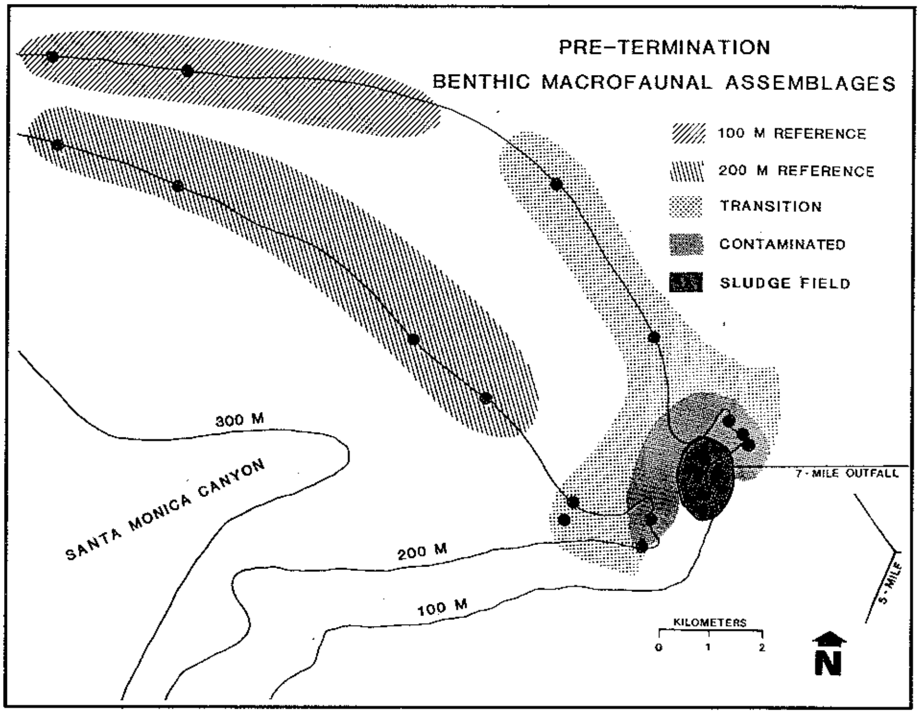
Temporal and Spatial Changes in Sediment Toxicity in Santa Monica Bay

The City of Los Angeles stopped discharging sludge into Santa Monica Bay in November 1987. As a result, the mass emissions of solids and contaminants into the bay have declined substantially (SCCWRP 1989). The effect of this event on the sediment quality and marine life in Santa Monica Bay was examined by SCCWRP from 1986 to 1990 (Thompson 1991).

This report describes the results of one component of the study, sediment toxicity in Santa Monica Bay. The objectives were to document temporal changes in sediment toxicity at the sludge discharge site, and to compare the relative sediment toxicity with the chemical characteristics of the sediments.

Figure 1.

Locations of sampling stations and the Hyperion sludge outfall in Santa Monica Bay.



Methods

Sediment samples were collected from eight stations in Santa Monica Bay (Figure 1). A pair of stations was located in each of four zones characterized by different benthic macrofaunal assemblages: reference (Stations 5 and 6), transition (3 and 4), contaminated (1 and 2), and canyon (sludge field; 50 and 51) (Thompson 1991). All stations, except those in the canyon, were located at a depth of 100 m.

Temporal changes in the toxicity of canyon zone sediments were determined by laboratory tests on samples collected five times between November 1987 and August 1990. Spatial changes in toxicity among the four zones were investigated by laboratory tests with sediments collected in August 1989 and August 1990.

Surface sediment (upper 2 cm) was removed from the Van Veen grab sample, placed in a plastic jar, and stored refrigerated for up to two weeks before the toxicity tests. Short-term (10 days) and long-term (28 days) tests were conducted with the amphipod, *Grandidierella japonica*, using methods described in Nipper *et al.* (1989). Survival was measured in the short-term test, while survival and growth (change in body length per 28 days) were measured in the long-term test. Sediment from the amphipod collection site in Newport Bay was included in each test as a laboratory control.

