details of the techniques have been given by Eagle *et al.* (1978a).

The distribution of coliforms and *E. coli* found from the most intensive study in 1976 (Ayres, 1977) are shown in Figure 6. The highest concentrations found in the dumping area extended eastwards towards the River Mersey, and are consistent with the results of the earlier radioactive tracer study (Crickmore, 1972a). Both studies thus indicate that initial settlement of the dumped sludge is predominantly along the east/west axis of the main tidal currents with little evidence of any pronounced residual movement to the south as might be predicted from the results of the current meter observations described in Section 4. Further bacteriological surveys were conducted in 1978 and 1980. The former took place after a break of four days in dumping and in rough seas: only low concentrations of faecal bacteria (near or below the detection limit of the method) were found, suggesting that waves can disperse much of the freshly settled sludge within a few days. The distribution of coliforms found in the 1980 survey (Figure 6c) was consistent with those of the 1976 survey.

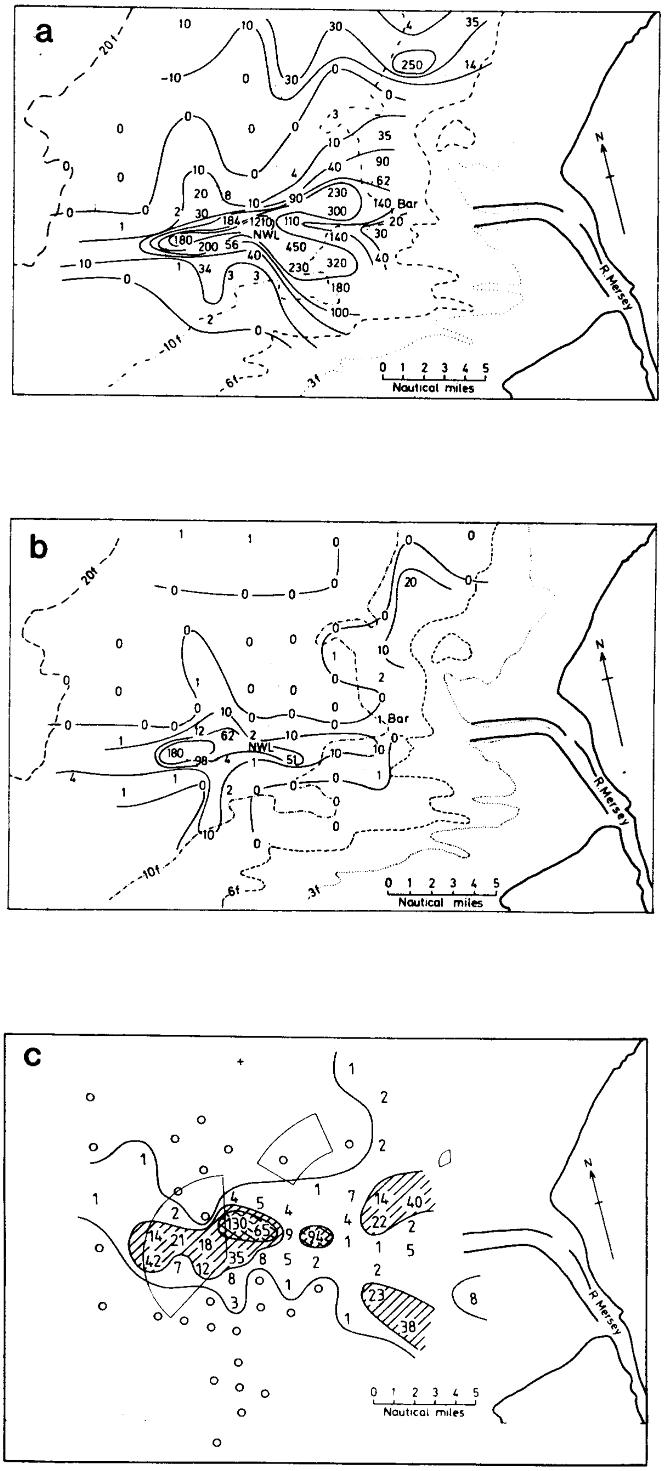


Figure 6 The distribution of faecal bacteria in Liverpool Bay sediments: (a) Coliform bacteria (number per gram sediment) in 1976 (from Ayres, 1977); (b) *E. coli* (number per gram sediment) in 1976 (from Ayres, 1977); (c) Coliform bacteria (number per gram sediment) in 1980.

6. The sediments

6.1 The distribution of natural sediment types

Sediments were sampled during each survey using a Day grab. The surface 1-2 cm of sediment was sampled in all cases, but in 1975 and 1976 short (10-15 cm) cores were also taken from the grab. All samples were analysed for particle size, organic carbon and trace metal content (Section 3).

Other investigations of the physical nature of the sediment in Liverpool Bay have taken place over the past fifteen years based on the analysis of vibrocorer, borehole or grab samples (DOE 1972b, 1973, 1976; DOE/NWC, 1979a; Pantin, 1978). Side-scan sonar surveys have also been carried out in limited areas of the dumping ground/Bar region by Rees (in press) and as part of the MAFF 1980 survey.

These studies show that in the vicinity and seawards of the dumping ground, the underlying gravel basement is frequently exposed. Elsewhere cover by sands and/or muds is extensive with depths of up to 10-20 m observed in the buried channels of the Bar area (Rees, in press). Sands in the deeper offshore areas occur as interstitial material or thin surface veneers on gravel substrates, or as thicker deposits with bedforms in areas of more copious sand supply. Inshore of the 18 m isobath, sands tend to occur as featureless sheets, while off the mouths of the Rivers Dee, Mersey and Ribble they occur in association (intermixed and layered) with mud deposits.

6.1.1 *Gravel deposits*

The distribution of gravel (material coarser than 4 mm diameter) during the MAFF surveys is shown in Figure 7. Gravel-sized material retrieved in the Bar area was composed largely of clinker and other waste material derived from ships at anchor and does not reflect the naturally-occurring sediments. Naturally-occurring gravels were located at the sewage dumping ground and to the west and south-east of the ground. The gravel fraction here was composed of unencrusted quartz particles, generally containing shell debris and often accompanied by very coarse sand and granule (2-4 mm diameter) populations. Large stones and cobbles were rare, although they have been encountered in sub-surface deposits. Core samples showed superficial deposits to be thin in this region, suggesting that the gravels are winnowed remnants of the underlying boulder clays. Side-scan sonar records showed that patches of gravel were frequently exposed between areas of thin sand cover or in the troughs of sand waves and megaripples, which accounts for the wide variations in the concentration of gravels in the surface sediment samples.

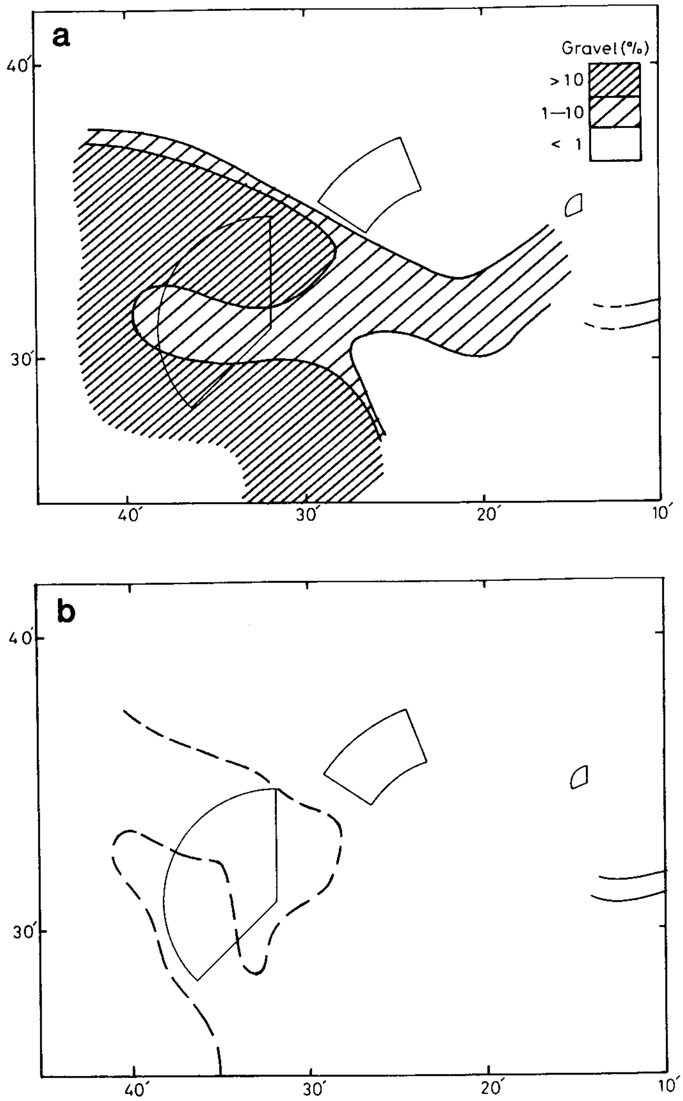


Figure 7 The distribution of coarse (>4 mm) sediments in Liverpool Bay: (a) gravel; (b) eastern boundary of coarse sand population (500-2 000 μm)

6.1.2 *Sand deposits*

Particle-sized analysis of the sand fraction of the sediment (63 μm-4 mm) showed that many samples exhibited complex polymodal particle-size distributions. Examples of discrete size-frequency curves, showing the component lognormal populations, are given in Figure 8. These suggest that sands over the whole area were comprised of different combinations of up to four basic populations – a course sand population, two populations within the medium sand range and a find sand population. The distribution of these sands is shown in Figure 9 based on the detailed particle-size analysis of the 1976 and 1977 surveys. Results of the 1975, 1978 and 1980 surveys were also consistent with the distribution shown, and no major changes over the period 1975-80 are suggested.

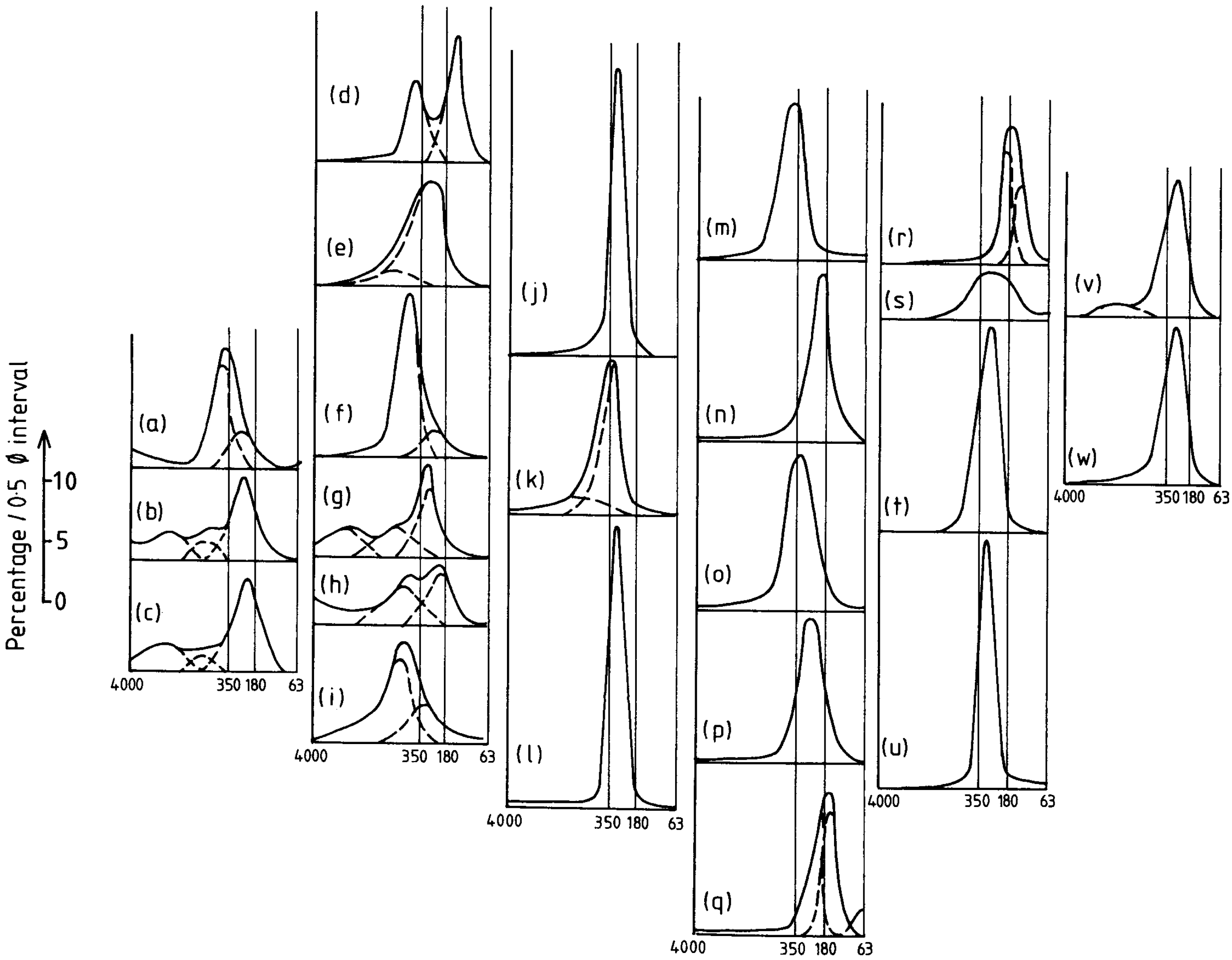


Figure 8 Particle-size (μm) distribution of the sand fraction at selected stations (for sampling positions see Figure 9).

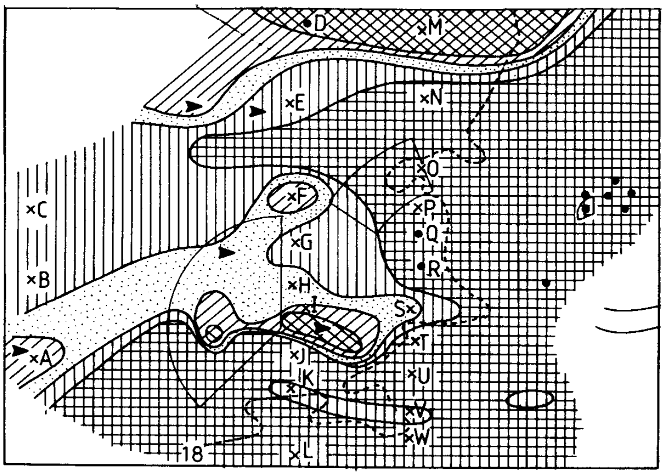


Figure 9 The distribution of sand populations: ˣA = position of samples whose particle sizes are given in Figure 8; ● = stations containing a fine sand population; ➤ = position of sandwaves (Rees, in press; Belderson and Stride, 1969; MAFF survey); ▨ = medium A population predominates; ▨ = medium A sands with subordinate medium B population; ▒ = equal amount of two populations; ▥ = medium B sands with subordinate medium A population; ▦ = medium B population predominates.

The coarse sands (mean diameters between 800 μm and 2 mm) which are typified by samples B, C, G and I in Figure 8 occurred mostly as interstitial particles in gravel substrates west of the dumping ground.

In most areas of the Bay the sediments were comprised of two distinct particle populations in the medium sand range (mean diameter of 350-500 and 180-350 μm, denoted medium A and B respectively in Figure 9). The finer population (medium B) predominated throughout the zones inshore of the 18 m isobath, including the muddy zones off the estuary mouths, and extended seawards to depths of 30 m in two east-west aligned tongues north and south of the sludge dumping ground. In the latter areas the medium B population was extremely well sorted (Figure 8, samples J, L, N, O) and side-scan sonar records from the area south of the dumping ground showed a plane bed giving a very strong return signal, characteristic of densely packed sands of a narrow size range.

The highest concentrations of the coarser population (medium A) were found well offshore and in two east-west aligned zones running towards the Mersey and Ribble. The medium A population was mostly found in admixture with a medium B population (Figure 8, samples A, H, I) and coincided with areas where sandwaves, megaripples and sand ribbons were found (Figure 9).

The fine sand population was of mean diameter 125-180 μm (Figure 8; samples D, Q, R) and was only encountered in the vicinity of site Z, in restricted localities on the Bar and in the extreme north of the surveyed area (Figure 9). This population was often associated with above average concentrations of mud.

6.1.3 *Fine sediment deposits*

The amount of fine material (<90 μm) in the sediments is plotted for each of the four full surveys in Figure 10. Sediments containing a high proportion (up to 80%) of fines occurred in all the surveys in an east-west elongated zone off the entrance to the Mersey. The extent of this zone varied between 1975 and 1980, with the amounts of fines increasing from 1975 to 1978 before falling between 1978 and 1980. Short cores collected within the central region of the Mersey Bar mud area showed the mud and sand comprising the sediment to be thoroughly intermixed. Cores from peripheral areas, however, often exhibited distinct layers, several centimetres thick, of clean sands alternating with dark cohesive muds; mud balls were also common within the areas containing 5-10% fines. These deposits thus appear to be in a zone of active sediment transport, as can also be seen from the variations in the boundaries of the Bar mud area between surveys. Some of the surveys showed a further zone off the entrance to the Dee containing fines concentrations between 5 and 50% and consistent with the isolated and variable mud deposits observed by Eagle (1977).

6.2 Sediment dynamics

6.2.1 *Gravel and coarse sand disturbance*

The threshold velocities for movement of sediment particles of different sizes are shown in Figure 11. It can be seen that peak shear velocities near the dumping ground, 3.0 cm s^{-1} for both tidal and wave-induced flows (see Table 5), are capable of initiating movement in quartz material of up to 1-2 mm diameter. Acting in conjunction, the two processes may in rare instances disturb coarser gravels. Thus the gravel deposits offshore with their associated coarse sand populations will only be rarely disturbed.

6.2.2 *Sand transport*

The medium sands described in Section 6.1.2 have been shown to originate from the west of the Bay where tidal currents are strongest, several authors having demonstrated the eastward movement of sand bodies in this sector of the Irish Sea (Harvey, 1966; Belderson and Stride, 1969; Williams *et al.*, 1981). The shear velocities required to initiate motion in the two medium sand populations are similar (1.5 and 1.4 cm s^{-1} for medium A and B sands respectively) and both populations will be in motion for approximately 40% of the time (Figure 3) but no sand transport will occur during neap tides when peak shear velocities are 1.0 cm s^{-1}. Ramster (1973) has shown that application of threshold speeds for sand motion to the dumping ground current meter data indicates a residual drift towards ESE, suggesting that movement of sand at the dumping ground is similar to the eastward movement elsewhere in the Bay.

