

University of New Hampshire

University of New Hampshire Scholars' Repository

Jackson Estuarine Laboratory

Institute for the Study of Earth, Oceans, and
Space (EOS)

8-8-1991

Effects of organic enrichment on estuarine macrofaunal benthos: a comparison of sediment profile imaging and traditional data

Raymond E. Grizzle

University of New Hampshire, ray.grizzle@unh.edu

Clayton A. Penniman

Narragansett Bay Project

Follow this and additional works at: <https://scholars.unh.edu/jel>

Recommended Citation

Grizzle, R.E. and C.A. Penniman. 1991. Effects of organic enrichment on estuarine macrofaunal benthos: a comparison of sediment profile imaging and traditional data. *Marine Ecology Progress Series* 74:249-262.

This Article is brought to you for free and open access by the Institute for the Study of Earth, Oceans, and Space (EOS) at University of New Hampshire Scholars' Repository. It has been accepted for inclusion in Jackson Estuarine Laboratory by an authorized administrator of University of New Hampshire Scholars' Repository. For more information, please contact nicole.hentz@unh.edu.

Effects of organic enrichment on estuarine macrofaunal benthos: a comparison of sediment profile imaging and traditional methods

Raymond E. Grizzle¹, Clayton A. Penniman²

¹ Natural Sciences & Mathematics, Livingston University, Livingston, Alabama 35470, USA

² Narragansett Bay Project, 291 Promenade Street, Providence, Rhode Island 02908-5767, USA

ABSTRACT: Sediment profile imaging (SPI) is a technique that involves collecting data in the field on sedimentary conditions, particularly vertical variations at and below the sediment/water interface. SPI approaches were developed nearly 20 yr ago, but have not been widely used in routine pollution monitoring efforts, although their potential for saving time and money is great. The present report is a comparison of SPI and 'traditional' (which involves excising a sediment sample, removing the organisms, and identifying and further processing them in the laboratory) benthic sampling techniques used along an organic enrichment gradient in a temperate coastal lagoon. Twenty-four core samples were taken in summer 1985 along an enrichment gradient mainly resulting from the discharge of sewage effluents, and processed by traditional macrofaunal benthic methods. Coincidental with these cores, sediment samples were taken for various physicochemical analyses, and optically clear acrylic box core samples were taken and analyzed in the field for changes in redox potential, dominant benthic taxa and their relative abundances, and sedimentary features. Various univariate and multivariate analyses of the benthic and sedimentary data showed substantial, and expected, changes along the pollution gradient. The changes were similar to those reported from other coastal waters affected by organic wastes: an area nearest the pollution source with lowest taxa numbers (S), abundances (A) and biomass (B); and increased S, A and B with increasing distance from the pollution source. Small near-surface polychaetes (capitellids and *Streblospio benedicti*) and the amphipod *Ampelisca* sp. were numerical dominants at all sites, and large, deep-burrowing species (e.g. *Asychis elongata*) were only found at the least-enriched sites. SPI allowed identification of the dominant benthic taxa to useful levels (e.g. 'small near-surface dwelling annelids', 'large deep-burrowing polychaetes'), and location of the apparent color redox potential discontinuity (RPD) layer depth. The RPD ranged from about 1 cm at the site nearest the pollution source to ≥ 4 cm at the 2 sites farthest away. The SPI data were as useful as the traditional data in delimiting the spatial extent of the pollution-impacted benthos. It is suggested that a variety of coring devices presently exist for quickly and efficiently obtaining SPI data, especially in shallow waters.

INTRODUCTION

Pollution surveys in coastal waters historically have relied heavily on benthic data collected and processed by 'traditional' methods involving removal of sediment samples, separation of organisms from the sediment, identification of species present, and various numerical analyses of the resulting data. But these traditional methods are very time-consuming and expensive, and have been dropped from some monitoring programs because of cost considerations (Bilyard 1987). In response to the need for improved benthic sampling and data assessment methods, particularly for survey-type work, D.C. Rhoads and colleagues designed and

developed a sediment profile camera for obtaining benthic data in the field (Rhoads & Cande 1971, Rhoads et al. 1981, Rhoads & Germano 1982, 1986). Their approach to sediment profile imaging (SPI) involves an apparatus which pushes an optical prism with camera into the sediment where photographs (or video images) of the sediment/water profile are taken. This approach appears to represent a substantial advancement in benthic sampling, especially for pollution assessment work, yet it has not been widely used.

Rhoads & Germano (1982) suggested that their approach had not gained wide acceptance primarily because of lack of an adequate theoretical basis for interpretation of the sediment profile images. They

proposed a successional model (similar to the enrichment model of Pearson & Rosenberg 1978 discussed below) as this theoretical basis, and described the basic rationale and methods used in gathering and interpreting SPI data. Nonetheless, to date very little SPI data have appeared in the published literature. Only a few researchers own an in situ sediment profile imaging device, and most have not published (in refereed journals) on their studies. The results of these studies are found mainly in technical reports (see 'Discussion') which are available from the agencies funding the work.

Exceptions to this generalization include some freshwater studies (e.g. Boyer & Shen 1988, Boyer & Hedrick 1989), and 2 recent papers on organic enrichment in coastal waters (O'Connor et al. 1989, Krieger et al. 1991). O'Connor et al. (1989) demonstrated the potential for the use of SPI in quickly estimating the spatial extent of organic enrichment from mariculture operations in a coastal embayment, but they provided only a brief comparison of SPI data with samples analyzed by traditional techniques. They concluded that SPI is a powerful approach for pollution investigations in organically enriched areas. Krieger et al. (1991) used SPI data in a geographic information system to produce maps of Narragansett Bay (Rhode Island, USA) related to degradation by pollutants. Nonetheless, we suggest that much more comparative research in different types of polluted environments, involving SPI data and benthic data collected using traditional methods, is needed for full development of the approach. Furthermore, at least in the initial stages of the development and verification of this approach for different types of polluted areas, there may be a need to develop ways to collect SPI data other than with an in situ sediment profile imaging camera because of its size, initial cost, and the fact that it does not remove a sample for processing by traditional techniques.

Herein we compare SPI data collected with an acrylic corer to macrofaunal and physicochemical data obtained by traditional methods, from an organic enrichment gradient in a temperate coastal lagoon. Our study showed that SPI data alone could have been used to delimit the extent of pollution-induced macrofaunal benthic changes along the pollution gradient, and that such data can be collected quickly in shallow waters using inexpensive hand-held coring devices.

MATERIALS AND METHODS

Study area. Great Sound is a high-salinity coastal lagoon in southern New Jersey, USA (Fig. 1). Detailed environmental data on the Sound are presented in a special volume of 'Marine Geology' (Vol. 82, No. 1/2,

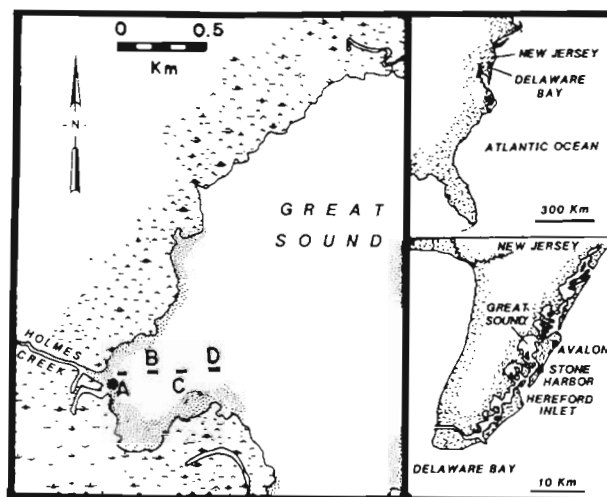


Fig. 1. Study area in Great Sound, New Jersey, USA. The shape, size and orientation of the boxes at each of the 4 study sites represent the actual areas from which samples were taken. Dot near Site A shows location of Royer's (1980) Site ST1 (see text). Stippled areas are intertidal

1988) devoted to hydrodynamical and sedimentological studies in the lagoon-salt marsh system. Briefly, Great Sound has a tidal range about 1.6 m, and most areas are < 1.0 m deep at low water. There is little constant freshwater discharge to the lagoon because of its small drainage basin. The marsh and associated tributary streams extend only a few kilometers from the lagoon proper and a narrow barrier island separates the lagoon from the Atlantic Ocean. The Sound is connected to the ocean by 2 large (5 km and 10 km long) tidal channels which enter at opposite ends. Hence, salinities approach full-strength seawater in most areas most of the time, with means near 30 ppt (Royer 1980, Ashley & Grizzle 1988). Water temperatures show seasonal fluctuations typical of temperate estuaries, ranging from about 0 to 30 °C (Allen et al. 1978, Royer 1980).