The stations sampled and the number of replicates varied between collections because of sampling difficulties and limited availability of amphipods. Three replicates of one composite sediment sample from station 50 were tested in 1987 and 1988.

Single grab samples (not replicated) from Stations 1, 2, 4, 5, 6, 50, and 51 were tested in August 1989. Replicate grab samples from stations 50 and 51 were tested in February 1990. Replicate grab samples from stations 1, 2, 3, 4, 5, 6, 50, and 51 were tested in August 1990.

The sediment samples were analyzed for grain size (sand, silt, and clay), total organic carbon (TOC), total nitrogen (TN), dissolved sulfides, cadmium, copper, zinc, silver, DDT, DDE, DDD, Aroclor 1242, Aroclor 1254, hexachlorobenzene, and 22 polynuclear aromatic hydrocarbons (PAHs). Sediment analyses and toxicity tests were conducted on subsamples of sediment from the same grab sample. The chemistry data and analytical methods are found in Anderson *et al.* (1988) for the 1987-88 samples and Thompson (1991) for the 1989-90 samples.

Data for chlorinated hydrocarbons and polynuclear aromatic hydrocarbons were normalized to sediment organic carbon content prior to statistical analyses. Nonionic hydrocarbon concentration data are more highly correlated with biological effects when normalized to organic carbon than when expressed on a dry weight basis (Di Toro *et al.* 1991). Trace metal concentrations, grain size, TOC, and TN were expressed on a dry weight basis.

The data were tested for differences among zones and the laboratory control by single factor analysis of variance and Dunnett's multiple comparison test. The relations between changes in sediment characteristics and toxicity were examined with Spearman rank correlation coefficients.

Results

Temporal Changes in Toxicity

The 10-day survival of amphipods in laboratory control sediment (Newport Bay) varied between experiments, especially in February 1990 (Table 1). Survival was expressed as percent of the laboratory control to compensate for variability in control survival. The adjusted survival of *G. japonica* doubled between 1988 and 1989 suggesting a dramatic reduction in the toxicity of canyon zone sediment (Table 1). Survival in canyon sediment remained high in subsequent experiments, although survival in the August 1990 experiment was significantly less than the laboratory control.

Table 1.

Survival of Grandidierella japonica following 10-day exposure to sediments from the Canyon Zone (Stations 50 and 51) in Santa Monica Bay. Data are mean and one standard deviation (in parentheses). Three samples from each station were analyzed in 1987 and 1988, two samples were analyzed in 1989, and four samples were analyzed in 1990.

	10-Day Survival (%)				
	Canyon Zone		Lab Control		Adjusted survival % of control
Nov. 1987	35	(35)*	92	(7)	38
Apr. 1988	8	(14)*	88	(7)	9
Aug. 1989	90	(0)	98	(4)	92
Feb. 1990	53	(10)	65	(6)	82
Aug. 1990	72	(21)*	94	(6)	76

*Means significantly different from lab control (ANOVA and Dunnett's test; $p<0.05$).

Spatial Changes in Toxicity

Amphipod growth rates were similar between the 28-day exposures in 1989 and 1990 (Table 2) and were not significantly different among the zones (ANOVA, $p>0.05$). Growth rates in 1990 were less variable among zones than growth rates in 1989. Growth rates were expressed as percent of the laboratory control to compensate for variations between experiments.

Amphipod survival during the 28-day tests was variable and not significantly different among the zones (Table 2). Animals exposed to reference zone sediment in 1990 had the lowest survival, while amphipods exposed to contaminated zone sediment had the highest survival.

Toxicity and Sediment Characteristics

The relation between toxicity and sediment characteristics was examined with the data from five experiments with canyon zone sediments. Correlations and *post hoc* hypotheses were used to identify which of the measured sediment variables corresponded with reductions in toxicity during the course of the study.

