Toxicity assessment of sediment cores from Santa Monica Bay

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ABSTRACT

During the summer of 1997, sediment core samples were taken at 25 stations in Santa Monica Bay. Toxicity testing was performed on 4 cm sections of the entire length of each core using purple sea urchin fertilization and amphipod survival tests. The sea urchin test identified sections as being toxic at 6 stations, all located near current or former Hyperion Treatment Plant (HTP) wastewater outfall locations. The amphipod test identified sections from 17 of the stations, scattered throughout the bay and at numerous core depths, as having toxic sediments. Spatial and temporal patterns indicated that toxicity was most strongly associated with the historical disposal of sludge. Many of the sections toxic to the amphipods did not have chemical levels expected to cause toxicity and were in locations where a source of toxicity was not apparent.

INTRODUCTION

The population of the area bordering Santa Monica Bay has increased from less than 200,000 in 1890 to its present level of more than 8 million (Schafer 1989). As the population increased, so did the input of contaminants to the bay. During this time period, inputs of various chemicals from wastewater discharged to the bay have fluctuated, depending upon treatment levels, chemicals common to the time period (i.e., DDTs from the 1940s through early 1970s), and placement of the discharge pipes (near the beach in the early part of the century to 5 miles offshore presently). Relative input from other sources, such as rivers and storm drains, has also changed over time.

Many of the contaminants entering the bay are associated with particles. The particles are deposited into the sediments, which then provide a sink for the chemical inputs and provide a record of inputs over time. These sediment-borne chemicals may, in turn, cause toxicity to marine organisms. Many studies of the chemistry and toxicity of sediments in Santa Monica Bay have been conducted over the past few decades. However, most of these studies focused on the spatial patterns of the surface sediments only. Little information exists that describes sediment conditions before the early 1970s.

In 1994, core samples were collected from several stations in Santa Monica Bay. Chemical analysis of sections of these cores focused on man-made chemicals (linear alkylbenzenes and chlorinated hydrocarbons) in the sediment profiles whose appearance in the sediment record occurred at known intervals (Southern California Coast Water Research Project 1995). Independent dating techniques or other associated analysis such as metals chemistry or toxicity were not conducted. By testing the toxicity of different sections of a vertical sediment core, we can get a view of toxicity back through time at a given location. A coring study conducted near the Sanitation Districts of Los Angeles County's wastewater outfall was successful in documenting changes in both sediment chemistry and toxicity over time (Swartz *et al.* 1991).

In June of 1997, a multidisciplinary study of the temporal trends in the sediments of Santa Monica Bay was initiated. Sediment cores from 25 stations were analyzed for a wide variety of parameters, including toxicity. The objective of the toxicology element of this study was to document changes in sediment toxicity that have occurred over time throughout Santa Monica Bay. Secondarily, we wanted to determine whether changes in toxicity were associated with changes in sediment chemistry. The analyses were focused on several key time points in recent history when different sources and levels of inputs had occurred in the bay. These time points were 1900 (before most of man's inputs), 1945 (as population

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and discharge increased but before operation of the Hyperion Treatment Plant [HTP] 5- and 7-mile outfalls), 1970 (near the peak period of contaminant discharge), 1985 (as treatment practices improved, but before termination of the HTP sludge outfall), and present-day surface sediments. This article reports on the toxicity results for this study and how the toxicity compares to contamination patterns.

METHODS

Field Sampling

Twenty-six sites in water depths between 10 m and 200 m within Santa Monica Bay, California, were sampled in June 1997. Twenty-two of these sites were selected using a stratified random design with the strata corresponding to three concentric circles around the City of Los Angeles' 7-mile treated wastewater outfall. The smallest circle had a radius of 2 km, with an area of 12 km^2. The next circle had a radius of 6.6 km, with an area of 112 km^2 between the 2 km and 6.6 km circles. The area outside the 6.6 km circle encompassed the remainder of Santa Monica Bay and included an area of 361 km^2. In addition to the random sites, four non-random sites were selected to assess conditions around specific sources. The first two were sites that historically have been monitored by the City of Los Angeles about 2 km from the 7-mile outfall (City site designations C6 and E5). The other two sites were located at depths of 25 m immediately offshore from Ballona and Malibu creeks, the major freshwater outlets to Santa Monica Bay.

Within strata, sample sites were selected randomly, but a systematic component was added to the selection process to minimize clustering. The systematic element was accomplished using the sampling design of the Environmental Protection Agency's (EPA's) Environmental Monitoring and Assessment Program (Stevens 1997). In this design, a hexagonal grid is randomly placed over a map of the sampling area, a subsample of hexagons is chosen from this population, and a random site is selected from each grid cell. The hexagonal grid structure ensures systematic separation of the sampling, while the random selection of sites within grid cells ensures an unbiased estimate of condition.

At each site, box core samples were taken using a standard NEL box corer, which has a surface area of 20 cm by 30 cm and a maximum penetration of 60 cm (Rosfelder and Marshall 1967). Subsamples of the upper 2 cm were taken and preserved for textural analysis and foraminifera. Four tubular subcores were taken for (1) geochemistry, (2) toxicity, (3) geochronology, and (4) stratigraphy. Subcores were taken by driving sharpened 8.7-cm outside diameter, 0.3-cm thick plastic tubes into the intact box core sample at a constant speed with an electric-powered actuator. Each tube contained a piston that was firmly attached to the subsampling rig and maintained at the sediment surface. Such an approach leads to virtually no core shortening and preserves the proper sub-bottom depth reference in the subcore. The geochemistry subcore was taken with a polycarbonate liner and placed in a freezer maintained at a temperature below -30° C. The toxicity subcore was taken with a polycarbonate liner and refrigerated. The stratigraphy and geochronology subcores were taken with polybutyrate liners. The geochronology subcore was extruded incrementally on board the ship. Two-cm-long sections of sediment from this subcore were placed in plastic bags and transported to the Skidaway Institute of Oceanography for radioisotope analysis.