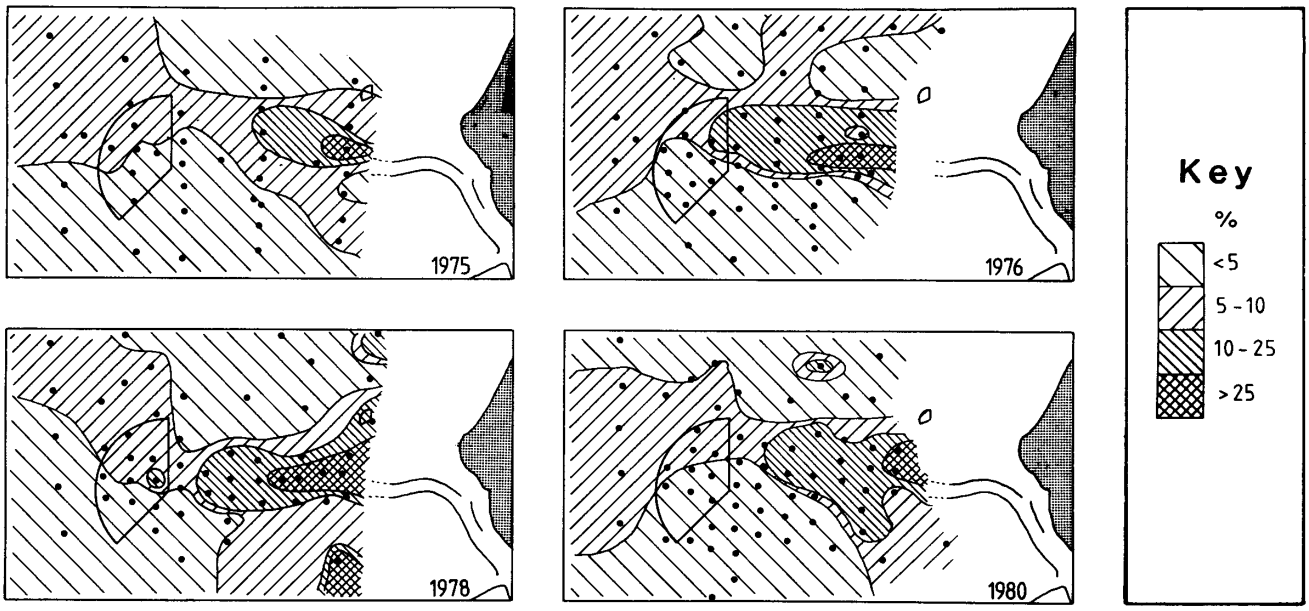


Figure 10 The concentration (%) of fines (<90 μm) in the sediment: sampling stations shown • (from Norton *et al.*, 1984).

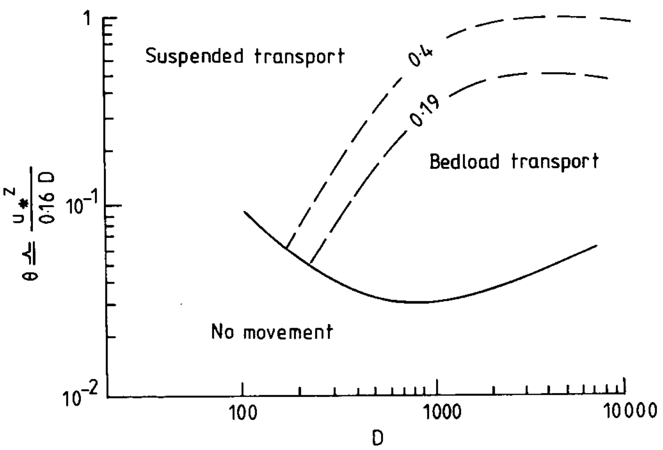


Figure 11 Threshold velocity (Shields, 1936) and mode of transport (Bagnold, 1966; McCave, 1971) for sand particles: 0.4 and 0.19 are the upper and lower transition zones between suspended and bedload transport according to Bagnold and McCave respectively; U_* = friction velocity (cm s^{-1}); D = particle diameter (μm); θ = Shields' criterion for quartz particles.

The distribution of the two distinct medium sand populations described in Section 6.1.2 is due to the comparative ease with which the finer (medium B) sands can be put into suspension. Figure 11 shows this to occur at shear velocities above 2.1 cm s^{-1} , at which velocity the medium A population moves only as bedload. Figure 3 shows that when tidal currents flow in a westerly direction both sands move as bedload, but during the period of the faster easterly currents, the medium B sands can move in suspension. As transport in suspension is far more rapid than bedload movement, some of the medium B population is swept into the shallow inshore areas where movement is further enhanced by the increased effects of wave activity. In the Bar area, however, the supply of estuary-derived suspended sediments enables thin mud deposits to accumulate over neap periods, leading to the formation of cohesive deposits which inhibit sand transport on the successive spring tides. Strong wave action is then necessary to break up the mud deposits (Section 6.2.3).

6.2.3 *Suspended sediment dynamics*

Suspended sediment concentrations in Liverpool Bay have been measured in the routine surveys of the LWSFJC and by MAFF. The MAFF water quality surveys (Norton *et al.*, 1984) showed seston levels to be around 5-10 mg l^{-1} in surface waters during a winter period, reaching concentrations of 30 mg l^{-1} in bottom waters. The regular LWSFJC surveys (Vivian *et al.*, 1979) showed that concentrations of suspended solids inshore were highest in areas influenced by the Mersey outflow, stations in the region of the Bar containing up to 70 mg l^{-1} in surface and up to 50 mg l^{-1} in bottom waters. Nearer the dumping ground, surface concentrations were usually 10 mg l^{-1} or below but patches of higher concentrations (up to 40 mg l^{-1}) were often present. Bottom concentrations at about 10-40 mg l^{-1} were generally higher than surface concentrations, except in some of the cases where elevated surface concentrations were observed.

In Section 4.5 it was shown that conditions in the dumping area allow accumulation of fine particles for approximately 50% of the time, with extended periods of accumulation during neap tides. The thickness of material that accumulates over such periods is directly proportional to the amount of material in suspension; thus the greatest amount of mud is expected to accumulate in the Bar area. During spring tides, the shear velocities of 1.5-2.0 cm s^{-1} found over mud substrates (Figure 3) cause very slow erosion of slightly consolidated mud deposits (Partheniades, 1965). These rates of erosion may be sufficient to remove thin mud veneers that may have accumulated over neap periods in most parts of the Bay, but not the thicker mud deposits that are likely to have built up in the Bar area. In the latter zone, thick mud layers may accumulate over successive neap periods, with the influx of sand in the peripheral mud areas producing characteristic layered deposits (Section 6.1.3).

Mud substrates can be rapidly eroded when shear stresses exceed a critical value. This varies with the properties of the mud but available information suggests that mass erosion may occur once $U_* >$4.0-7.0 cm s^{-1} (Pierce *et al.*, 1970; Nunny, 1980). Tidal currents never reach these velocities in Liverpool Bay but peak wave-induced stresses in 10 m depth reach values of 6-7 cm s^{-1} (Figure 4, Section 4.4), leading to mobility of the mud deposits in the Mersey Bar and Dee areas during storms. During these storms resuspended material can be widely dispersed. Some of the mud may reach the offshore areas and settle in the deeper waters of the Bay, but some may be carried slowly back into the shallow water zones during quiescent periods of the year by residual near-bed water movements (Section 4.5).

6.2.4 *The implication of natural sediment movements to the dispersal of sewage sludge*

In Section 4 it was shown that sludge particles could reach the sea bed within 2 h of discharge, and that conditions favourable for settlement occur for about 50% of the time, including extended unbroken periods during neap tides and low wave activity. Much of the sludge dumped during these extended periods will settle at the sediment surface but its subsequent behaviour will depend on the sediment type.

On the coarser substrates found in the region of the dumping ground and further offshore, sludge particles may be incorporated within the sediment by biological activity, by settlement within the interstices of coarse sands or gravels, or by burial after settlement within a mobile sand body.

The capacity of the coarse sediments to retain sludge particles is limited, however, and most of it is likely to be resuspended during the stronger spring tides or high wave activity and continue to be dispersed by the onshore residual bottom water movements.

Sludge particles reaching the areas of finer sands and muds inshore may become intimately associated with these deposits and, since a relatively high threshold velocity is required for erosion, these deposits probably remain intact through the spring tide period. This will be the case particularly in the Bar area where fine material from several sources (including sludge dispersed into the Bar area) settles during neap periods. The sludge is thus likely to become incorporated with the naturally-accumulating muds which resist subsequent tidal erosion. Considerable quantities of sludge may accumulate in this fashion over long periods, but during heavy storms the velocity in these shallower waters may be sufficient to erode these deposits and so result in their redispersion within the Bay.

There are few areas within the Bay which can provide a permanent sink for large quantities of fine particles, and consequently sludge is subject to multiple cycles of accumulation and resuspension, which ultimately lead to wide dispersion of sludge particles throughout the Bay sediments. Surveys of the dispersion of radioactively labelled sludge (Crickmore, 1972a) confirmed that after 3-4 months sewage particles had become widely distributed throughout the sediments of the Bay. Contamination was most apparent in muddy areas where the potential for accumulation is high, e.g. the Bar area, and least evident in mobile sand areas containing little fine material, notably the shallower waters along the North Wales coast.

6.3 Organic matter in the sediments

The organic carbon content of three particle-size fractions of the sediment (<90 μm, 90-500 μm, 500-4000 μm) has been measured on samples from the 1975-80 MAFF surveys. The approximate organic content of the two coarser fractions was determined by loss on ignition at 550°C; the carbon and nitrogen content of the <90 μm fraction was determined using a Carlo Erba CHN analyser, after removal of carbonates by treatment with sulphurous acid. The carbon content of the whole sediment was estimated by multiplying the ignition loss figures by a factor of 0.6, and combining the resulting percentages in the ratios that their respective fractions contributed to the total sediment. During the 1975 and 1976 surveys the Redox potential (Eh value) of the sediment was also measured at sea on split cores taken from the grab immediately after collection. Further details of the methods used have been published (Eagle *et al.*, 1978a).

6.3.1 *Spatial variation in organic carbon concentrations*

The organic carbon content of the surface sediments in the five MAFF surveys (Figure 12) varied between 0.2 and 1.0% of the total sediment over most of the Bay, rising to maximum values of 2-5% in a restricted tongue off the entrance to the Mersey. Carbon concentrations in the whole sediment in the vicinity of the sludge dumping ground ranged from 0.3 to 0.8% and did not appear significantly elevated.

The organic content of a sediment tends to be related to its specific surface area, due to the similar hydrodynamic properties of fine organic and mineral particles and because degraded organic matter tends to occur as grain coatings

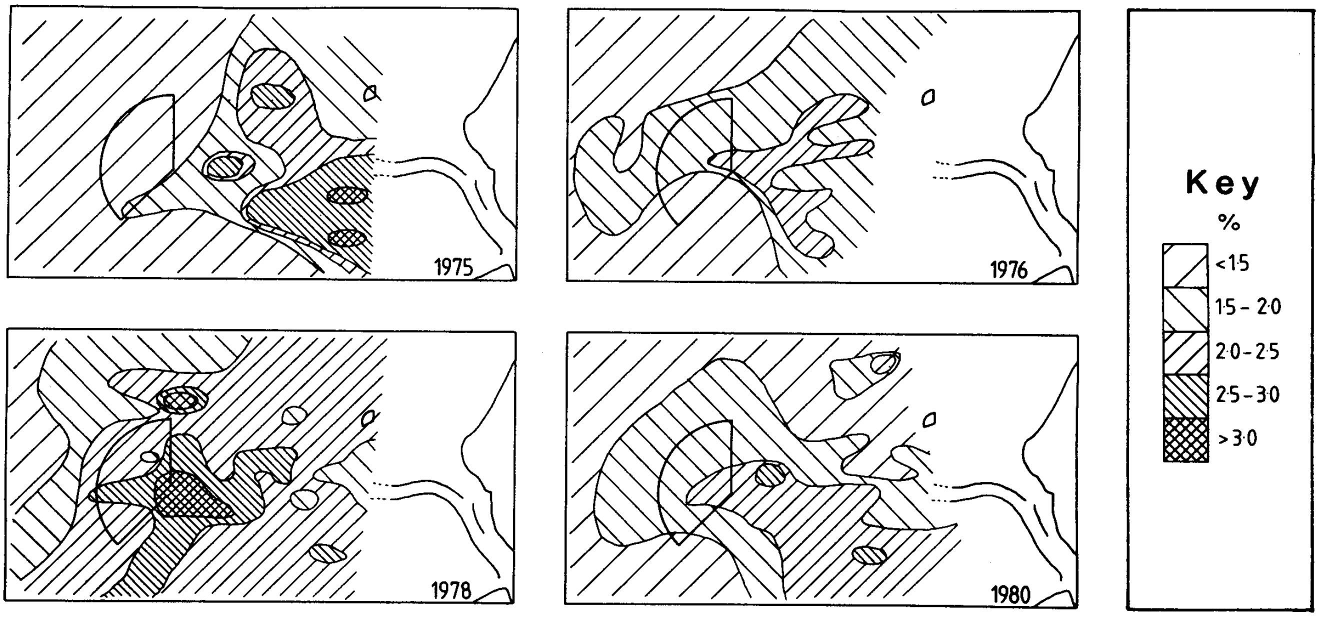


Figure 12 The distribution of organic carbon (%) in the <90 μm fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

and be complexed within clay lattices (Trask, 1932). In natural sediments, therefore, a good correlation usually exists between the fines and carbon concentrations of the whole sediment. Such a correlation was found to exist (Table 6) in samples taken during all the surveys, which suggests that the distribution of carbon in the whole sediment is related to the fines content of the sediment rather than the location of specific sources of carbon. In order to resolve the effects of different carbon inputs in an area of such heterogeneous sediments, we have investigated further the distribution of carbon within the <90 μm fraction of the sediment. The results of four of the surveys are plotted in Figure 12.

Table 6 Correlation between total organic carbon and fines (<90 μm) content of total sediment

	n	r≠
1975	80	0.76
1976	111	0.76
1977 (site Z)	24	0.67
1978	70	0.69
1980	70	0.50

n = number of degrees of freedom
r = correlation coefficient
≠ All were significant at the P = 0.001 level

It is apparent in all four surveys that levels were elevated generally throughout the eastern inshore parts of the survey area, with carbon concentrations at their lowest (<1.0%) in the offshore and south-western sectors of the survey area. Superimposed on this pattern were zones of elevated concentrations (2.0-4.5%) at, or immediately inshore, of the sludge dumping ground which varied from year to year but were most apparent in the 1978 survey. Concentrations were also elevated (1.5-3.5%) in the Bar area, off the mouths of the Dee and Ribble, and at the dredged spoil disposal ground (site Z).

Sludge dumping, discharges from the rivers (particularly the Mersey) and, on a local basis, dredged spoil dumping have been identified as significant sources of organic matter in the sediments. However, in the Bar area particularly the effects of these inputs are superimposed, making it impossible to assess from carbon measurements the degree of sludge dispersion into this area. The relative importance of these sources cannot be estimated from the input data since the amount of particulate organic matter (POM) has been estimated only for sewage sludge. The BOD inputs from discharges listed in Table 4 cannot be used to estimate POM inputs, since the proportion due to particulate rather than dissolved organic matter is not known.

6.3.2 *Redox potential (Eh) of the sediments*

The degradation of organic particles from sewage sludge dumping exerts an oxygen demand on the sediments, particularly in the first 20 days or so after discharge during which time, under aerobic conditions, up to 50% of the organic matter may degrade (Barrett *et al.*, 1972b). The oxygen demand of the sediment as measured by the Redox potential (Eh) depends on the organic content, the temperature, the sediment permeability and the extent of flushing of pore waters by wave-induced water movements at the sea bed. Under fully aerobic conditions, the Eh has a value of +470 mv, but it reaches levels of zero or less under anaerobic conditions.

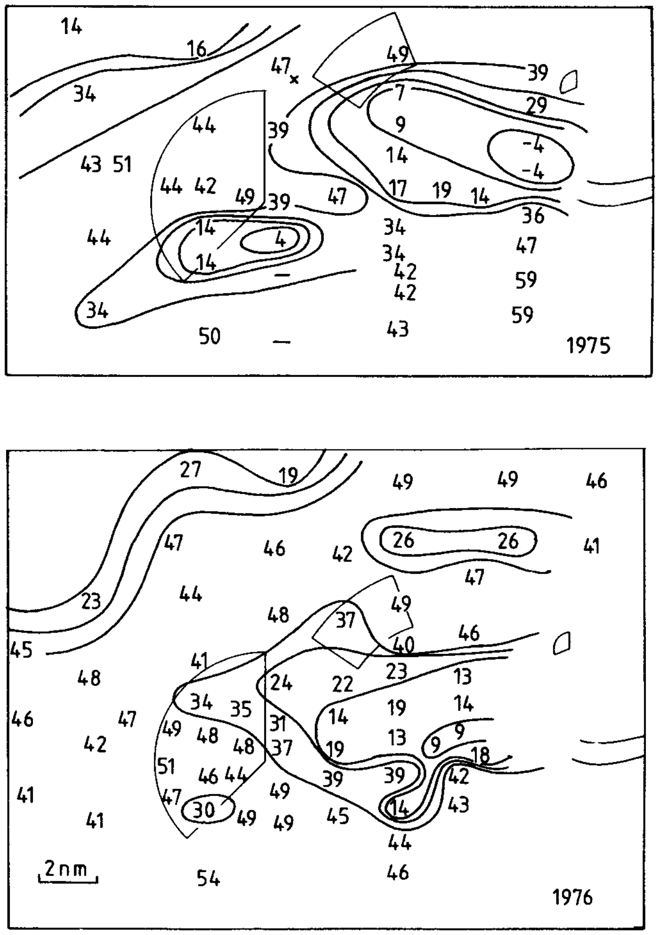


Figure 13 Redox potential (Eh) of sediment 5 cm below the sediment/water interface, 1975 and 1976 (mV).

