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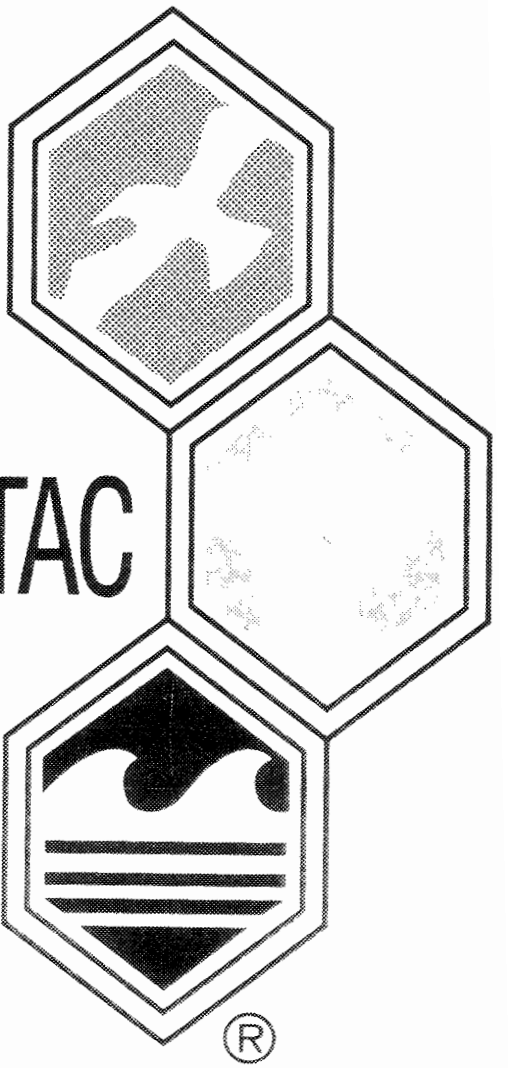

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SEDIMENT TOXICITY, CONTAMINATION AND AMPHIPOD ABUNDANCE AT A DDT- AND DIELDRIN-CONTAMINATED SITE IN SAN FRANCISCO BAY

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Abstract—Sediment toxicity to the amphipod *Eohaustorius estuarius*, sediment contamination, and the abundance of amphipods were examined along a contamination gradient in the Lauritzen Channel and adjacent parts of Richmond Harbor, California. Dieldrin and DDT were formulated and ground at this site from 1945 to 1966. Sediment contamination by both dieldrin and the sum of DDT and its metabolites (ΣDDT) was positively correlated with sediment toxicity and negatively correlated with the abundance of amphipods excluding *Grandidierella japonica*. The maximum dieldrin and ΣDDT concentrations in toxic units were 0.018 and 9.43, respectively, indicating that ΣDDT was the dominant ecotoxicological factor. Concentrations of PAHs, PCBs, and metals were not sufficient to cause appreciable toxicity, except at one PAH-contaminated station. Relations between ΣDDT, sediment toxicity, and amphipod abundance are similar at three ΣDDT-contaminated sites. The 10-d LC50 for ΣDDT in field-collected sediment was 2,500 μg/g organic carbon (OC) for *Eohaustorius estuarius* in this study, 1,040 μg/g OC for *Rhepoxynius abronius* exposed to Palos Verdes Shelf, California, sediment, and 2,580 μg/g OC for *Hyalella azteca* exposed to sediment from a freshwater stream system near Huntsville, Alabama. The threshold for 10-d sediment toxicity occurred at about 300 μg ΣDDT/g OC. The abundance of amphipods (except *Grandidierella japonica*) was reduced at ΣDDT concentrations >100 μg/g OC. Correlations between toxicity, contamination, and biology indicate that acute sediment toxicity to *Eohaustorius estuarius*, *Rhepoxynius abronius*, or *Hyalella azteca* in lab tests provides reliable evidence of biologically adverse sediment contamination in the field.

Keywords—Sediment toxicity DDT Dieldrin San Francisco Bay Amphipods

INTRODUCTION

A portion of San Francisco Bay, California, including the Lauritzen Channel, Santa Fe Channel, and part of Richmond Inner Harbor, has been designated by the U.S. Environmental Protection Agency (EPA) as a Superfund site because of historic contamination by organochlorine compounds [1]. Property adjacent to the Lauritzen Channel was used by various corporations to formulate and grind DDT and dieldrin from approximately 1945 to 1966 [2]. This activity resulted in substantial contamination of the soils and sediments of the Lauritzen Channel and adjacent Santa Fe Channel and Richmond Harbor. Remedial action in 1990 removed the worst contamination from the embankment of the Lauritzen Channel, where in some areas the DDT concentration in the soil was virtually 100%. Despite this action, the sediments in the area remained contaminated by DDT and dieldrin, with a gradient of decreasing concentrations from the Lauritzen Channel to the Santa Fe Channel to the Richmond Harbor [2].

The first objective of the present research was to examine relations between sediment contamination by DDT and dieldrin; acute sediment toxicity to the amphipod, *Eohaustorius estuarius*; and the field abundance of amphipods at nine sites in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor area. The second objective was to identify the lowest DDT and dieldrin concentrations that were associated with effects on the survival of amphipods in lab toxicity tests and the abundance of amphipods in the field. The final objective was to evaluate the relative contribution of DDT, dieldrin, PAHs, PCBs, and metals to sediment toxicity and effects on amphipod abundance in the study area.

