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Effects of metal contamination on macrobenthos of two North Sea estuaries

John A. Hall, Christopher L. J. Frid, and
Roger K. Proudfoot

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Time series of faunal data for the Tyne (1978–1989) and the Tees (1981–1992) estuaries were analysed using Bray–Curtis similarity indices and dendrogram plots, multidimensional scaling, and principal components analysis (PCA). These data show that benthic community structure changed during the 1980s. Data on sediment heavy metal concentrations in the Tees estuary for the period 1971–1986 show significant reductions in contamination across this period, and also between-station differences. PCA of faunal data showed grouping of stations into marine, estuarine, and contaminated clusters. PC axis 1 appeared to be controlled by position of sampling site in relation to sources of contamination, and PC axis 2 by salinity. However, sediment metal concentrations were not significantly correlated with PC axis 1 scores. Reasons for this are discussed. Changes in the benthos indicate improving environmental quality, suggesting that clean-up initiatives in both estuaries have been effective.

Key words: benthic macrofauna, heavy metals, Tees estuary, time series, Tyne estuary.

J. A. Hall and C. L. J. Frid: Dove Marine Laboratory, University of Newcastle upon Tyne, Cullercoats, North Shields NE30 4PZ, England, UK; R. Proudfoot: Environment Agency, Northumbria Area Office, Tyneside House, Newcastle Business Park, Skinnerburn Road, Newcastle upon Tyne NE4 7AR, England, UK.

Introduction

Some estuaries represent a significant source of contaminants entering the North Sea (Clark, 1992; North Sea Task Force, 1993), leading to pollution of coastal and offshore sediments (Kröncke, 1987). For example, Lewis (1989) showed that a plume of metal-contaminated water from the Tees estuary in north-east England potentially affected an area of 70 km^2 of North Sea sediments. Initiatives to improve the environmental quality of the North Sea, such as those established by the Oslo and Paris Commissions in 1972 and 1974, have been partially directed to reducing the loads of hazardous substances in estuaries in order to prevent potentially toxic compounds from entering the sea (North Sea Task Force, 1993).

The north-east coast of England, particularly the Tyne and Tees estuaries (Fig. 1a, b, c), has been identified as a "hot-spot" in terms of heavy metal contamination (Topping, 1983; North Sea Task Force, 1993). A history of urbanization, heavy engineering, shipbuilding and chemical manufacture with numerous wastewater discharges to these estuaries has left a legacy of severe sediment contamination (Middlebrook, 1950; Le Guillou, 1978; NRA, 1994a, b). Discharges from chemical and sewage treatment works, leachate from contaminated land, high levels of zinc, cadmium, and mercury from a tributary and mineral mining in the upper catchment all contribute contaminants to the Tyne estuary (NRA, 1994b). A sewage interceptor and treatment scheme was built to intercept 200 major outfalls in this area over the period 1980–1996. The Tees estuary is particularly contaminated as a result of industrial activity, especially at Wilton, and Teesport, contributing to the sediment and water column pollution load (NRA, 1994b). The implementation of measures to reduce sewage inputs to the Tees has been confined mainly to the upper and middle reaches of the estuary. However, there are plans to install an interceptor sewer system similar to that on the Tyne.

Monitoring of these areas is necessary in order to assess the efficiency of any remedial action. The use of benthic macrofauna as an indicator of sediment quality is well described (e.g. Pocklington and Wells, 1992). However, the value of such observations may be enhanced by concurrent monitoring of sediment quality

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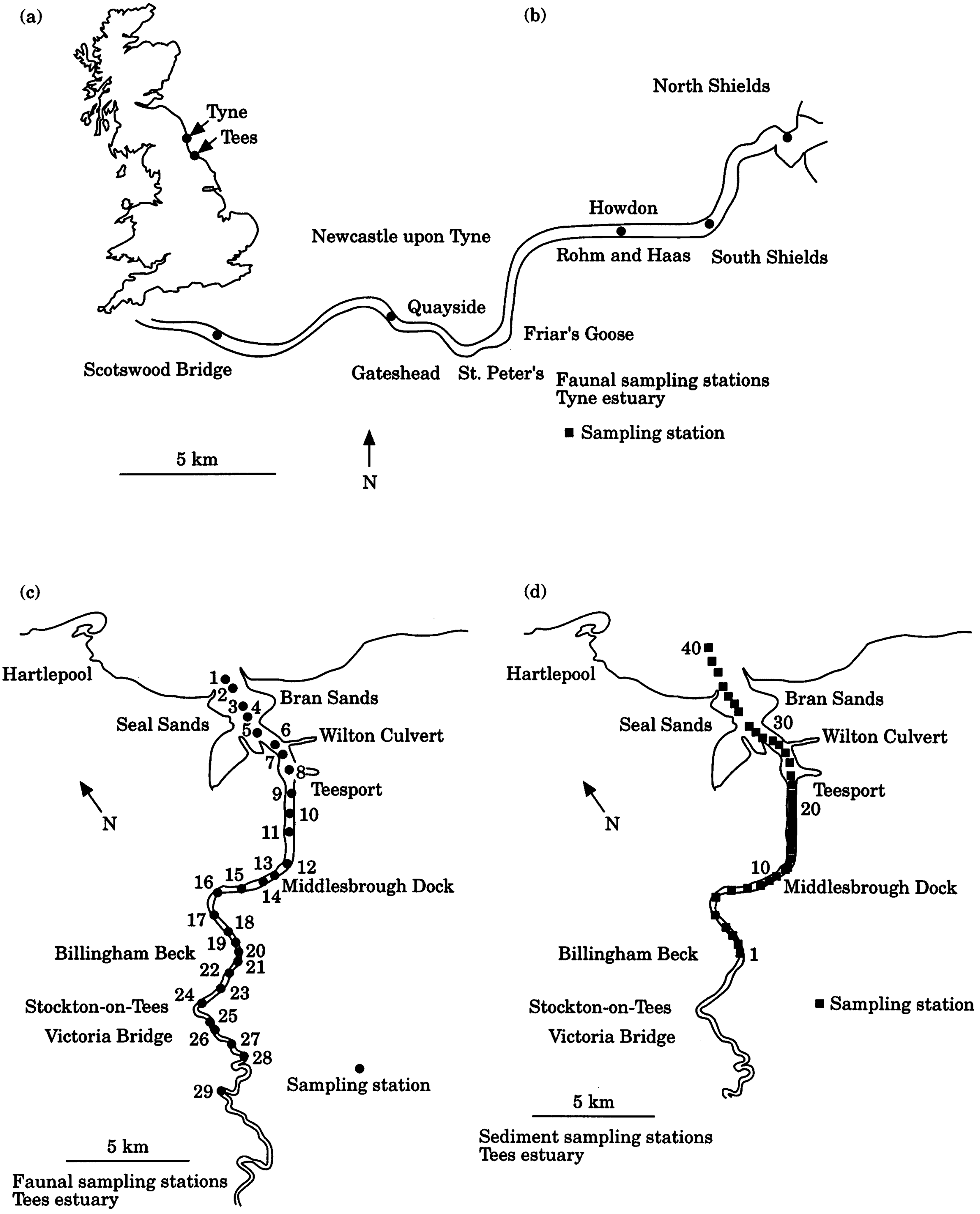