Spearman rank correlations were calculated between amphipod toxicity and sediment characteristics (Table 3). We assumed that sediment constituents exerting toxic effects would have negative coefficients; a correlation of -1.0 is the strongest possible relation (i.e., reduced concentration accompanied by increased survival). Organic carbon, organic nitrogen, naphthalene, C1 substituted naphthalenes, and sulfide had high negative correlations. Hexachlorobenzene was the only chlorinated hydrocarbon that was negatively correlated with survival. Correlation coefficients for DDTs and PCBs were positive, indicating that amphipods had higher survival at higher concentrations. Zinc had the largest negative correlation among the trace metals.

In the second analysis, sediment data for the canyon zone were divided into low toxicity samples (1989 and 1990 tests) and high toxicity samples (1987 and 1988 tests) (Table 3). Mean chemical concentrations in the low toxicity group were compared to those in the high toxicity group (hydrocarbons were TOC normalized). We assumed that only contaminants with a substantial concentration reduction from 1987/88 to 1989/90 would have played a role in temporal toxicity changes.

The greatest change between the high and low toxicity groups was the reduction in sulfide concentration (Table 3). Concentrations of anthracene, C2 and C3 substituted phenanthrenes, and perylene declined by more than 50%. Concentrations of 29 of the 36 measured constituents declined

Figure 2.

Scatterplot of Spearman rank correlations and relative concentration change for canyon zone sediments in the 10-day amphipod survival test. The correlations were calculated between toxicity and sediment chemistry. The concentration change compared sediments collected in 1989/90 to sediments collected in 1987/88. Labelled points in the lower left quadrant are the sediment characteristics that had the greatest relation to temporal toxicity changes. C1NAPH = C1 substituted naphthalenes, C1-C3PHEN = C1-C3 substituted phenanthrenes, HCB = hexachlorobenzene, NAPH = naphthalene, PCB42 = Aroclor 1242, PCB54 = Aroclor 1254, PERY = perylene, PPDDD = p,p'DDD, S = dissolved sulfide, TN = total nitrogen, TOC = total organic carbon.

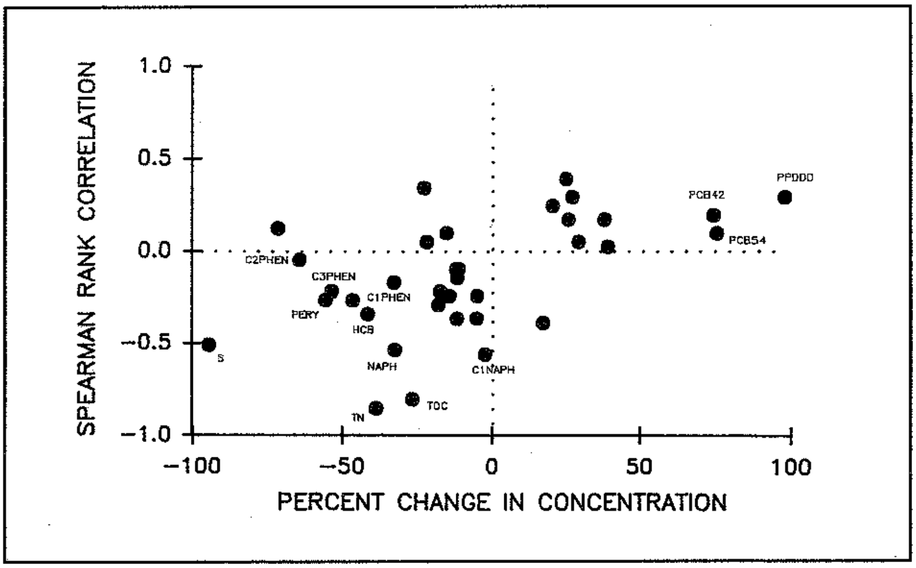


Table 2

Mean growth (mm/28 days) and survival (percent) of Grandidierella japonica following 28-day exposure to sediments from Santa Monica Bay and Newport Bay (laboratory control). N = 2 for each zone in 1989; N = 4 for each zone in 1990 (except transition where N = 2). Data are mean and one standard deviation (in parentheses).