The present study was done in a cove off the mouth of Holmes Creek in the southwest corner of Great Sound (Fig. 1). A 2 yr benthic study done in the late 1970's showed that a site in the cove area near our Site A (see Fig. 1) had benthic communities typical of organically enriched coastal waters, and that benthic communities farther north of the cove area were not as affected (Royer 1980). For 10 yr prior to Royer's study (1969 to 1979), Holmes Creek received the effluent of a 190 000 l d⁻¹ (50 000 gal d⁻¹) wastewater treatment plant providing secondary treatment via contact stabilization followed by chlorination (Royer 1980, Tom Vernam, New Jersey Department of Environmental Protection, pers. comm.). During the present study Holmes Creek received runoff from various types of land uses, and reportedly received overflows from enriched fresh-

water ponds (Tom Vernam pers. comm.). Water column chlorophyll *a* data collected in summer and fall 1985 as part of another study from Site D averaged $17.7 \mu\text{g l}^{-1}$, and were substantially higher than at 4 other sites in the lagoon system; but sediment total organic matter (TOM) levels were similar to other sites in Great Sound with muddy sediments (Grizzle 1988; also see data in Grizzle & Lutz 1989). Hence, the Holmes Creek area of Great Sound was enriched relative to other areas of the lagoon when samples were collected for the present study, and had probably been enriched for at least the previous 15 yr. It should also be noted that the above data on water column chlorophyll concentrations suggest that Site D was probably not in an 'unaffected' area; i.e. our transect did not include a complete range

of sites based on relative enrichment for all of Great Sound.

Field methods. Duplicate samples of the upper 2 to 3 cm of sediment for various physicochemical analyses (see below) were collected (and composited in some cases) from each of 4 sites (Fig. 1) in August and September 1985 using a 5 cm ID PVC piston corer. These samples were frozen until analyzed, except for the chlorophyll samples which were freeze-dried on the day of collection. Temperature, salinity and dissolved oxygen in the water column were measured 10 cm above the bottom using meters on one day in August and one in September.

Benthos/sediment samples were collected coincidentally with the above sediment samples along a transect

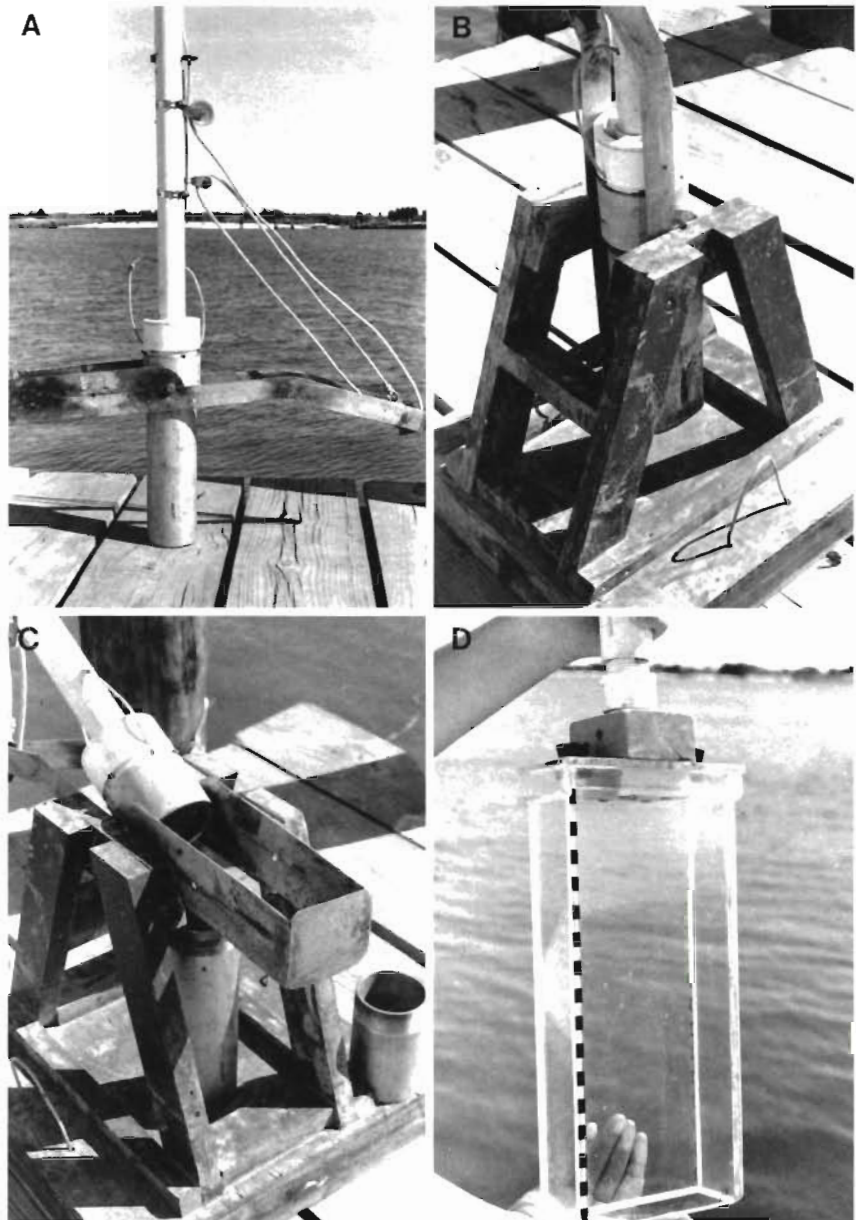


Fig. 2. Spade corer (A to C) and acrylic box corer (D). Acrylic corer is flow-through with hole (not shown) at base of outer PVC pipe at top; hole is sealed after insertion of box core into bottom by inner (not shown) PVC pipe with rubber stopper attached to end. Bottom of corer is sealed with rubber block before removal of corer from the water. Rubber stoppers on top plate are for insertion of redox probe

extending outward from the mouth of Holmes Creek (Fig. 1). Twenty-four benthos/sediment samples were collected in August with a custom-made corer similar in design to a spade box corer with a 10 cm ID by 40 cm long stainless steel core tube (Fig. 2A, B, C). Six core samples were taken at each of 4 sites (A, B, C and D; Fig. 1). The samples were taken at randomly chosen points along a marked, 50 m rope positioned in-line with the overall sampling transect at each site. Each sample was placed in a plastic bag and 37 % buffered formaldehyde with rose bengal was added as a fixative. The fixed samples were later washed on a 0.5 mm mesh sieve and the residue was preserved in 70 % isopropanol.

Coincidentally with the above benthic cores, 2 to 4 acrylic box core samples (inside dimensions: 4 × 12 × 30 cm long; Fig. 2D) were taken at each site. In most cases only one core from each site was fully analyzed in the field for sediment profile imaging (SPI) data following the methods described in Rhoads & Germano (1982). Box cores for SPI data also were collected in September. Redox potential (Eh) measurements were made directly using a probe with an Ag/AgCl reference system and platinum indicating element on one core sample (with duplicate profiles taken on each core) from each site in August and September following the methods in Pearson & Stanley (1979).

Laboratory methods. Sediment samples were analyzed for grain size distribution using an Elzone Analyzer (particle counter) after digestion of the organic matter with concentrated hydrogen peroxide. Sediment total organic matter was determined by ashing at 475 °C (Byers et al. 1978). Sediment chlorophyll *a* was determined spectrophotometrically (Parsons et al. 1984) after extraction from freeze-dried samples with 90 % alkaline acetone. Benthic faunal samples were sorted under 2× magnification, identified to species where possible, counted and weighed.

Data analysis methods for benthos.

Univariate analyses: One-way ANOVAs followed by Duncan's multiple-range tests were done on the mean numbers of taxa ('S'), and the mean abundances of individuals ('A') collected per core (0.008 m²) for the total benthic community. These means, and biomass (alcohol-preserved wet weights; 'B') per core (the replicate cores were composited before weighing so no statistical analyses were possible for the biomass data) were also plotted by site. *K*-dominance curves (Warwick 1986, Gray et al. 1988), which show cumulative abundance and biomass patterns, were calculated for each of the 4 sites.

Multivariate analyses: Classification analysis was done in order to determine the degree of similarity among each of all possible pairs of core samples with respect to the abundances of the taxa collected, and to arrange the entire set of samples into a hierarchy of increasing

similarity. To accomplish this, a matrix was computed using Pearson product-moment correlation coefficients of log(*x*+1) transformed abundances for all possible pairwise comparisons of core samples. Clustering of the data in this matrix was done using unweighted pair-group average (UPGMA) linkage, and a dendrogram was produced using CLUSTAN (Wishart 1987, Clarke & Green 1988).