Figure 1. (a) Locations of Tyne and Tees estuaries in the United Kingdom. (b) Faunal sampling stations and areas of anthropogenic activity on the Tyne estuary. (c) Faunal sampling stations and areas of anthropogenic activity on the Tees estuary. (d) Approximate locations of sediment sampling stations in the Tees estuary.

(Gray *et al.*, 1991a). In the Tees estuary, Zeneca plc has carried out annual monitoring of benthic communities in the period 1978–1995. Sediment metal concentrations were also measured intermittently.

In contrast, until recently, monitoring of the Tyne estuary has been less comprehensive. Environmental parameters have been measured only occasionally, and not alongside benthic samples, thus making any comparisons difficult (e.g. MAFF, 1992). However, to monitor the improvements brought about by the Tyneside Sewerage Scheme beginning in 1980, limited sampling of benthos was carried out between 1978 and 1989 by Northumbrian Water (Sinton, 1978).

Changes in environmental policies on discharging wastes to the Tees estuary, and the installation of treatment works on the Tyne mean that during the 1970s and 1980s improvements in environmental quality were expected. This paper examines the changes in benthic macrofauna in the Tees estuary against a background of declining sediment heavy metal concentrations, and changes in the Tyne estuary against decreasing discharges of untreated sewage over the past two decades.

Data sources and methods

All faunal data for the Tyne estuary were obtained from reports held by the Environment Agency, Newcastle upon Tyne). Three samples were taken at five stations (Fig. 1b) using a 0.1 m^2 Day grab. Sediments were then passed through a 1 mm mesh, and retained fauna identified to species level where possible and enumerated. The sites were visited annually in October or early November between 1978 and 1989, with the exceptions of 1982 and 1987.

Data for the Tees estuary were obtained, via the State of the Natural Environment of Teesside project (SONET), from Zeneca plc. Single faunal samples were taken using a 0.1 m^2 Smith–McIntyre grab at up to 30 sites in the main channel, with sampling of some additional sites in areas of particular interest (Shillabeer and Tapp, 1990). These samples were sieved through a 1 mm mesh, and fauna was enumerated and identified to species level where possible. Faunal data were obtained from a number of sites in September or early October, but not every site was sampled every year. Therefore, data for only 26 sites extending from the mouth to the upper estuary (sites 1–26; Fig. 1c) were included in the analysis. In 1971, 1976, 1981, and 1986, sediment samples were taken for metal analysis at 80 locations at the northern and southern sides of the channel throughout the middle and lower estuary and in Tees Bay using a conical dredge. Whole wet sediments were digested using an aqua regia mixture, followed by atomic absorption spectrophotometry analysis of Cd, Cu, Cr, Hg, Ni, Pb, and Zn (Taylor, 1974). For statistical analysis, sediment data from the north and south banks at any one location were pooled, giving 40 values for sites along the estuary (Fig. 1d).

Analyses

The mean abundance of individuals in all samples taken at any one station on a sampling occasion was used for analysis. Faunal data from all sites within an estuary were combined for assessing changes through time. To reduce between sampling error due to taxonomic inconsistencies, all data were combined at the genus level. Taxonomy followed Howson (1987).

To elucidate changes in community structure through time, cluster analysis and multi-dimensional scaling (MDS) was carried out using PRIMER 4.0β (Plymouth Marine Laboratory, Plymouth, UK), based on the Bray–Curtis similarity index (Bray and Curtis, 1957), on 4th root transformed data and group averaged clustering for plotting of dendrograms (Clarke and Warwick, 1994). Principal components analysis (PCA) was carried out on $\log(x+1)$ transformed faunal data using MVSP (Kovach Computing Services, Anglesey, UK). Analysis of variance of between-site and between-year differences in sediment concentrations of metals was performed using Excel 4.0, while regression analyses were carried out using Minitab Release 10 (Minitab Inc., Pennsylvania, USA).