The Eh values of the sediments measured in the 1975 and 1976 surveys are shown in Figure 13. The short equilibration period of 4 min used before measurement (Eagle *et al.*, 1978a) may have led to the Eh being overestimated by up to 50 mV. Even allowing for this, however, it is clear that aerobic conditions prevail in at least the top 5 cm of sediment over most of the survey area. Values were depressed at some stations near the sludge dumping ground and, at one station in 1975, near anaerobic conditions were observed. Depressed values (Eh below +100 mv) were also found in the sediments of the Bar area, with anoxic conditions occurring at localised sites of mud accumulation off the entrance to the Mersey.

6.3.3 *Carbon/nitrogen ratios*

The carbon/nitrogen ratios of marine organic detritus differ according to the source. Those in recent shelf sediments generally lie in the range from 5:1 to 12:1 (Degens and Mopper, 1976). The C/N ratios of sewage sludges vary considerably but normally range between 5:1 and 10:1. Organic detritus derived from terrigenous souces is known to contain high C/N ratios, e.g. 30:1 to 50:1 in the case of saltmarsh debris, (Haines *et al.*, 1976) and > 12 : 1 for estuarine muds (Murray *et al.*, 1980).

The C/N ratios of fine sediments collected during the 1975-80 surveys are plotted in Figure 14. During 1975 and 1976 samples containing ratios higher than 10:1 were few and restricted to the Bar area. In 1978 a marked change in C/N ratios was observed, with ratios below 7:1 occurring only at one station in the sludge dumping area. Other changes included the extension of a narrow tongue of >10:1 values west from the Queen's Channel and the development of an extensive zone in which ratios exceeded 15:1 north-west of the sludge dumping area. In 1980 a return to C/N ratios in the lower range was recorded. There were no spatial distributions which were consistent between the survey. Variations in C/N ratios between surveys had also been noted in studies of Thames Estuary sediments (Norton *et al.*, 1981) where the possibility of seasonal factors was considered. All the Liverpool Bay surveys were conducted in the autumn and it seems unlikely that seasonal factors such as contributions from phytoplankton blooms were responsible. The exact cause of the changes observed from year to year is thus unclear.

6.4 Metals in sediments

The distribution of metals in the sediments of Liverpool Bay has been investigated by several authors (Winter and Barrett, 1972; Wood *et al*, 1976; Gould, 1976). Their observations, which were until 1974, showed that it was possible to identify areas of consistently increased metal

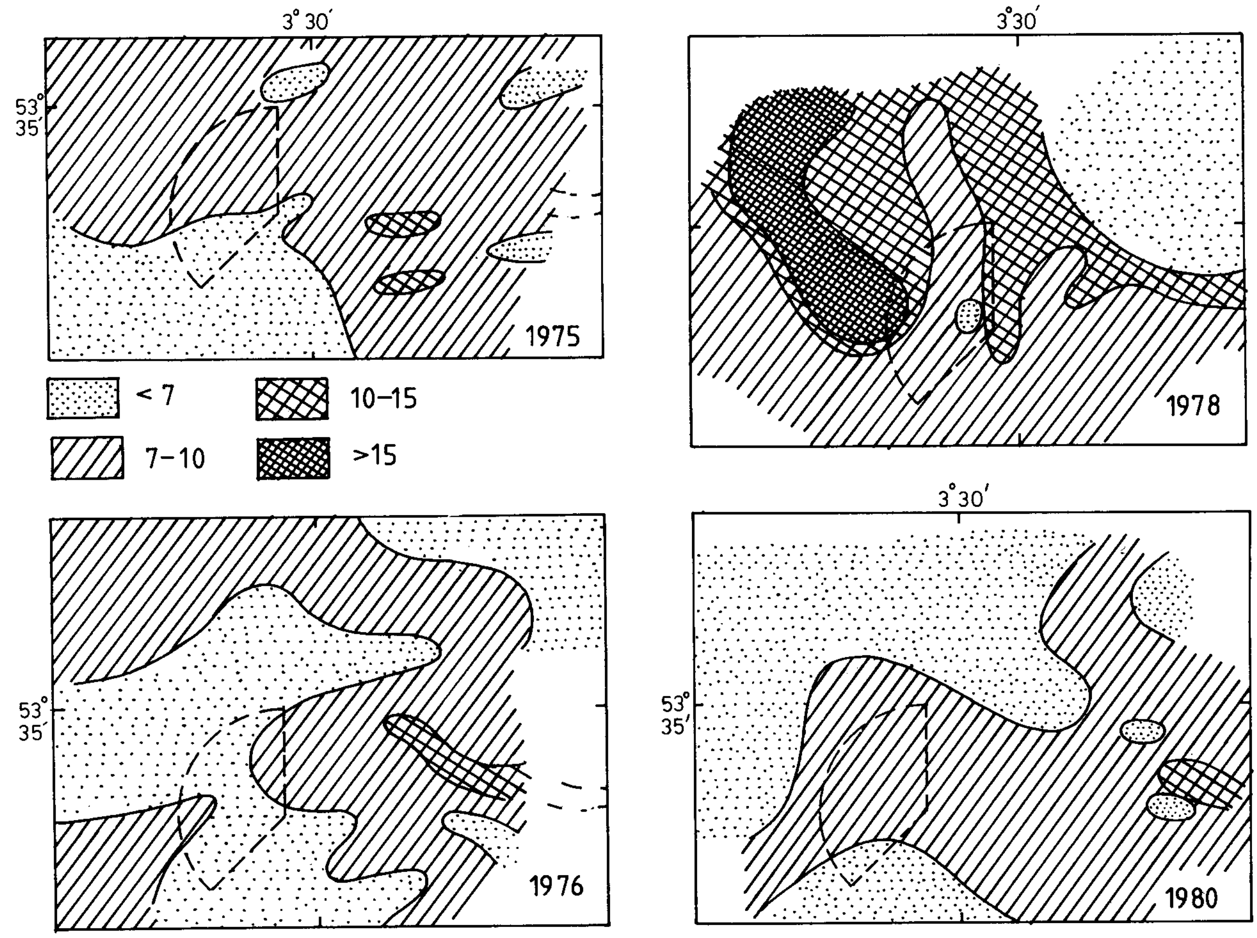


Figure 14 Carbon: nitrogen ratios in the <90 μm fraction of the sediment 1975-80.

concentrations near the Mersey Bar, but that metal concentrations elsewhere varied considerably so that it was difficult to isolate effects that were clearly due to sewage sludge dumping. The results of further surveys carried out by the Hydraulics Research Station (HRS) in 1975 (Nunny and Kiff, in press) showed that some metal concentrations were elevated at the sludge dumping ground as well as near the entrance to the Mersey, suggesting that the effects of dumping were becoming detectable, possibly as a result of the increased rate of dumping in that year (Head, 1981).

Samples taken on the five MAFF surveys in 1975-80, a period when there was an increase in the quantities dumped (Section 2), provide a considerable body of reliable data with which both spatial and temporal trends can be examined. During this same period HRS carried out several surveys, but due to differences in methodology the two sets of results are not comparable (Norton and Rowlatt, 1981), so the HRS results are not included in the present consideration.

Concentrations of mercury, copper, zinc, lead, nickel, chromium and cadmium were measured in both the <90 μm and 90-500 μm fractions of the sediment. The metals were extracted from the sediment sub-samples using a rigorous acid leaching technique and the concentrations were determined by atomic absorption spectrophotometry (Eagle *et al.*, 1978a). The spatial variations in metal concentrations in the coarse fraction are plotted in Figure 15 for all but the 1977 survey of the dredged spoil site. Concentrations in the fine fraction are plotted for each metal for the years 1975, 1976, 1978 and 1980 in Figures 16 to 22. The results of the 1977 survey of site Z are given in Figure 23.

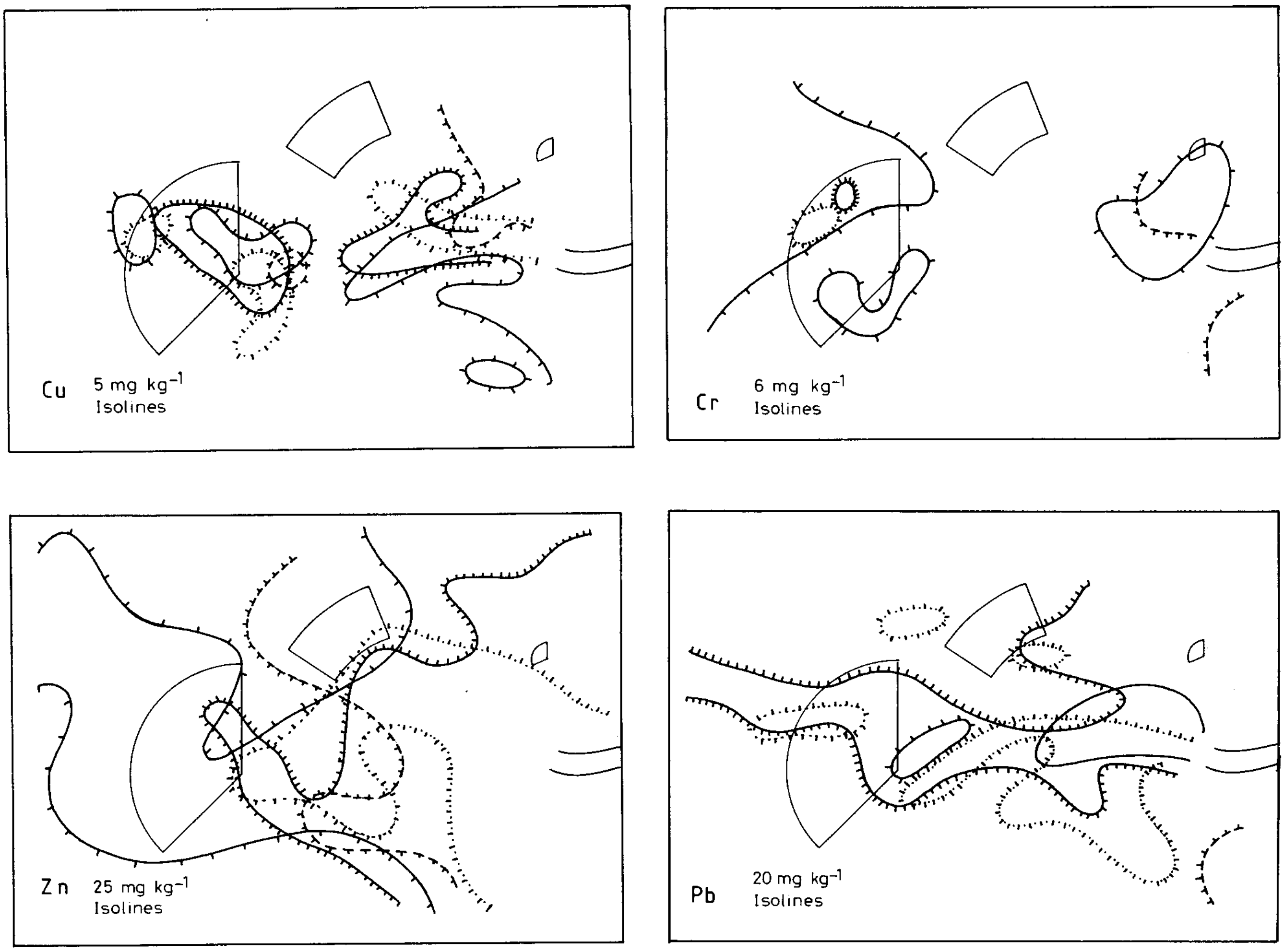
6.4.1 *Spatial variations in metal concentrations in the coarse fraction*

Copper, zinc, mercury, chromium and lead : In Figure 15, for each metal, a single contour line has been drawn separating stations where values during each survey were above or below a selected concentration. The values of each contour line, and the average metal concentrations in each of the so-defined areas are shown in Table 7. All the distributions exhibit a zone of elevated metal levels extending westwards from the Mersey towards the sludge dumping ground. The distributions of copper in all years, and of mercury, lead and chromium in some years, reveal a separate zone of elevation at the sewage sludge dumping site. Localised elevated concentrations also occur in the vicinity of site Z.

Table 7 Average metal concentrations (mg kg^{-1}) in coarse (90-500 μm) sediment fractions (see Figure 15)

	Stations with concentrations greater than contour value				Stations with concentrations less than contour value			
	1975	1976	1978	1980	1975	1976	1978	1980
Mercury contour (0.05 mg kg^{-1})	0.09 (11)	0.12 (11)	0.08 (33)	0.15 (25)	0.02 (29)	0.03 (52)	0.03 (34)	0.02 (49)
Copper contour (5 mg kg^{-1})	6.4 (5)	7.1 (8)	8.0 (25)	21.0 (8)	2.7 (35)	2.9 (55)	3.4 (42)	2.0 (66)
Zinc contour (25 mg kg^{-1})	37.3 (17)	33.9 (22)	40.0 (49)	62.3 (30)	19.9 (23)	19.9 (41)	21.5 (18)	15.6 (44)
Lead contour (20 mg kg^{-1})	24.3 (12)	24.3 (22)	27.6 (9)	80 (1)	14.5 (18)	15.4 (41)	11.0 (58)	8.1 (72)
Chromium contour (6 mg kg^{-1})	6.0 (1)	6.0 (1)	7.1 (33)	16 (4)	3.4 (39)	3.9 (62)	4.6 (34)	2.6 (70)

( ) Number of stations for each average



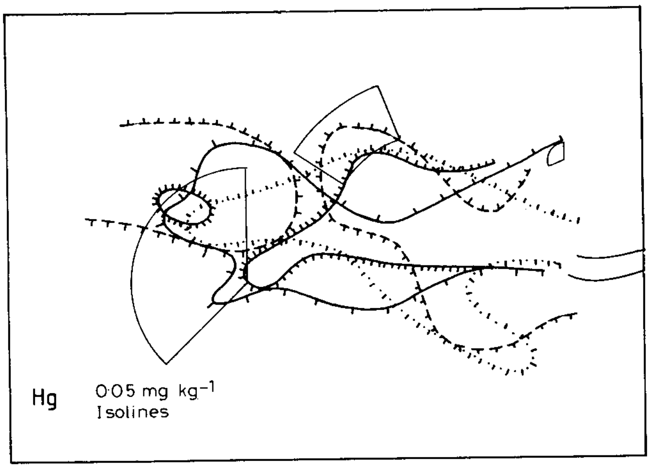


Figure 15 Concentrations of copper, zinc, mercury, lead and chromium (mg kg^{-1}) in the 90-500 μm fraction of the sediment: isolines = ⊥⊥⊥ 1980; └┴┘ 1978; ⊔⊔⊔⊔ 1976; .I.I.I. 1975; (note: spurs on lines are towards lower concentrations).

Nickel and cadmium : In view of the limited spatial variations observed, the distributions of these two metals are not illustrated here. Nickel concentrations in the coarse fraction (all in the range 10-23 mg kg^{-1}) bore no apparent relationship to the Mersey or dumping input zones; the highest levels occurred at the most seaward stations. All cadmium concentrations were below the detection limit of 0.2 mg kg^{-1}.

6.4.2 *Spatial variations in metal concentrations in the fine fraction*

Copper : Figure 16 shows that the highest copper concentrations always occurred in the vicinity or a little to the east of the dumping ground with concentrations as high as 100-140 mg kg^{-1}, compared with offshore concentrations which ranged between 30 and 40 mg kg^{-1}. Elsewhere, less marked and more localised increases were occasionally detected at the dredged spoil dumping grounds. Concentrations of around 70 mg kg^{-1} near the entrance to the Mersey were generally lower than in the dumping areas.

Zinc : In the offshore parts of the survey area (Figure 17) zinc concentrations in fine sediments ranged from 180 to 220 mg kg^{-1} and steadily increased towards the shoreline, reaching 350-540 mg kg^{-1} at the mouths of the Mersey and the Dee. Superimposed on this gradient were areas of increased concentrations at the sewage sludge and dredged spoil dumping sites. The highest zinc concentration at the dumping ground (1100 mg kg^{-1}) was detected in the 1980 survey when other localised areas of high concentration (up to 1200 mg kg^{-1}) were also found inshore of the sludge dumping ground.