METHODS

Sediment and macrobenthos samples were collected from October 7 to 10, 1991, at a total of nine stations: four in the Lauritzen Channel (LC1–4), two in the Santa Fe Channel (SF5 and 6), and three in the Richmond Harbor Channel (RH7–9) (Fig. 1). Five sediment grabs were taken with a 0.1-m^2 van Veen grab at each station except LC1, where eight grabs were taken. The following cores were taken from each grab: three (8 cm diameter, 10 cm deep, plastic corer) for macrobenthos; one (7.6 cm, 10 cm deep, glass corer) for sediment toxicity; and one (7.6 cm, 10 cm deep, glass corer) for dieldrin, DDT, and total organic carbon (OC) analyses. Sediment in the three macrobenthos cores from each grab was combined and sieved through stacked 1.0- and 0.5-mm-mesh screens. Material retained on the screens was preserved in the field in 10% buffered formalin. The toxicity and chem-

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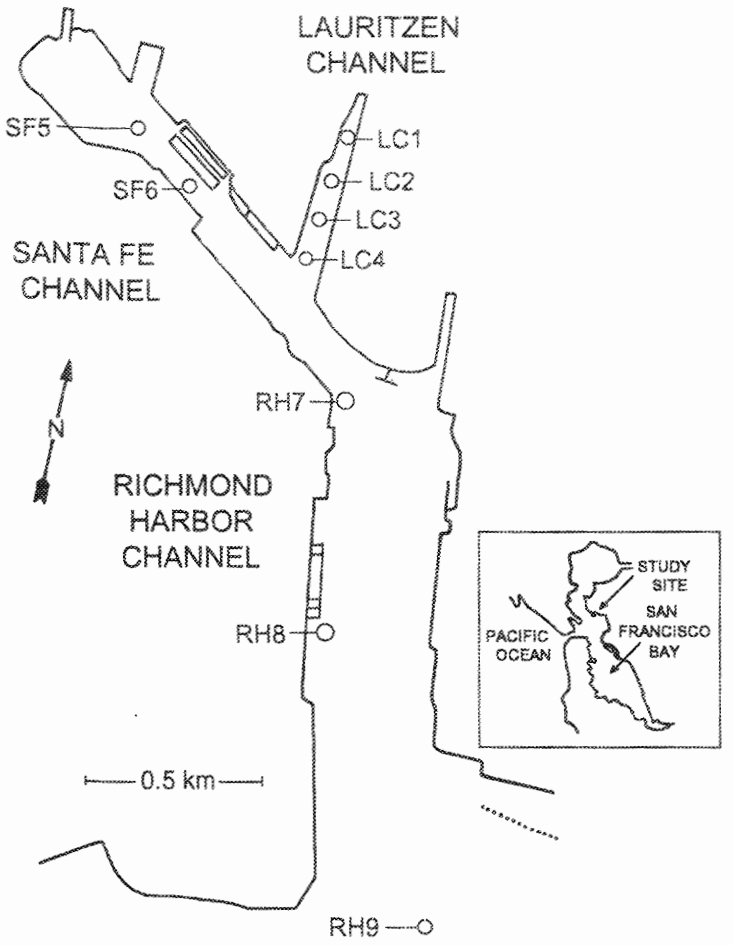

Fig. 1. Location of sediment sampling stations in the Lauritzen Channel, Santa Fe Channel, and Richmond Inner Harbor.

istry samples were placed in glass jars and transported in ice chests (~4°C) to the lab for analyses.

Macrobenthos specimens were transferred in the lab to 70% ethanol. Amphipods were identified to the species level and enumerated. The combination of three macrobenthos cores from each grab represented the fauna from an area of 0.015 m^2. Data presented in this report are for the combination of specimens retained on the 0.5- and 1.0-mm screens.

Ten-day sediment toxicity tests were conducted with the amphipod *E. estuarius* according to the American Society for Testing and Materials (ASTM) [3] guide. On October 14, 1991, each sediment sample was mixed by hand with a spatula. Large pieces of shell or other debris were removed, and approximately 175 ml sediment was placed in a 1-L glass beaker to form a 2-cm-deep layer. The sediment was covered by 775 ml filtered, 28-ppt seawater, which was aerated through a 1-ml glass pipette. The beakers were placed in a 15°C water bath.

On October 9, 1991, approximately 2,000 *E. estuarius* (3–5 mm total length) were collected from the shallow subtidal area along the northern bank of Beaver Creek at Ona Beach State Park, Oregon. The amphipods were counted and placed in glass bowls containing 2-cm-deep layers of 0.5-mm-screened sediment from the Ona Beach site. The bowls were submerged in a flowing seawater table until the initiation of the experiment.

On October 15, 1991, the amphipods were recovered from sediment by sieving the contents of the bowls through a 1.0-mm screen. Amphipods were sequentially sorted into lots of 20 individuals and recounted, then each lot was placed into a randomly assigned 1-L experimental beaker. Most amphipods quickly swam or sank to the bottom and buried in the sediment. After 10 d the contents of the beakers were sieved through a 0.5-mm screen and the survivors counted. Individuals that appeared to be dead were examined under a microscope for signs of life. Missing amphipods were counted as dead [4]. Survivors were subjected to a behavioral test to see if they were able to rebury in clean sediment [3]. Because 99.2% of the survivors were able to rebury, we report only the mortality data.

A toxicity test beaker was prepared for each sediment sample from each grab. Thus, there were eight replicates for station LC1 and five replicates for the other eight stations (LC2–RH9). In addition, negative and positive controls were

conducted. Negative controls were 10-d mortality tests in five beakers with amphipod collection site (Ona Beach) sediment. The QA/QC requirement for the negative control is ≤10% mean mortality [3]. Positive controls were 4-d mortality tests in an unreplicated dilution series of cadmium (spiked as $CdCl_2$) in 28-ppt seawater (without sediment) at nominal concentrations of 30, 15, 7.5, 3.8, 1.9, 0.94, and 0.00 mg Cd/L. The QA/QC requirement for the positive control is that the Cd 4-d LC50 be within 2 SDs of the mean Cd 4-d LC50 in previous tests. The mean Cd 4-d LC50 for the previous 16 positive control tests with *E. estuarius* in our lab is 11.8 mg/L (mean ± 2 SD = 6.3–17.2 mg/L). The negative and positive control beakers were processed exactly the same as the beakers containing sediment from stations LC1 to RH9, except for the lack of sediment and shorter test duration of the positive control.