Results and discussion

Estuaries tend to be areas of naturally low species diversity, and many of the taxa present are indicative of high levels of organic matter (Pearson and Rosenberg, 1978). The Tyne estuary proved to be no exception (Table 1). A total number of 66 taxa has been distinguished, but the majority of these were rare, with only a limited number found regularly. Six taxa—*Tubificoides* (44%), Nematodes (19%), *Capitella* (13%), Oligochaeta (11%), *Polydora* (6%), and *Tharyx* (4%)—accounted for 96% of the overall numbers of individuals in the Tyne estuary. The number of taxa recorded was variable, but tended to increase through time (Table 1).

Similarity dendrograms of the faunal data (Fig. 2a) show that two year groups (1979–1985 and 1986–1989) can be identified at the 50% similarity level, with 1980 taking up an aberrant position. An MDS plot of the same data (Fig. 2b) shows that 1980 was distinct from the other groups and that the community was more variable in the period 1986–1989. The change in the community was not reflected in the abundance of the six most dominant taxa (Table 1), but was due to changing densities of the rarer taxa, and to an increase in number of taxa present.

The classic model proposed by Pearson and Rosenberg (1978) predicts an increase in species diversity as levels of organic matter fall. This pattern is

Table 1. Abundances of dominant taxa, number of taxa present, and total abundances (per 0.1 m^2) at five stations in the Tyne estuary (Fig. 1), 1978–1989. Taxa identified by Pearson and Rosenberg (1978) as being abundant in organically polluted areas are indicated by an asterisk*.

Taxon	1978	1979	1980	1981	1983	1984	1985	1986	1988	1989
North Shields										
Tubificoides spp.*	49	32	11	26	0	0	0	6	28	12
Nematoda	490	35	0	2	55	1	3	0	100	1
Capitella spp.*	38	4	0	12	766	0	0	3	1	4
Oligochaeta indet.*	98	12	0	0	0	2	5	0	0	0
Polydora spp.*	561	25	0	0	0	0	0	0	0	0
Tharyx spp.	0	0	0	0	0	0	3	1	8	0
No. of taxa	21	16	12	13	15	13	22	14	17	28
Total abundance	1405	202	37	132	860	44	138	36	266	173
South Shields										
Tubificoides spp.*	81	115	1084	280	0	0	0	4913	846	79
Nematoda	288	7	2353	2907	1041	9	223	737	0	15
Capitella spp.*	50	28	731	260	62	11	108	1197	194	0
Oligochaeta indet.*	98	72	0	0	0	110	1028	0	0	0
Polydora spp.*	244	93	0	0	854	0	19	0	0	0
Tharyx spp.	0	0	0	0	0	0	0	2525	142	0
No. of taxa	13	14	8	18	12	9	13	14	6	16
Total abundance	893	416	4241	3818	2022	145	1459	9421	1230	152
Howdon										
Tubificoides spp.*	223	3463	2338	656	0	0	0	415	331	733
Nematoda	104	3253	470	815	237	449	81	15	2	175
Capitella spp.*	24	2336	195	21	23	26	28	8	76	43
Oligochaeta indet.*	0	0	0	0	820	563	1480	0	0	0
Polydora spp.*	26	464	0	2	212	6	37	0	0	0
Tharyx spp.	0	0	0	0	0	0	0	2	16	1
No. of taxa	10	6	10	14	17	16	14	19	19	17
Total abundance	416	9541	3019	1556	1349	1109	1739	528	496	1028
Quayside										
Tubificoides spp.*	64	103	329	371	0	0	0	15 778	11	158
Nematoda	10	15	0	12	19	71	10	152	2	24
Capitella spp.*	72	599	854	415	382	364	155	245	0	29
Oligochaeta indet.*	0	0	0	0	658	2394	790	0	0	0
Polydora spp.*	1465	164	0	6	138	24	15	251	0	0
Tharyx spp.	0	0	0	0	0	0	0	0	0	1
No. of taxa	5	7	4	6	6	8	6	5	4	19
Total abundance	1617	891	1194	813	1204	2862	976	16 432	17	304
Scotswood Bridge										
Tubificoides spp.*	0	2	0	0	0	0	0	2	0	283
Nematoda	0	0	0	1	2	0	0	0	0	1
Capitella spp.*	0	0	0	1	0	0	0	0	0	1
Oligochaeta indet.*	0	0	0	13	2	47	0	0	0	0
Polydora spp.*	0	0	0	0	0	0	0	0	0	12
Tharyx spp.	0	0	0	0	0	0	0	0	0	0
No. of taxa	2	2	0	4	3	3	0	2	2	12
Total abundance	2	4	0	19	6	49	0	3	1	351

present in data subsequent to 1986, which coincided with the interception of a large number of sewers on the south bank of the estuary. This last large single interception of effluents in the Tyneside Sewerage Scheme amounted to the equivalent of trade and domestic sewage generated by 75 000 people and potentially represents the beginning of the period of greatest estuarine recovery. Further changes in benthic infaunal quality are expected as a result of further interception of sewage. However, the sediment concentrations of organic chemicals in the Tyne, and also the Tees, are currently high enough to cause toxicity (Matthiessen *et al*., 1993), and are possibly responsible for the impoverished faunas observed.