Principal components analysis (PCA) was used on log(*x*+1) transformed abundance data to ordinate core samples in multi-dimensional space according to their relative similarities. The PCA plots were scaled by size using the sediment TOM and chlorophyll data to show how the sediment characteristics were related to the spatial arrangement (Clarke & Green 1988).

RESULTS

Physicochemical data

Although sufficient replicates were not available for a meaningful statistical analysis, the sediment data

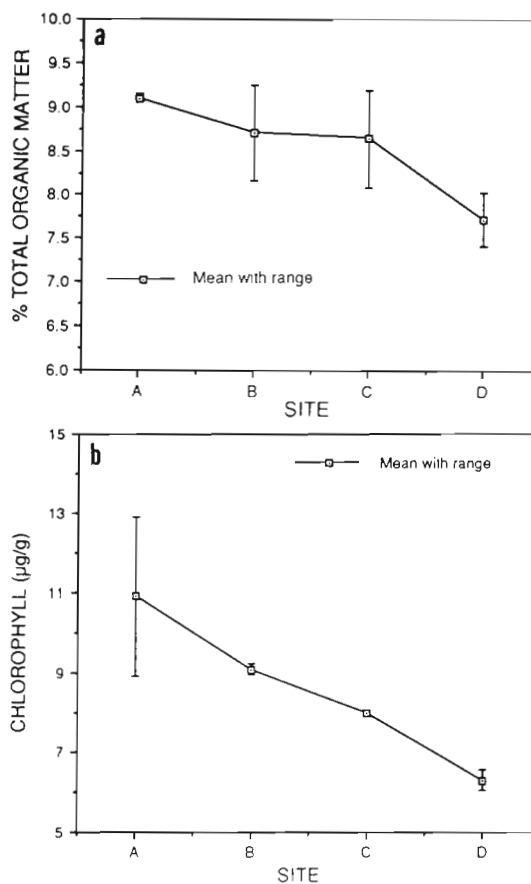


Fig. 3. Sediment characteristics from samples taken in August and September 1985; bars show range. (a) Mean (*n* = 2) TOM concentrations in % dry weight. (b) Mean (*n* = 2) chlorophyll *a* concentrations in µg g⁻¹ dry weight

Table 1. Grain size distribution in volume percent for one core sample from each site

Site	Grain size (μm)							
	> 125	> 63	> 32	> 16	> 8	> 4	> 2	> 1
A	0.0	0.5	6.3	21.6	24.6	17.3	15.1	14.6
B	0.0	1.1	6.3	19.2	23.7	18.5	14.7	16.5
C	0.2	2.2	7.7	20.1	22.2	16.4	14.8	16.4
D	0.9	2.7	3.8	8.6	12.1	15.1	17.0	39.8

indicated an enrichment gradient extending along the sampling transect, with Site A being most enriched and Site D least. Mean (from 2 samples per site) total organic matter (TOM) concentrations ranged from 9.1 % at Site A to 7.7 % at Site D, with Sites B and C intermediate between these values (Fig. 3a). Sediment chlorophyll *a* concentrations ranged from a mean of $10.9 \mu\text{g g}^{-1}$ dry wt at Site A to $6.3 \mu\text{g g}^{-1}$ at Site D, with Sites B and C intermediate (Fig. 3b). Grain size distributions showed no trends along the gradient. Sites A, B and C were very similar for all size classes (Table 1). Site D was similar to the other 3 for most size classes, but had a much greater percentage of clay-size particles than the other sites. In the present study, any trends in differences in grain size among the study sites would have been considered a 'nuisance' variable (see Clarke & Green 1988) with respect to interpretation of benthic changes along the pollution gradient because benthic community changes can be caused by sediment grain size changes.

Sediment redox potentials also showed strong trends related to the enrichment gradient. The rate of decrease in Eh values with increase in sediment depth was greatest at Sites A and B, and least at Site C (Fig. 4). The relative depths of the apparent color redox potential discontinuity (RPD) layers at the 4 sites compared well with the probe measurements, and indi-

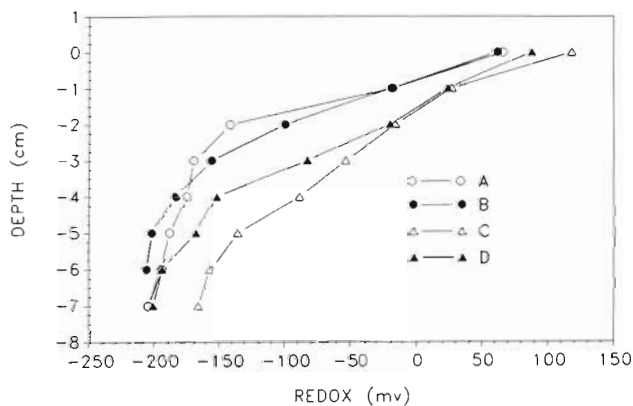


Fig. 4. Mean values of redox (Eh) measurements made by probe on one core sample (duplicate profiles on each core) per site in August and September 1985

cated that the apparent color RPD layer occurred between -110 and -150 mV (cf. Figs. 4 & 5). Redox potentials are related to changes in benthic communities, and interpretation of these trends is discussed in more detail below.

At the temporal scale measured, water column characteristics were not expected to show trends relative to the pollution gradient, and they did not (Table 2). If substantial differences in temperature or salinity had been indicated they would have been considered 'nuisance' variables, as mentioned above for sediment grain size. Over all measurements, temperature ranged from 25.0 to 27.5°C , and salinity

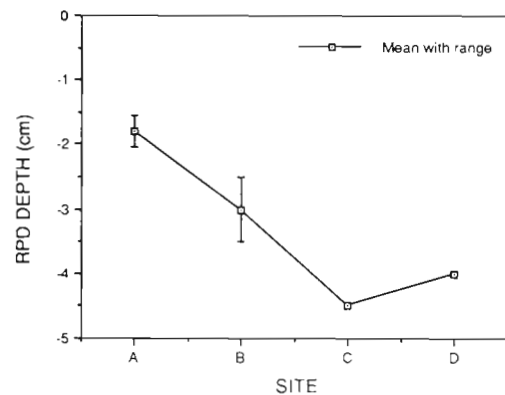


Fig. 5. Depth of apparent color RPD layer as determined by color changes in one acrylic box core sample per site in August and September 1985. Bars show range

Table 2. Near-bottom water parameters. Temp.: temperature in $^\circ\text{C}$; Sal.: salinity in ppt; DO: dissolved oxygen in mg l^{-1}

Site	Date	Temp.	Sal.	DO
A	Aug	27.0	27.5	5.4
	Sep	25.5	29.5	7.2
B	Aug	27.0	29.0	5.6
	Sep	25.5	30.5	6.9
C	Aug	27.5	29.5	5.0
	Sep	25.5	30.5	6.6
D	Aug	27.0	29.5	2.8
	Sep	25.0	30.5	6.5

Table 3. Abundances (no. 0.05 m⁻²) and biomasses (preserved alcohol wet weight in g 0.05 m⁻²) for all species, by site