Percentage sand–silt–clay in sediment samples from all grabs at each station was determined by the sieve and pipette methods [5]. Cadmium concentrations in the positive control seawater were determined by flame AA (Perkin Elmer [Norwalk, CT] 5100 AA spectrophotometer). Sediment OC was determined by the combustion method using a Perkin Elmer 2400 CHN elemental analyzer after the samples were acidified to pH <2 with HCl to remove carbonates [6]. Sediment samples were centrifuged at ~3,200 *g* for 90 min at 4°C, and the interstitial water (IW) was collected by aspiration (R.J. Ozretich and D.W. Schults, in preparation). The IW was split into two 12-ml subsamples for analysis of the total (IW-total) and bound (IW-bound) phases. The IW-bound subsample was passed through a cartridge containing a C_{18} solid-phase resin to retain the freely dissolved compounds and allow the compounds bound to dissolved organic matter to pass through. Both phase subsamples were processed by gentle overnight liquid–liquid extraction to avoid emulsions. The freely dissolved compounds (IW-free) were quantified as the difference between IW-total and IW-bound. Organic chemicals were extracted from bulk sediment samples with acetonitrile using sonication, and were cleaned using C_{18} solid-phase columns [7]. All IW and bulk extracts were quantified by capillary GC (Hewlett Packard 5890II; Avondale, PA) with selected mass detection (Hewlett Packard 5970 MSD). Although the degradation of DDT within the GC averaged only 8% (range 3–10%, with 75% as DDD and the remainder as DDE), a peak area correction was made to the isomers of DDD to compensate for the creation of DDD from the dehalogenation of DDT in the instrument. The presence of potentially high and variable DDT concentrations necessitated this correction, which is not accomplished by standardization alone. All concentrations were corrected on the basis of the recovery of internal standards added before extraction. Concentrations of organic compounds are reported in dry-weight-normalized (micrograms per dry kilogram) and organic-carbon-normalized (micrograms per gram OC) units, because the latter may best reflect bioavailability [8]. DDT concentrations in sediment and interstitial water are reported as the sum of DDT and its metabolites (ΣDDT: Σ(2,4′-DDE, 4,4′-DDE, 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT); ΣDDT-OC = organic-carbon normalized ΣDDT sediment concentration; ΣDDT-IW = interstitial water ΣDDT concentration). ΣDDT and dieldrin were determined in samples from all grabs at all stations because previous studies indicated that these chemicals were the dominant sediment contaminants [2,9]. PAH and PCB (Aroclor mixtures) sediment concentrations were determined at stations LC2, SF6, and RH9 to represent sites of high, intermediate, and low ΣDDT and dieldrin contamination. SF6 was selected for PAH analyses also because of the presence of oil on the surface of the sediment sample from that station. The analyzed PAH compounds were naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, and benzo[*a*]pyrene. Aroclor 1242, Aroclor 1254, and Aroclor 1260 concentrations (free of contributions from other Aroclor mixtures) were estimated using the weight percentages of individual PCB congeners in standard solutions in specific Aroclor mixtures [10].

Acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) were determined in a sediment sample from each station. The samples were placed in a set of sediment toxicity test beakers set up for the purpose of determining AVS and SEM at the initiation of the toxicity tests. Overlying water in these beakers was removed on day 0, then a sediment sample was drawn into an open-barrel 10-cc plastic syringe and frozen. AVS was determined by converting the solid-phase sulfide to hydrogen sulfide (H_2S) using cold 1 M HCl. The released H_2S was trapped in sulfide antioxidant buffer, and the sulfide was measured using a sulfide-specific electrode. The SEM were determined by ICP spectrometry from a filtered sample of the sediment/acid solution after the AVS was released [11]. The SEM analysis included six metals expected to be AVS-reactive (Cu, Cd, Ni, Pb, Zn, and Ag).

The relative toxicological importance of different sediment contaminants can be evaluated by converting chemical concentrations to toxic units. A toxic unit (TU) is the concentration of a chemical that kills 50% of test specimens in a toxicity test [12]. The number of TUs is, therefore, the measured chemical concentration divided by the LC50. Toxicological data for the amphipods *Hyalella azteca* and *Rhepoxynius abronius* were used to estimate TUs because LC50s for the tested chemicals were not available for *E. estuarius*. The mean 10-d LC50s of dieldrin in sediment and interstitial water (IW-total) to *H. azteca* are 1,955 μg/g OC and 245.2 μg/L, respectively [13]. The mean 10-d LC50s of DDT in sediment and interstitial water (IW-total) to *H. azteca* are 371 μg/g OC and 1.07 μg/L, respectively [14]. The mean 10-d LC50 of DDT in interstitial water (IW-total) to *R. abronius* is 2.65 μg/L [15]. Combining the data for *H. azteca* and *R. abronius* gives a mean 10-d LC50 of DDT (IW-total) of 1.86 μg/L, which was used in subsequent calculations and discussion. Use of DDT LC50s to calculate the ΣDDT TU concentration assumes that the toxicities of DDT, DDD, and DDE in sediment are approximately equal (supported by Hoke et al. [16]) and additive (supported by Swartz et al. [17] for other sediment contaminants). The 10-d LC50 of PCB (Aroclor 1254) in sediment to *R. abronius* is 2,600 μg/g OC [17]. Sediment concentrations of the 13 PAH compounds listed above were converted to TUs by the ΣPAH model developed by Swartz et al. (in review). This model pre-

dicts (a) 10-d LC50 (IW-free) for PAH compounds from a QSAR regression, (b) PAH TUs by dividing interstitial-water PAH concentrations by the predicted 10-d LC50 (IW-free), and (c) total TUs for the 13 PAH compounds by addition.