	Growth				Survival			
	mm/28 day		% of control		Percent		% of control	
	1989	1990	1989	1990	1989	1990	1989	1990
Lab control	3.4 (0.1)	2.8 (0.3)	100	100	42.(12)	74.(6)	100	100
Reference	2.2 (0.5)	2.0 (0.8)	65	71	50.(8)	26.(21)	119	35
Transition	2.3 (0.7)	1.6 (0.4)	68	57	20.(19)	31.(29)	48	42
Contaminated	2.9 (0.2)	1.8 (0.4)	85	68	22.(8)	47.(13)	52	64
Canyon	1.8 (0.3)	1.8 (0.4)	53	64	25.(4)	44.(50)	60	59

by less than 25%. The mean concentrations of 16 contaminants, including DDTs, PCBs, and silver, increased.

Ten contaminants had the greatest expected responses (negative correlation and reduced concentration): sulfide, TOC, TN, hexachlorobenzene, naphthalene, C1 substituted naphthalenes, phenanthrene, C2 and C3 substituted phenanthrenes, and perylene (Figure 2). The patterns of change in sulfide and organic enrichment (TOC) at Station 50 matched the temporal change in toxicity (mortality) most closely (Figure 3). The change in TN was similar to that of TOC (data not shown). The concentrations of the hexachlorobenzene, naphthalenes, phenanthrenes, and perylene declined between 1987 and 1988, a period during which amphipod toxicity remained high (Figures 4 and 5).

Figure 3.

Temporal changes in 10-day amphipod (Grandidierella japonica) mortality, dissolved sulfide, and sediment TOC for Station 50 (canyon zone).

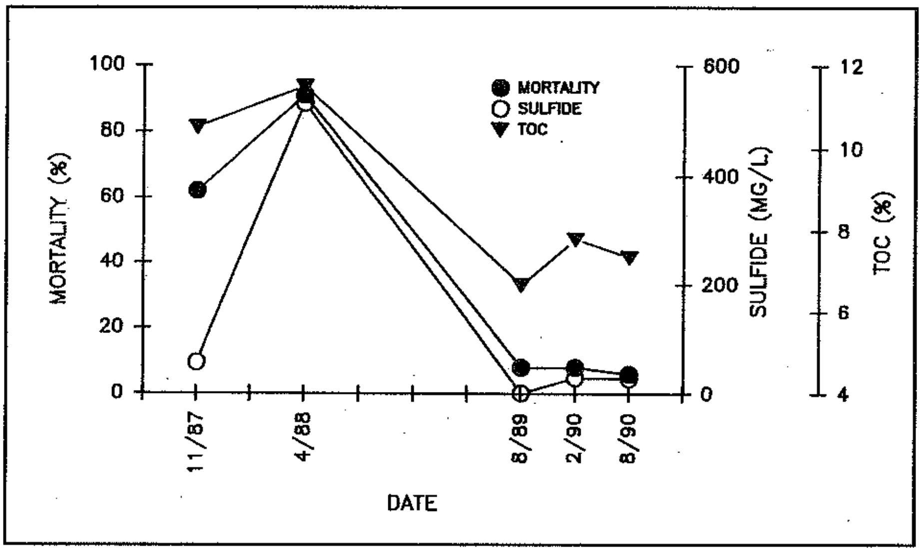
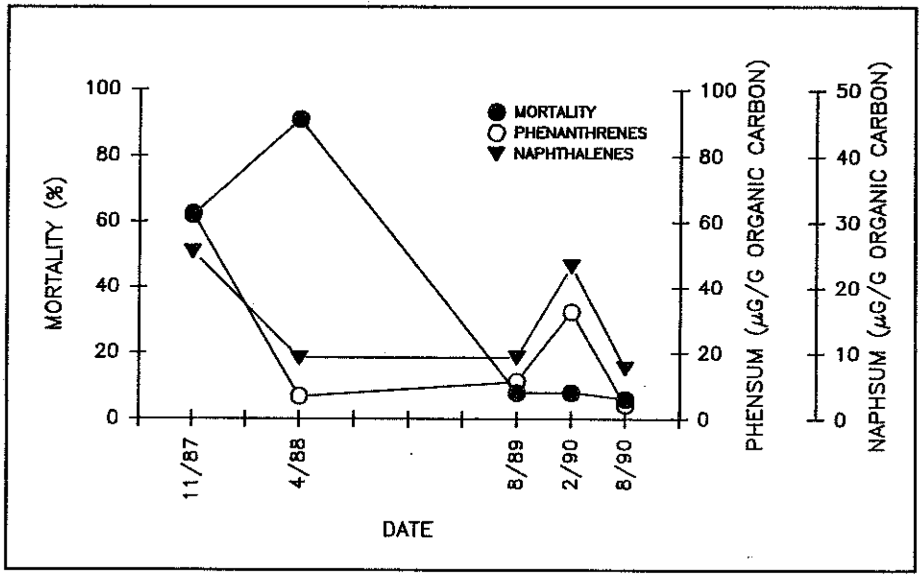