Toxicity Testing

Each subcore was sliced into 4 cm sections for toxicity testing. All sections of each core were analyzed since the dating information was not available at the time of testing. Four cm sections were selected to ensure sufficient sediment to make two replicates per section for the amphipod test. Slicing was achieved using a manual, piston-like extruder and a Teflon knife. Approximately 100 mL of homogenized sediment from each section was transferred to each of two 800 mL Tripour beakers for testing with the amphipod *Grandidierella japonica*. Methods were based upon those of the American Society for Testing and Materials (1996). Approximately 600 mL of laboratory seawater was added over the sediment, and then the contents were allowed to equilibrate overnight with aeration. Juvenile amphipods were collected from Upper Newport Bay less than a week before testing. Twenty animals were added to each test chamber. The test duration was 10 d under static conditions at 15º C with constant aeration. Temperature was continually monitored and pH, salinity, dissolved oxygen, and ammonia was measured at the beginning and end of the exposure. At the end of the exposure period, the sediment from each chamber was screened to remove surviving animals. Toxic effects were expressed as a reduction in the percentage of animals surviving relative to the control. With each experiment, a sample of the amphipod collection site sediment was tested as a negative control. Concurrently with each sediment exposure, a reference toxicant series of cadmium was tested.

Sea urchin fertilization tests were performed on samples of the water overlying the sediment in the amphipod exposure beakers. On day 1 of the amphipod exposures, 20 mL of overlying water was removed from each amphipod replicate for use in the fertilization test. Methods

for the exposure followed those of the U.S. Environmental Protection Agency (1995). The test consists of a 20-min exposure of sperm to the samples. Eggs are then added and given 20 min for fertilization to occur. The eggs are then preserved and examined later with a microscope to assess the percentage fertilized. Toxic effects are expressed as a reduction in fertilization percentage. Purple sea urchins (*Strongylocentrotus purpuratus*) used in the tests were collected from the intertidal zone in northern Santa Monica Bay. The tests were conducted in glass shell vials containing 10 mL of solution at a temperature of 15° C. Four replicates were tested from each core section, two from each amphipod replicate. Water quality parameters were measured on a subsample of water from the amphipod beakers. With each experiment, a sample of the amphipod collection site overlying water and a sample of laboratory seawater were tested as a negative control. Concurrently with each sediment exposure, a reference toxicant series of copper was tested.

Radiochemical Analysis

Measurements of ^{210}Pb were used to define sediment accumulation rates at 24 of the coring stations (Alexander *et al*. 1993). Activities of radionuclides were determined using intrinsic germanium detectors, computer-based multichannel analyzers, and ORTEC Maestro software. Sediments were ground to a powder, sealed in 30-mL polypropylene jars and equilibrated for 20 d to allow ingrowth of ^{222}Rn. Samples were counted for 24-72 h. Total ^{210}Pb activity was directly determined by measuring the 46.5-KeV gamma peak (Cutshall *et al*. 1983). Supported levels of ^{210}Pb were determined by measuring the gamma activity of ^{214}Pb (295 and 352 KeV) and ^{214}Bi (609 KeV). Self-absorption corrections for ^{210}Pb were made on each sample following the technique of Cutshall *et al*. (1983).

^{137}Cs activities were determined by measurement of its 661.6-KeV gamma peak (Kuehl *et al*. 1986). ^{137}Cs (half-life 30.0 y) is an impulse tracer (produced from atmospheric nuclear tests), which was first introduced into the environment in significant amounts around 1954 and had peak input in 1963. Penetration depth and peak location in ^{137}Cs profiles provide an independent check on calculated ^{210}Pb accumulation rates.

Chemical Analysis

Subcores for the analysis of metal and organic compounds were stored frozen at –20º C until after the dating information was available. Then the cores were thawed and 1 or 2 cm sections were taken at the depth of the date horizons of interest. These sections were cut in a manner similar to the toxicity samples, except that beveled glass pieces were used for the cutting. Chemical analysis for metals (As, Be, Cd, Cr, Cu, Pb, Hg, Ni, Ag, Zn, and Fe), DDTs, and PCBs were performed on sections for all of the date horizons. Measurements of PAHs were only made on sections from the surface, 1985 and 1970 date horizons. The analyses were conducted as described in Zeng *et al*. (2001).

Data Analysis

The sea urchin results for each section were tested against the laboratory water control for that experiment using t-tests. Sections having mean fertilization significantly lower ($p<0.05$) than the control were deemed to be toxic. Sample pairs having unequal variance were tested using the Mann-Whitney Rank Sum Test. Again, sections having significantly lower fertilization than the control were identified as toxic.

Having only two replicates per section for the amphipod test precluded tests of significant difference relative to the control. In other studies using amphipods, it has become common practice to identify samples as being toxic if they have significantly reduced survival relative to the controls and also show a response that is less than 80% of the control value. For this study, stations were designated as toxic if both replicates had less than 80% of the control survival.

Calculations of the percent area toxic were based upon the area weighting of each station associated with the probabilistic sampling design of the study (Stevens 1997). Area weighting calculations for 1970 were made with the knowledge that due to lack of penetration by the sampler, five stations did not extend back to 1970. Since these stations were lost in a non-biased manner, it was assumed that their loss did not greatly affect the accuracy of the percent area calculations.