Taxon	Site A		Site B		Site C		Site D	
	No.	Wt	No.	Wt	No.	Wt	No.	Wt
Oligochaeta	7	<0.01	119	0.01	110	<0.01	19	<0.01
Polychaeta								
Ampharetidae								
<i>Melinna</i> sp.	0	0	2	0.06	0	0	0	0
Capitellidae								
<i>Heteromastus filiformis</i>	46	0.02	138	0.03	51	0.03	6	0.01
<i>Mediomastus</i> sp.	0	0	30	0.01	85	0.01	143	0.02
Cirratulidae								
<i>Tharyx acutus</i>	22	0.03	12	0.02	5	0.01	2	<0.01
Cossuridae								
<i>Cossura</i> sp.	0	0	21	<0.01	105	0.01	122	0.01
Glyceridae								
<i>Glycera dibranchiata</i>	0	0	1	0.02	0	0	0	0
Goniadidae								
<i>Glycinde solitaria</i>	0	0	0	0	0	0	1	0.01
Hesionidae								
<i>Gyptis brevipalpa</i>	0	0	0	0	0	0	1	<0.01
<i>Microphthalmus aberrans</i>	0	0	6	<0.01	1	<0.01	4	<0.01
<i>Parahesionia luteola</i>	28	0.01	0	0	0	0	0	0
<i>Podarke obscura</i>	0	0	1	<0.01	0	0	0	0
Lumbrineridae								
<i>Lumbrineris tenuis</i>	0	0	1	<0.01	13	0.03	7	0.01
Maldanidae								
<i>Asychis elongata</i>	0	0	2	2.07	9	5.48	2	0.15
<i>Asychis</i> sp.	0	0	0	0	0	0	6	0.01
Nereidae								
<i>Nereis succinea</i>	1	0.02	0	0	0	0	1	0.01
Onuphidae								
<i>Diopatra cuprea</i>	0	0	2	0.02	0	0	0	0
Orbiniidae								
<i>Haploscoloplos</i> sp.	36	0.19	34	0.05	50	0.56	48	0.29
Phyllodocidae								
<i>Eteone heteropoda</i>	4	0.01	14	<0.01	1	<0.01	4	0.05
Spionidae								
<i>Polydora</i> sp.	0	0	2	<0.01	2	<0.01	0	0
<i>Streblospio benedicti</i>	132	0.06	262	0.12	171	0.09	191	0.05
<i>Scololepis squamata</i>	0	0	2	0.01	0	0	2	<0.01
Syllidae								
Unident. sp.	0	0	1	<0.01	0	0	0	0
Amphipoda								
<i>Ampelisca</i> sp.	85	0.04	302	0.13	138	0.05	210	0.05
<i>Listriella barnardi</i>	0	0	0	0	28	0.01	9	<0.01
<i>Corophium</i> sp.	0	0	0	0	0	0	1	<0.01
Unident. caprellid	0	0	0	0	0	0	3	<0.01
Cumacea								
<i>Cyclaspis varians</i>	5	<0.01	6	<0.01	4	<0.01	10	<0.01
Decapoda								
<i>Callinassa atlantica</i>	0	0	0	0	1	2.87	0	0
<i>Crangon septemspinosa</i>	0	0	0	0	0	0	1	0.04
Isopoda								
<i>Cyathura</i> sp.	0	0	0	0	1	0.01	3	0.04
<i>Edotea triloba</i>	4	0.01	4	0.01	0	0	0	0
Pycnogonida								
<i>Anaplodactylus lentus</i>	0	0	0	0	0	0	1	<0.01
Bivalvia								
<i>Gemma gemma</i>	0	0	6	0.01	9	0.02	26	0.05
<i>Mulinia lateralis</i>	0	0	0	0	0	0	1	0.13

Table 3 (continued)

Taxon	Site A		Site B		Site C		Site D	
	No.	Wt	No.	Wt	No.	Wt	No.	Wt
Gastropoda								
<i>Ilyanassa obsoleta</i>	0	0	2	6.07	0	0	1	1.45
Nemertea								
Unident. sp.	0	0	0	0	1	0.05	0	0
Totals:	370	0.39	970	8.66	785	9.25	825	2.38
Total species:	11		23		18		27	

Table 4. Means for no. of taxa (S) and abundances (A) per core (0.008 m²). Lower case letters show the significance relations determined by Duncan's multiple-range tests; those with same letter were not significantly different (p > 0.05), those with different letters were significantly different (p < 0.05)

Site	Taxa per core	Site	Ind. per core
D	14.0 ^a	B	161.5 ^a
C	11.2 ^{ab}	D	138.2 ^a
B	10.0 ^b	C	130.2 ^a
A	5.8 ^c	A	61.7 ^b

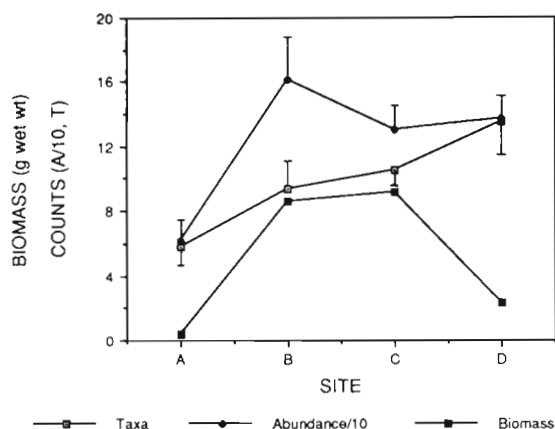


Fig. 6. Mean taxa numbers (S), mean community abundances (A), and mean community biomasses (alcohol-preserved wet weights; B) per core (0.008 m²). Bars show 1 standard error for S and A

ranged from 27.5 to 30.5 ppt. Dissolved oxygen ranged from 5.0 to 7.2 mg l⁻¹, except for a reading of 2.8 mg l⁻¹ at Site D.

Benthic macrofauna

Univariate analyses. A total of 37 taxa (most identified to species level) of macrofauna was collected (Table 3). The total number of taxa (S) collected at each

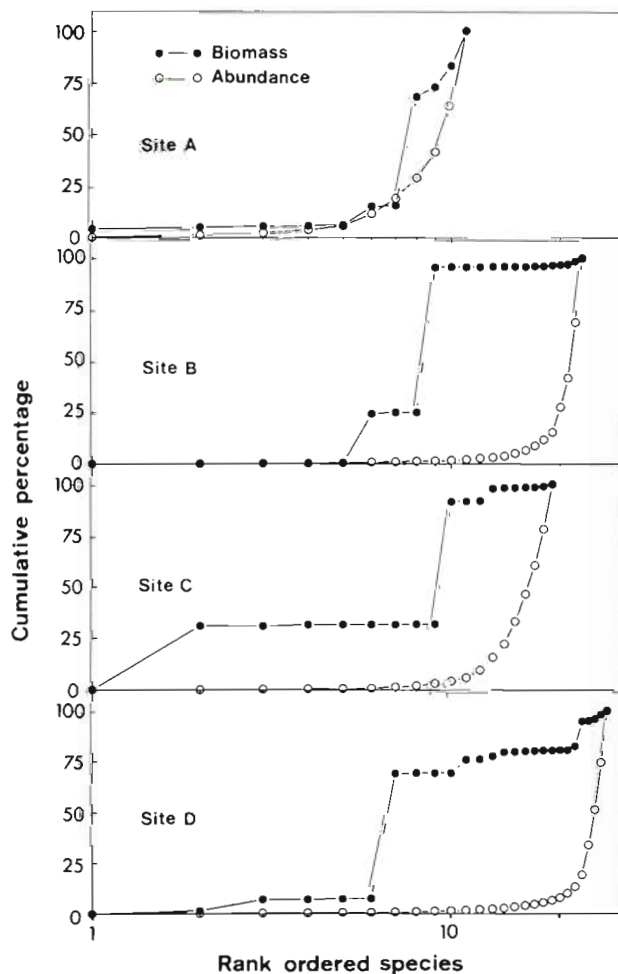


Fig. 7 K-dominance curves comparing cumulative abundances and biomasses

station was lowest at Site A (11) and highest at Site D (27). Total community abundances (A) ranged from 370 ind. 0.05 m⁻² at Site A to 970 ind. 0.05 m⁻² at B. Total wet weights (B) ranged from 0.39 g 0.05 m⁻² at Site A to 9.25 g 0.05 m⁻² at C. One-way ANOVAs indicated that there were significant differences between the mean number of taxa (S) collected per core (p = 0.001), and the mean number of individuals (A) collected per core

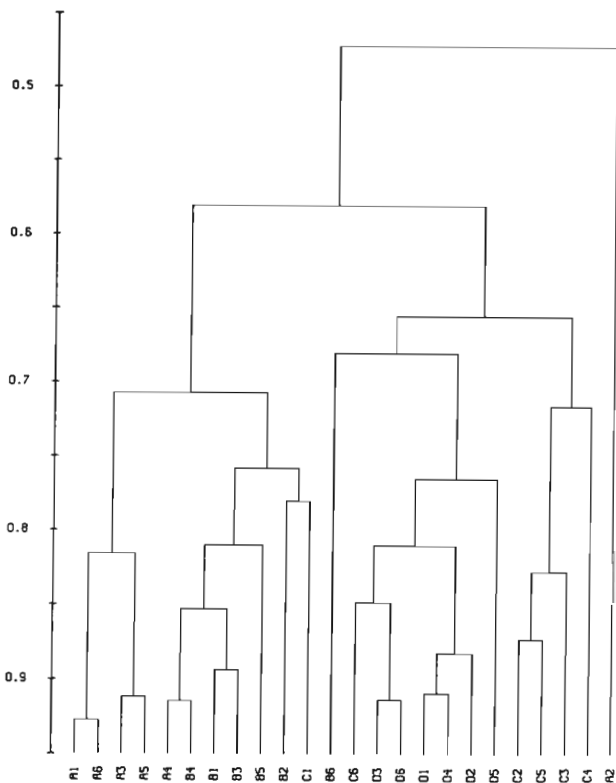


Fig. 8. Cluster diagram (dendrogram) of log(x+1) transformed abundances using Pearson product-moment correlation coefficients and UPGMA linkage

($p = 0.013$). Table 4 shows the S and A means on a per-core basis (0.008 m^{-2}), and the significance relationships indicated by Duncan's multiple-range tests; the means are plotted in Fig. 6. Site A had significantly and substantially lower S and A values than the other 3 sites, but there was no clear separation of Sites B, C and D. These 3 sites showed no significant differences in mean abundances (A), and only Sites B and D were significantly different based on number of taxa per core (S), with Site D being highest. The biomass (B) data suggested an 'enhanced' zone at Sites B and C, but no statistical analyses were possible for these data (see 'Methods'). Small near-surface dwelling polychaetes were numerical dominants at all 4 sites, with *Heteromastus filiformis* and *Streblospio benedicti* most abundant at Sites A and B, and *Mediomastus* sp. and *S. benedicti* most abundant at Sites C and D (Table 3). The amphipod *Ampelisca* sp. was also abundant at all 4 sites. Large deep-burrowing taxa (e.g. *Asychis elongata* and *Diopatra cuprea*) and several additional species (e.g. *Mediomastus* sp., *Cossura* sp.) were only collected from Sites B, C and D.