In the Tees estuary, concentrations of Cd, Cu, Cr, Hg, Ni, Pb, and Zn differed significantly between sites in all years, with generally lower sediment concentrations of metals in the upper and very high concentrations in the lower estuary. Concentrations of Cd, Cu, and Zn (Fig. 3) are of particular concern to monitoring groups (e.g. North Sea Task Force, 1993). Concentrations in the sediment differed significantly between the years

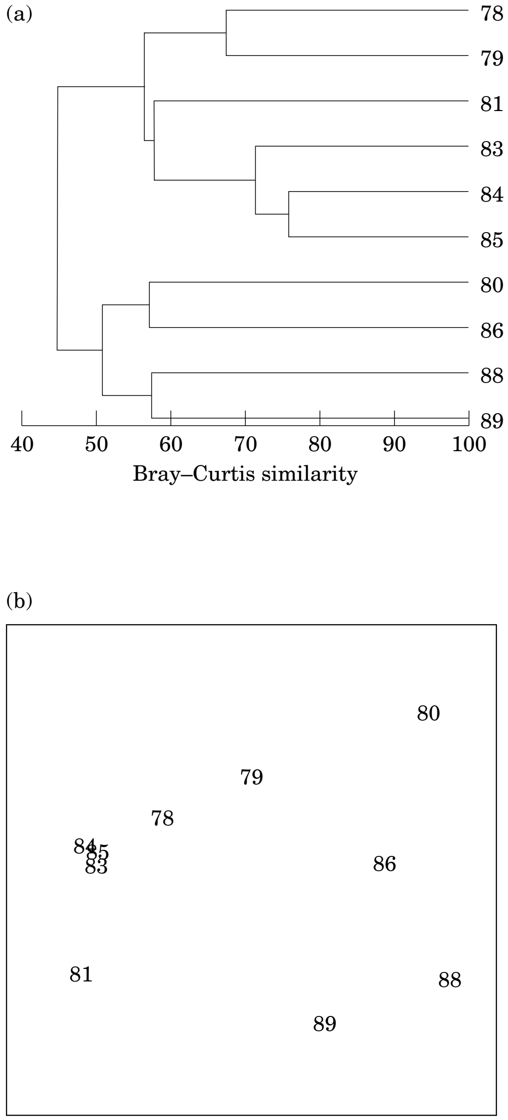


Figure 2. Bray–Curtis similarity indices of $\sqrt{\sqrt{}}$-transformed abundances of Tyne estuary fauna across year of sampling (five stations combined). (a) Dendrogram using group averaged clustering. (b) MDS ordination (stress value=0.07).

1971–1986 (ANOVA $p<0.05$), and decreased for all metals over this period (Tukey test $p<0.05$).

The fauna in the Tees estuary (107 genera) was more diverse than that of the Tyne, with 95% of all individuals belonging to 13 taxa (Table 2). This may be partially explained by the fact that, in the Tees estuary, sampling was across a greater salinity and contamination gradient than in the Tyne estuary. The four most common taxa also appeared in the list of commonest taxa for the Tyne (*Capitella* 39%, *Polydora* 21%, Oligochaeta 19%, *Tubificoides* 6%).

Using cluster analysis, the sites formed five faunal groups at the 50% similarity level (Fig. 4a). These can be attributed to the gradients of salinity and contamination present within the estuary. Group C stations are all near the mouth, where there is lower contamination and higher salinity. Group E is in the less contaminated upper estuary, and groups A, B, and D are in the central parts, with D being more contaminated than A and B. Other studies of the Tees have also identified groups based on these variables, e.g. Shillabeer and Tapp (1990) and Tapp *et al.* (1993). Separation of groups was caused by *Capitella*, the most abundant species present. However, different numbers of groups have been described in previous studies. Examining data from the 1970s and early 1980s, Shillabeer and Tapp (1990) described only three groups, and suggested that grouping was controlled by densities of the polychaete *Spiophanes bombyx*. Analysis of data from the early 1990s by Ashman and Shillabeer (pers. comm.) gave five groups for 1991 and six groups in 1992. These groups were defined in terms of species abundance, and also affinity of the species for either marine or estuarine conditions.

To assess changes in benthic faunal composition through time, MDS was performed on data from two sites in the most contaminated part, Teesport and Wilton (sites 6 and 8, respectively; Fig. 1c) and a site upstream of these areas where contamination was lower (site 11; Fig. 1c). The MDS plot indicates that changes at the Wilton and Teesport sites between 1986 and 1992 were greater than at the less contaminated site (Fig. 4b). Tapp *et al.* (1993) recorded improvements in benthic infaunal communities in the period 1979–1985, and this study shows that these have continued up to at least 1992.

Under PCA of faunal data, sites in the lower estuary adjacent to Teesport and the numerous effluent discharges were found to have higher PC1 scores than either marine sites or those upstream of the most contaminated areas (Fig. 5). There was no significant correlation between PC1 scores and gradients of concentrations of any particular metal (Pearson product–moment correlation coefficient $r<0.264$; $p>0.05$ for all metals), though gradients of metal contamination did exist (Fig. 3). A lack of significant correlations between crude determination of concentrations of individual metals and the community structure as given by PCA scores is not surprising, since these measurements do not take into account possible synergistic or antagonistic toxicity effects of the different metals. The bioavailability of metals in estuarine sediments is also highly variable (Luoma, 1983), depending upon not only the concentration of the metal, but also the salinity of the overlying water column, temperature (McLusky *et al.*, 1986), the organic matter concentration of the sediment, and the redox state (Luoma *et al.*, 1995). Such data are not available for the Tyne or Tees estuaries for the