Correlation analysis was performed on the chemistry and toxicity data to identify what constituents might be causing toxicity. Because the toxicity data were found to be not normally distributed, the non-parametric Spearman Rank Correlation was used for analysis.

Sediment chemistry and toxicity data were also compared using effects range-median (ERM) values. The ERM value for each chemical is the median concentration associated with effects based upon the results of laboratory and field tests conducted throughout the country (Long *et al*. 1995). For comparison of the joint effects of chemicals, mean ERM quotients were calculated. The concentration of each chemical was divided by its ERM value. This ERM quotient was then averaged for all of the chemicals in each section. Since total polyaromatic hydrocarbons (PAHs) were not measured on all of the sections, their ERM

quotient was not calculated. Mean ERM quotients were only calculated for sections where data for metals, total DDTs, and total PCBs were present.

RESULTS

We successfully tested 169 separate core sections from the 25 stations with each species in three sets of exposures. The fertilization percentage in the controls was, in all cases, above the minimum standard of 70%. Survival in the reference site sediment for one amphipod test was above the minimum of 90%, but was slightly below (89% and 88%) for the other two.

Water quality parameters (dissolved oxygen, total ammonia, pH, and salinity) were within acceptable levels for all exposures. The results of all reference toxicant exposures for both species were within the expected ranges.

Spatial Distribution of Toxicity

Two stations, those closest to HTP's former sludge outfall, had surface sediments that caused reductions in sea urchin fertilization (Figure 1). An additional four stations had reduced fertilization success in at least one section below the surface. All of these stations were located within a 6.6 km radius of the end of the former sludge outfall. The sea urchin fertilization test identified 1.4% of the bay as containing toxic sediments.

The amphipod test also identified two stations as toxic at the surface (Figure 2). These stations were located at the extreme north and south ends of the sampling area. Another 15 stations demonstrated toxicity in at least one section below the surface. The amphipod test detected toxicity at or below the surface at 68% of the stations located throughout the bay. The amphipod test identified 14.5% of the bay as containing toxic sediments.

Temporal Distribution of Toxicity

Radiological dating was successfully completed on all but one of the cores (Station 30) for which a toxicity subcore was collected. The sea urchin test found toxicity at each of the target years for Stations 10 and 53, but only a few sections were toxic at the target horizons of other stations (Table 1; Figure 1). Subsurface toxicity to the amphipods was variable both with time and location (Figure 2). Sediment accumulation rates differed throughout the bay. Therefore, the date associated with a given depth was highly variable between cores. In most cases, the cores did not penetrate deeply enough to include the 1900 time horizon. In more than half of the cores, the first section of the core contained both the 1997 and 1985 time points. For Stations 51 and 52, the uppermost section also included 1970.

A negative correlation was found between amphipod survival ($r=-0.160$, $p=0.038$) and sea urchin fertilization ($r=-0.360$, $p<0.001$) with depth, indicating that deeper sediments were more toxic than surficial sediments. No consistent pattern for the profiles was observed between stations, however. Station 10, located near the now-unused HTP 7-mile sludge outfall, showed the most toxicity throughout the core of any station. All of the sections were toxic to the sea urchin and all but the surface and 1945 sections were toxic to the amphipods (Figure 3). At Station 16, more offshore and north of Station 10, very little toxicity was detected (Figure 3). The date associated with the bottom of this core is before 1900. Station 50 shows a pattern for the sea urchin test of no toxicity over the past 15 years, but high toxicity in the 1970s (Figure 3). Station 53, located near the terminus of the sludge outfall, showed toxicity throughout the core. Because of the high deposition rate of this area, the bottom section of the core does not reach to 1970 (Figure 3).

Because the surface section of so many cores contained both the 1997 and 1985 horizons, no comparisons between these dates can be made. Since the 1970 sediments were deposited at the period of highest input of contaminants of any of the time horizons of focus, it is of interest to compare them to the present-day sediments. The area of the entire bay that was found to be toxic for the 1970 horizon was 13.1% for the amphipods and 4.8% for the sea urchins, compared with 14.5% and 1.4% at the surface.

On the maps for 1945 and 1970, the historical location of the HTP outfalls is represented. On the 1945 maps, only the 1-mile pipe was in operation, yet toxicity was identified for both sea urchin and amphipod tests in the vicinity of where the 5- and 7-mile pipes were later located (Figures 1 and 2). In 1900, the small amount of discharge was located near the beach, yet toxicity was detected at this horizon for Station 10, near the future location of the 5- and 7-mile outfalls.

Between-species Comparisons

Of the 27 sections that were identified by the sea urchin fertilization test as being toxic, 17 were also identified as toxic by the amphipod test. A positive correlation ($r=0.460$, $p<0.001$) was also found when comparing the toxic response between the two species. The amphipod test found 13 more core sections to be toxic than the sea urchin test and toxicity at 11 more stations than the sea urchin test. However, the degree of response was much greater in the sea urchin test (Figure 4). While the amphipod test had only 1 section that caused less than 40% survival relative to the control, the sea urchin test had 19. The lowest percent

FIGURE 1. Stations designated as toxic or non-toxic by the sea urchin fertilization test on core sections at date horizons of interest. Stations marked as toxic had fertilization success that was significantly lower (p>0.05) than the control response. Dashed ring is demarcation of near outfall stations.