The ABC plots (K-dominance curves) were similar at Sites C and D, with the biomass curves lying above the abundance curves (Fig. 7). This indicates 'unpolluted' conditions according to Warwick's (1986) criteria. The

biomass and abundance curves were nearly coincident at Site A, indicating 'moderately polluted' conditions by Warwick's criteria. At Site B the 2 curves were coincident for the first few species, then the biomass curve was higher.

Multivariate analyses. The dendrogram produced by classification analysis of the abundance data showed that most replicates clustered by site (Fig. 8). Sample A2 is an outlier because 2 polychaete species were abundant that were not present, or were rare, in the other replicates at Site A, and at the other sites. Overall, Sites A and C were least similar, and Site A was more similar to Site B than to D. Thus, the classification analysis also indicated a relationship among sites potentially related to the pollution gradient; with Sites A and B (closest to the pollution source) being most

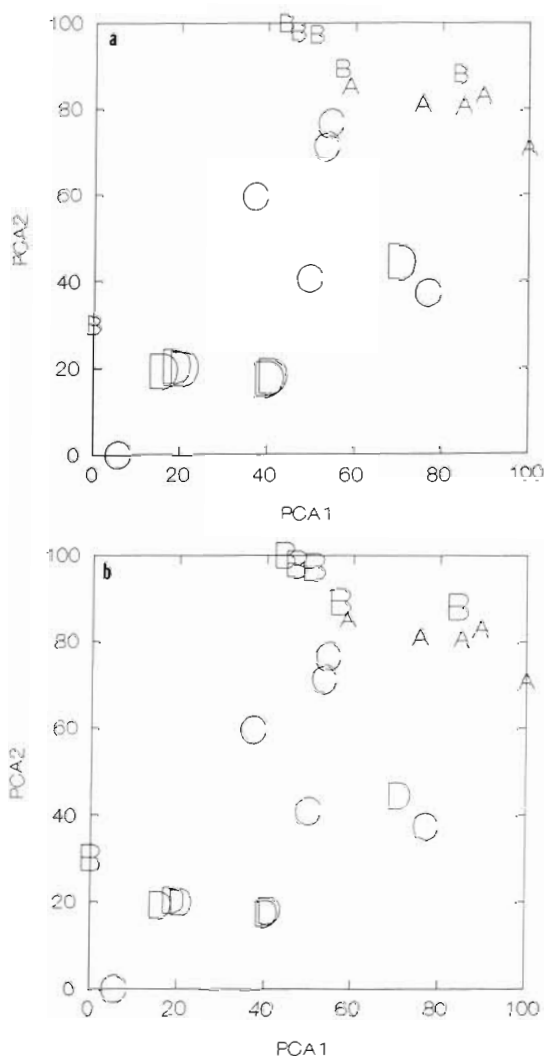


Fig. 9. Principal components analysis (PCA) of log(x+1) transformed abundances showing PCA1 vs PCA2 plots, with size of plot (site) symbols proportional to: (a) % TOM of sediments; (b) chlorophyll a concentration of sediments. 73.8% of variance accounted for by PCA axis 1, and 8.5% by axis 2

similar to each other, and Sites C and D (farthest from the pollution source) clustering together.

PCA showed a separation of Sites A and B from C and D, and indicated that differences among sites in sediment TOM concentrations were related to the separation (Fig. 9a). Sediment chlorophyll *a* concentrations were not as well correlated to the site clusters as were TOM values (Fig. 9a, b). The first PCA axis explained 73.8% of the variance and the second explained 8.5%.

SPI data. Fig. 10 is composed of 4 photographs of acrylic box cores, one taken from each of the 4 sites. Fig. 11 is composed of close-ups of the sediment surface from the photographs in Fig. 10. The most important aspects of these photographs include changes in apparent color RPD layer depth, and features that indicate species composition of the benthic communities. Although sufficient replicates were not available for a statistical analysis (see 'Methods'), the apparent color RPD was consistently shallowest at Site A, deeper at Site B, and deepest at Sites C and D during August and September. This RPD trend extending along the pollution gradient was easily and quickly detected in the field (Figs. 5, 10 & 11). Actual Eh measurements made on the intact cores verified the data obtained by SPI (cf. Figs. 4 & 5).

The characteristic, flattened muddy tubes built by the amphipod *Ampelisca* sp. were seen at Sites B, C and D, but not at Site A (Figs. 10 & 11). The benthic/sediment cores revealed that *Ampelisca* sp. were pres-

ent at Site A, but probably in lower abundances than at the other sites (Table 3). Worm burrows and some tubes < 1 mm wide were abundant at Site A, but absent or much less abundant at the other sites. The sediment surface was more flocculent at Sites A and B compared to the other sites (Fig. 11). No large worm tubes or evidence of large burrows were visible at Site A, indicating that benthic fauna capable of reworking the sediment more than a centimeter or so were absent. One large tube was visible in one of the acrylic cores taken at Site B (but not shown in Figs. 10 & 11), and 2 specimens of the polychaete *Asychis elongata* were collected (Table 3). Several large specimens of *A. elongata* were collected from both Sites C and D, and a tube is visible at Site C in Figs. 10 & 11 (a tube of *Diopatra cuprea* is visible at Site D, but none was collected in the cores for benthos). Several aerated (as evidenced by a light brown color compared to the dark gray of surrounding sediments) tubes and/or burrows several centimeters below the sediment surface were at Site C (Fig. 10). The increased depth of the RPD layer at Sites C and D was evidence of the abundant large polychaetes at these sites (Table 3).

Comparison of traditional and SPI benthic data

Both univariate and multivariate benthic data analyses showed changes related to the enrichment gradient. The most important univariate results were the

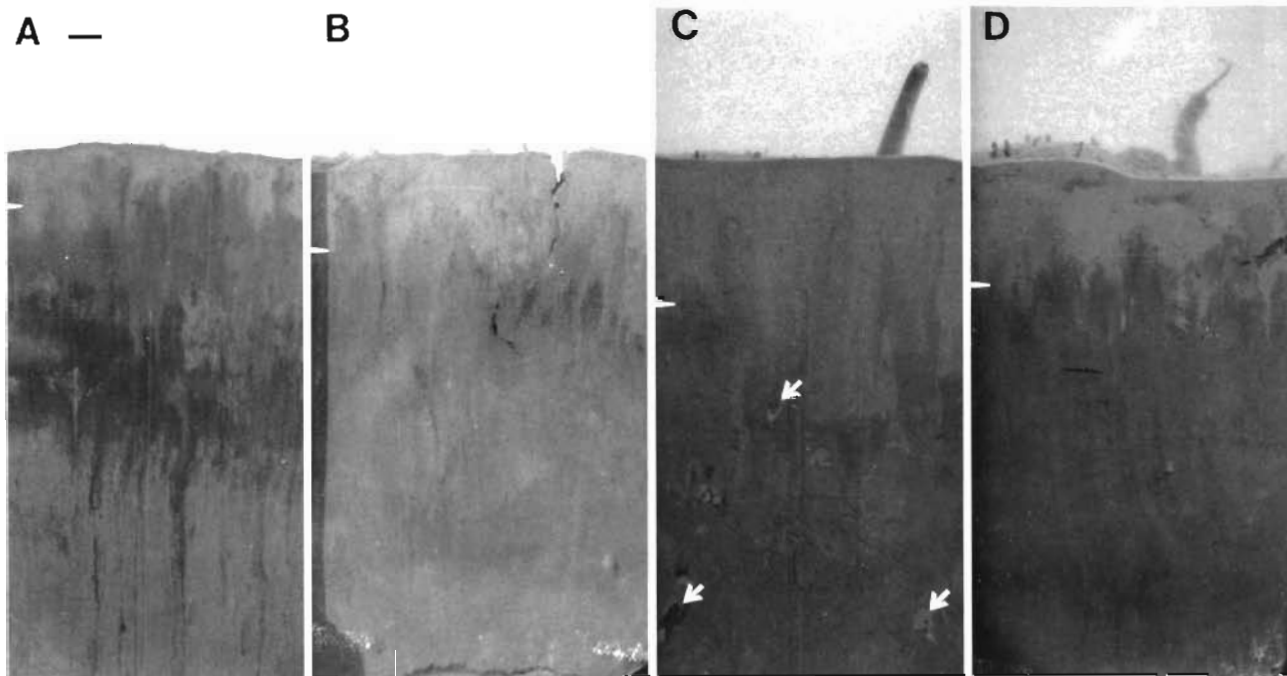


Fig. 10. Representative acrylic box cores from Sites A to D along the organic enrichment gradient. Arrows on left side of each photo show location of RPD depth as determined by sediment color changes. Arrows in C show broken aerated polychaete tubes and burrows. Scale bar = 1 cm