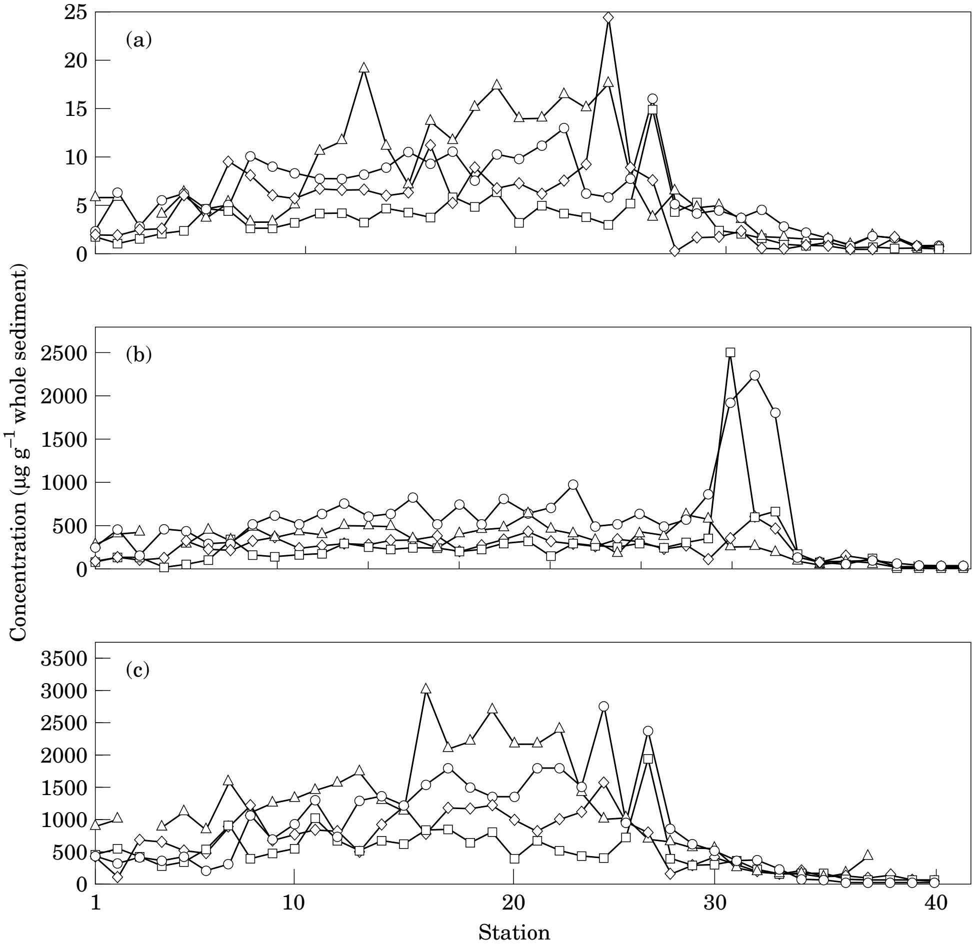


Figure 3. Tees estuary whole fraction wet sediment concentrations sampled in 1971 (○), 1976 (△), 1981 (◇), and 1986 (□) of: (a) Cadmium. (b) Copper. (c) Zinc.

period under consideration, but would be valuable in future monitoring work.

An alternative approach to establishing a relationship between sediment quality and benthic fauna is by using a tripartite method (discussed in Chapman *et al.*, 1992), using sediment toxicity in addition to benthic community and sediment contaminant data. Such observations have been proven to be effective in many studies (e.g. Chapman and Long, 1982; Hall and Frid, in press).

Benthic monitoring, similar to that undertaken in the Tyne and Tees, has been of considerable value in assessing the effectiveness of remediation programmes (Gray *et al.*, 1991b). With the advent of more sophisticated chemical analysis methods, the value of faunal sampling through time may be underestimated. The response of many species to contamination is well described (e.g. Pearson and Rosenberg, 1978; Pocklington and Wells, 1993), and even simple present/absence data, or species lists, may suffice to allow determination of sediment quality, as opposed to taking numerous chemical assay samples. Some such lists form a historic record stretching back to before the start of the 20th century (e.g. Alexander *et al.*, 1935), pre-dating even the simplest analyses of environmental concentrations of many toxic substances. There remains scope for incorporating such data sets with contemporary studies, although this raises the issue of comparability of samples.

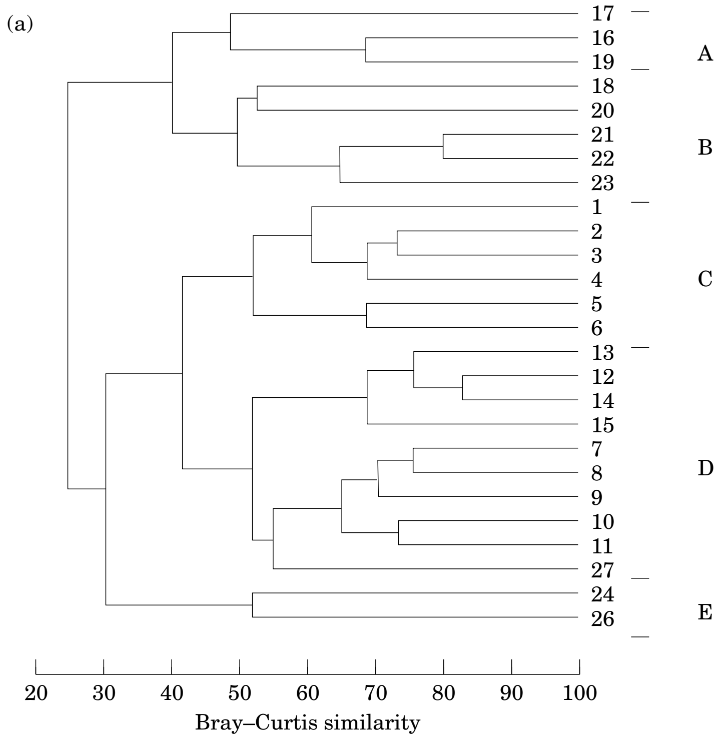
During the monitoring periods analysed here, methods were kept the same. However, methods used to sample Tyne estuary benthos have changed since 1992. In such cases it is of primary importance to establish whether changes in methodologies have biased the results. Comparison is needed between, for example, grab types (e.g. Gallardo, 1955) and changes in sieve mesh sizes used (Rees, 1984). Even changes in personnel potentially give variability in identification (MAFF, 1993). To overcome the problem at species level, combination at genus level is possible, as in this study, but