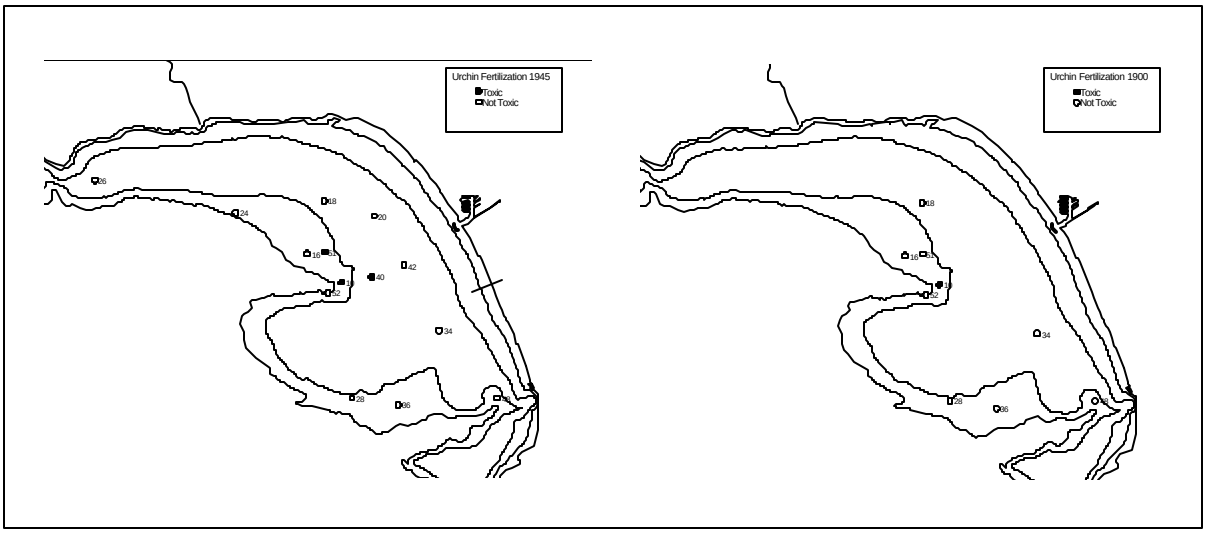


FIGURE 2. Stations designated as toxic or non-toxic by the amphipod survival test on core sections at date horizons of interest. Stations marked as toxic had survival in both replicates that was less than 80% of the control response. Dashed ring is demarcation of near outfall stations.

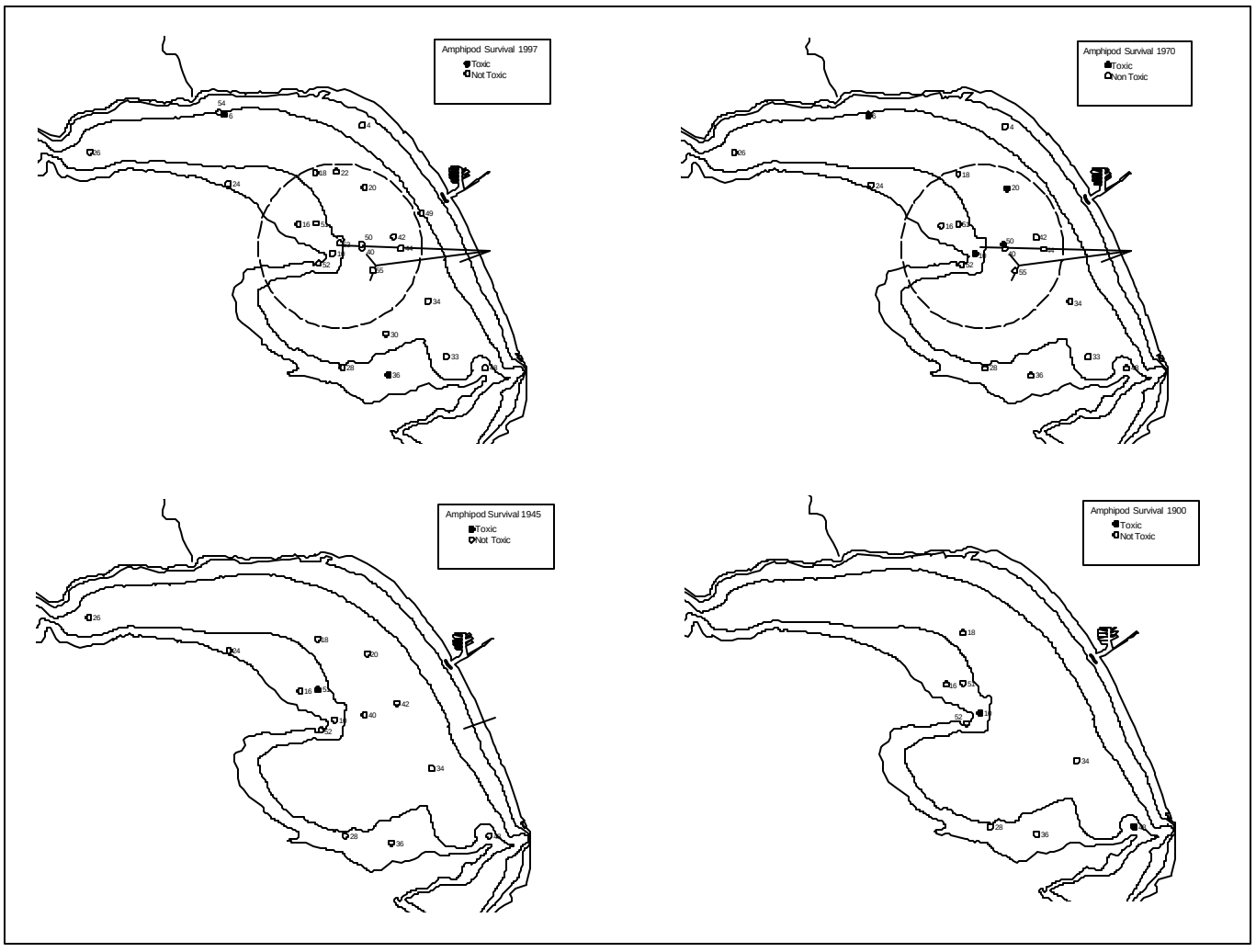


TABLE 1. Results of toxicity testing for urchin fertilization and amphipod survival at selected time horizons of core sections. Sea urchin data are expressed as percent successful fertilization normalized to the control response. Amphipod data are expressed as percent survival normalized to the control response. Data enclosed in boxes represent sections containing multiple date horizons.