Figure 4.

Temporal changes in 10-day amphipod (Grandidierella japonica) mortality and the TOC-normalized sediment concentrations for naphthalenes (naphsum) and phenanthrenes (phensum) for Station 50 (canyon zone).



Discussion

Laboratory tests with *Grandidierella japonica* documented: 1) a decrease in the toxicity of sediments in the canyon zone (sludge field) between April 1988 and August 1989, and 2) sediment quality at 100 m in the transition, contaminated, and canyon zones in 1989 and 1990 was similar to sediment quality in the reference zone. The reduction in toxicity of sediments in the sludge field occurred during a period of marked changes in animal species composition and abundance (Thompson 1991). This indicates that laboratory toxicity tests with *G. japonica* included relevant sediment qualities that were important to organisms living in Santa Monica Bay.

Changes in the concentration of hydrogen sulfide, TOC, and TN (sediment characteristics associated with organic enrichment) were strongly related to the changes in toxicity (Table 3). Sea urchins exposed to sulfide at concentrations comparable to those measured in canyon zone sediments suffer reduced growth and increased mortality (Thompson *et al.* 1991).

Naphthalenes, phenanthrenes, perylene, and hexachlorobenzene were identified as potential

Table 3

Changes in Santa Monica Bay canyon sediment characteristics and their correlation (Spearman rank correlation coefficient, r_s) to 10-day amphipod survival. Percent change in sediment concentration is the difference between means of the high toxicity group (Station 50 in 1987 and 1988) and the low toxicity group (Station 50 in 1989 and 1990). Negative change indicates a reduction in concentration from the high to the low toxicity group. Reference concentrations are means for Stations 5 and 6 between 1987 and 1990. Hydrocarbon concentrations are expressed relative to the organic carbon (oc) content of the sediment; remaining sediment constituents are expressed on a dry weight basis.