Table 2. Abundances of dominant taxa, number of taxa present, and total abundances (per 0.1 m^2) in regions identified as being dissimilar in the Tees estuary (Figure 4a), 1981–1992. (See also Table 1.)

Year	81	82	83	84	85	91	92	81	82	83	84	85	91	92
					Region A						Region B			
Capitella spp.*	1	0	1	6	29	1	4	0	0	0	0	0	0	1
Polydora spp.*	0	0	0	6	12	1	1	0	0	0	0	56	0	2
Oligochaeta indet.*	0	0	0	1	1	1	1	6	0	0	6	0	1	0
Tubificoides spp.*	0	0	0	0	0	0	0	0	0	0	0	0	0	4
*Malacoceros ful.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0
*Hydrobia ulvae**	0	0	0	0	0	0	7	0	0	0	0	0	0	1
Cirriformia tent.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nephtys spp.*	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Nematoda	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Mytilus spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Abra abra		0	0	0	0	0	0	0	0	0	0	0	0	0
Phyllodoce spp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
*Spio filicorni**	0	0	0	0	0	0	0	0	0	0	0	0	0	0
No. of taxa	1	1	2	3	4	4	5	4	2	1	4	6	2	5
Total abundance	1	0	1	12	42	3	14	7	1	0	7	57	1	7

				Region C							Region D			
Capitella spp.*	1	2	25	1	10	0	3	193	189	126	378	384	79	328
Polydora spp.*	49	2	12	2	292	0	2	168	10	117	24	68	0	166
Oligochaeta indet.*	0	1	0	6	2	1	1	23	238	34	15	152	16	23
Tubificoides spp.*	1	6	4	5	36	9	8	8	88	5	9	41	6	46
*Malacoceros ful.**	0	0	1	0	0	0	0	6	3	15	5	10	24	90
*Hydrobia ulvae**	1	0	0	1	0	0	1	2	0	1	2	1	26	85
Cirriformia tent.	1	1	0	1	0	1	0	1	22	1	2	7	44	11
Nephtys spp.*	14	14	18	2	7	9	12	0	1	0	1	1	1	1
Nematoda	2	4	4	2	27	0	0	0	10	1	0	8	2	1
Mytilus spp.	1	1	13	24	2	1	0	0	0	0	15	0	0	0
Abra abra	2	0	2	1	0	19	33	0	0	0	0	0	1	0
Phyllodoce spp.	4	4	5	1	10	2	0	3	7	3	1	1	3	6
*Spio filicorni**	34	3	1	0	2	0	6	0	0	1	0	1	0	0
No. of taxa	37	55	44	33	64	53	54	20	21	24	17	24	30	35
Total abundance	132	84	116	55	456	77	129	408	573	311	453	679	214	775

				Region E			
Capitella spp.*	1	1	0	1	1	3	0
Polydora spp.*	0	0	0	0	0	0	0
Oligochaeta indet.*	1	63	3	1	1	63	3
Tubificoides spp.*	0	0	0	0	0	4	4
*Malacoceros ful.**	0	0	0	0	0	0	0
*Hydrobia ulvae**	0	0	0	0	0	0	1
Cirriformia tent.	0	0	0	0	0	0	0
Nephtys spp.*	0	0	0	0	0	3	0
Nematoda	0	0	0	0	0	0	0
Mytilus spp.	0	0	0	0	0	0	0
Abra abra	0	0	0	0	0	0	0
Phyllodoce spp.	0	0	0	0	0	0	0
*Spio filicorni**	0	0	0	0	0	0	0
No. of taxa	3	2	1	2	2	7	4
Total abundance	3	64	3	2	2	73	9