Station Number	Sea Urchin					Amphipod				
	1997	1985	1970	1945	1900	1997	1985	1970	1945	1900
4	115		107			91		106		
6	98	94	83			68[a]	71[a]	66[a]		
10	28[a]	18[a]	11[a]	11[a]	52[a]	97	68[a]	55[a]	76	73[a]
16		102	101	102	102	82		90	79	88
18	119	112	118	118	117	100	94	97	111	106
20	100		99	102		92		71[a]	73	
22	118	115				83	60[a]			
24	111		115	117		100		102	94	
26	82		101	96		83		75	83	
28	101		93	99	96	86		73	89	76
30[b]	111					106				
33	99		100			86		86		
34	101		102	102	102	88		104	96	84
36	117		113	113	117	38[a]		108	111	85
40	100	95	73	25[a]		99	82	76	88	
42	101		101	98		96		99	96	
44	95	99	101			84	90	82		
48	118		117	118	113	97		97	106	43[a]
49	117	117				85	91			
50	101	95	23[a]			93	84	73[a]		
51	102			48[a]	99	82			71[a]	88
52	117			115	115	74			108	72
53	47[a]	22[a]				78	47[a]			
54	101	100				89	81			
55	100	93	71[a]			106	93	79		

[a]Identified as toxic. Either significantly different from the control for the sea urchin fertilization test or less than 80% of control survival for the amphipod test.
[b]Unable to date core below surface level.

survival relative to the control for the amphipod test was 38%, while six sections resulted in less than 10% fertilization relative to the control for the sea urchin test.

DISCUSSION

Little toxicity was detected in the surface layers by either species. This result matches well with recent studies of surface sediment toxicity in Santa Monica Bay. The Southern California Bight 1994 Pilot Project (SCBPP) found no toxicity using exposures of the amphipod *Ampelisca abdita* to whole sediment (Bay *et al.* 1998). None of the stations in Santa Monica Bay tested using the amphipod *Eohaustorius estuarius* for the Bight'98 project were identified as being toxic (Bay *et al.* 2000, *in prep*). The SCBPP identified five stations where interstitial water samples were toxic to the purple sea urchin development test (Bay *et al.* 1998). All of the stations were to the south of the HTP outfall. The sea urchin development test is generally considered to be a more sensitive test than the fertilization test, and contaminants in the interstitial water would be expected to be more concentrated than those found in the overlying water that was tested in the current study.

The percent area of the bay found to be toxic to the sea urchin test was relatively small and concentrated near the HTP outfall. A previous sediment toxicity study using area weighting methods focused on enclosed bays, harbors, and estuaries (Anderson *et al.* 1997). That study found 58% of the study area in selected Southern California bays, estuaries, and lagoons was significantly toxic to the amphipod *Rhepoxynius abronius*. An examination of a large set of

FIGURE 3 A-D. Depth profiles of toxicity test results for stations 10, 16, 50 and 53. Toxicity test results are expressed as percentage of control response (fertilization success for the sea urchins and survival for the amphipods). Horizontal dashed lines indicate the depth of the selected time horizons as identified through radiological dating.

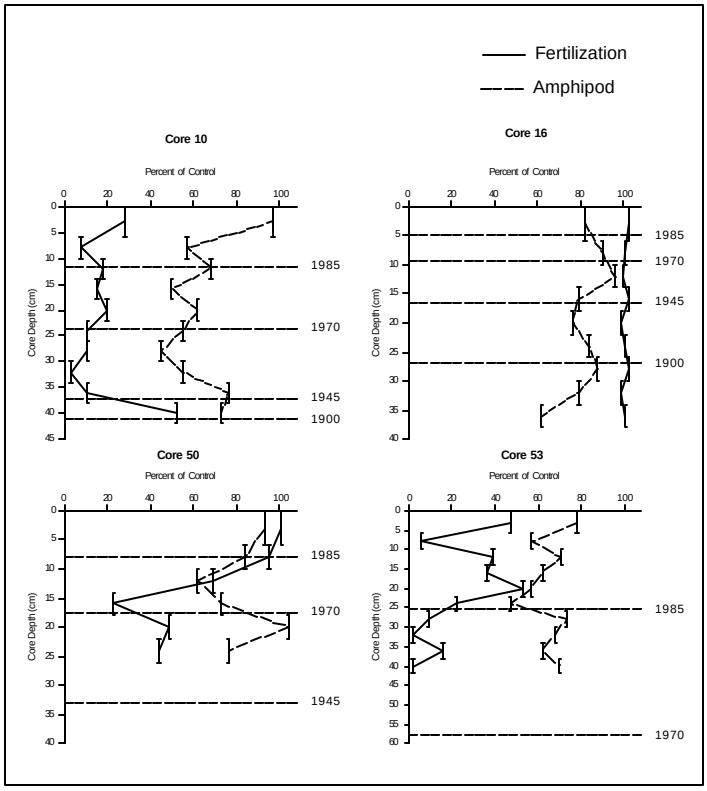
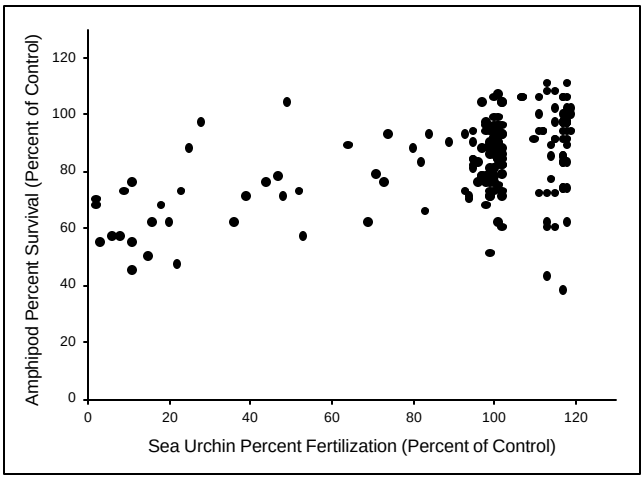