Conditions at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations are compared in this report with similar surveys of sediment toxicity, chemistry, and amphipod abundance in a marine system on the Palos Verdes Shelf, California, and with sediment toxicity and chemistry in a freshwater stream system, Huntsville Spring Branch–Indian Creek, Alabama. ΣDDT is a major contaminant in all three systems [2,9,16,18–21]. The major differences between the present survey and those on the Palos Verdes Shelf were the use of *R. abronius* as the toxicity test species for Shelf sediments and the use of a 1.0-mm screen, rather than 0.5-mm screen, to collect amphipods on the Shelf. The freshwater amphipod *H. azteca* was used as the toxicity test species in the Huntsville study. Also, there were differences in the relative concentrations of DDT metabolites; 4,4′-DDE was dominant on the Palos Verdes Shelf [18–21], whereas 4,4′-DDT and 4,4′-DDD were dominant at Huntsville [16] and the Lauritzen Channel [22].

The organic carbon/water partition coefficient for ΣDDT (K_{oc}) was estimated using the equilibrium partitioning model [8]:

$$K_{oc} = C_s/[C_{iw} \times f_{oc}],$$

where

K_{oc} = organic carbon/water partition coefficient for ΣDDT, (L/kg OC)
C_{iw} = ΣDDT-IW (mg/L)
C_s = ΣDDT concentration in sediment (mg/dry kg)
f_{oc} = OC in sediment expressed as a fractional mass, (kg OC/dry kg sediment).

ΣDDT K_{oc} estimates were derived separately from C_{iw}, C_s, and f_{oc} data for individual ΣDDT-contaminated sediment samples in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor and in the Huntsville streams [16]. Statistical analyses were based on arithmetic expression of K_{oc} values. Confidence limits and mean K_{oc} values are given as log-transformed values.

Relations among chemical, toxicological, and biological variables from grab samples ($n = 48$) were determined by product–moment correlation (r) or, when paired variable distributions were not bivariate normal, by Kendall's (τ_b) coefficients [23,24]. In the latter case the tested hypotheses were (a) there is no association between the presence and absence of *Grandidierella japonica* and the presence and absence of all other amphipods; and (b) there is no association between the presence and absence of amphipods, excluding *G. japonica*, and low (≤20%) and high (≥25%) mortality in sediment toxicity tests with *E. estuarius*. Differences among stations in mean percentage mortality in the sediment toxicity tests were determined by ANOVA and Newman–Keuls multiple-range test [25]. The significance level was $p < 0.05$ in all tests. LC50s were calculated by probit analysis [26].

Our evaluations of the interstitial water data are based primarily on IW-free rather than IW-total concentrations because IW-free is the better indicator of bioavailability [8]. However, IW-total data were used to compare our results with those of studies that reported only IW-total concentrations [13–16].

RESULTS

Sediment toxicity

The control results of the sediment toxicity test with *E. estuarius* met all QA/QC requirements. Mean mortality in the negative control sediment was 3.0%, which is well within the ASTM [3] requirement of ≤10% negative control mortality (Table 1). The Cd 4-d LC50 in seawater was 16.9 mg/L, which is within the acceptable range.

Mean mortality of *E. estuarius* exposed to sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations ranged from 23.0 to 66.2% (Table 1). Mean mortality in sediment from station LC1 was significantly higher than the negative control sediment and sediment from all other stations. Sediment from two of the eight grab samples from station LC1 caused 100% mortality of test specimens. Mean mortality in sediment from stations LC1, LC2, LC3, and SF6 was significantly higher than that in the negative control sediment.

The bioassay data indicate a toxicity gradient along four stations from the head to the mouth of Lauritzen Channel (Table 1). Also, the sequence in mean mortality was Lauritzen Channel (42.0%) > Santa Fe Channel (30.0%) > Richmond Harbor Channel (23.7%).

Amphipod fauna

Eight species of amphipods were collected in this study (Table 2). Three species were found in the Lauritzen Channel, two in the Santa Fe Channel, and seven in the Richmond Harbor Channel. Only two individuals of a single species were collected at station RH8, a site at which the bottom sediment is greatly disturbed by prop scour [27]. Only one amphipod species, *G. japonica*, was abundant in the Lauritzen Channel. *Corophium heteroceratum* was abundant in the Santa Fe and Richmond Harbor Channels. *Ampelisca abdita* was abundant at station RH9.

The distribution of *G. japonica* was very different from that of other amphipods (Fig. 2 and Table 2). Where *G. japonica* was abundant or common (stations LC1–4), all other amphipods were rare or absent. Where other amphipods were abundant or common (stations SF5 and 6, RH7 and 9), *G. japonica* was rare or absent.

Sediment and IW chemistry

The mean sediment concentrations of dieldrin, ΣDDT, OC, and sediment fines (silts + clays) at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations are given in Table 3. The percentage of fines at all stations was ≥79%, with no obvious gradient. OC was highest at stations SF6 (2.98%) and LC1 (2.38%), and lowest in the Richmond Harbor stations (0.87–1.18%). About 90% of the total IW concentrations of both dieldrin and ΣDDT were freely dissolved. There was a very strong gradient of chemical contamination. The first three stations in the Lauritzen Channel (LC1-3) were highly contaminated by both dieldrin (25.8–35.2 μg/g

Table 1. Mortality of the infaunal amphipod *Eohaustorius estuarius* during 10 d exposure to sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

Station	No. dead ($N = 20$) for grab no.								Mean no. dead	Student–Newman–Keuls multiple-range test[a] (percent mortality)
	1	2	3	4	5	6	7	8		
LC1	20	14	9	11	12	13	7	20	13.2	66.2
LC2	3	11	8	9	12				8.6	43.0A
LC3	5	10	7	8	6				7.2	36.0A
LC4	4	7	3	4	5				4.6	23.0AB
SF5	5	4	5	9	5				5.6	28.0AB
SF6	5[b]	12	8	3	4				6.4	32.0A
RH7	2	5	2	6	8				4.6	23.0AB
RH8	5	2	8	2	7				4.8	24.0AB
RH9	1	5	7	4	7				4.8	24.0AB
Negative control	0	1	0	0	2				0.6	3.0B

Positive control—Cd (mg/L)	35	18	8.7	4.6	2.2	1.1	0.0
No. dead ($N = 20$)	19	6	7	3	2	1	1

4-d LC50 (±95% CL) = 16.9 (13.1–21.7) mg/L Cd.
[a]Means with the same letter are not significantly different ($p \geq 0.05$).
[b]Mortality adjusted for double seeding.