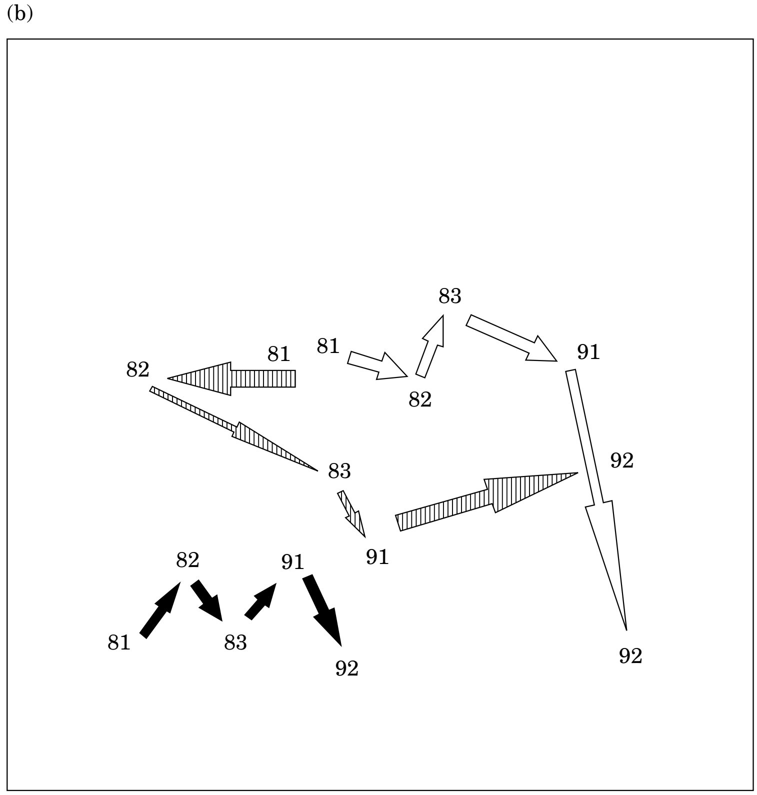


Figure 4. Bray–Curtis similarity indices of √√-transformed faunal abundances of Tees estuary fauna. (a) Dendrogram using group averaged clustering across 26 stations (years combined) with the five groups separated at a 50% threshold indicated (A–E). (b) MDS plots of abundances at contaminated (Teesport (hatched arrow), Wilton (open arrow) and a cleaner (black arrow) station in the lower estuary during the period 1981–1992 (stress value=0.14).

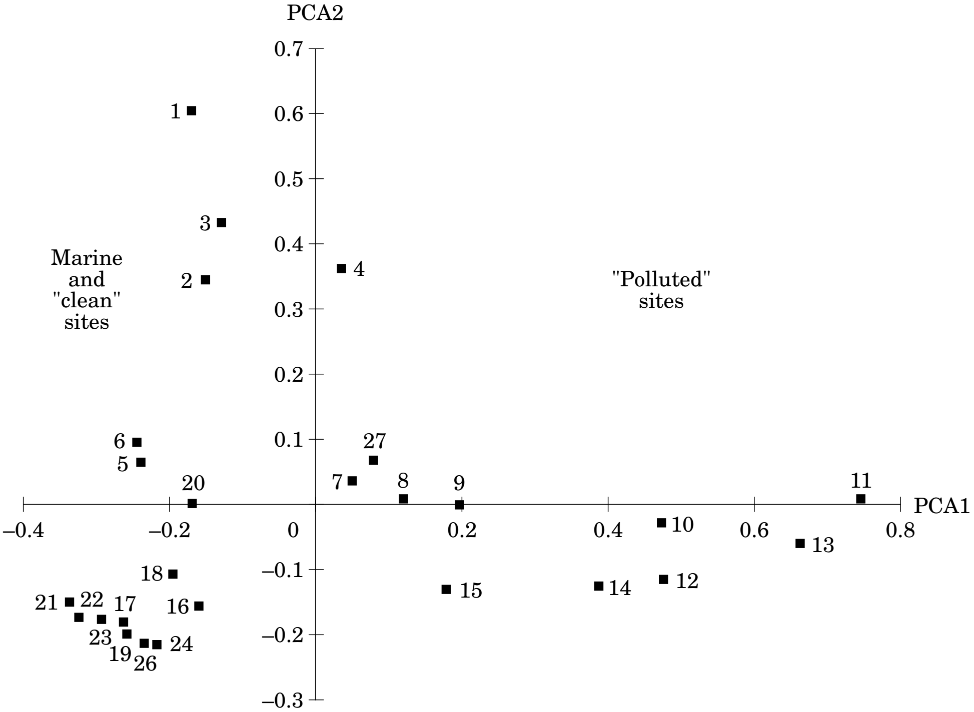


Figure 5. PCA plots of log(x+1) transformed data for Tees estuary fauna across 26 stations (years combined), PC axis 1 and 2 explaining 43.8% and 20.5% of the variation, respectively.

combination of taxa to as high as family level, or phylum has also been advocated (Warwick, 1988).

This study has identified changes in the benthic infauna of two heavily industrialized North Sea estuaries over a 16-year period. Although data on sediment and water column quality are not easily interpreted, the communities in both estuaries appear to have responded positively to large reductions in the inputs of contaminants. In the Tees estuary, this is reflected in the extent to which communities in the most contaminated part have become more like to those present in less contaminated areas (Fig. 4). In the Tyne, abundances and number of species increased, particularly in the upper estuary. These observations are in accord with the predictions of Pearson and Rosenberg's (1978) model.

Environmental improvement measures have been implemented by Water Authorities, more latterly by water companies and the Environment Agency, driven by a variety of national and international initiatives designed to protect the health of the aquatic environment. It is only through direct measurements of the biota that the success of such strategies can be fully assessed. In spite of the intermittent nature of some of the sampling the application of appropriate procedures allows temporal comparisons to be made. While considerable improvement in the benthic faunas has been achieved, they must still be considered as impoverished and there remains scope for further environmental remediation (Matthiessen *et al.*, 1993). However, this may possibly be hindered by slow leaching of contaminants from the sediments and further action to remediate sediments may be worthwhile, making dredging of contaminated areas worthy of consideration.

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References

Alexander, W. B., Southgate, B. A., and Bassindale, R. 1935. Survey of the River Tees. Part II. The estuary – chemical and biological. Department of Scientific and Industrial Research. Water Pollution Research. Technical Paper No. 5. HMSO, London. 171 pp.

Bray, J. R., and Curtis, J. T. 1957. An ordination of the upland forest communities of southern Wisconsin. Ecological Monographs, 27: 325–349.