FIGURE 4. Comparison of the response of the amphipod and sea urchin tests to the same core sections.



toxicity data found that 11% of the area within selected bays and estuaries on the Pacific, Gulf, and Atlantic coasts of the United States were toxic to the amphipods *Ampelisca abdita* or *Rhepoxynius abronius* (Long *et al.* 1996). Our estimation of approximately 14.5% for all of Santa Monica Bay is consistent with that study.

Only one other study has analyzed cores from Santa Monica Bay for toxicity (Swartz *et al.* 1991). Two stations were analyzed in that study, one in the southern part of the bay (distant from our stations) and one in the northern part of the bay (close to our Station 26). No toxicity to the amphipod *Rhepoxynius abronius* was found at any core depth at either station. We also found no toxicity to either species at Station 26. The previous study also analyzed two cores from near the Sanitation Districts of Los Angeles County's outfall and found a pattern of little toxicity near the surface, high toxicity in the middle of the core, and lower toxicity at the bottom of the core. This pattern was in agreement with the chemistry values and input history of the area for that study.

The most consistent pattern of toxicity was observed at three stations (10, 50, and 53) located within 2 km of the former HTP sludge outfall. The sections within these cores accounted for 89% of the sections identified as toxic by the sea urchin test and 42% of the sections identified as toxic by the amphipods for the entire study. The cores for Stations 50 and 53 extend back only to the mid-1950s and 1970s, respectively, so the toxicity observed through their length is not surprising. However, the core for Station 10 extends to before 1900. Elevated concentrations of total DDT and total PCB were also found at the 1900 horizon for Station 10. The HTP 5- and 7-mile outfalls were not operational until the late 1950s, and DDT was not manufactured and dumped in the area until 1947 (Schafer 1989).

Several explanations may account for the presence of toxicity and elevated chemistry values in sediments near the outfall location but dated as earlier than 1950. The accumulation rates upon which the date model is based may have been in error; samples may have been contaminated during subsampling; or contaminants may have migrated downward in the sediment column by biological or geochemical processes. Due to the nature of radiological dating techniques, errors in the accumulation rate would tend toward rates that are too high as opposed to too low. Therefore, if the actual accumulation rate was lower than calculated, the actual date would be even older than reported, making the problem worse. This cannot then be the correct explanation. Since the toxicity sections and chemistry sections came from entirely different subcores, sectioned months apart, it is unlikely they could have each been contaminated in such a manner as to allow the toxicity and chemistry data

to match up as well as they do. Therefore, the likely explanation is migration of contaminants downward in the seabed. Biological mixing is a probable cause, especially considering that we noted the presence of animals in the cores that in some cases stretched through tens of centimeters of depth. A study in New Zealand found a similar occurrence of organochlorine pesticides at a core depth dated before such chemicals were introduced to the area (Hendy and Peake 1996). These researchers offered an explanation of bioturbation and/or physical mixing for this anomaly.

Other than the wastewater outfalls, the major source of contaminants to Santa Monica Bay is stormwater discharges. Three stations in our study were located near the largest sources of stormwater input, Ballona and Malibu creeks. No toxicity was observed for either species in any section of Station 49 near Ballona Creek. For the Malibu Creek area, Station 54 had only one section that indicated toxicity to the amphipod test and none to the sea urchin test. However, at Station 6 toxicity to the amphipods was observed throughout the core; again, no toxicity was found for the sea urchins. The differing results given the proximity of Stations 6 and 54 and lack of agreement between the two species indicate that something other than influence from Malibu Creek may be causing the toxicity. A previous study of the sediment quality offshore of Ballona and Malibu creeks found no toxicity in the sediments near either creek (Bay *et al.* 1997).

The toxicity results showed a partial correspondence with patterns in sediment contamination (Zeng *et al.* 2001). The DDTs were the most prevalent contaminants present. Nearly all of the sections analyzed had detectable levels of DDT or its metabolites, and the highest concentrations were found near the HTP outfalls. The highest concentrations of other contaminants were also present at stations located in the vicinity of the outfalls, primarily at Stations 10 and 53. This general pattern corresponds closely with the toxicity data, which also indicated that the greatest toxicity to amphipods and sea urchins was present at Stations 10 and 53 (Figure 3). Contaminant concentrations in some sections were above ERM sediment quality guidelines for eight metals, PCBs, and DDTs, indicating that the sediment contaminant concentrations were likely to be biologically significant (Long *et al.* 1995).