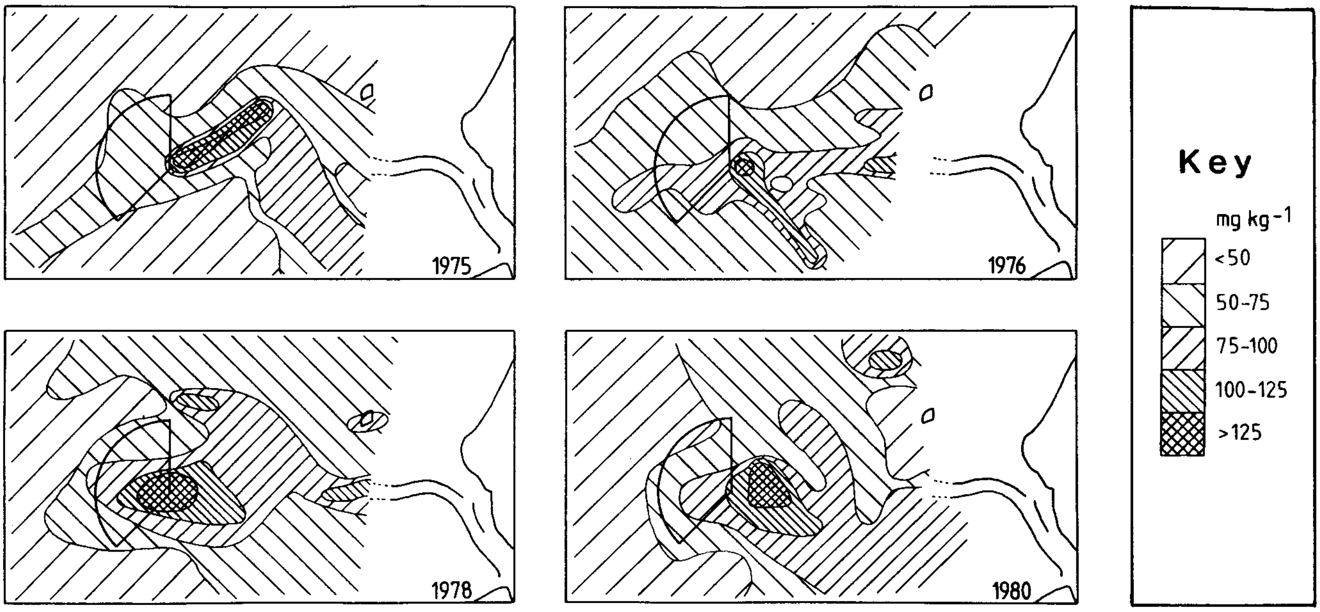


Figure 16 The concentration of copper (mg kg^{-1}) in the fine (< 90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

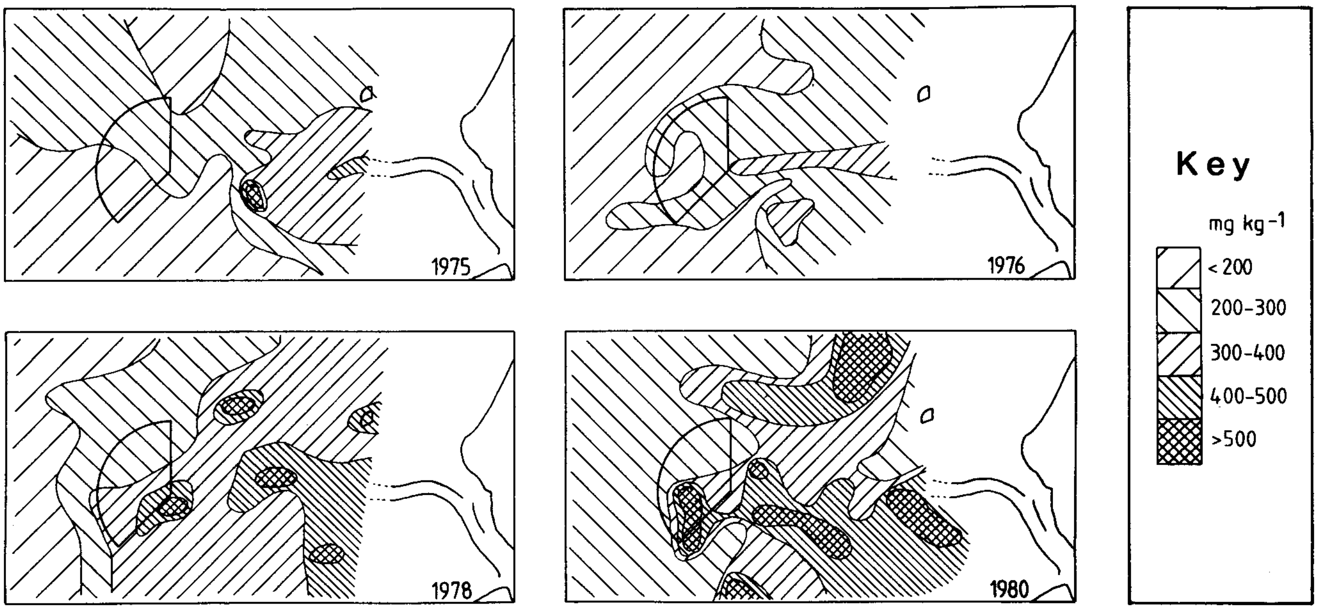


Figure 17 The concentration of zinc (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

Mercury : The distribution of mercury in fine sediments (Figure 18) was similar to that of zinc. In the offshore zone average mercury concentrations were below 1.0 mg kg^{-1}, and as low as 0.3 mg kg^{-1} in some sediments. Concentrations increased steadily towards the coast, reaching 2-2.5 mg kg^{-1} at the entrance to the Mersey. Superimposed on this gradient were higher concentrations at the dumping grounds, particularly near the sludge dumping ground where the highest individual concentrations (4.0 mg kg^{-1}) occurred.

Lead : The effects of dumping on the lead concentration in the fine fraction of sediments are clearly distinguishable from the background lead distribution (Figure 19). Elevated concentrations were found near the sludge dumping site during each survey, with peak values of 380 mg kg^{-1} in 1975 and 220 mg kg^{-1} in 1976. In 1978 the maximum concentration near the dumping ground was 680 mg kg^{-1}; in 1980 it reached 1000 mg kg^{-1}. Concentrations inshore were generally below 200 mg kg^{-1} and those near the Mersey were consistently lower than at the dumping ground. The 1980 survey also revealed high lead concentrations at two stations in the north-east of the survey area which cannot be associated with any known input of this metal.

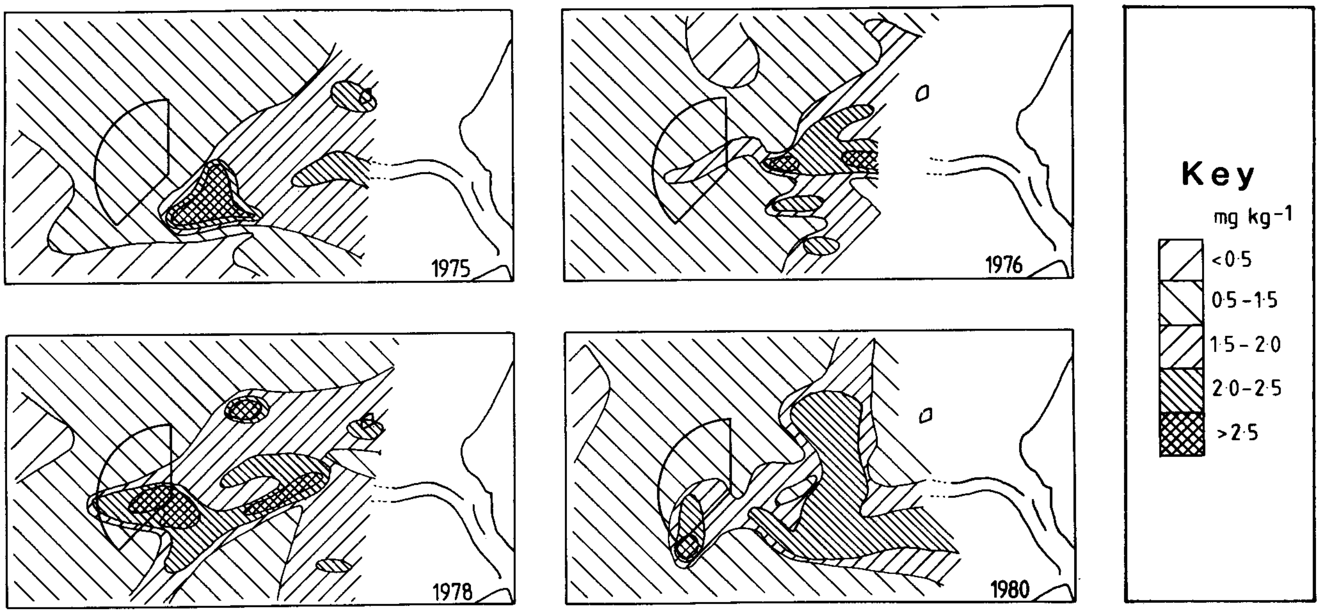


Figure 18 The concentration of mercury (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

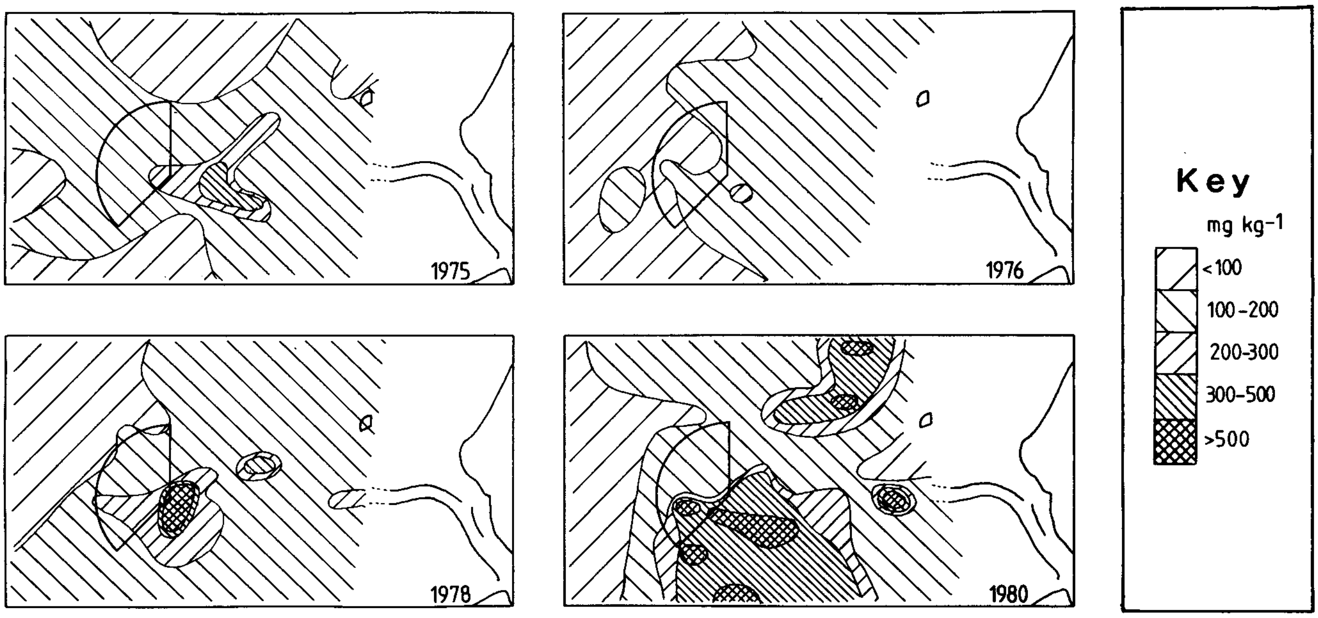


Figure 19 The concentration of lead (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

Chromium: The effects of dumping and estuary outflows on the chromium content of the sediments varied considerably between surveys (Figure 20). Background concentrations of chromium were in the range 40-50 mg kg^{-1}, and the maximum concentration at the dumping ground was 150 mg kg^{-1}.

Nickel: Background concentrations of nickel (Figure 21) ranged from 40 to 70 mg kg^{-1}. Higher concentrations (up to 110-140 mg kg^{-1}) generally occurred in the vicinity of the sludge dumping site. There appeared to be little evidence of elevated nickel concentrations near the Mersey or Dee.

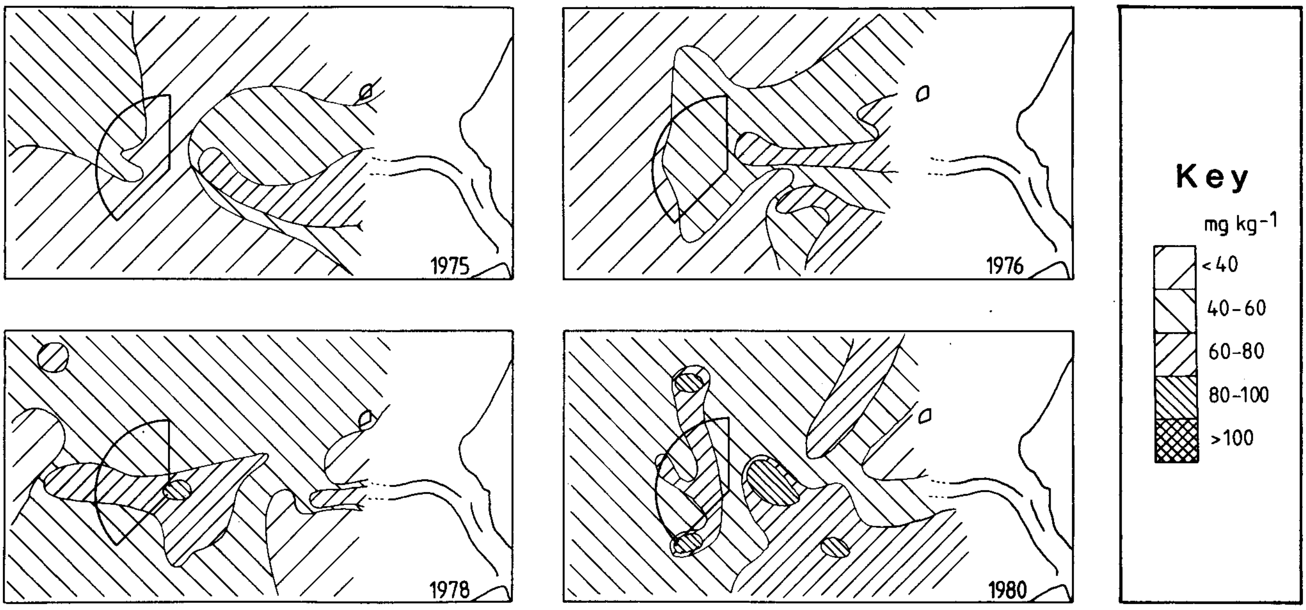


Figure 20 The concentration of chromium (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

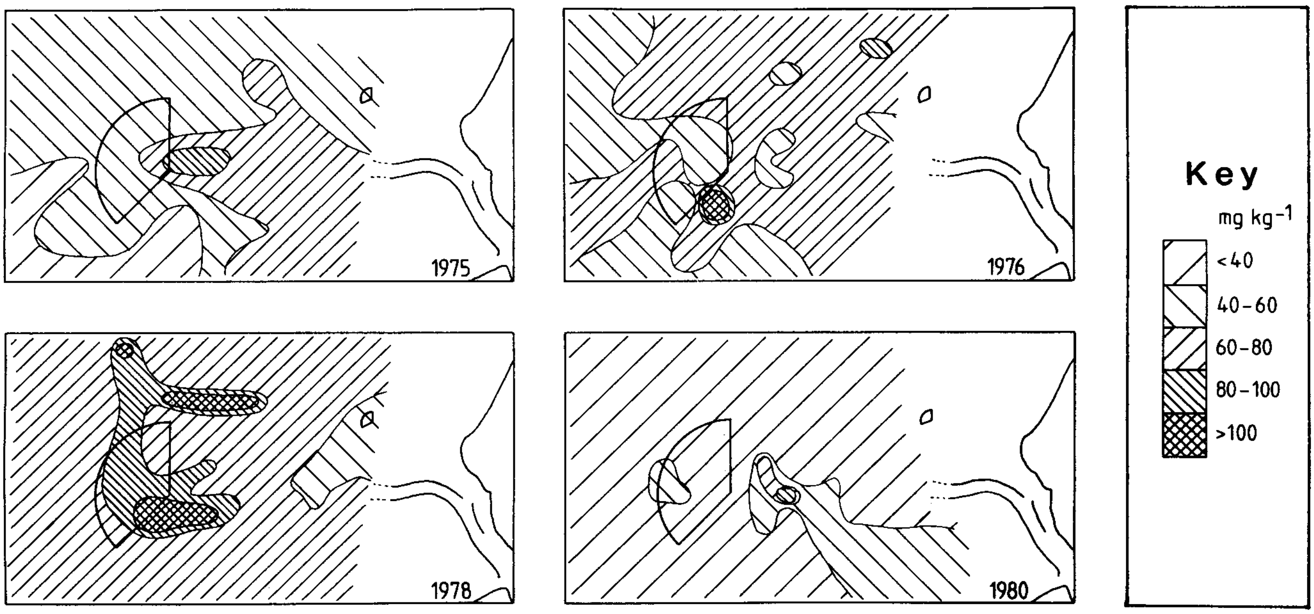


Figure 21 The concentration of nickel (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments 1975-80 (from Norton *et al.*, 1984).

Cadmium : The detection limits of all but the 1976 survey were too high to enable any conclusions to be drawn about this metal's distribution in sediments. The limited data from 1976 (Figure 22) suggest that cadmium concentrations in the sediments of the inner parts of Liverpool Bay were relatively high with maximum concentration at 4.5 mg kg $^{-1}$. The spatial distribution suggests that inshore sources are more important than inputs to the dumping zone.

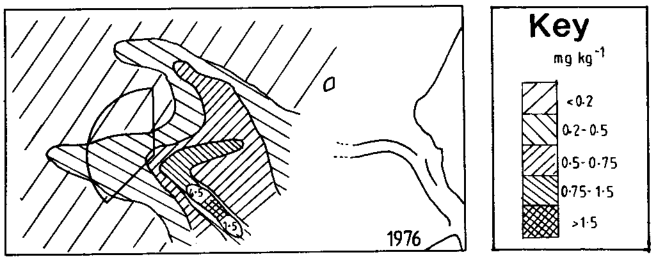


Figure 22 The concentration of cadmium (mg kg^{-1}) in the fine (<90 μm) fraction of the sediments in 1976 (from Norton *et al.*, 1984).

1977 survey of site Z : The more localised survey in 1977 of site Z (the main dredged spoil dumping site) showed concentrations of carbon and metals to be elevated near the dumping site (Figure 23). Contours generally extended in a south-westerly direction from the dumping ground. This is in the approach path of the dumping vessels, so does not necessarily indicate the direction of residual movement of material after dumping.