OC in sediment, 0.067–0.251 μg/L IW-free) and ΣDDT (1,520–3,500 μg/g OC, 0.89–4.49 μg/L IW-free). The dieldrin and ΣDDT concentrations at the other six stations were about an order of magnitude or more below the concentrations at stations LC1 to LC3. However, a sediment contamination gradient is evident even at these six stations, with lowest concentrations of dieldrin (0.07 μg/g OC, 0.000 μg/L IW-free) and ΣDDT (1.34 μg/g OC, 0.003 μg/L IW-free) occurring at station RH9, farthest from the Lauritzen Channel.

Application of the equilibrium partitioning model to the C_{iw}, C_s, and f_{oc} data for ΣDDT-contaminated sediment in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor sediment (Table 3) resulted in an estimated log K_{oc} = 6.15 (95% confidence limits 6.04–6.24, $n = 41$) for ΣDDT-IW-free and log K_{oc} = 6.07 (95% C.L. 5.97–6.16, $n = 41$) for ΣDDT-IW-total. We estimated a log K_{oc} = 6.14 (95% C.L. 5.94–6.27, $n = 10$) for ΣDDT-IW-total for the Huntsville stream sediments [16]. There was no significant difference between the estimates of log K_{oc} for ΣDDT-IW-total for the Lauritzen and Huntsville sediments ($p = 0.52$). Data were not available to estimate log K_{oc} for ΣDDT-IW-free for Huntsville stream sediments.

The Aroclor 1254 concentration was 120 μg/kg (6.7 μg/g OC) at station LC2 and 610 μg/kg (20 μg/g OC) at station SF6. The Aroclor 1260 concentration was about half that of Aroclor 1254 at stations LC2 and SF6 (Table 4). The presence of neither Aroclor 1254 nor Aroclor 1260 was confirmed in the sediment from station RH9. Aroclor 1242 was not detected at stations LC2, SF6, or RH9. PAH concentrations were substantially higher at station SF6 than at station LC2, and lowest at station RH9 (Table 4).

The concentrations of AVS and most metals were highest at stations LC1 and SF6, and lowest at stations RH8 and RH9 (Table 5). The ratio of total SEM/AVS on a molar basis ranged from 0.05 at station LC1 to 0.48 at station RH9.

Relations among chemistry, toxicity and amphipod abundance

ΣDDT-OC, ΣDDT-IW-free, dieldrin-OC, and dieldrin-IW-free at the nine stations were highly correlated ($r = 0.83$–0.93, $p < 0.001$; Table 6). OC was significantly correlated with ΣDDT-OC, ΣDDT-IW-free, dieldrin-OC, and dieldrin-IW-free ($r = 0.31$–0.61, $p < 0.05$). The fine sediment frac-

Table 2. Species composition and abundance of amphipods at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

Amphipod species	No. of individuals/0.075 m^2 at stations								
	LC1	LC2	LC3	LC4	SF5	SF6	RH7	RH8	RH9
Grandidierella japonica	28	64	14	6			2		1
Caprella mutica		2							
Corophium heteroceratum				1	7	34	70	2	18
Ampelisca abdita						1	3		77
Listriella goleta							1		
Dulichia rhabdoplastis									6
Caprella incisa									2
Aoroides columbiae									1

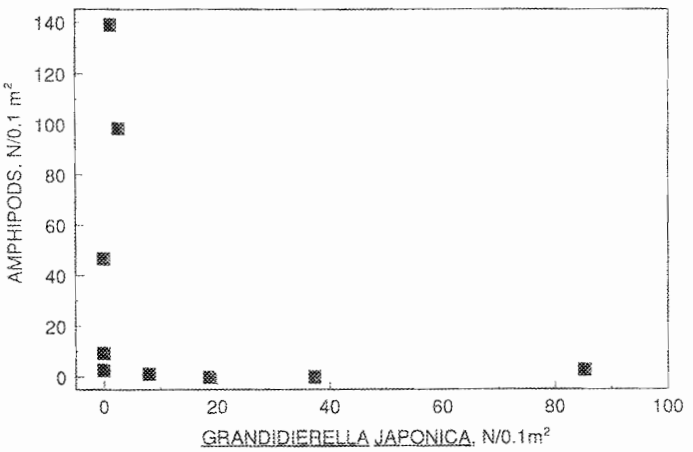

Fig. 2. The mean abundance of *Grandidierella japonica* in relation to the mean abundance of other amphipods.

tion was not significantly correlated with any variable except the abundance of amphipods other than *G. japonica*.

Because the pattern of dieldrin and ΣDDT sediment contamination was virtually identical, relations between dieldrin and ΣDDT and sediment toxicity and amphipod field abundance were also almost identical (Table 6). There were no substantial differences in mortality of *E. estuarius* at dieldrin concentrations <3 μg/g OC (<0.05 μg/L IW-free) or at ΣDDT concentrations <200 μg/g OC (<0.4 μg/L IW-free) (Tables 1 and 3; Figs. 3 and 4). Sediment toxicity was significantly greater at stations LC1, -2, and -3, where the dieldrin and ΣDDT concentrations ranged from 25 to 35 μg/g OC (0.07–0.25 μg/L IW-free) and 1,500 to 3,500 μg/g OC (0.9–4.5 μg/L IW-free), respectively (Tables 1 and 3). The field-derived 10-d LC50 for ΣDDT-OC to *E. estuarius* was 2,500 μg/g OC, and the 10-d LC50 for ΣDDT-IW-free was 2.70 μg/L (2.88 μg/L for IW-total). There were highly significant, positive correlations between sediment toxicity and the concentrations of both dieldrin and ΣDDT in sediment and interstitial water (Table 6).