The DDT contamination, though widespread, did not show a strong relationship with toxic effects. No significant correlation was found between amphipod survival or sea urchin fertilization and DDT concentration (Table 2). Furthermore, although 34 core sections exceeded the DDT ERM (including some with total DDT concentrations an order of magnitude above the ERM), less than one-quarter of these sections caused substantial toxicity to the amphipods (Figure 5). Some populations of *Grandidierella japonica* can be quite tolerant to DDT (Swartz *et al.* 1994). Therefore, while high levels of DDT were found throughout the bay, it is unlikely that it was responsible for the amphipod toxicity found in this study.

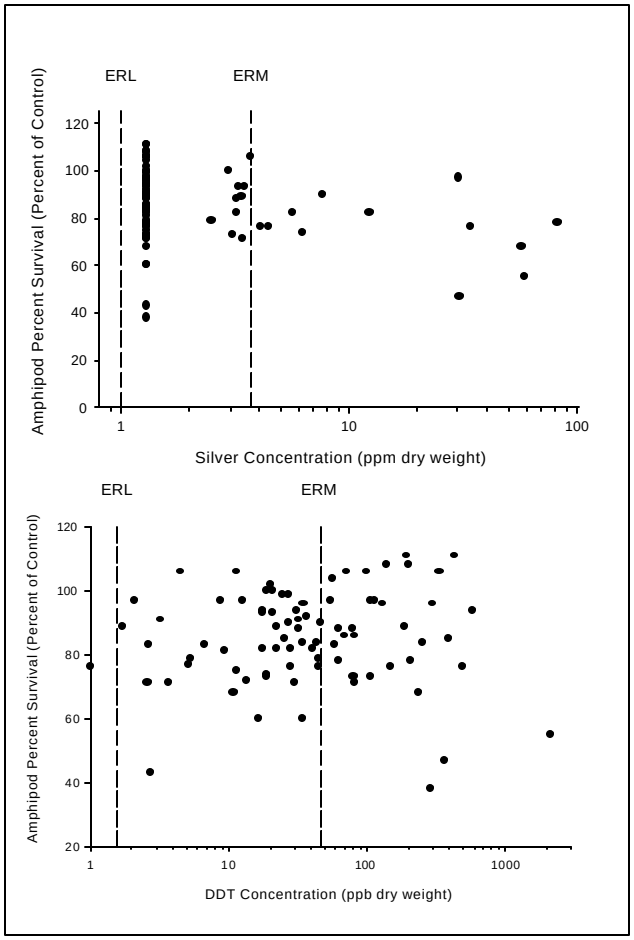
Toxicity to sea urchins showed a stronger relationship to sediment contamination gradients than did amphipod survival. Significant ($p<0.05$) negative correlations were found between sea urchin fertilization and the sediment concentrations of zinc, silver, copper, cadmium, total PCBs, and total PAHs (Table 2). Amphipod survival was negatively correlated only with silver. The sea urchin results were also consistent with responses predicted from the comparison of results to ERM sediment quality guidelines. As shown in representative plots for silver and PCB (Figure 6), sea urchin toxicity was always present in the most contaminated samples and toxicity was usually detected in samples containing contaminant concentrations greater than the ERM. The severity and frequency of amphipod toxicity showed little difference relative to ERM values (Figure 5).

TABLE 2. Spearman rank correlations for toxicity results and sediment chemistry constituent concentrations for core sections. Also included are the effects range-median (ERM) values, range of ratios between the ERM and chemistry values, and the number of sections exceeding the ERM value.

	Sea Urchin Fertilization		Amphipod Survival		Effects Range Median		
Constituent	r_s	p_s	r_s	p_s	Value (mg/L)	Ratio	Number Above
Zn	-0.249	0.020*	-0.082	0.448	410	0.0-3.5	4
Ag	-0.426	<0.001*	-0.242	0.024*	3.7	0.0-22.3	12
Ni	-0.047	0.663	-0.176	0.103	51.6	0.2-3.8	4
Hg	-0.211	0.05	0.036	0.739	0.7	0.0-7.1	7
Pb	-0.088	0.417	0.034	0.751	2.8	0.0-2.4	3
Cu	-0.37	<0.001*	-0.12	0.266	270	0.0-3.5	5
Cr	-0.191	0.076	-0.037	0.733	370	0.1-3.0	4
Cd	-0.444	<0.001*	-0.142	0.188	9.6	0.0-8.2	6
Total PCB	-0.218	0.048*	0.014	0.897	0.18	0.0-24.0	5
Total DDT	-0.09	0.419	0.095	0.39	0.046	0.0-46.6	34
Total PAH	-0.292	0.028*	-0.244	0.067	44.8	0.0-0.1	0
Percent Sand	-0.094	0.484	0.055	0.686			
TOC	-0.147	0.182	-0.173	0.115			

* Indicates significant correlation ($p<0.05$).

FIGURE 5. Amphipod survival relative to silver and total DDT concentration of the core sections. The effects range-low (ERL) and effects range-median (ERM) values for each constituent are plotted for reference.