Table 8 Correlation coefficients between metal and organic carbon concentrations in the <90 μm fraction of the sediments

Metal	Correlation coefficient				
	1975	1976	1977	1978	1980
Mercury	0.55	0.70	0.81	0.66	0.31
Cadmium	–	0.33	–	–	–
Lead	0.62	0.45	0.62	0.72	0.06
Copper	0.47	0.51	0.69	0.74	0.53
Zinc	0.57	0.65	0.46	0.54	0.35
Chromium	0.68	0.64	0.64	0.57	0.38
Nickel	0.61	0.28	0.54	0.44	0.48
(n-2)	69	109	28	58	67
(P 0.001)	0.380	0.321	0.554	0.408	0.380

6.4.3 *Metal/organic carbon relationships*

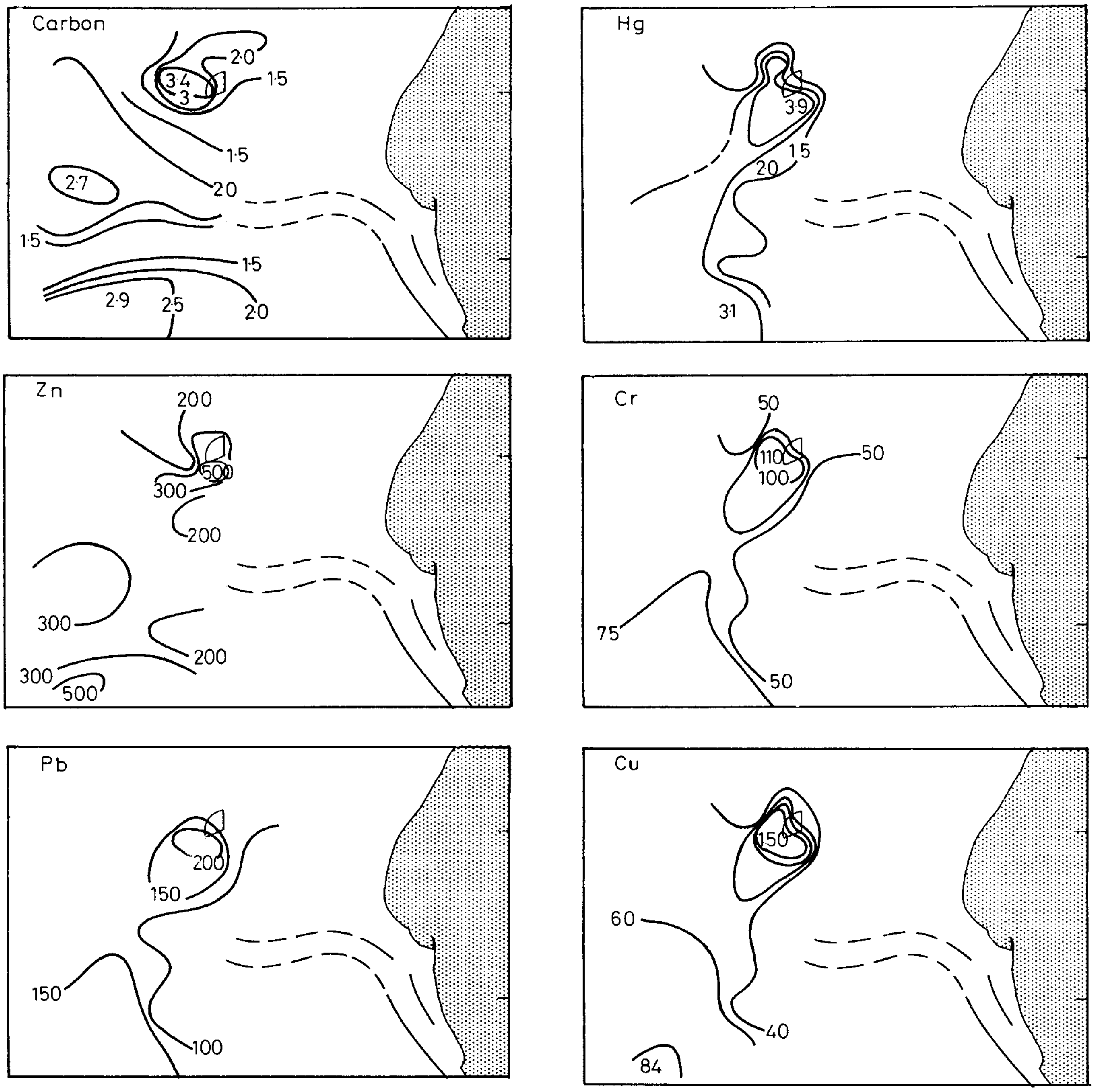
Linear regression analyses (Table 8) show that there was a good correlation between metal and organic carbon concentrations in the <90 μm fraction of the sediments for most of the surveys. These correlations suggest that, despite the high degree of mineralisation in the North Wales catchments (Nichol *et al.*, 1970), the bulk of the trace metals within the local fine sediments is derived from biological sources such as sewage discharges or uptake from solution by aquatic organisms or organic particles.

Most of the correlations listed in Table 8 are significant at P<0.001, but one notable exception is that of lead in the 1980 survey. This element shows little correlation with carbon, suggesting that either a source of lead other than that associated with organic matter influenced the sediment in 1980 or that a post-depositional process has caused lead to dissociate from the organic phase. The fact that lead concentrations at some stations exceeded the average concentrations in sludge particles suggests that an additional source of inorganic lead may have been present prior to the 1980 survey. If this were the case, it is not possible to tell whether this was due to a transient increase in sludge lead concentrations or to the dumping of an industrial waste contaminated with high concentrations of lead.

6.5 Temporal trends

The results for the 1975, 1976, 1978 and 1980 surveys have been examined further to determine whether any temporal trends exist in the concentrations of organic carbon and metals in the fine fraction of the sediments. The identification of such trends is, however, complicated by the spatial variability of the sediments and the imprecision of sampling locations at sea, which render impossible the valid comparison between results from single stations in different years. To overcome this, four 8 x 8 km square areas have been defined in the central area of the Bay (Figure 1b). Each area includes several stations and so enables mean metal concentrations to be calculated that are representative of each area rather than a single location (Norton and Rowlatt, 1982). The mean concentrations of carbon and metals for each square in each year, expressed as multiples of the 1975 values, are given in Figure 24. The significance of differences between annual means has been assessed using t-tests.

Square A includes the western extremity of the dumping ground and an area offshore from it. Most of the sludge is dumped near the inshore apex of the licensed dumping area and this square is therefore unlikely to receive any direct input of sludge; however, sludge has been shown to be dispersed into this area by tidal currents (Crickmore, 1972a; Barrett *et al.*, 1972a). Concentrations of metals and carbon there rose from 1975 to 1978 and subsequently decreased, but due to the low number of stations sampled in this square only three of these changes can be shown to be



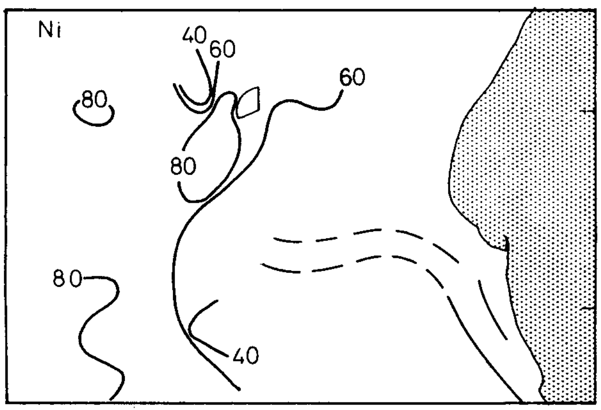


Figure 23 The concentrations of carbon (%) and metals (mg kg^{-1}) in the fine (< 90 μm) fraction of the sediment near Site Z in 1977.

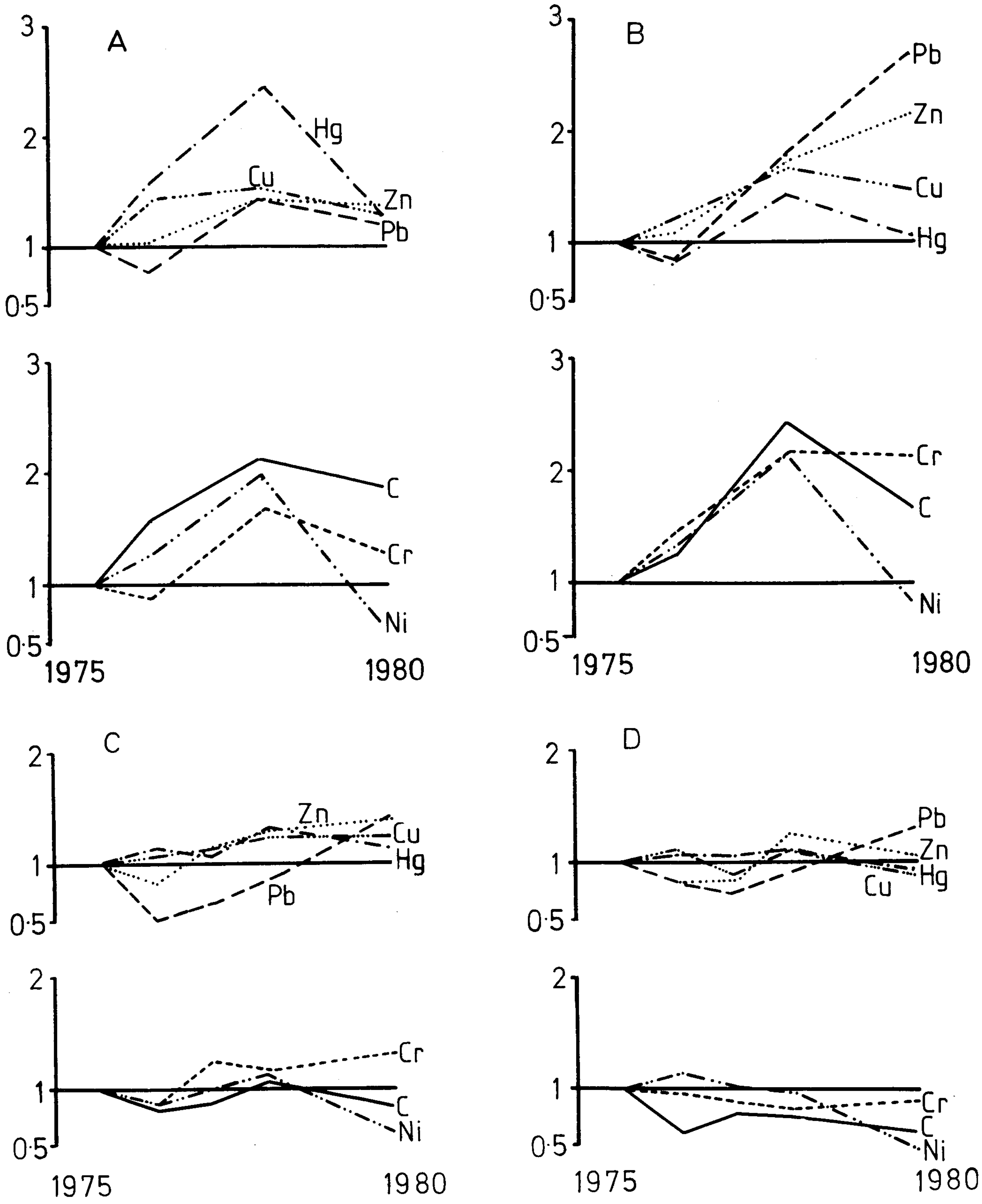


Figure 24 Temporal trends in concentrations of organic carbon and metals in the four square areas A-D (see Figure 1b). 1975-80. All values expressed as multiples of the 1975 value (from Norton *et al.*, 1984).

significant at the $P <0.05$ level (C, Ni and Cr from 1975 to 1978).

Square B includes the area in which sewage sludge is usually dumped. Trends in this square can be placed into two groups: firstly, carbon, copper, nickel, mercury and chromium, of which concentrations increased from 1975 to 1978 and subsequently decreased; secondly, lead and zinc which showed a sustained increase from 1975 to 1980. Several of the increases between 1975 and 1978 were significant: copper ($P <0.05$); zinc, nickel and chromium ($P <0.01$); carbon ($P <0.001$). The increases shown by lead and zinc between 1975 and 1980 were both significant ($P <0.01$). Decreases between 1978-80 were significant for nickel ($P <0.05$), mercury ($P \approx 0.05$) and carbon ($P \approx 0.002$).

The variations in metal concentrations within squares C and D situated between the sludge dumping site and the mouth of the Mersey were smaller than in square B. The changes in square B were more strongly correlated with those in square C than with those in Square D, suggesting that sludge dumping may also affect sediments inshore of the dumping ground, which is consistent with the transport paths of dumped sludge discussed in Section 4.

Of the four squares considered, dumping of sewage sludge is likely to most affect the sediments of square B, while discharges from the Mersey are most likely to affect sediments in square D. The latter exhibited only small changes over 1975-80, whereas carbon and metal concentrations in the dumping ground squares (A and B) changed substantially. It is thus of interest to compare these changes with those in dumping inputs over the same period. In this context, Table 1 shows that the input from sludge dumping of mercury, copper and chromium peaked in about 1977 and then declined, while the input of lead and zinc remained approximately constant. Nickel inputs varied erratically. When these changes are compared with the trends in sediment concentrations in square B, it is seen that the metals whose inputs changed little (lead and zinc) exhibited steady increases in sediment concentrations, while those whose inputs fell (mercury, copper and chromium) showed a subsequent fall or levelling off in sediment concentrations. Nickel concentrations in the sediment varied considerably as did the inputs. The delay between the drastic fall in mercury inputs and the fall in sediment concentrations suggests that the latter do not respond fully to changes in inputs in much less than a year.

Inputs of organic carbon derived from dumping rose from 1975 to 1977 and have fallen only slightly since then. Carbon concentrations in the sediments of square B rose until 1978, probably reflecting the increased input of sludge, but in 1980 concentrations fell back halfway towards their 1975 value, despite the relatively unchanged input.

One factor likely to affect the carbon content of the sediment is the age of any sludge deposit, because sludge has been shown to degrade quite rapidly (Barrett *et al.*, 1972b) losing up to 30% of its carbon in 7 days under aerobic conditions. Studies of the fate of metals during degradation of a digested sludge in sea water over 14 days (J P Daruvala, personal communication) have shown that metals other than cadmium and nickel tended to remain on the particulate phase, causing up to a 50% reduction of the carbon: metal ratio.

It was pointed out earlier that sediments sampled during neap periods are likely to contain a surface layer of freshly settled sludge in addition to any older sludge deposits within the sediments, so that for a given degree of metal contamination the carbon content will tend to be higher. The 1975, 1976 and 1978 surveys were carried out during neap periods when freshly settled sewage was probably present, while the 1980 survey took place 2-4 days after spring maximum flows when probably only older deposits were sampled. The survey timing may thus account in part for the anomalously low carbon concentrations in 1980.

Although the concentrations of metals in the fine sediment fraction appear to correlate generally with sludge inputs, it is apparent that concentrations of metals whose input has remained constant have risen, while substantial reductions in the inputs of other metals have led to only a limited fall or levelling off in sediment concentrations. This suggests that variations arising from changes in the composition of dumped sewage sludge (Section 2 showed the input of industrial wastes to be very small) may be superimposed on a rising background, possibly caused by an overall increase in the degree of contamination of the fine fractions of the sediment by dumped particles.

6.6 Conclusions on the effectiveness of dispersive processes

In general, the spatial distribution of organic carbon and metals confirms the conclusions reached from the study of natural sediment behaviour that substances entering the study area (whether from dumping or from estuarine discharges) cause local contamination but eventually become widely dispersed. Correlations show that dispersing organic matter is generally found in the fine fraction of the sediments and that the metals are mostly associated with organic carbon.

In Section 6.2 it was concluded that settlement of fine sediment occurs over much of the Bay but is subject in many areas to reworking by storms. Thus inputs of organics and metals tend to lead to a general contamination of the fine sediment fraction rather than to steady depositional sequences. This is confirmed by the analysis of short (10-15 cm) sediment cores taken in 1975/76 which showed good mixing and generally uniform concentrations throughout their depth.

At the rates of dumping observed until 1973-74 the natural dispersive characteristics of the area appear to have succeeded in effectively dispersing dumped sludge away from the dumping ground to become part of the fine sediment system of the Bay. Because of this and because the main sites of accumulation (e.g. Mersey Bar) were strongly influenced by other inputs, it was difficult then to distinguish effects specifically due to dumping activities. The results of surveys since 1974 have shown that elevations of carbon and metal concentrations in the fine fraction of the sediments near the dumping ground have become readily detectable. In the region of the dumping ground there also appears to have been a steady increase in the degree of contamination of the fine fraction of the sediments between 1975 and 1980, in contrast to other areas of the survey grid. This suggests that the natural dispersive processes are incapable of adequately dispersing sludge particles at the rate of dumping which has taken place since 1975 (Table 1).

Although this trend seems to have been maintained over the 5-year study period, the sediments in this area are subject to varying degrees of disturbance from year to year. Continued monitoring would thus be necessary to determine if dispersion processes with recurrence intervals in excess of five years are capable of disrupting this trend.

7. The Macrobenthos

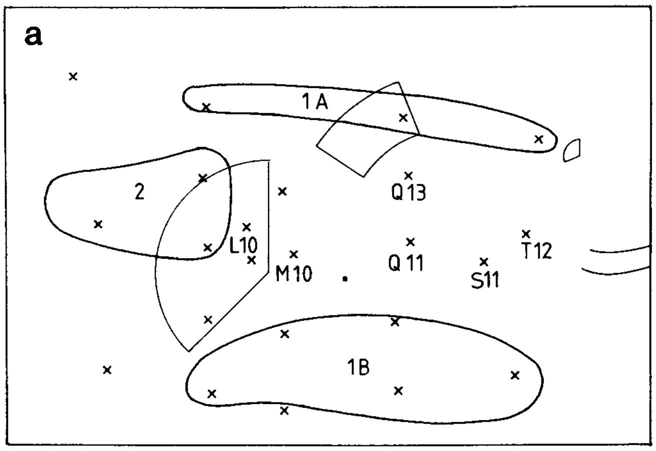
7.1 Methods

The macrobenthos has been studied in 1975, 1976, 1978 and 1980. In the 1975 and 1976 surveys a single 0.1 m^2 Day grab sample was taken at each station. In the 1978 survey two 0.1 m^2 samples were taken at each station. In 1980 a single sample was taken with a 0.14 m^2 grab. In all surveys, the animals retained by a 1 mm sieve were identified in the laboratory as far as possible to species level. Nomenclature basically follows the Plymouth Marine Fauna (MBA, 1957). Samples of sediment for laboratory analysis (Section 6) were taken from separate grab hauls, except in 1980 when the larger grab provided material for both benthos and sediment analysis.

The faunal data have been analysed using the group average classification method of Lance and Williams (1967) with the Bray Curtis index of similarity (Bray and Curtis, 1957) and the method of ordination using reciprocal averaging (Hill, 1973a). These techniques enable stations with a similar faunal content to be identified and our use of them has been described by Eagle *et al.* (1978a). Diversity was calculated by the formula of Hill (1973b) which gives a compound measure of the species richness (number of species) and the evenness of distribution of individuals between species. Multiple correlation analysis between the main faunal indices (abundance, species richness and diversity) and characteristics of the sediment was carried out for the major faunal associations identified from the classification and ordination analyses, to assist in the interpretation of the results and the identification of any effects attributable to dumping.

7.2 Results of the 1975 and 1976 surveys

Relatively few stations were sampled for benthos during the 1975 and 1976 surveys (26 and 30 stations respectively) and coverage was fairly poor in the region between the dumping ground and the Mersey, the area shown by tracer experiments to be that most likely affected by sludge. The classification analysis which was carried out resulted in limited sample clustering, most of the 'associations' consisting of single, isolated stations. Only a brief summary of the results from these two years will therefore be given.