Sediment toxicity was significant at station SF6, even though dieldrin and ΣDDT concentrations were about an order of magnitude below those at stations LC1, -2, and -3 (Tables 1 and 3). Concentrations of Aroclor 1254; Aroclor 1260; and, especially, PAHs were substantially higher at station SF6 than at station LC2 (Table 4).

The abundance of amphipods (excluding *G. japonica*) in the field was reduced at dieldrin and ΣDDT concentrations lower than those at which *E. estuarius* mortality began to increase in lab sediment toxicity tests. Amphipod abundance was variable, but often >40 individuals per 0.1 m^2 at dieldrin and ΣDDT concentrations <3 μg/g OC (<0.02 μg/L IW-free) and <100 μg/g OC (<0.4 μg/L IW-free), respectively (Tables 2 and 3; Fig. 5). The rarity of amphipods at stations SF5 and RH8 (Table 2), where dieldrin and ΣDDT concentrations were relatively low (Table 3), may be associ-

Table 3. Mean concentration of sediment fines,[a] total organic carbon (OC), dieldrin, and ΣDDT[b] at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

	Sediment						Interstitial water (μg/L)			
			Dieldrin		ΣDDT		Dieldrin		ΣDDT	
Station	% Fines	% OC	(μg/dry kg)	(μg/g OC)[c]	(μg/dry kg)	(μg/g OC)[c]	total	free	total	free
LC1	92.3	2.38	748	35.2	77,700	3,500	0.257	0.251	4.75	4.49
LC2	85.2	1.78	528	28.7	47,800	2,710	0.168	0.168	2.12	1.95
LC3	85.9	1.73	442	25.8	26,000	1,520	0.102	0.067	0.977	0.890
LC4	89.5	1.46	35.7	2.46	2,740	189	0.017	0.016	0.123	0.102
SF5	99.6	1.48	5.54	0.37	420	28.3	0.001	0.000	0.058	0.052
SF6	79.0	2.98	78.4	2.59	2,340	90.2	0.002	0.002	0.463	0.381
RH7	83.2	1.08	9.46	0.89	368	34.8	0.001	0.001	0.028	0.024
RH8	94.6	1.18	1.73	0.15	82.5	6.99	0.000	0.000	0.007	0.007
RH9	81.7	0.87	0.63	0.07	11.6	1.34	0.000	0.000	0.004	0.003

[a] % silt + % clay
[b] Σ(2,4′-DDE, 4,4′-DDE, 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT).
[c] OC-normalized concentration is the mean of the OC-normalized concentrations for all grabs taken at each station.

Table 4. Concentrations of 13 PAHs and three Aroclor mixtures of PCB congeners at three of the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

Compound/mixture	Concn. (μg/g OC)		
	Station LC2 (OC = 1.78%)	Station SF6 (OC = 2.98%)	Station RH9 (OC = 0.87%)
Naphthalene	8.6	490	2.8
Acenaphthylene	8.2	19	1.2
Acenaphthene	1.7	340	0.82
Fluorene	4.0	370	0.66
Phenanthrene	16	660	4.8
Anthracene	19	160	1.2
Fluoranthene	41	590	13
Pyrene	69	550	18
Benz[*a*]anthracene	25	500	7.1
Chrysene	57	340	8.2
Benzo[*b*]fluoranthene	97	250	9.6
Benzo[*k*]fluoranthene	51	69	8.4
Benzo[*a*]pyrene	72	190	12
Aroclor 1242	ND	ND	ND
Aroclor 1254	6.7	20	ND
Aroclor 1260	3.4	8.5	ND

OC = total organic carbon.
ND = not detected.

Table 5. Acid-volatile sulfide (AVS) and simultaneously extracted metals (SEM) at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations[a]

Station	AVS (μmol/g)	Metal concn. (μg/g sediment)					Total SEM (μmol/g)	SEM-AVS ratio
		Cu	Zn	Pb	Ni	Cd		
LC1	135	68.1	326	92.8	29.4	1.1	7.02	0.05
LC2	27.8	60.4	174	65.9	28.2	0.2	4.40	0.16
LC3	44.0	51.6	142	52.8	34.7	0.2	3.83	0.09
LC4	22.1	30.3	102	37.7	22.7	0.0	2.60	0.12
SF5	20.2	33.4	90.0	35.2	17.9	0.0	2.38	0.12
SF6	113	47.6	298	111	25.3	1.4	6.29	0.06
RH7	29.9	38.9	119	41.7	27.9	0.1	3.10	0.10
RH8	11.0	29.5	89.9	30.2	26.7	0.0	2.44	0.22
RH9	2.3	10.1	49.9	11.2	8.0	0.0	1.11	0.48

[a]Ag was not detected in any sample.

Table 6. Product-moment correlation (r) or Kendall's (τ_b) coefficients between $\log_{10}(\Sigma DDT^a + 1)$ freely dissolved in interstitial water (IW; ng/L), $\log_{10}\Sigma DDT$ in sediment (μg/g OC), $\log_{10}$(dieldrin + 1) freely dissolved in interstitial water (IW; ng/L), $\log_{10}$(dieldrin + 1) in sediment (μg/g OC), sediment toxicity (percentage mortality of *Eohaustorius estuarius* if r; low [0–20%] or high [≥25%] if τ_b), amphipods excluding *Grandidierella japonica* (number/0.015 m^2 if r; present or absent if τ_b), and *G. japonica* (number/0.015 m^2 if r; present or absent if τ_b), organic carbon (OC), and sediment fines (% silt + % clay) at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