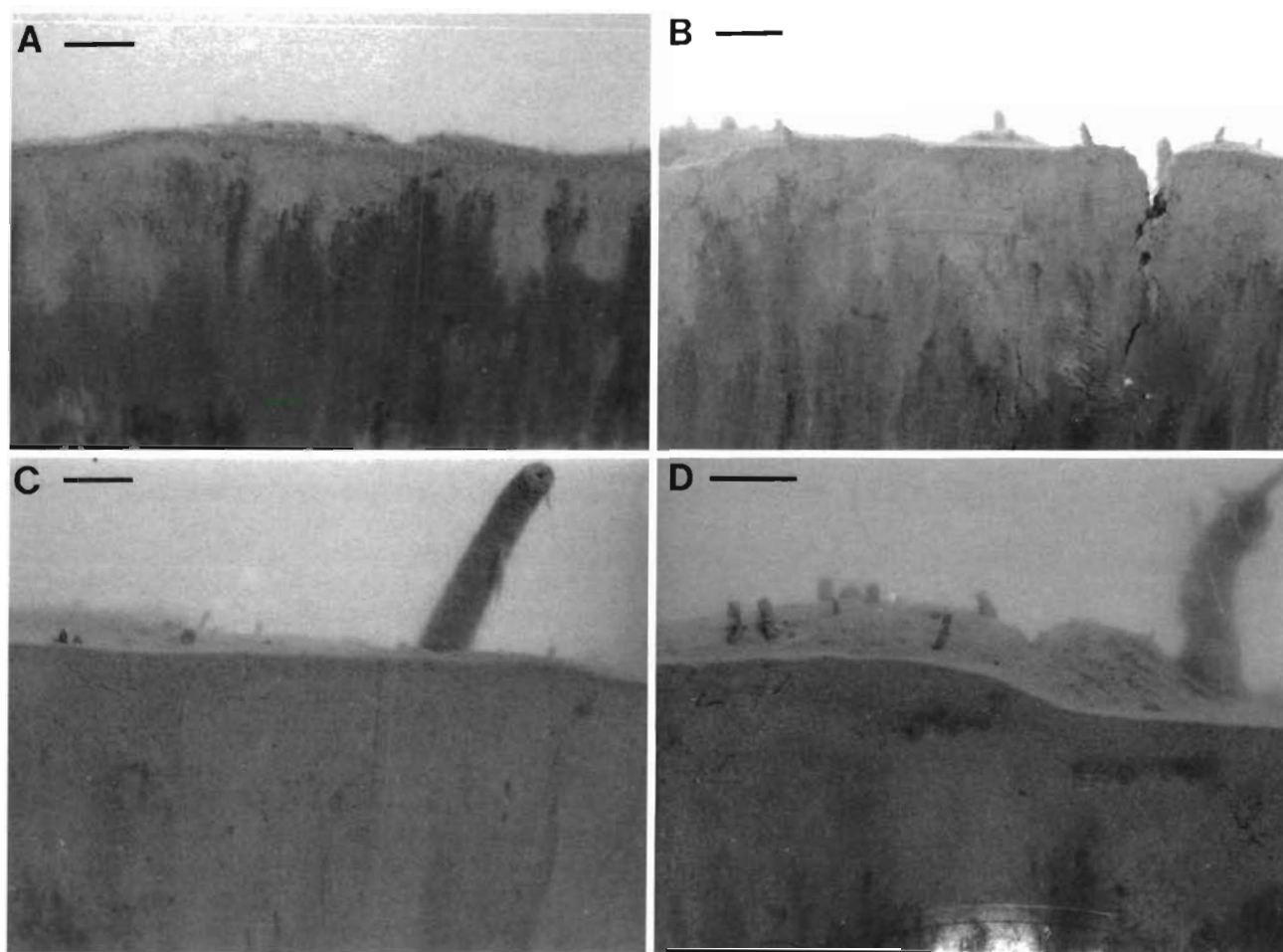


Fig. 11. Close-ups of box cores in Fig. 10. Flat tubes projecting a few mm above the surface in (B), (C) and (D) are of *Ampelisca* sp. Large tube in (C) is of *Asychis elongata*. Large tube in (D) is of *Diopatra cuprea*. Scale bars = 1 cm

substantially lower numbers of taxa (S) and total abundances (A) per core at Site A, the most enriched site, compared to the other 3 sites. The biomass (B) data suggested an 'enhanced' zone at Sites B and C, but no statistical analyses were possible for these data. ABC plots (*K*-dominance curves) and the multivariate PCAs generally corroborated the univariate statistical analyses. The major among-site differences in species composition were the absence of large deep-burrowing fauna at Site A, and the consistent and common occurrence of several additional species at Sites B, C and D compared to Site A. In sum, the 'traditional' analyses showed benthic changes correlated to pollution-related changes in sediment characteristics, but the only definitive changes were in the benthos at Site A.

SPI data collected in the field showed changes in benthos, and changes in apparent color RPD layer depth, along the transect. The most useful SPI data were the substantially more shallow apparent color RPD layer depth (and Eh measurements made with a probe) at Site A compared to the other 3 sites. The SPI

data also indicated that no deep-burrowing taxa were present at Site A, and the site was numerically dominated by small near-surface-dwelling annelids. In sum, the SPI data were as useful as the more detailed 'traditional' data on species composition and abundances for delimiting the approximate extent of the bottom area affected by organic wastes.

DISCUSSION

In summarizing a recent workshop on methods for measuring the biological effects of pollutants, Underwood & Peterson (1988) outlined a logical framework for pollution investigations. They suggested the initial stages should consist of a determination of whether or not there were changes in the ecosystem that were correlated with pollutants, and not with other environmental variables. The traditional and SPI data indicated this to be the case for the macrofaunal benthos in the present study. Assuming that both approaches

were successful, we can address the primary question in the EVALUATION section of the framework of Underwood & Peterson (1988): 'Which method(s) is (are) best?' We could see no differences in the ability of one approach over the other. However, because SPI data can be collected and processed in such a short time relative to traditional techniques, we suggest that SPI techniques will be clearly superior for some types of studies. However, Underwood & Peterson (1988) noted that a simple cost-benefit analysis should not be the only criterion for the EVALUATION process. It should also include information from their INTERPRETATION and PREDICTION stages (which are concerned with the consequences of the pollutants at hierarchical levels in the ecosystem other than those measured). But, as discussed below, these are outside the immediate concern of our study. Furthermore, at the present time our understanding of the mechanisms responsible for even well known pollution-induced changes in benthos is minimal. For example, determining the causal mechanisms for the dominance of so-called 'pollution indicator species' which dominate the 'polluted' zone in the Pearson & Rosenberg (1978) model (see discussion below), and what combinations of conditions cause such dramatic changes in community composition, represent important areas of future research (also see Grizzle 1984, Chesney 1985, Rhoads & Germano 1986). We are well aware that traditional techniques will continue for some time as the mainstay for benthic studies, and indeed must be used to get an understanding of the environmental changes causing such dramatic alterations of benthos. Nonetheless, we suggest that SPI represents a potentially very useful approach that should be more fully developed.

Our study was aimed primarily at comparing SPI and traditional techniques, but it also provided an assessment of the predictive capabilities of a widely cited (e.g. Gray 1981, Rhoads & Germano 1982, 1986, Spies 1984) conceptual model of Pearson & Rosenberg (1976, 1978 – their Fig. 15) showing a generalized benthic response to enrichment. Briefly, extending outward from a 'grossly polluted' zone with no macrofauna, it includes: (1) an impoverished (low S, A and B) 'polluted' area typically dominated by small polychaetes that live near the sediment surface, and a shallow RPD layer depth; (2) a 'transitory' zone with increased numbers of species and larger, deeper burrowing species, and increasing RPD layer depths; and (3) a 'normal' area dominated by large deep-burrowing fauna, and an RPD layer depth of several centimeters. We believe the Pearson & Rosenberg model provides an important conceptual framework for organizing benthic research, yet it has apparently received little attention with respect to explicit tests of predictability.