			Concentration		
	r_s	% Change	1987/88	1988/89	Reference
TN (%)	-0.85	-39	1.16	0.70	0.01
TOC (%)	-0.80	-27	10.99	8.02	0.95
C1-Naphthalenes ($\mu g/g_{oc}$)	-0.56	-3	7.24	7.05	0.28
Naphthalene ($\mu g/g_{oc}$)	-0.53	-33	2.72	1.83	0.24
Sulfide (mg/L)	-0.51	-94	293.75	16.07	0.06
Benzo(b)fluoranthene ($\mu g/g_{oc}$)	-0.39	17	4.39	5.14	1.12
Benzo(e)pyrene ($\mu g/g_{oc}$)	-0.36	-6	1.96	1.85	0.49
Benzo(a)pyrene ($\mu g/g_{oc}$)	-0.36	-13	2.42	2.12	0.43
Hexachlorobenzene ($\mu g/g_{oc}$)	-0.34	-42	0.04	0.02	0.03
Clay (%)	-0.29	-18	9.45	7.71	7.36
C1-Phenanthrenes ($\mu g/g_{oc}$)	-0.26	-47	6.78	3.60	0.36
Perylene ($\mu g/g_{oc}$)	-0.26	-56	1.99	0.88	0.98
Benz(a)anthracene ($\mu g/g_{oc}$)	-0.24	-5	2.41	2.29	0.26
Chrysene ($\mu g/g_{oc}$)	-0.24	-15	3.13	2.66	0.54
Zinc (mg/kg)	-0.22	-18	491.00	403.33	60.70
C3-Phenanthrenes ($\mu g/g_{oc}$)	-0.22	-54	15.94	7.35	0.26
Biphenyl ($\mu g/g_{oc}$)	-0.17	-33	15.32	10.24	0.53
C3-Naphthalenes ($\mu g/g_{oc}$)	-0.14	-12	2.59	2.28	0.16
Copper (mg/kg)	-0.09	-12	389.95	341.83	22.53
Benzo(ghi)perylene ($\mu g/g_{oc}$)	-0.09	-12	1.24	1.09	0.39
C2-Phenanthrenes ($\mu g/g_{oc}$)	-0.04	-65	9.66	3.43	0.27
Sand (%)	0.02	39	35.75	49.73	42.44
Silt (%)	0.04	-22	54.75	42.53	50.12
Silver (mg/kg)	0.04	29	24.78	32.01	1.18
Cadmium (mg/kg)	0.09	-16	22.07	18.60	0.27
Aroclor 1254 ($\mu g/g_{oc}$)	0.09	76	4.55	7.99	5.85
Anthracene ($\mu g/g_{oc}$)	0.12	-72	0.88	0.25	0.06
9,10-Diphenylanthracene ($\mu g/g_{oc}$)	0.12	290	0.03	0.14	0.06
p,p'-DDT ($\mu g/g_{oc}$)	0.17	327	0.03	0.14	0.09
C2-Naphthalenes ($\mu g/g_{oc}$)	0.17	38	4.72	6.52	0.35
Dibenz(ah)anthracene ($\mu g/g_{oc}$)	0.17	25	0.37	0.46	0.08
Aroclor 1242 ($\mu g/g_{oc}$)	0.19	74	1.59	2.78	1.03
Pyrene ($\mu g/g_{oc}$)	0.19	107	3.58	7.42	1.37
Fluoranthene ($\mu g/g_{oc}$)	0.24	20	3.80	4.57	0.74
Benzo(k)fluoranthene ($\mu g/g_{oc}$)	0.26	648	0.03	0.24	0.07
p,p'-DDE ($\mu g/g_{oc}$)	0.29	26	1.63	2.07	7.88
p,p'-DDD ($\mu g/g_{oc}$)	0.29	97	0.22	0.43	0.98
Phenanthrene ($\mu g/g_{oc}$)	0.34	-23	2.24	1.72	0.41
o,p'-DDE ($\mu g/g_{oc}$)	0.39	24	0.13	0.17	1.30
o,p'-DDT ($\mu g/g_{oc}$)	0.53	334	0.06	0.28	0.12
2,3-Benzofluorene ($\mu g/g_{oc}$)	0.56	100	0.44	0.89	0.74

contributors to reductions in the toxicity of canyon zone sediments. Hexachlorobenzene concentrations, however, were similar in all four zones (Table 3) and were unlikely to have a significant influence on toxicity.

Draft sediment quality criteria have been proposed for phenanthrene based on equilibrium partitioning theory. Concentrations less than 120 µg/g organic carbon (95% confidence limits: 74-340 $\mu g/g_{oc}$) are "protective" of marine benthic organisms. Phenanthrene concentrations in excess of 3,000 $\mu g/g_{oc}$ produce 50% amphipod mortality in spiked sediment experiments (U.S. E.P.A. 1992). The mean phenanthrene sediment concentration in the high toxicity group in this study was 2.2 $\mu g/g_{oc}$, three orders of magnitude below the toxic levels in spiked sediment tests. The combined concentrations of naphthalenes and phenanthrenes in the high toxicity group was 52 $\mu g/g_{oc}$, approximately half of the draft E.P.A. criterion.