Chapman, P. M., and Long, E. R. 1983. The use of bioassays as part of a comprehensive approach to marine pollution assessment. Marine Pollution Bulletin, 4: 81–84.

Chapman, P. M., Power, E. A., and Burton, G. A. 1992. Integrative assessments in aquatic ecosystems. *In* Sediment toxicity assessment, pp. 313–339. Ed. by G. A. Burton. Lewis Publishing, Boca Raton, Florida, USA. 459 pp.

Clark, R. B. 1992. Marine pollution. Clarendon Press, Oxford. 172 pp.

Clarke, K. R., and Warwick, R. M. 1994. Changes in marine communities: an approach to statistical analysis and interpretation. Natural Environment Research Council, UK. 144 pp.

Gallardo, V. A. 1955. Observations on the biting profiles of three 0.1 m^2 bottom-samplers. Ophelia, 2: 319–322.

Gray, J. S., Calamari, D., Duce, R., Portmann, J. R., Wells, P. G., and Windom, H. L. 1991a. Scientifically based strategies for marine environmental protection and management. Marine Pollution Bulletin, 22: 432–440.

Gray, J. S., McIntyre, A. D., and Stirn, J. 1991b. Manual of methods in aquatic environment research. Part II. Biological assessment of marine pollution with particular reference to benthos. FAO Fisheries Technical Paper No. 324. FAO, Rome, 1991. 49 pp.

Hall, J. A., and Frid, C. L. J. in press. Benthic community structure and sediment heavy metal concentrations in two estuaries in N.E. England. Transactions of the Northumberland Natural History Society.

Howson, C. M. 1987. Directory of British marine Flora and Fauna. Marine Conservation Society, Ross-on-Wye, UK. 471 pp.

Kröncke, I. 1987. Lead and cadmium contents in selected macrofauna species from the Dogger Bank and the eastern North Sea. Helgoländer Meeresuntersuchungen, 41: 465–475.

Lewis, R. E. 1989. The nature of outflows from the North East estuaries. Hydrobiologia, 195: 1–11.

Le Guillou, M. 1978. A history of the River Tees. Cleveland County Libraries, Middlesbrough, UK. 182 pp.

Luoma, S. N. 1983. Bioavailability of trace metals to aquatic organisms: a review. Science of the Total Environment, 28: 1–22.

Luoma, S. N., Ho, Y. B., and Bryan, G. W. 1995. Fate bioavailability and toxicity of silver in estuarine environments. Marine Pollution Bulletin, 31: 44–54.

MAFF. 1992. Monitoring and surveillance of non-radioactive contaminants in the aquatic environment and activities regulating the disposal of wastes at sea, 1990. Directorate of Fisheries Research, Lowestoft, UK. 66 pp.

MAFF. 1993. Analysis and interpretation of benthic community data at sewage sludge disposal sites. Aquatic Environment Monitoring Report Number 37, Ministry of Agriculture, Fisheries and Food, Lowestoft, UK.

Matthiessen, P., Thain, J. E., Law, R. J., and Fileman, T. W. 1993. Attempts to assess the environmental hazard posed by complex mixtures of organic chemicals in estuaries. Marine Pollution Bulletin, 26: 90–95.

McLusky, D. S., Bryant, V., and Campbell, R. 1986. The effects of temperature and salinity on the toxicity of heavy metals to marine and estuarine invertebrates. Oceanography and Marine Biology: An Annual Review, 24: 481–520.

Middlebrook, S. 1950. Newcastle upon Tyne – its growth and achievement. Newcastle Chronicle and Journal, Newcastle upon Tyne, UK. 361 pp.

NRA. 1994a. River Tees catchment management plan consultation report. NRA Northumbria and Yorkshire region, Darlington. 112 pp.

NRA. 1994b. River Tyne catchment management plan consultation report. NRA Northumbria and Yorkshire region, Gosforth, UK. 71 pp.

North Sea Task Force 1993. North Sea Quality Status Report 1993. Oslo and Paris Commissions, London. Olsen and Olsen, Fredensborg, Denmark. 132 pp.

Pearson, T. H., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology: An Annual Review, 16: 229–311.

Pocklington, P., and Wells, P. G. 1992. Polychaetes. Key taxa for marine environmental quality monitoring. Marine Pollution Bulletin, 24: 593–598.

Rees, H. L. 1984. A note on mesh selection and sampling efficiency in benthic studies. Marine Pollution Bulletin, 15: 225–229.

Shillabeer, N., and Tapp, J. F. 1990. Long-term studies of the benthic biology of Tees Bay and the Tees Estuary. Hydrobiologia, 195: 63–78.

Sinton, A. J. W. 1978. Studies of benthic macroinvertebrates and sediments of the Tyne estuary in relation to the Tyneside Sewerage Scheme – Preliminary investigations. BSc Report, University of Salford. 124 pp.

Tapp, J. F., Shillabeer, N., and Ashman, C. M. 1993. Continued observations of the benthic fauna of the industrialised Tees estuary. Journal of Experimental Marine Biology and Ecology, 172: 67–80.

Taylor, D. 1974. Natural distributions of trace metals in sediments from a coastal environment. Tor Bay, England. Estuarine and Coastal Marine Science, 2: 417.

Topping, G. 1983. Guidelines for the use of biological material in first order pollution assessment and trend monitoring. Department of Agriculture and Fisheries for Scotland. Aberdeen Marine Laboratory. 28 pp.

Warwick, R. M. 1988. Analysis of community attributes of the macrobenthos of Frierfjord/Langesfjord at taxonomic levels higher than species. Marine Ecology Progress Series, 46: 167–170.