	ΣDDT-IW	ΣDDT-OC	Dieldrin-IW	Dieldrin-OC	Toxicity	Amphipods	*G. japonica*	OC
ΣDDT-OC	$r = 0.93$***							
Dieldrin-IW	$r = 0.83$***	$r = 0.91$***						
Dieldrin-OC	$r = 0.89$***	$r = 0.93$***	$r = 0.93$***					
Toxicity	$r = 0.59$***	$r = 0.58$***	$r = 0.59$***	$r = 0.64$***				
Amphipods	$r = -0.39$**	$r = -0.48$***	$r = -0.38$*	$r = -0.37$**	$\tau_b = -0.34$*			
G. japonica	$r = 0.49$***	$r = 0.54$***	$r = 0.60$***	$r = 0.58$***	$r = 0.29$*	$\tau_b = -0.32$*		
OC	$r = 0.61$***	$r = 0.51$***	$r = 0.31$*	$r = 0.46$***	$r = 0.40$**	$r = -0.26$ NS	$r = 0.13$ NS	
Fines	$r = -0.04$ NS	$r = 0.04$ NS	$r = -0.04$ NS	$r = -0.11$ NS	$r = 0.10$ NS	$r = -0.30$*	$r = -0.14$ NS	$r = -0.06$ NS

NS = not significant.
[a]Σ(2,4′-DDE, 4,4′-DDE, 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT)
*$p < 0.05$, **$p < 0.01$; ***$p < 0.001$

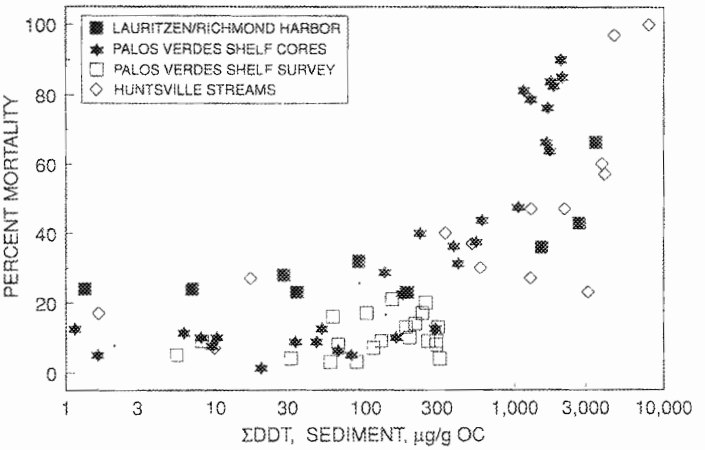

Fig. 3. Mean mortality of *Eohaustorius estuarius* in Lauritzen Channel/Santa Fe Channel/Richmond Harbor sediment, *Rhepoxynius abronius* in Palos Verdes Shelf sediment, and *Hyalella azteca* in Huntsville stream sediment in relation to the mean sum of the organic-carbon-normalized concentrations of DDT and its metabolites in sediment.

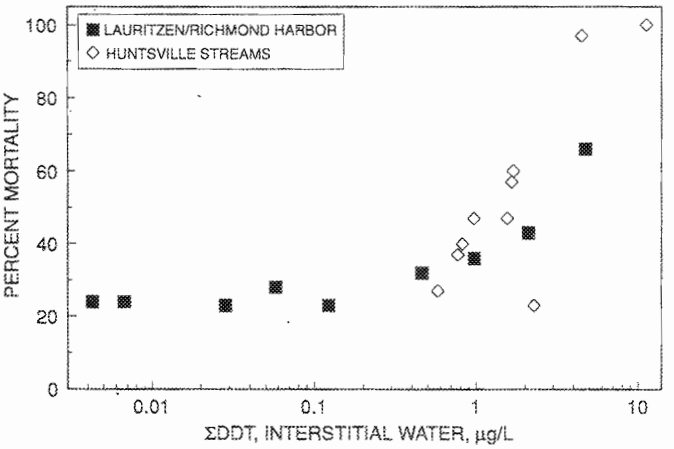

Fig. 4. Mean mortality of *Eohaustorius estuarius* in Lauritzen Channel/Santa Fe Channel/Richmond Harbor sediment, and *Hyalella azteca* in Huntsville stream sediment in relation to the mean sum of the concentrations of DDT and its metabolites in interstitial water.

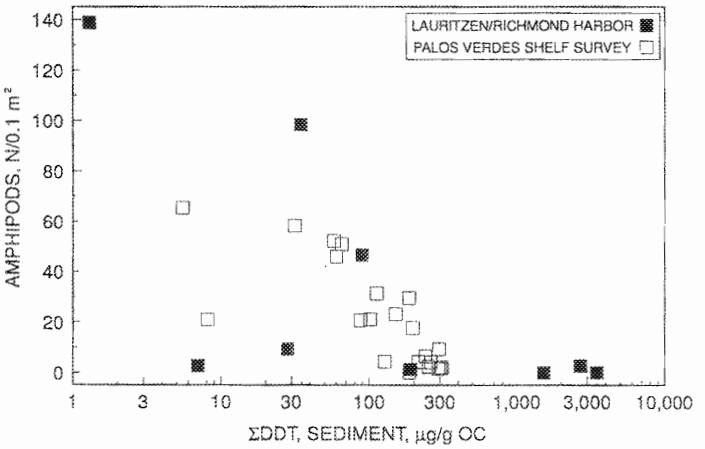

Fig. 5. The mean abundance of amphipods (excluding *Grandidierella japonica*) in relation to the mean sum of the organic-carbon-normalized concentrations of DDT and its metabolites in Palos Verdes Shelf and Lauritzen Channel/Santa Fe Channel/Richmond Harbor sediment.

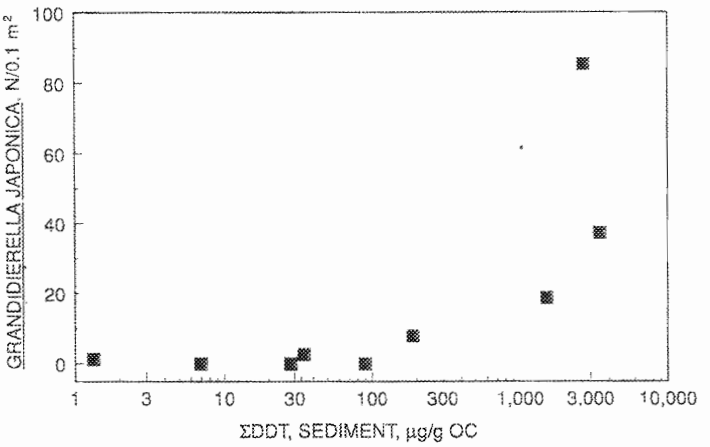

Fig. 6. The mean abundance of *Grandidierella japonica* in relation to the mean sum of the organic-carbon-normalized concentrations of DDT and its metabolites in sediment.

ated with factors other than sediment contamination, such as prop scour at station RH8 [27]. Few or no amphipods (except *G. japonica*) were collected at stations LC1 to LC4, where dieldrin and ΣDDT concentrations were very high (dieldrin, 2.46–35.2 µg/g OC [0.02–0.25 µg/L IW-free]; ΣDDT, 189–3,500 µg/g OC [0.10–4.5 µg/L IW-free]).