It is unlikely that the low molecular weight PAH compounds were the principal factor in the temporal toxicity changes in Santa Monica Bay sediments. These compounds may have contributed to the reductions, but additional laboratory testing with pure chemicals is needed before conclusions can be drawn. Insufficient data were available to evaluate the influence of perylene on toxicity in this study.

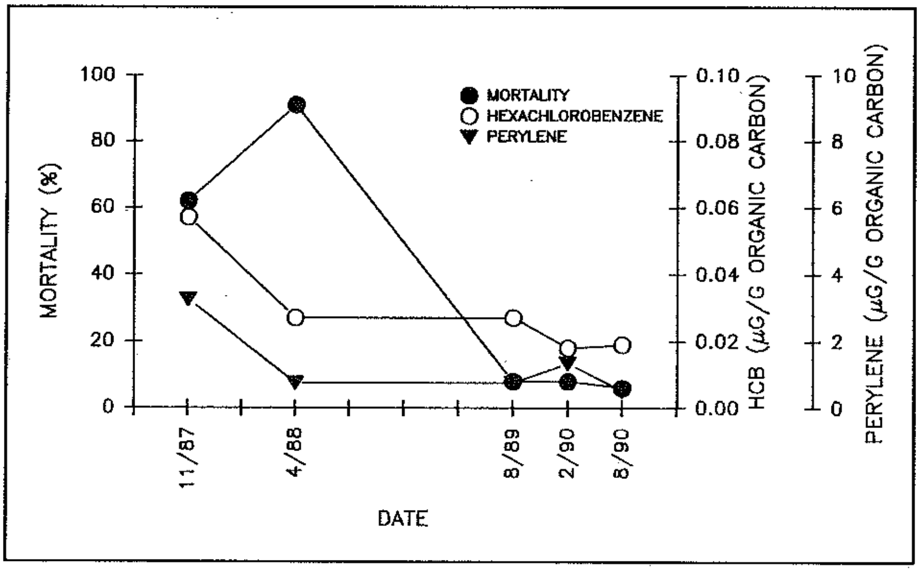
The lack of negative correlations between toxicity and DDT and PCB concentrations was not surprising; TOC-normalized concentrations in the canyon zone were similar to concentrations in the reference area (Table 3).

Conclusions

The reductions in sediment toxicity in 1989 and 1990 did not indicate that the effects of sludge discharge had disappeared in Santa Monica Bay. The assemblages of benthic macrofauna in the canyon, contaminated, and transition zones were still adversely affected in 1990 (Thompson 1991) and the concentrations of many contaminants in the canyon zone were 5-10 times higher than in reference areas (Table 3). We do not know if the differences in composition of macrofauna among the zones were caused by toxic sediments, or if they were the product of variables not represented in the toxicity test (e.g., sediment preferences, interactions among species, etc.). In any case, sediment toxicity tests with *G. japonica* were a less sensitive indicator of sediment quality in Santa Monica Bay than the composition of the benthic macrofauna.

Figure 5.

Temporal changes in 10-day amphipod (Grandidierella japonica) mortality and the TOC-normalized sediment concentrations of hexachlorobenzene and perylene for Station 50 (canyon zone).