Although it was not possible to delimit 3 (or more)

zones of impact, our data clearly showed a gradient of sediment enrichment and benthic changes along the sampling transect similar to the Pearson & Rosenberg model, and possibly an 'enhanced' zone with respect to biomass. However, our data indicated 2 apparent and perhaps important differences from their model. First, small polychaetes (capitellids and *Streblospio benedicti*) were abundant at all 4 of our study sites (Table 3). Pearson & Rosenberg (1978) cited several studies showing these taxa as abundant in polluted areas. However they are typically reported as numerical dominants in estuaries (e.g. Maurer et al. 1978, Holland 1985, Nichols & Thompson 1985), and are euryhaline (Boesch 1977). They are 'opportunists', capable of rapid invasion of disturbed areas (see reviews by Pearson & Rosenberg 1978, Gray 1979). Furthermore, Germano (1983) showed that the opportunistic species typically dominant in early successional stages may persist after the arrival of larger, longer-lived (see McCall 1977) species more characteristic of later successional stages. Hence, 'mixed' communities of 'opportunistic', 'pollution indicator' species and species not typically found in heavily polluted areas can be expected to occur (see also Gray 1981, Nichols & Thompson 1985).

The second departure in our data from the Pearson & Rosenberg model is that the RPD layer depth may have been deeper at Site C than at the other sites. Both the apparent color RPD, and the rate of decrease in Eh measured by probe, indicated a deeper RPD layer at Site C. This can be explained by the abundant, large malmanid polychaete *Asychis elongata* (Table 3). *A. elongata* builds a mucus/mud tube and can penetrate to 50 cm sediment depth (Reinharz et al. 1982); it attains a length of 30 cm (Gosner 1971). *A. elongata* occurs in high densities in the deeper mud-bottom areas of San Francisco Bay (California, USA) with *Streblospio benedicti*, *Heteromastus filiformis*, and *Ampelisca abdita* (Nichols & Thompson 1985). It also occurs commonly in Chesapeake Bay (USA) (Reinharz et al. 1982). Abundant *A. elongata* and other species capable of penetrating decimeters into the sediment caused a deep RPD layer at Site C (and D; Figs. 4 & 5). These data (and the biomass data; Table 3) suggest that there may have been an 'enhanced' zone along our study transect where large, deep-burrowing species were most abundant, pushing the RPD layer deeper into the sediment. Weston (1990) recently quantified the increased biomass vertically in the sediment along an organic enrichment gradient.

A final study that warrants discussion with respect to our findings concerning a pollution gradient is Royer (1980). He sampled the macrofauna from a site (his 'ST1'; cf. Fig. 1) near our Site A, and at 3 others farther north in Great Sound during 1977 to 1979. He reported

lowest species numbers from Site ST1, and on most sampling occasions it was dominated numerically by *Streblospio benedicti*, which was also dominant at Site A during the present study (Table 3). *Asychis elongata* was not collected by Royer (1980) from Site ST1, but occurred at all other sites, as in the present study. Amphipods were rare at Site ST1 but abundant at the other sites, and *Ampelisca* sp. was the numerical dominant at these sites; *Ampelisca* sp. was most abundant at our Sites B, C and D. Hence, Royer's (1980) study, done over 8 yr earlier, showed spatial trends in dominant benthic species very similar to those reported in the present study.

The remaining 2 stages in the Underwood & Peterson (1988) framework are INTERPRETATION: 'Does the observed effect of pollution matter to the biological system?'; and EVALUATION: 'Will other worse consequences happen?' As mentioned above, these questions are not appropriate for our study. They do, however, point to important considerations for future research on SPI techniques. It will be important to determine if pollution-induced changes in benthic populations cause changes at higher levels in the system. In particular, do benthic changes cause changes in populations of commercially-important species such as decapod crustaceans and flatfish that prey on infauna? Surprisingly little research has been published on this topic (see Boesch 1982, Spies 1984, Rhoads & Germano 1986, Becker & Chew 1987). Benthic ecologists will never be able to provide comprehensive input to environmental managers unless they have such an understanding and can make predictions concerning higher-level effects. As part of these determinations it would be best to know how 'general' the trophic relationships are. In other words, it would be most useful to know if the predator-prey relationships elucidated extended to prey of a certain life habit mode, or were restricted to certain taxa. Such knowledge would be needed for making predictions, as well as for developing SPI techniques to an appropriate level of resolution. It may be that these topics and more have already been addressed by researchers who own SPI devices. But until more published information is available, we must conclude that much remains to be done to develop SPI techniques, and much more descriptive/correlative work comparing SPI data and traditional data in waters affected by different combinations of pollutants needs to be accomplished.

Recent research (Herman & Heip 1988, Warwick 1988a, b, Warwick et al. 1990) has indicated that the cost of traditional benthic methods may be greatly reduced by identifying animals collected only to the level of family, or perhaps even higher taxonomic levels. O'Connor et al. (1989), Krieger et al. (1991), and the present study indicate that SPI techniques hold the

potential for providing a much more cost-effective approach than any form of traditional methods for assessing the impacts of organic pollutants on benthos (also see Chapman et al. 1986 for mention of the use of SPI data to replace traditional benthic data in their 'sediment quality triad' approach). As mentioned above, SPI techniques must be based upon information provided by traditional approaches, and thus must be further developed in conjunction with such approaches. Nonetheless, SPI techniques have been effectively and routinely used for years in assessing the impacts of dredge and fill activities (e.g. Diaz et al. 1986, Diaz & Schaffner 1988, Nichols et al. 1990, Parker & Revelas 1989, 1990), and their use should be extended to other types of pollution studies.

We also suggest that the SPI approach can be further developed using a variety of sampling devices. Acrylic core tubes have been used by others for obtaining benthic samples, and have been used routinely for many years as liners for various small-diameter corers mainly for geological studies. Hence, simple techniques such as the one we used in the present study for collecting SPI data have been available for some time. Nonetheless, we are not aware of published SPI data from pollution studies that have been collected with anything other than a sediment profile camera. We believe the development of a low-cost, lighter sampling device (compared to the large, heavy devices now in use for collecting SPI data; e.g. see Rhoads & Germano 1982) would be useful in allowing further development of the SPI approach for benthic studies.

Acknowledgements. This paper is largely the result of the skillful help of Linda Harnden, who did the initial processing/identification of all benthic samples. We very much appreciate her help. Frederick Ferry sorted the samples. Andy Megliss did the grain size analyses. Timothy Jacobsen, Francesco Trama, and Alan McIntosh provided laboratory space for various analyses. We thank Don Rhoads for his pioneering interdisciplinary work on the ecology of the benthos of coastal waters. His research has always been a source of inspiration and direction. The manuscript was greatly improved by a detailed review by Bob Diaz and 2 anonymous reviewers. The field portion of the study was funded by the Center for Coastal and Environmental Studies at Rutgers University. The Jackson Estuarine Laboratory of the University of New Hampshire, and Livingston University, funded the remainder of the study.

LITERATURE CITED

- Allen, D. M., Clymer, J. P., Herman, S. S. (1978). Fishes of the Hereford Inlet estuary, southern New Jersey. The Wetlands Institute, Lehigh University, Stone Harbor, New Jersey.
- Ashley, G. M., Grizzle, R. E. (1988). Interactions between hydrodynamics, benthos and sedimentation in a tide-dominated coastal lagoon. *Mar. Geol.* 82: 61–81.
- Becker, D. S., Chew, K. K. (1987). Predation on *Caprell* spp.