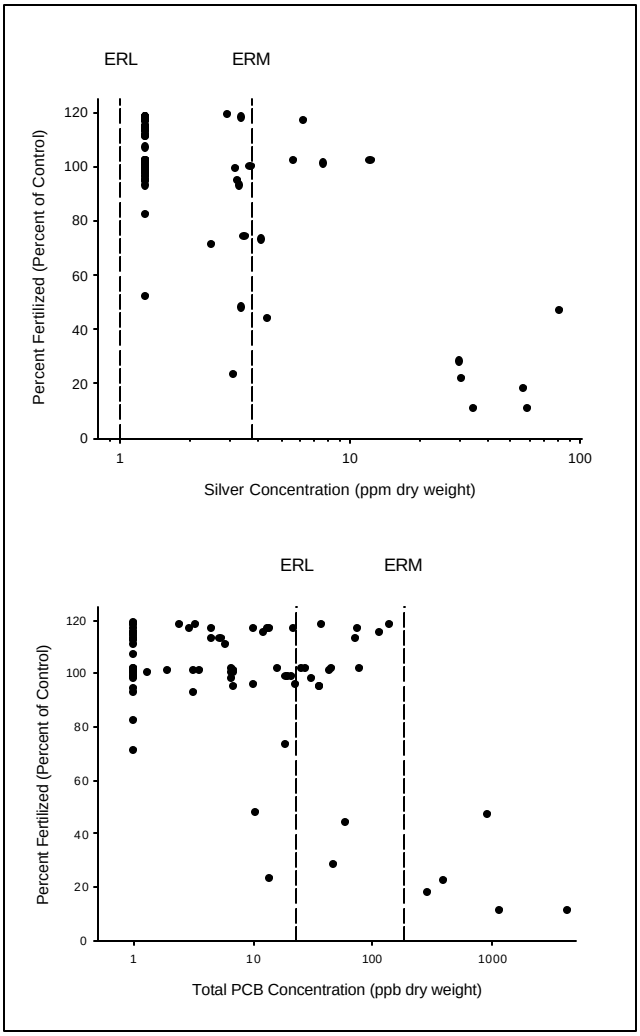


The comparison of the mean ERM quotient (a measure of cumulative contamination effects) to toxicity produced similar results. The mean ERM quotients were not efficient at predicting toxicity to the amphipods. Only 38% (3 out of the 8) of the sections that had ERM quotients greater than 1 were found to be toxic. A survey of amphipod sediment toxicity data collected from locations on the Atlantic, Gulf, and Pacific coasts found that when the mean ERM quotient exceeded 1 at a given site, that site was found to be marginally toxic or highly toxic in more than 75% of the cases (Long *et al.* 1998). Of the remaining 73 sections for which we have toxicity data and were able to calculate a mean ERM quotient, 66 had a mean ERM between 0.1 and 1. Of these, only 12% were found to be toxic, compared to approximately 48% in the Long *et al.* (1998) data set. The mean ERM quotient was an effective predictor of the sea urchin toxicity results, however.

A few caveats regarding comparisons of the toxicity and chemistry data should be noted. The sections used for toxicity analysis were 4 cm long, whereas the chemistry sections were only 2 centemeters. Therefore, the toxicity samples may have had different contaminant loads than indicated by the chemistry data. This may account for some of the unexplained toxicity detected in our samples. It should also be noted that while correlations were found between several chemical constituents and sea urchin toxicity, these chemical constituents also were highly correlated with one another, making it difficult to identify an individual chemical as the causative agent.

The two species used in this study appear to be responding to the sediment samples in a different manner. The sea urchin fertilization test, while identifying fewer sections as toxic, had a stronger response, as evidenced by the number of sections having very low fertilization success. This wide range of response was coupled with strong

FIGURE 6. Fertilization success of the sea urchins relative to the silver and total PCB concentration of the core sections. The effects range-low (ERL) and effects range-median (ERM) values for each constituent are plotted for reference.



correlations with several chemical constituents. Tests of pore water with invertebrate gametes are generally found to be more sensitive than amphipod sediment tests (Long *et al.* 1998). However, for our study, the sea urchin test was performed on overlying water, where lower concentrations of contaminants would be expected than in pore water. The amphipod test identified more sections as toxic, but had a smaller range of response and only correlated with one chemical constituent. Many of the sections identified as toxic by the amphipods did not correspond to contamination patterns. It is a possibility that the low replication and the variability associated with the amphipod testing led to some sections being falsely identified as toxic. The reduced amphipod survival may also have been the result of unmeasured chemical constituents or variation in physical characteristics of the sediment such as grain size. Sediment grain size has been found to affect survival in another amphipod species (DeWitt *et al.* 1988). All of these factors, when taken together, may help to explain why the two species' toxicity results match well only at the stations located closest to the former sludge outfall and having the highest concentrations of contaminants.

Based upon the spatial and temporal pattern of toxicity, it appears that the most likely source of toxicity is the historical inputs from the HTP 7-mile sludge outfall, rather than the 5-mile wastewater outfall. This is consistent with the cluster of toxicity near the sludge outfall and greater toxicity at depth in the cores as opposed to near the surface. Following the termination of use of the HTP 7-mile sludge outfall in November of 1987, SCCWRP conducted sediment toxicity tests at stations near the end of the pipe each year until 1990. These tests found that the sediments went from being highly toxic at the start of the project to marginally or non-toxic by the end (Southern California Coastal Water Research Project 1992). This finding matches with the pattern observed in the present study for Station 10, but not as well for Station 53, where we found toxicity in the surface sediment for the current study. Remixing of the sediments may account for this discrepancy.

With the termination of the sludge outfall and improvements in wastewater treatment, it would be expected that the percent area of toxicity would have declined between 1970 and 1997. The changes were not dramatic for the bay as a whole, with sea urchin toxicity decreasing from about 5% of the area in 1970 to approximately 1% of the surface sediments. The amphipod test actually identified slightly more toxic area in 1997 sediments compared to 1970 sediments. This small increase could be an artifact of the area weighting calculation, since four stations were identified as toxic in 1970 and only two were identified as toxic in 1997 (although these two stations had high area weights). The overall assessment for the bay is that the toxic area near the sludge outfall has decreased considerably, while only a small percentage of the rest of the bay has toxicity today, similar to the conditions found in 1970.

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