Results based on chemical analyses of field sediments are not definitive, but can guide the selection of contaminants for laboratory research. Additional laboratory tests with sediments from Santa Monica Bay are planned to complement the data reported herein. We will use longer exposures to examine effects on reproduction of *G. japonica* and we will examine the effects of organic enrichment and grain size on benthic organisms. This research will provide some of the data needed to determine the relative influence of the various sediment characteristics on the benthic macrofauna in Santa Monica Bay. ■

References

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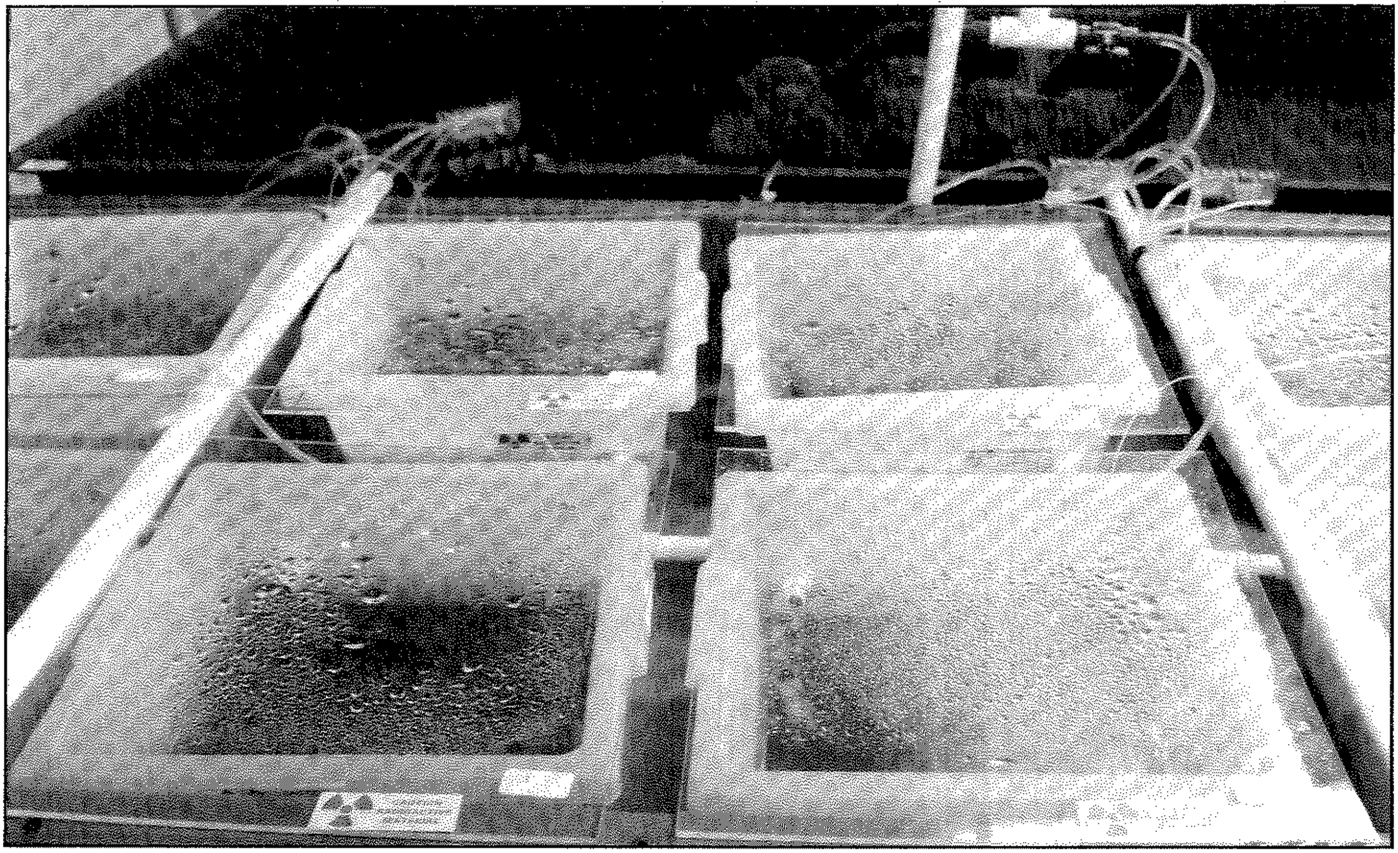

Sediment toxicity tests