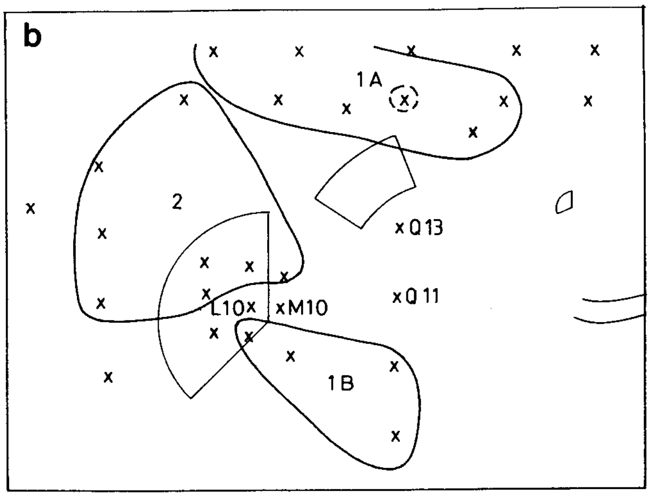


Figure 25 Sampling stations in the biological surveys carried out in (a) 1975 and (b) 1976:
1 faunal association identified by classification; Q13 sample station referred to in text; x sampling station.

Two major associations were identified in 1975. The first consisted geographically of two parallel bands, running roughly east-west, to the south and north of the dumping ground (as shown in Figure 25a). This association was characterised by the presence of the polychaetes *Nephthys cirrosa* and *Spiophanes bombyx*, the former being relatively more abundant in the cleaner sand present in the southern area, the latter commoner in the muddier substrates in the north. A second smaller association was formed in the coarse sand/gravel basement area in the north-western sector of the dumping ground, with the polychaete *Scalibregma inflatum* a common dominant. These two associations are typical of the main community types found in the areas north and south of the dumping ground and have appeared in most of the MAFF surveys covered by this report.

All the remaining survey stations were isolated by the classification. A number of these single station associations occurred between the dumping ground and the Mersey. The fairly coarse substrate at station L10 just inside the dumping ground apex had a sparse though fairly diverse fauna. Samples taken at station M10 just to the east and at Q13 to the north-east of the dumping ground were both dominated by the bivalve *Abra alba*, although the substrates were very dissimilar, being basically of medium-coarse sand at M10 and of much finer material at Q13. The fauna at station Q11 mid-way between the dumping ground and the Mersey was dominated by *Cultellus pellucidus*, while nearer the Mersey itself stations S11 and T12 were characterised by progressively finer substrates dominated by *Spiophanes bombyx* and *Nephthys hombergii* respectively.

A somewhat similar picture resulted from classification analysis of the 1976 data (Figure 25b). Two associations were again present, one to the south of the dumping ground dominated by *Nephthys cirrosa* and the other to the north dominated by *Spiophanes bombyx*. Stations in the north-western part of the dumping ground were again linked; there, although *Scalibregma inflatum* was common as in 1975, the association was dominated in 1976 by amphipod species. Near the dumping ground apex stations L10 and M10 were isolated as before, but in 1976 both had sparse faunas. The only stations sampled to the east of the dumping ground were at Q11 and Q13 which were linked by the presence of large numbers of *Scalibregma inflatum*.

7.3 Results of the 1978 survey

Classification assisted by ordination analysis of the 61 stations at a similarity level of 0.4 produced 20 associations. The fauna and sediment characteristics of these associations are listed in Table 9 and their geographical positions shown in Figure 26. Although above the chosen similarity level, the classification linkage of the two stations assigned to association 3 was not supported by ordination, so these stations have been subdivided in Table 9 into associations 3A and 3B. Table 9 also shows associations 9 and 10 similarly split into their geographically separate parts. The twenty associations fell broadly into four groups in the classification dendrogram (associations 1-8, 9-13, 14-17 and 18-20) the characteristics of which are now discussed.

7.3.1 *Associations 1-8*

These associations had in common low faunal abundance and predator species as a fairly important constituent. Although not closely related by the classification, associations 1 and 4 were grouped by the ordination, and these appear to be the equivalent of the northern *Nephthys cirrosa/Spiophanes bombyx* association found in the previous surveys. The diversity index was high in associations 1 and 4, but this was due to fairly high species richness within a sparse fauna. The substrate of association 1 appeared to be basically medium sand, but the fauna suggests that mixed microhabitats were present – *Ophelia borealis* would indicate clean sand, but the supply of organics must have been substantial to support *Scoloplos* and the cirratulids. The sparseness of the fauna in association 4 would appear to be due to a relatively large fraction of the substrate being of coarse sand which is likely to be mobile (Section 6).

Association 2 was also impoverished and large numbers of blackened shells were present, suggesting low Eh as a possible cause. Association 3 consisted of two stations, one in the north-east of the survey area and the other at the offshore extremity of the dumping ground. The latter contained a coarser (and from the appearance of *Nephthys cirrosa*) cleaner substrate, though both had fair numbers of deposit feeders in common.

Association 5 appeared to be the same *Nephthys cirrosa/Spiophanes bombyx* association as appeared consistently in previous surveys, located in a band of less muddy sediments between much richer, siltier areas to both north and south.

Associations 6, 7 and 8 appeared to have little in common, except for the sparseness of the fauna and the presence of *Tharyx marioni*. Association 6 to the south of the dumping ground was found in a gravelly substrate and only a poor sample was obtained, contributing to the low faunal abundance. The low numbers of animals in association 7, lying at the apex of the dumping ground, could have been the result of the exclusion of species due to deposition of fine organic-rich material. Association 8 was geographically separate from other stations in group 1, lying at the centre of the large, fairly coarse sand association 16 to the west of the dumping ground.

Table 9 1978 Liverpool Bay Survey – Association Characteristics of Benthos

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Diversity Index (Hill)	Characteristic species (density≥25 m^{-2})		Sediment type *	Organic carbon in fines (%)
1	1	16	210	11.6	*Lumbrinereis gracilis*	25	2/1/18/71/8/0	2.4
					Ophelia borealis	25		
					Scoloplos armiger	25		
					Cirratulidae	25		
2	1	10	140	6.2	*Nephthys longosetosa*	35	2/0/25/62/10/1	2.4
					Ophelia borealis	30		
3A	1	22	400	11.6	*Nephthys longosetosa*	70	2/2/42/52/2/0	2.2
					Spiophanes bombyx	50		
					Pectinaria koreni	45		
					Bathyporeia sp.	35		
					Magelona papillicornis	30		
					Phyllodocidae	25		
3B	1	18	260	7.7	*Spio filicornis*	65	2/1/19/73/5/0	Not measured
					Spiophanes bombyx	50		
					Pectinaria koreni	25		
					Nephthys cirrosa	25		
4	1	22	165	11.0	*Spiophanes bombyx*	40	3/3/3/30/58/3	2.0
5	5	14	236	6.2	*Nephthys cirrosa*	75	2/1/30/62/3/2	2.9
					Spiophanes bombyx	25		
6	1	19	130	16.1	–		1/0/11/43/22/23	2.8
7	1	11	110	6.9	*Tharyx marioni*	30	7/4/5/43/40/1	2.9
8	1	17	255	5.3	*Tharyx marioni*	100	2/1/22/54/18/3	1.7
					Ophelia borealis	30		
9	10	31	1311	4.0	*Pectinaria koreni*	624	3/3/28/47/18/1	2.5
					Scalibregma inflatum	80		
					Lanice conchilega	51		
					Phyllodocidae	51		
					Abra alba	47		
					Spiophanes bombyx	45		
					Ampharete acutifrons	35		
					Lumbrinereis gracilis	27		
					Polydora antennata	26		
9A	4	32	1020	4.8	*Pectinaria koreni*	368	4/3/14/40/36/3	3.0
					Scalibregma inflatum	189		
					Ampharete acutifrons	69		
					Lanice conchilega	50		
					Tharyx marioni	35		
					Cirratulidae	28		
					Owenia fusiformis	25		
9B	6	31	1505	3.5	*Pectinaria koreni*	795	3/2/37/52/6/0	2.2
					Abra alba	78		
					Phyllodocidae	78		
					Spiophanes bombyx	72		
					Lanice conchilega	51		
					Polydora antennata	42		
					Lumbrinereis gracilis	32		
					Ophelia borealis	29		
					Eulalia sp.	25		
10	4	21	441	5.8	*Pectinaria koreni*	160	14/16/14/46/9/1	2.4
					Lumbrinereis gracilis	33		

* Percentage of clay/silt/fine sand/medium sand/coarse sand/gravel

Table 9 *continued ...*

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Diversity Index (Hill)	Characteristic species (density>25 m^{-2})		Sediment type *	Organic carbon in fines (%)
10A	2	14	347	3.2	*Pectinaria koreni*	185	21/25/16/33/4/1	2.4
					Nephthys hombergii	30		
10B	2	27	535	8.4	*Pectinaria koreni*	135	7/7/12/61/13/0	2.4
					Lumbrinereis gracilis	65		
					Ampharete acutifrons	33		
					Scalibregma inflatum	33		
					Lanice conchilega	30		
11	4	39	1131	8.4	*Lanice conchilega*	288	13/10/28/39/9/1	2.3
					Abra alba	96		
					Pectinaria koreni	93		
					Eulalia sp.	93		
					Ampharete acutifrons	73		
					Scalibregma inflatum	56		
					Spiophanes bombyx	36		
					Cultellus pellucidus	28		
12	3	27	569	11.8	*Scalibregma inflatum*	63	3/3/9/48/26/11	2.8
					Ampharete acutifrons	60		
					Spiophanes bombyx	57		
					Pectinaria koreni	43		
					Lanice conchilega	38		
					Nephthys cirrosa	33		
					Tharyx marioni	30		
					Lumbrinereis gracilis	28		
					Cultellus pellucidus	28		
					Ensis ensis	25		
13	1	32	885	8.1	Anthozoa *(*flat disc*)*	235	1/1/14/80/3/1	2.3
					Lanice conchilega	155		
					Anthozoa *(*bulbous*)*	75		
					Phyllodocidae	65		
					Polyzoa (encrusting)	40		
					Glycera lapidum	35		
					Lumbrinereis gracilis	30		
					Pectinaria koreni	30		
					Tharyx marioni	30		
					Alcyonidium sp.	30		
14	10	32	787	7.8	*Scalibregma inflatum*	234	4/4/23/23/36/10	2.0
					Pectinaria koreni	62		
					Lumbrinereis gracilis	61		
					Owenia fusiformis	54		
					Ampharete acutifrons	50		
					Cerianthus lloydi	25		
					Anthozoa *(*flat disc*)*	25		
15	1	32	1025	8.7	*Pomatoceros triqueter*	260	1/1/18/18/38/24	2.5
					Lumbrinereis gracilis	150		
					Scalibregma inflatum	100		
					Ampharete acutifrons	75		
					Cerianthus lloydi	70		
					Polyzoa (encrusting)	60		
					Hydrallmania falcata	45		
					Glycera convoluta	40		
16	8	29	404	10.7	*Scalibregma inflatum*	66	2/2/24/33/34/5	1.4
					Urothoe sp.	66		
					Pectinaria koreni	36		

* Percentage of clay/silt/fine sand/medium sand/coarse sand/gravel

Table 9 *continued ...*

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Diversity Index (Hill)	Characteristic species (density>25 m^{-2})		Sediment type *	Organic carbon in fines (%)
17	1	41	475	22.4	*Lanice conchilega*	50	2/2/17/32/42/5	1.9
					Polyzoa (encrusting)	40		
					Ascidiacea *(sandy)*	35		
					Urothoe sp.	30		
					Sertularia sp.	25		
18	1	26	3640	1.7	*Pectinaria koreni*	2745	10/0/78/11/1/0	2.1
					Phyllodocidae	375		
					Phyllodoce maculata	120		
					Owenia fusiformis	70		
					Magelona papillocornis	60		
					Nephthys sp.	45		
					Spiophanes bombyx	30		
					Scalibregma inflatum	25		
19	1	42	5020	6.3	*Pectinaria koreni*	1265	7/4/11/51/21/6	3.0
					Lanice conchilega	1040		
					Abra alba	980		
					Anthozoa *(flat disc)*	465		
					Ampharete acutifrons	215		
					Anthozoa *(bulbous)*	120		
					Mya arenaria	100		
					Polyzoa (encrusting)	100		
					Eulalia sp.	90		
					Scalibregma inflatum	85		
					Lumbrinereis gracilis	75		
					Sthenelais boa	65		
					Natica alderi	50		
					Nereis sp.	30		
					Harmothoe sp.	30		
					Phyllodoce maculata	25		
					Venus striatula	25		
					Lutraria lutraria	25		
					Phyllodocidae	25		
20	4	16	1458	2.3	*Abra alba*	1006	23/30/35/10/2/0	2.1
					Nephthys hombergii	136		
					Pectinaria koreni	129		

* Percentage of clay/silt/fine sand/medium sand/coarse sand/gravel

7.3.2 *Associations 9-13*

These associations were linked by the presence of the deposit feeders *Pectinaria koreni*, *Lanice conchilega* and *Scalibregma inflatum*, which often occurred in very large numbers; it was the variation in dominance of these three species which distinguished the individual associations formed. Associations 9 and 10 were both dominated by *Pectinaria koreni*: each occurred in two different areas of the Bay so have been distinguished in Table 9 as associations 9A, 9B and 10A, 10B respectively. Figure 26 shows that these associations were linked geographically; association 9A was located just inshore of the dumping ground connected via the two stations in association 10A with the six stations of association 9B north of the Mersey Bar. Association 10B contained two stations to the north and west of association 9B.

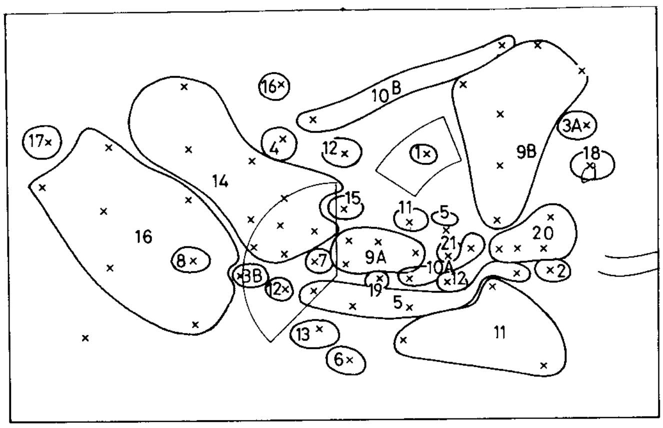


Figure 26 Faunal associations identified in the analysis of the 1978 benthos data: 20 boundary and number of the faunal association; x sampling station.

Association 9A, located just inshore of the dumping ground, contained almost the same number of species as association 9B (north of the Mersey outflow) but abundances were lower due to *Pectinaria koreni* being less dominant. The concentration of organic carbon in the fine sediment was higher near the dumping ground and *Scalibregma inflatum* was present in large numbers; the substrate in association 9A was also coarser than in 9B, as indicated by the presence of *Owenia fusiformis*.

Associations 10A and B contained fewer numbers and species than associations 9A and B. Although association 10A geographically linked associations 9A and B, sediments were finer and the abundance of *Pectinaria* was very much lower. The differences in the fauna may indicate that the associations were the result of differing causal factors. Association 9B could be linked to the outlow of particulate organics from the Mersey while association 9A may have responded to dumped organic matter. Association 10A is exposed to inputs from both the Mersey and the dumping ground, and is also in the area where fine sediments accumulate, as shown by the high silt and clay content (Section 6.1). It is possible that the supply of fine particulate matter was too high for some species and also above the optimum for *Pectinaria*, leading to a relatively impoverished fauna, but the greater instability of these fine sediments may also account for this. The related association 10B was present in mixed mud/sand substrates to the north and west of association 9B, and was richer and more diverse, with *Pectinaria* again the dominant species.

Association 11 to the south of the Mersey Bar had a rich, diverse fauna dominated by *Lanice conchilega*, but with other deposit feeders also well represented. Within this association, the most easterly station near the Burbo Bight (Figure 1) was isolated by ordination; sediments here were extremely muddy (60% sediment particles <63 μm) and *Abra alba* rather than *Lanice* was the most abundant deposit feeder.

Association 12 possessed no strong dominants at the three stations which were fairly widely separated geographically. Although not linked by classification, association 13 was grouped with association 12 by ordination. However, the substrates in these two associations were rather different, with association 12 on a mixed substrate with high organic carbon concentrations in the fines and association 13 on mainly fine/medium sand. Both associations, however, contained a number of deposit-feeding species at moderate densities, resulting in high diversities.

7.3.3 *Associations 14-17*

These four associations occupied a large area including the northern part of the dumping ground and to the west of it. They appear typical of this area of coarse sand and gravel basement, with *Scalibregma inflatum* generally a common dominant (as in 1975 and 1976). The fauna was moderately rich with fairly high diversities, the lowest being in association 14 where *Scalibregma* greatly outnumbered the other species. Association 15 was isolated by the ordination, due presumably to the presence of *Pomatoceros triqueter* and the polyzoans found on stones in the substrate. The latter was also found in association 17.

7.3.4 *Associations 18-20*

These, the remainder of the associations found by the 1978 survey, appeared unrelated, association 18 being separated from 19 by the ordination and association 20 being isolated by both classification and ordination. Very large numbers of *Pectinaria koreni* occurred in both association 18 and association 19, but the general faunal composition differed greatly and substrates were dissimilar. Association 18 was situated just to the north of the dredged-spoil dumping ground in a mostly fine sand substrate; *Pectinaria* was extremely dense, resulting in a very low diversity. *Pectinaria* was also numerous in association 19, lying to the east of the dumping ground, where other deposit feeders were also abundant, resulting in a very rich, fairly diverse fauna.

Association 20 was obviously different from the other two, occurring on very muddy substrates at the mouth of the Mersey. Diversity was very low due to the dominance of *Abra alba*. *Nephthys hombergii* appeared in this association, encouraged presumably by the muddy conditions.

7.3.5 *Comparison with other studies in 1978*

The benthos in Liverpool Bay was also examined in 1978 by the University College of North Wales (UCNW) whose survey concentrated on the central region from the dumping ground to the Mersey channel (Rees and Walker, in press).

A number of stations were common to both the UCNW and MAFF surveys, and a comparison of the fauna present at them indicated that the two sets of results were generally consistent even though the MAFF survey was carried out later in the year (end of October as opposed to late September for UCNW) leading to a noticeable overall reduction in the density of some species. This was especially true for *Pectinaria*, which was found in the dumping ground during the UCNW survey but not the MAFF survey: nevertheless, the MAFF results are consistent with the conclusion of Rees and Walker that there was an increase in the relative abundance of *Pectinaria koreni* in the vicinity of the dumping ground in 1978, which led to quantifiable changes in community diversity.