- by small-mouthed pleuronectids in Puget Sound, Washington. Fish. Bull. U.S. 85: 471-479
- Bilyard, G. R. (1987). The value of benthic infauna in marine pollution monitoring studies. Mar. Pollut. Bull. 18: 581-585
- Boesch, D. F. (1977). A new look at the zonation of benthos along the estuarine gradient. In: Coull, B. C. (ed.) Ecology of marine benthos. Univ. of South Carolina Press, Columbia, p. 245-266
- Boesch, D. F. (1982). Ecosystem consequences of alterations of benthic community structure and function in the New York Bight region. In: Mayer, G. F. (ed.) Ecological stress and the New York Bight: science and management. Estuarine Research Federation, Columbia, p. 543-568
- Boyer, L. F., Shen, E. J. (1988). Sediment-profile camera study of Milwaukee Harbor sediments. J. Great Lakes Res. 14: 444-465
- Boyer, L. F., Hedrick, J. (1989). Submersible-deployed video sediment-profile camera system for benthic studies. J. Great Lakes Res. 15: 34-45
- Byers, S. C., Mills, E. L., Stewart, P. L. (1978). A comparison of methods of determining organic carbon in marine sediments, with suggestion for a standard method. Hydrobiologia 58: 43-47
- Chapman, P. M., Dexter, R. N., Cross, S. F., Mitchell, D. G. (1986). A field trial of the sediment quality triad in San Francisco Bay. NOAA Tech. Mem. NOS OMA 25. Rockville, MD
- Chesney, E. J. Jr (1985). Succession in soft-bottom benthic environments: are pioneering species really out-competed? In: Gibbs, P. E. (ed.) Proc. 19th Eur. Mar. Biol. Symp. Cambridge University Press, Cambridge, p. 277-286
- Clarke, K. R., Green, R. H. (1988). Statistical design and analysis for a 'biological effects' study. Mar. Ecol. Prog. Ser. 46: 213-226
- Diaz, R. J., Schaffner, L. C. (1988). Monitoring environmental impacts associated with open-water thin-layer disposal of dredged material at Fowl River, Alabama, using sediment profile camera. VIMS, College of William and Mary, Gloucester Pt., VA. Final Rpt to TAI Environmental Sciences, Mobile, AL
- Diaz, R. J., Schaffner, L. C., Kiley, K., Phillips, R., Davis, L. (1986). Sediment profile camera survey of the Fowl River open-water thin-layer disposal area, Mobile Bay, AL. Preoperation survey - Cruise 1, June 20-21, 1986. VIMS, College of William and Mary, Gloucester Pt., VA. Rpt to TAI Environmental Sciences, Mobile, AL
- Germano, J. D. (1983). Infaunal succession in Long Island Sound: animal-sediment interactions and the effects of predation. Ph.D. dissertation, Yale University
- Gosner, K. L. (1971). Guide to identification of marine and estuarine invertebrates, Cape Hatteras to the Bay of Fundy. Wiley-Interscience, New York
- Gray, J. F. (1979). Pollution-induced changes in populations. Phil. Trans. R. Soc. Lond. B. 286: 545-561
- Gray, J. F. (1981). The ecology of marine sediments. Cambridge University Press, Cambridge
- Gray, J. F., Aschan, M., Carr, M. R., Clarke, K. R., Green, R. H., Pearson, T. H., Rosenberg, R., Warwick, R. M. (1988). Analysis of community attributes of the benthic macrofauna of Frierfjord/Langesundfjord and in a mesocosm experiment. Mar. Ecol. Prog. Ser. 46: 151-165
- Grizzle, R. E. (1984). Pollution indicator species of macrobenthos in a coastal lagoon. Mar. Ecol. Prog. Ser. 18: 191-200
- Grizzle, R. E. (1988). The relative effects of tidal currents, seston, and bottom sediments on individual growth of *Mercenaria mercenaria* (Linné): feeding ecology and aquacultural implications. Ph.D. dissertation, Rutgers University
- Grizzle, R. E., Lutz, R. A. (1989). A statistical model relating horizontal seston fluxes and bottom sediment characteristics to growth of *Mercenaria mercenaria*. Mar. Biol. 102: 95-105
- Herman, P. M. J., Heip, C. (1988). On the use of meiofauna in ecological monitoring: who needs taxonomy? Mar. Pollut. Bull. 19: 665-668
- Holland, A. F. (1985). Long-term variation of macrobenthos in a mesohaline region of Chesapeake Bay. Estuaries 8: 93-113
- Krieger, Y., Mulsow, S., Rhoads, D. C. (1991). Organic enrichment of the seafloor; impact assessment using a geographic information system. In: Pillman, W., Jaescheke, A. (eds.) Informatik für den Umweltschutz. Proc. Symposium of a meeting at Wien, Austria, September 1990. Springer, Berlin, p. 363-372
- Maurer, D., Watling, L., Kinner, P., Leathem, W., Wethe, C. (1978). Benthic invertebrate assemblages of Delaware Bay. Mar. Biol. 45: 65-78
- McCall, P. L. (1977). Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. J. mar. Res. 35: 221-266
- Nichols, F. H., Thompson, J. K. (1985). Time scales of change in the San Francisco Bay benthos. Hydrobiologia 129: 121-138
- Nichols, M. M., Diaz, R. J., Schaffner, L. C. (1990). Effects of hopper dredging and sediment dispersion, Chesapeake Bay. Environ. Geol. Water Sci. 15: 31-43
- O'Connor, B. D. S., Costelloe, J., Keegan, B. F., Rhoads, D. C. (1989). The use of REMOTS® technology in monitoring coastal enrichment resulting from mariculture. Mar. Pollut. Bull. 20: 384-390
- Parker, J. H., Revelas, E. C. (1989). 1985 monitoring surveys at the central Long Island Sound disposal site: an assessment of impacts from disposal and Hurricane Gloria. Disposal Area Monitoring System (DAMOS), Cont. 57 US Army Corps of Engineers, Waltham, MA
- Parker, J. H., Revelas, E. C. (1990). Monitoring cruise at the western Long Island Sound disposal site, July 1988. Disposal Area Monitoring System (DAMOS), Cont. 76. US Army Corps of Engineers, Waltham, MA
- Parsons, T. R., Maita, Y., Lalli, C. M. (1984). A manual of chemical and biological methods for seawater analysis. Pergamon Press, Oxford
- Pearson, T. H., Rosenberg, R. (1976). A comparative study of the effects on the marine environment of wastes from cellulose industries in Scotland and Sweden. Ambio 5: 77-79
- Pearson, T. H., Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. mar. Biol. A. Rev. 16: 229-311
- Pearson, T. H., Stanley, S. O. (1979). Comparative measurement of the redox potential of marine sediments as a rapid means of assessing the effect of organic pollution. Mar. Biol. 53: 371-379
- Reinharz, E., Nilsen, K. J., Boesch, D. F., Bertelsen, R., O'Connell, A. E. (1982). A radiographic examination of physical and biogenic sedimentary structures in the Chesapeake Bay. Dept. Natural Resources, Maryland Geological Survey, Rep. Investig. 36
- Rhoads, D. C., Cande, S. (1971). Sediment profile camera for *in situ* study of organism-sediment relations. Limnol. Oceanogr. 16: 110-114
- Rhoads, D. C., Germano, J. D. (1982). Characterization of organism-sediment relations using sediment profile imag-

- ing: an efficient method of remote ecological monitoring of the seafloor (REMOTS system). *Mar. Ecol. Prog. Ser.* 8: 115–128
- Rhoads, D. C., Germano, J. D. (1986). Interpreting long-term changes in benthic community structure: a new protocol. *Hydrobiologia* 142: 291–308
- Rhoads, D. C., Germano, J. D., Boyer, L. F. (1981). Sediment profile imaging: an efficient method of remote ecological monitoring of the seafloor (REMOTSTM system). *Oceans*, Sep 1981: 561–566
- Royer, D. F. (1980). The effects of a sewage treatment effluent on the benthic macrofauna of a salt marsh estuary. Ph.D. dissertation, Lehigh University
- Spies, R. (1984). Benthic-pelagic coupling in sewage affected marine ecosystems. *Mar. environ. Res.* 13: 195–230
- Underwood, A. J., Peterson, C. H. (1988). Towards an ecological framework for investigating pollution. *Mar. Ecol. Prog. Ser.* 46: 227–234
- Warwick, R. M. (1986). A new method for detecting pollution effects on marine macrobenthic communities. *Mar. Biol.* 92: 557–562
- Warwick, R. M. (1988a). Analysis of community attributes of the macrobenthos of Frierfjord/Langesundfjord at taxonomic levels higher than species. *Mar. Ecol. Prog. Ser.* 46: 167–170
- Warwick, R. M. (1988b). The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. *Mar. Pollut. Bull.* 19: 259–268
- Warwick, R. M., Platt, H. M., Clarke, K. R., Agard, J., Gobin, J. (1990). Analysis of macrobenthic and meiobenthic community structure in relation to pollution and disturbance in Hamilton Harbour, Bermuda. *J. exp. mar. Biol. Ecol.* 138: 119–142
- Weston, D. P. (1990). Quantitative examination of macrobenthic community changes along an organic enrichment gradient. *Mar. Ecol. Prog. Ser.* 61: 233–244
- Wishart, D. (1987). CLUSTAN user manual, 4th edn. Computing Laboratory, Univ. of St. Andrews, UK

This article was presented by K. L. Heck, Dauphin Island, Alabama, USA

*Manuscript first received: August 8, 1990
Revised version accepted: May 6, 1991*