7.3.6 *Correlations between faunal and sedimentological characteristics*

Statistical correlations between biological indices (faunal abundance, species richness, diversity) and the characteristics of the sediments were computed for the associations containing sufficient stations. The results are listed in Table 10. Statistical tests were also carried out on combinations of the basic associations that separated out at similarity levels of above 0.3 in the classification dendrogram.

Within association 9, species richness was negatively (though at a relatively low level of significance) correlated with the concentration of organic carbon in the fines. However, in the part of this association north of the Mersey (9B), species richness was positively related to the amount of organic carbon in the fine sediment fraction, which would infer a strong negative correlation between the number of species in association 9A and organic carbon. (Unfortunately there were too few stations in 9A to allow full correlation to be carried out, but the correlation coefficient was –0.85. The correlation coefficient between Hill diversity and organic carbon was –0.94). This suggests that the high carbon concentrations (Figure 12) arising from sludge dumping may be leading to a slight impoverishment of the fauna even though the dominant species, *Pectinaria*, may be taking advantage of the organic carbon input. In contrast, the more diffuse source of carbon in association 9B does not appear to have led to any reduction in species in a more abundant fauna.

In association 14, the medium sand fraction was positively correlated with the number of species, whereas the abundance of species was positively related to the silt/clay fractions of the sediment and more weakly to the organic carbon content. This indicates that, in this association, the dominant species *Scalibregma* was taking advantage of the organic input. Within associations 3, 4 and 5 combined, species richness was positively related to the amounts of silt and clay in the sediment and negatively to both the medium sand fraction and organic carbon levels. Abundance was also negatively related to organic carbon, so fauna in these associations could be sensitive to any increase in carbon loading.

Table 10 Correlations between the faunal data and physical/chemical characteristics of the sediments in 1978

Association No	Variables		Correlation	
	Fauna	Sediment	+ or –	Significance (%)
9	Species richness	Clay	+	95
	Species richness	Organic carbon	–	81
9B	Species richness	Organic carbon	+	90
14	Species richness	Medium sand	+	98
	Abundance	Clay	+	98
	Abundance	Silt	+	95
	Abundance	Medium sand	+	93
	Diversity	Organic carbon	–	91
	Abundance	Organic carbon	+	80
16	NIL			
3 + 4 + 5	Species richness	Clay	+	90
	Species richness	Silt	+	94
	Species richness	Medium sand	–	96
	Species richness	Organic carbon	–	93
	Abundance	Fine sand	+	93
	Abundance	Organic carbon	–	90
	Diversity	Medium sand	–	97
9 + 10	Species richness	Clay	–	90
	Species richness	Silt	–	93
	Abundance	Fine sand	+	94
11 + 12 + 13	Species richness	Fine sand	+	96
	Species richness	Organic carbon	–	98
	Abundance	Clay	+	99
	Abundance	Silt	+	99
	Diversity	Organic carbon	+	93
14 + 15 + 16 + 17	Species richness	Medium sand	+	90
	Abundance	Clay	+	99
	Abundance	Silt	+	98
	Diversity	Clay	–	90
	Diversity	Silt	–	95

Table 11 1980 Liverpool Bay Survey – Association Characteristics of Benthos

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Diversity Index (Hill)	Characteristic species (density≥25 m^{-2})		Sediment	Organic carbon in fines (%)
1	4	25	925	5.5	*Lanice conchilega*	353	1/1/32/50/10/6	1.6
					Urothoe marina	60		
					Lumbrinereis gracilis	38		
					Eulalia sanguinea	35		
					Ophelia borealis	30		
					Ampharete acutifrons	25		
					Tharyx marioni	25		
2	6	42	4675	4.0	*Lanice conchilega*	2285	5/7/55/26/5/2	1.7
					Ampharete acutifrons	397		
					Pectinaria auricoma	207		
					Eulalia sanguinea	177		
					Scoloplos armiger	163		
					Tubulanus polymorphus	153		
					Pholoe minuta	137		
					Mysella bidentata	120		
					Scalibregma inflatum	68		
					Abra alba	65		
					Edwardsia callimorpha	63		
					Lumbrinereis gracilis	62		
					Phyllodoce maculata	58		
					Phyllodoce lineata	48		
					Poecilochaetus serpens	35		
					Thyasira flexuosa	32		
					Spiophanes bombyx	25		
3	1	34	3480	5.3	*Owenia fusiformis*	1290	2/1/18/75/4/0	2.4
					Lanice conchilega	470		
					Ampharete acutifrons	420		
					Edwardsia callimorpha	390		
					Scoloplos armiger	180		
					Eulalia sanguinea	90		
					Tubulanus polymorphus	70		
					Pectinaria koreni	70		
					Philine sp.	60		
					Nephthys hombergii	50		
					Stenelais boa	40		
					Pholoe minuta	40		
					Scalibregma inflatum	30		
4	7	32	1644	9.7	*Ampharete acutifrons*	376	7/10/35/34/13/1	1.7
					Magelona alleni	141		
					Lumbrinereis gracilis	117		
					Edwardsia callimorpha	96		
					Amphiura filiformis	84		
					Scalibregma inflatum	76		
					Mysella bidentata	73		
					Tubulanus polymorphus	53		
					Owenia fusiformis	51		
					Nephthys hombergii	50		
					Lanice conchilega	49		
					Cultellus pellucidus	46		
					Upogebia deltaura	34		
5	8	35	1433	11.1	*Urothoe marina*	281	2/2/23/32/30/11	1.6
					Ampharete acutifrons	173		
					Lumbrinereis gracilis	78		
					Scoloplos armiger	70		
					Lanice conchilega	58		
					Photis longicaudata	48		
					Tharyx marioni	43		
					Poecilochaetus serpens	41		
					Dendrodoa sp.	40		

Table 11 *continued ...*

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Density Index (Hill)	Characteristic species (density>$25m^{-2}$)		Sediment	Organic carbon in fines (%)
5 cont.	8	35	1433	11.1	Anthozoa sp	38	2/2/23/32/30/11	1.6
					Ampelisca typica	31		
					Tubulanus polymorphus	29		
					Ammotrypane aulogaster	25		
					Nucula turgida	25		
6	2	30	845	15.2	*Photis longicaudata*	90	3/3/28/57/8/1	1.9
					Mysella bidentata	85		
					Amphiura filiformis	70		
					Scoloplos armiger	50		
					Tharyx marioni	50		
					Poecilochaetus serpens	45		
					Urothoe marina	40		
					Harpinia sp.	35		
					Lumbrinereis gracilis	30		
					Nephthys hombergii	30		
					Pectinaria auricoma	25		
7	1	28	1750	9.2	*Urothoe norvegica*	420	2/1/17/44/19/17	0.8
					Anthozoa sp.	220		
					Pholoe minuta	200		
					Paraonis lyra	160		
					Mysella bidentata	110		
					Ampharete acutifrons	80		
					Scoloplos armiger	70		
					Tubulanus polymorphus	60		
					Lumbrinereis gracilis	60		
					Oxydromus capensis	60		
					Lanice conchilega	40		
					Gattyana cirrosa	40		
					Nereis longissima	40		
					Caulleriella alatus	30		
					Tharyx marioni	30		
8	12	51	4480	6.1	*Dendrodoa* sp.	1959	3/3/19/24/39/12	1.8
					Lumbrinereis gracilis	480		
					Ampharete acutifrons	328		
					Photis longicaudata	208		
					Cerianthus lloydi	141		
					Lanice conchilega	134		
					Scalibregma inflatum	93		
					Pholoe minuta	83		
					Paraonis lyra	76		
					Owenia fusiformis	56		
					Urothoe marina	51		
					Nucula turgida	37		
					Pomatoceros triqueter	37		
					Tubulanus polymorphus	36		
					Upogebia deltaura	33		
					Mysella bidentata	32		
					Orchomene nana	28		
					Porcellana longicornis	28		
					Nereis longissima	25		
9	1	12	180	9.5	*Urothoe marina*	30	3/3/27/25/36/6	1.7
					Eulalia sanguinea	30		
10	2	19	345	11.0	*Urothoe marina*	75	2/1/6/44/36/11	1.4
					Anthozoa sp.	35		
					Lanice conchilega	25		

Table 11 *continued ...*

Association Number	No of Stations	Average No of species per station	Faunal density (m^{-2})	Density Index (Hill)	Characteristic species (density ≥25 m^{-2})		Sediment	Organic carbon in fines (%)
11	3	27	679	17.5	*Nucula turgida*	53	2/1/10/50/36/1	2.1
					Chaetozone setosa	53		
					Lanice conchilega	50		
					Dorvillea kefersteini	50		
					Lumbrinereis gracilis	47		
					Urothoe marina	47		
					Tubulanus polymorphus	37		
12	3	20	553	8.6	*Scoloplos armiger*	140	1/1/17/71/10/0	1.8
					Lanice conchilega	53		
					Chaetozone setosa	37		
					Ophelia borealis	33		
13	3	20	1030	3.5	*Scoloplos armiger*	533	1/1/33/33/27/5	1.7
					Urothoe elegans	100		
					Bathyporeia sp.	40		
					Tharyx marioni	33		
14	1	20	1210	7.7	*Pholoe minuta*	280	8/15/63/11/2/1	2.5
					Scoloplos armiger	220		
					Spiophanes bombyx	200		
					Scalibregma inflatum	90		
					Nephthys hombergii	70		
					Lanice conchilega	50		
					Edwardsia callimorpha	50		
					Phyllodoce laminosa	40		
					Poecilochaetus serpens	40		
					Chaetezone setosa	30		
					Tubulanus polymorphus	30		
15	4	8	175	4.4	*Scoloplos armiger*	55	1/0/18/74/6/1	2.2
					Nephthys longosetosa	33		
16	6	10	261	5.7	*Nephthys cirrosa*	77	1/0/18/71/9/1	2.1
					Scoloplos armiger	28		
17	1	12	160	10.7	–	–	1/1/14/74/10/0	1.6
18	1	9	100	8.3	–	–	1/0/3/66/30/0	2.2
19	1	12	160	8.5	*Pectinaria koreni*	40	41/57/2/0/0/0	2.4
20	2	20	1500	2.3	*Abra alba*	980	21/24/52/1/1/1	1.2
					Nephthys hombergii	115		
					Nucula turgida	60		
					Owenia fusiformis	45		
					Scoloplos armiger	30		
					Nucula hanleyi	25		

Combining associations 9 and 10 did not provide much useful information additional to that obtained from association 9 alone. The combination resulted in a negative correlation between species richness and clay/silt and a positive correlation between abundance and the fine sand fraction. Species richness was negatively correlated with organic carbon but at a far less significant level than in association 9 alone.

Combination of associations 11, 12 and 13 showed that the number of species was correlated positively with the fine sand fraction and negatively with organic carbon levels. Abundance was positively related to both clay and silt fractions while diversity was positively related to the organic carbon level. It would appear that the number of species was dependent on the fine sand fraction, but dominants such as *Lanice* preferred more clay and silt in the substrate. The role of organic carbon in this association appears complex, since its correlation was positive with diversity yet strongly negative with species richness. It may be that the quantities of organic carbon present in the substrate had a negative effect on many species (including *Lanice*) so that their distribution was more even, thus increasing the diversity.

The combination of associations 14, 15, 16 and 17 produced correlations little different from association 14 alone.

7.4 Results of the 1980 survey

Classification and ordination analysis of the fauna at the 66 stations sampled resulted in the formation of twenty associations at a 0.35 level of similarity. The fauna and sediment characteristics of these associations are listed in Table 11 and their geographical positions are shown in Figure 27. The twenty associations in the classification dendrogram could be split broadly into five groups (associations 1-3, 4-8, 9-11, 12-16 and 17-20), the characteristics of which are now discussed.

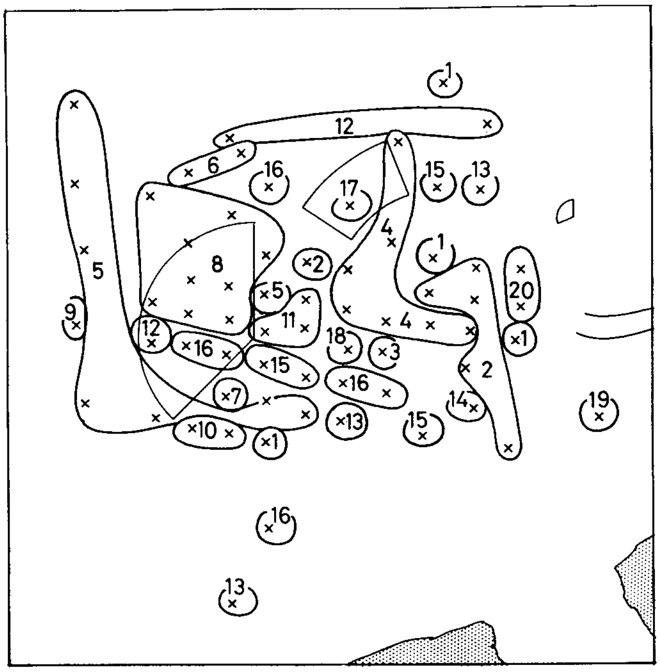


Figure 27 Faunal associations identified in the analysis of the 1980 benthos data: ② boundary and number of the faunal association; x station position.

7.4.1 *Associations 1-3*

These associations occurred mostly in the eastern part of the survey area and were characterised by the presence of large numbers of tube-building polychaetes, notably *Lanice conchilega* with its associate *Eulalia sanguinea*. Association 1 consisted of four isolated stations containing a moderately rich fauna on fairly coarse substrates, diversity being somewhat reduced by a predominance of *Lanice*. Association 2 lay in an area of muddy sand sediments to the west of the Mersey outflow which supported an abundant fauna with many species, though diversity was again low due to the high numbers of *Lanice*. *Pectinaria auricoma* was also abundant in association 2, being suited by the muddy-sand conditions as were *Mysella bidentata* and *Thyasira flexuosa*; *Ampharete acutifrons* was also common, but was, like *Lanice*, apparently able to survive in a variety of habitats, for it was equally numerous in the coarser substrate of association 3, an isolated station located mid-way between the dumping ground and the Mersey and containing an abundant and rich fauna dominated by *Owenia fusiformis* which was able to use the coarser sand in the sediment for tube-building. The richness of the fauna in all three associations can be largely attributed to the stability resulting from the formation of a 'mat' of polychaete tubes, although the availability of carbon-enriched fines may also have contributed.

7.4.2 *Associations 4-8*

The associations in this grouping were rather varied. Association 4 lay to the east of the dumping ground adjacent to association 2. The muddy sand substrate supported a fairly rich, diverse fauna dominated by *Ampharete acutifrons* with populations of *Amphiura filiformis*, *Nephthys hombergii* and *Cultellus pellucidus* also present. *Magelona alleni*, which is typical of this area of Liverpool Bay, was also common.

Ampharete was also common in association 5, a group of stations in a band to the west and south of the dumping ground (plus a single station just inshore of the dumping ground). Here, the coarser, more gravelly substrate was also inhabited by *Urothoe marina* and such sedentary species as *Dendrodoa*. *Ampharete* appears to have been very successful in 1980; Rees and Walker (1981) noted that it was then one of the few species which occurred in densities above its long-term median.

Association 6 lay to the north of the dumping ground on a mixed sand substrate, the range of sediments present supporting a very diverse community. Association 7 was a single, gravelly station to the south of the dumping ground, dominated by *Urothoe norvegica*. As in previous years, the stations in the northern part of the dumping ground and to the north-west of it were in a single association, association 8, which contained an extremely rich fauna though diversity was low due to the dominance of the sedentary tunicate *Dendrodoa*. The presence of large numbers of *Dendrodoa* together with species such as *Pomatoceros* indicates a considerable amount of exposed hard substrate.

7.4.3 *Associations 9-11*

These associations occured in areas of coarse sediment around the dumping ground, and were characterised by a rather impoverished fauna with the amphipod *Urothoe* *marina* a prominent species. Association 9, a single station in the west of the survey area, and association 10 to the south, both contained sparse, though reasonably diverse faunas. Species numbers were low and the environment appeared rather hostile. The fauna was more abundant in association 11 which was located just inshore of the dumping ground apex. Organic carbon levels were high, as would be expected from its position, and perhaps supported the cirratulids and *Nucula turgida* which are usually found in a siltier environment. Association 11 is interesting in that it occurs in roughly the same geographical area as the fairly rich association 9A of the 1978 survey. The association characteristics were obviously very different there in 1980, due essentially to a loss of fines and the virtual disappearance of *Pectinaria koreni* and *Scalibregma inflatum*: the absence of these dominants led to very high diversities within the sparser fauna.

7.4.4 *Associations 12-16*

These associations shared the common factor of the presence of the deposit-feeding polychaete *Scoloplos armiger*. They differed in other respects, however, varying considerably in the richness of the fauna and in sediment characteristics. Associations 12 and 13, consisting of widely separated stations, both contained moderately rich faunas on clean well-sorted sediments, as indicated by the presence of *Ophelia borealis* in association 12 and *Bathyporeia* in association 13. Association 14, an isolated station towards the Burbo Bight area, was much muddier and organically rich. *Scoloplos* was still common, but the dominant species was *Pholoe minuta*. Muddy sand species present were *Spiophanes*, *Scalibregma* and *Nephthys hombergii*. Associations 15 and 16 both had sparse faunas on medium sand substrates and appeared to represent the southern *Nephthys cirrosa* association of earlier surveys, occupying an east-west band south of the dumping ground.

7.4.5 *Associations 17-20*

Associations 17 and 18 both had extremely impoverished faunas, with no species occurring at densities exceeding 25 m^{-2}. Association 17 occurred to the north-east of the dumping ground, in an area which also supported a poor fauna in 1978. Association 18, however, like nearby association 11, had a richer fauna in 1978 than in 1980, with considerable numbers of *Pectinaria koreni* in a substrate containing a higher percentage of fine sediment than in 1980. Association 19 was a single station well to the east, near the Burbo Bight area. Although the fauna was sparse, this was the only station where *Pectinaria koreni* occurred at any considerable density. The sparseness of the fauna may have been due to smothering by the very large quantities of fine material in the sediment. However, Rees and Walker (1981) found high densities of *Pectinaria* in this area when they sampled it two months earlier in the year, and it seems possible that the substrates are of unstable muds, leading to a destruction of the fauna by autumnal gales.

Association 20 was the *Abra alba* community located, as in 1978, just offshore of the Mersey in a muddy sediment. As in 1978, this association was completely isolated by the ordination/classification analysis from the other associations in the survey area. Diversity was very low due to the dominance of *Abra*, although the muddy conditions also supported populations of species such as *Nephthys hombergii* and *Nucula turgida*.

7.4.6 *Correlations between faunal and sedimentological characteristics*

Statistical correlations between biological indices (faunal abundance, species richness, diversity) and the characteristics of the sediments were computed for the associations containing sufficient stations and are listed in Table 12.

Within association 2, diversity was negatively correlated with the clay fraction in the sediment, though the level of significance was low. Combining associations 2 and 3 strengthened this correlation. Since species richness in these two associations was very high, the negative effect on diversity must have been associated with increased numbers of the dominants, possibly related to the retention of fine sediments within the stable 'mats' of *Lanice* tubes.

In association 4, both species richness and abundance were positively correlated with the fine sand fraction suggesting that species in this association had a strong preference for this type of sediment, as was also indicated by the negative correlation of abundance with medium sand. There was no relation between organic carbon levels and the faunal characteristics.

In association 5, situated to the west and south of the dumping ground, the numbers of species were also positively related to the amount of fine as opposed to medium sand. Diversity was positively related to the finer fractions (clay, silt and fine sand), an obviously *physical* effect, since the correlation with organic carbon was negative. A similar effect was observed in association 8 in the northern part of the dumping ground and to the north-west of it where species richness and abundance were related positively to the amount of clay but negatively to organic carbon. There was also a positive relationship between species richness and fine sand content in association 8, especially for stations in the southern part of the asociation. In association 16 all biological indices were positively related to the clay content of the sediment, though within a sparse fauna.

The general picture which seems to emerge from these correlations is the importance of the finer sediment fractions to the structure and abundance of the benthic fauna in 1980. A rich fauna occurred in the finer sediments, whereas in areas where fines had been lost, such as inshore of the dumping ground apex, the fauna was impoverished.

Table 12 Correlations between the faunal data and physical/chemical characteristics of the sediments in 1980

Association No	Variables		Correlation	
	Fauna	Sediment	+ or –	Significance (%)
2	Diversity	Clay	–	86
2 + 3	Diversity	Clay	–	94
4	Species richness	Fine sand	+	92
	Abundance	Fine sand	+	94
	Abundance	Medium sand	–	90
5	Species richness	Fine sand	+	90
	Species richness	Medium sand	–	98
	Diversity	Clay	+	99
	Diversity	Silt	+	99
	Diversity	Fine sand	+	97
	Diversity	Organic carbon	–	92
8	Species richness	Clay	+	95
	Abundance	Clay	+	94
	Abundance	Organic carbon	–	98
16	Species richness	Clay	+	91
	Abundance	Clay	+	99
	Diversity	Clay	+	98

7.4.7 *Comparison with other studies in 1980*

The annual UCNW benthos survey of Liverpool Bay took place in 1980 between September and November, the survey period being rather extended due to adverse weather conditions. The overall conclusions were the same as those from the MAFF survey, namely that there had been a considerable fall in faunal abundance at stations near the dumping ground. The UCNW survey showed a decline in the dominants such as *Pectinaria*, though it was observed that the paucity of the fauna was due to low numbers of a whole range of the more sessile species. Only *Ampharete acutifrons* and *Urothoe elegans* (also prominent in the MAFF survey) had abundances in 1980 substantially above their long-term median values.

Rees and Walker (1981) speculated that one factor which could have caused such a decline was "trophic group ammensalism" in this case the result of dense populations of a deposit feeder loosening the sediment, creating an unstable environment and adversely affecting other species either by preventing successful recruitment or by clogging the filtering mechanisms of existing filter feeders. It was suggested that the build up of the large population of *Pectinaria* inshore of the dumping ground in the late 1970s could have led to such a situation, with a subsequent impoverishment of the fauna following a collapse in the *Pectinaria* populations. Examination of Figure 10 shows that the fines content of the sediment in the area dominated by *Pectinaria* in 1978 had fallen by 1980, which would be consistent with this hypothesis. However, the natural mobility of the sands in this area renders them subject to considerable variability from physical factors alone and the exact cause of the faunal changes must remain unresolved. Nevertheless, it is interesting to note that there was some indication from the results of the 1978 survey that the dense *Pectinaria* populations could have been promoted by the sewage sludge dumping. If this was the case, then dumping activity may have been indirectly responsible for the subsequent fauna changes.

7.5 Effects of dumped wastes on the benthic fauna

The descriptions of the benthic fauna in 1975, 1976, 1978 and 1980 show the survey area to be very heterogeneous in both sediments and fauna within any one year and that substantial changes occur from year to year. With such spatial and temporal variability due to essentially natural fluctuations, the detection of any effects due to the dumping of sewage sludge and industrial wastes is difficult.

It is clear from the discussions of each year's results that no gross adverse changes attributable to dumping have occurred in the benthic fauna and that the physical conditions and the vagaries of larval settlement appear to be the dominant influence on the fauna. Only a few associations contained sufficient stations for a statistical link with sediment carbon levels to be sought, but in most of these there was either no correlation or it was negative, suggesting that increased carbon concentrations in the sediment are not necessarily beneficial to the fauna. In some areas near the dumping ground, however, particularly association 9A in 1978, the possibility remains that the dominant species, *Pectinaria*, may have responded to organic matter in suspension from sewage sludge dumping. This in turn may have contributed to the subsequent impoverishment of the fauna in the same area in 1980 following the collapse of the *Pectinaria* population.

8. Results of other MAFF studies in Liverpool Bay

In addition to the results of the surveys described in Sections 5-7, MAFF has also investigated the effects of dumping on water quality and on the degree of contamination of fish and shellfish in Liverpool Bay by metals, organochlorine pesticides and PCBs. The results of the water quality study have been presented by Norton *et al* (1984) and of the study of contaminants in fish and shellfish by Murray and Norton (1982), Preston and Portmann (1981) and Norton and Murray (1983). In order to provide a complete description of the extent and the results of MAFF's work in the Bay, their conclusions will be summarised here.

8.1 Water quality studies

In 1978 and 1979 the concentrations of zinc, copper, cadmium and nickel in solution and in suspension were determined at a series of stations from the mouth of the River Mersey to an area offshore of the sewage sludge dumping ground.

The concentrations of dissolved metals were found to be correlated with salinity, which suggested that a major source of dissolved metals was from river and estuarine discharges. In 1979 a study of the stations nearest to the dumping ground suggested that dumping may have led to local and patchy effects on concentrations of dissolved cadmium and, to a lesser extent, nickel.

The concentrations of metals on suspended particles were highest at the dumping ground where, relative to those found inshore, zinc concentrations were greater by 50%, those of copper and cadmium showed a two-fold increase and those of nickel a ten-fold increase. These observations, made towards the end of a period in which 40 000 t of sludge were dumped, demonstrated that dumping had caused a major increase in the concentrations of metals present on suspended matter near the dumping ground. The contrast with the 1978 results, when no such effects were observed after 3 days without dumping, suggests that elevated concentrations of metals in suspension in the dumping area are relatively short-lived.

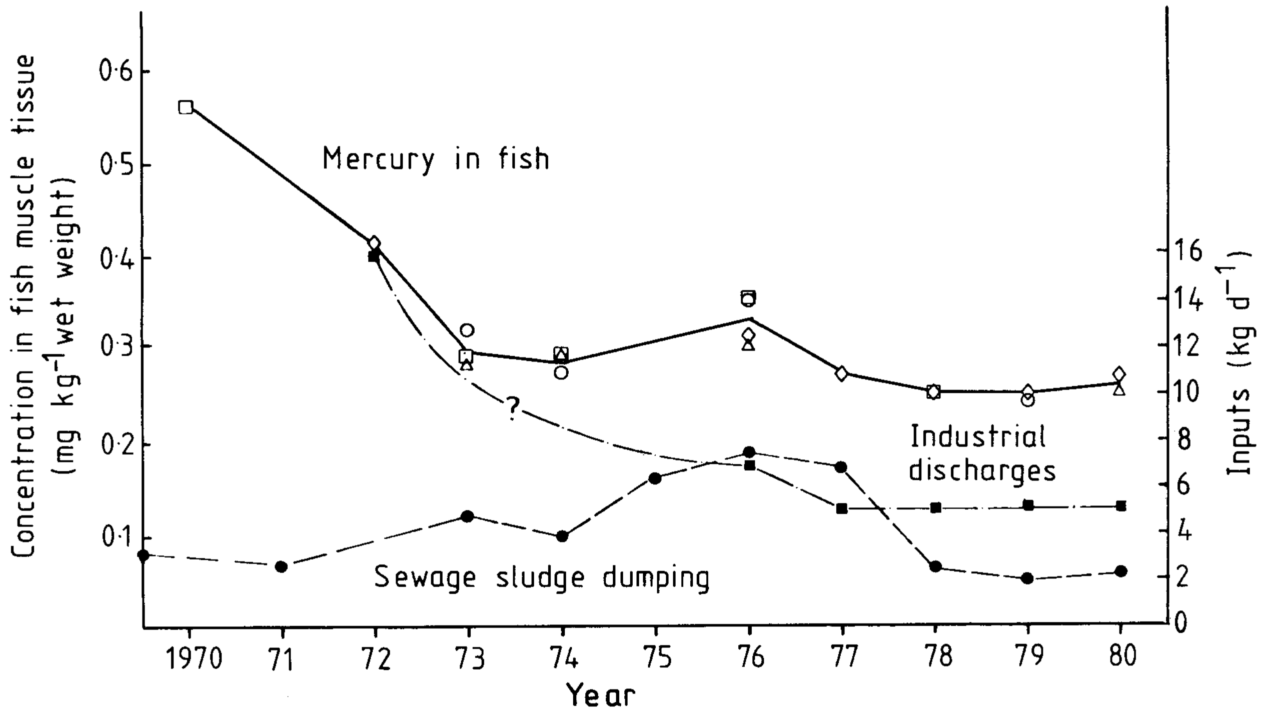


Figure 28 Concentrations of mercury in fish muscle tissue and inputs of mercury to Liverpool Bay from industrial discharges and sewage sludge dumping (from Norton and Murray, 1983): △ mean concentration of mercury in dab, cod, plaice, sole; □ mean concentration of mercury in dab, whiting, plaice, sole; ○ mean concentration of mercury in whiting, cod, plaice, sole; ◇ mean of a sample reflecting dietary intakes (Preston & Portmann, 1981); —— mean of all sets of measurements of mercury concentrations in fish; – ● – input of mercury from sewage sludge dumping (from Head (1981) & MAFF records); –·■·– input of mercury from industrial discharges (from Preston & Portmann, 1981).

8.2 Concentrations of metals in fish and shellfish

The concentrations of mercury, cadmium, lead, zinc and copper in fish and shellfish from Liverpool Bay have been measured frequently by MAFF from 1970 to date. A review of the results of this programme (Norton and Murray, 1983) showed that the concentration of mercury in fish muscle had declined from the highest levels in 1970. These changes were plotted against changes in the input of mercury from industrial discharges and sewage sludge dumping to illustrate the effects of different sources of mercury (Figure 28).

Figure 28 shows that between 1970 and 1980 the average concentration of mercury in fish flesh was approximately halved and that this in general reflects the reduced input of mercury from industrial discharges and sewage sludge. The major reduction of mercury concentrations in fish which occurred between 1970 and 1974 was most probably related to a fall in the amount of mercury from industrial discharges. However, between 1974 and 1976 mercury levels in fish appeared to rise against a background of falling industrial input but increasing inputs from sewage sludge dumping. Between 1976 and 1978 levels of mercury in fish fell, in line with a decrease in the mercury content of sewage sludge. From 1978 to 1980 mercury concentrations in fish remained constant at about 0.25 $mg\,kg^{-1}$ (wet weight): this concentration is still slightly above the level found in fish from areas free of anthropogenic inputs but it has been shown (Preston and Portmann, 1981) that such concentrations have no significance from the public health viewpoint.

There was no evidence of widespread elevated concentrations of the other metals studied (cadmium, lead, zinc and copper) in fish and shellfish from the Bay, but there was some evidence of local contamination by these metals (Norton and Murray, 1983). Some samples of shrimp and queens from the vicinity of the dumping ground contained cadmium and lead at about twice the concentration found in other UK waters. Some samples of elasmobranch fish were also found to contain elevated concentrations of lead. Zinc and copper concentrations in fish and shellfish were within the normal range for the same species caught elsewhere; the only exceptions to this were in some samples of queens and hermit crabs.

8.3 Concentrations of organochlorine pesticides and PCBs in fish and shellfish

Concentrations of organochlorine pesticides and PCBs in fish and shellfish from Liverpool Bay have been measured over the period 1973-76 (Murray and Norton, 1982). This limited study suggested that, compared with those found in other UK inshore waters, the concentrations of PCBs and the DDT metabolites DDE and DDD were elevated significantly in several species of fish and shellfish, particularly in fish liver tissue. Concentrations of DDT and α and γ HCH showed no elevation, whereas dieldrin concentrations appeared slightly higher in a few species relative to concentrations found in other inshore waters. No temporal trends were apparent over the 1973-76 period.

9. Conclusions

Surveys of Liverpool Bay carried out during the period 1975-80 have allowed the natural dispersive processes affecting the fate of dumped wastes to be described, and the effects of dumping on the sediments and benthos to be determined. Because of the extent of each survey and the time scale of the investigation, it has been possible to identify both spatial and temporal trends in the results.

Most effort has been devoted to studying the area used for the dumping of 1.6-1.8 million tonnes of sewage sludge and 0.05-0.3 million tonnes of industrial waste per year, but some sampling was also carried out near the site where 3.5-5.7 million tonnes of dredged spoil is dumped annually. Near the dumping grounds dumping is the dominant source of particulate matter and of its contaminants, but estimates of the amounts of organic substances and metals entering the Bay from other sources show that the outflow of the rivers Mersey and Dee are also important to the Bay as a whole.

At the sewage sludge/industrial waste dumping area, located about 30 km west of the mouth of the river Mersey in depths of 20-30 m, currents are sufficient to disperse the dumped sludge rapidly during spring tides and periods of storms, but for up to 50% of the time, predominantly during neap tides, conditions allow the rapid settlement of dumped material to the sea bed. The main direction of initial dispersion is along the east-west axis of the tidal streams but the residual movement of particulate matter in the bottom part of the water column is strongly inshore towards the Mersey. Radiotracer releases have demonstrated that after dumping most of the sludge particles move towards the Mersey but that dispersion of some of the sludges also occurs offshore and to the north and south of the dumping area. These initial dispersion paths have been confirmed by studies of the distribution of faecal bacteria in the sediments.

The distribution of the natural sediments throughout the survey area is very heterogeneous ranging from gravels offshore, through sands, to muddy areas inshore. A study of natural sediment dynamics shows there to be only a limited capacity for the accumulation of dumped material near the dumping ground but that substantial amounts of sludge-derived material may be incorporated within the muddier inshore sediments, although even these are subject to reworking during storms.

The degree of contamination of the sediments by dumping has been assessed by examination of the fine (<90 μm) fraction of the sediment because the variability in sediment grain size precludes the use of whole sediment data to identify spatial trends. In all the surveys, concentrations of organic carbon and metals in the fine sediment were elevated in areas associated with the two main sources of these materials – sludge dumping and the discharge of the Mersey. Elevated concentrations were also found near the dredged spoil disposal site and in the south-eastern extremity of the survey area near the Burbo Bight.

As the concentrations of some metals in sediments near the dumping ground appeared to increase from 1975 to 1980, the mean concentrations of metals in four zones along the main transport path from offshore of the dumping ground to the Mersey estuary were calculated and the statistical significance of trends between surveys was evaluated. These showed that mean concentrations of zinc and lead in the zone where dumping took place had risen consistently from 1975 to 1980 whereas concentrations of carbon, mercury, copper, chromium and nickel had risen from 1975 to 1978 and fallen from 1978 to 1980. In contrast, changes in the zone nearest the Mersey were slight. The changes in the dumping ground zone appeared to follow changes which took place in the inputs of these substances from sludge dumping, and there was some evidence that variations in metal concentrations in the sediment arising from changes in sludge inputs were superimposed on a general rise in background concentrations which may also have been caused by an increase in the degree of contamination of the fine fraction by sludge particles.

Effects on the benthos of the area were sought by objective analysis of the faunal distributions. Classification identified up to twenty different faunal associations in each year, reflecting the wide variations in sediment type and conditions. The spatial and temporal variability of the area's fauna made the effects of dumping difficult to detect; in no area could gross changes be attributed to sewage sludge or industrial waste dumping. Statistical analysis of trends within the larger faunal associations showed the basic faunal indices of abundance and species richness to be more dependent on the physical than on the chemical properties of the sediment. This is consistent with the conclusion that the dominant influence affecting the benthos was the natural environment. There was no evidence of increased carbon levels in the fine sediment fraction leading to increased numbers of animals or species; indeed in some zones increased carbon levels tended to be associated with fewer animals and species. It is possible that sludge deposits may have assisted the growth of an association near the dumping ground dominated by *Pectinaria* by providing a source of organic matter in suspension. The growth of this species may, however, have led ultimately to a loss of fine sediment which resulted in an impoverishment of the fauna following the collapse of the *Pectinaria* population.

Studies on water quality and fish and shellfish quality in the Bay, published elsewhere, have shown dumping to have little effect on concentrations of dissolved metals in the dumping area, but that substantial increases in the concentrations of metals associated with suspended particles occurred immediately after dumping; these effects disappeared within 3 days or so following a cessation of dumping. Mercury concentrations in fish flesh were found to be elevated in the Bay; they declined from 1970 to 1980 and appeared to vary with the two major inputs of mercury to the Bay, i.e. sewage sludge dumping and industrial discharges. There was no evidence of widespread elevations in cadmium, lead, copper or zinc concentrations in fish and shellfish from Liverpool Bay, but there was some evidence of localised contamination of species taken from near the sludge dumping ground. Nevertheless, none of the elevated concentrations was significant from a public health viewpoint.

The results of this series of studies thus demonstrates that Liverpool Bay has a substantial capacity to receive and disperse dumped waste without significant effects on the health or quality of the biota. However, there are indications that the rate of dumping in recent years may be leading to an increasing degree of contamination of the fine sediment fraction by metals. Equally it is apparent that the chemical quality of fish and shellfish has been affected by inputs of metals and other substances of which sewage sludge is a significant contributor. It is important, therefore, that monitoring continues in order to establish future trends. It would appear unwise for the inputs of metals from dumping to be increased until trends over a longer period have been established.

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