The abundance of *G. japonica* was positively correlated with sediment toxicity and dieldrin and ΣDDT contamination (Table 6 and Fig. 6). The abundance of amphipods other than *G. japonica* was negatively correlated with sediment toxicity and dieldrin and ΣDDT contamination (Table 6; Figs. 5 and 7). Mortality of *E. estuarius* was highest at stations LC1 to LC3, where few or no amphipods except *G. japonica* were collected (Tables 1 and 2). *Grandidierella japonica* was most abundant at stations LC1 to LC3, where the ΣDDT concentration exceeded 1,500 µg/g OC (0.89 µg/L IW-free), the dieldrin concentration exceeded 25 µg/g OC (0.07 µg/L IW-free), and the mortality of *E. estuarius* in sediment toxicity tests exceeded 35% (Tables 2 and 3; Fig. 6). The presence/absence of *G. japonica* was inversely related to the presence/absence of all other amphipods (Table 6 and Fig. 2).

Sediment fines and sediment OC were not significantly correlated with the abundance of *G. japonica* (Table 6). Sediment OC was not but sediment fines were significantly correlated with the abundance of all other amphipod species. Sediment toxicity was significantly correlated with sediment OC but not with sediment fines.

DISCUSSION

Relative contributions of ΣDDT and dieldrin to biological impacts

The correlations between sediment contamination, sediment toxicity, and amphipod abundance that we observed in samples from the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations are typical of complex sediment

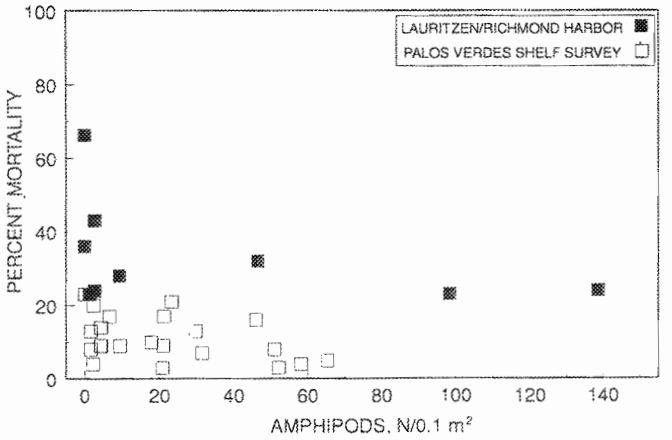

Fig. 7. Mean mortality of *Eohaustorius estuarius* in Lauritzen Channel/Santa Fe Channel/Richmond Harbor sediment and *Rhepoxynius abronius* in Palos Verdes Shelf sediment in relation to the mean abundance of amphipods (excluding *Grandidierella japonica*)

contamination gradients in which variables are affected by interrelated factors [18]. The correlations demonstrate covariance, not causality. Experimental evidence is necessary to discriminate the relative contributions of covarying chemicals to biological effects. Fortunately, experimental and supporting field data are available to make this discrimination. The experimental data come from sediment toxicity tests in which DDT, dieldrin, Aroclor 1254, PAHs, or metals were spiked into uncontaminated sediment [11,13–15,17]. The supporting field data come from the studies of the DDT contamination gradient on the Palos Verdes Shelf [18–21] and the Huntsville stream system [16].

The relative toxicological importance of ΣDDT, dieldrin, Aroclor 1254, and ΣPAH can be compared by converting chemical concentrations to toxic units (Table 7). For example, there were about nine sediment TUs of ΣDDT-OC at station LC1 [(chemical concentration)/LC50 = (3,500 μg/g OC)/(371 μg/g OC) = 9.43]. The sediment TUs of ΣDDT-OC at stations LC1, LC2, and LC3 ranged from 4.10 to 9.43, more than sufficient to account for the observed mortality of *E. estuarius* in the sediment collected at these stations. In contrast, the concentration range for dieldrin-OC at stations LC1–3 was 0.013 to 0.018 sediment TUs. At station LC2 there were 0.003 sediment TUs of Aroclor 1254-OC and 0.284 sediment TUs of ΣPAH-OC (Table 7). Thus, at station LC2, the ΣDDT-OC TU concentration was about 25× that of ΣPAH-OC, 500× that of dieldrin-OC, and 2,500× that of Aroclor 1254-OC.

AVS is the sediment phase that determines the toxicity of metals that form insoluble sulfides [11]. No metal toxicity is expected when the sum of the molar concentrations of divalent metals is less than that of AVS. Because the metal-to-AVS molar ratio in this study was always ≤0.48 (Table 5), metals (Cu, Zn, Pb, Ni, Cd, Ag) are unlikely causes of the observed sediment toxicity (Table 1).

The ΣDDT-OC TUs at stations LC4 and SF6 were 0.51 and 0.24, respectively (Table 7). Such TU values reflect levels of contamination that might cause a slight increase in sediment toxicity. Significant toxicity at station SF6 (Table 1), however, is probably due primarily to PAH contamination, as there were 1.66 TUs of ΣPAH-OC in the sediment at that station (Table 7). Sediment toxicity at stations SF5, RH7, RH8, and RH9 was not significantly different from the control (Table 1), and the sediments from these stations each contained <0.1 TU of ΣDDT-OC and dieldrin-OC (Table 7). Sediment at station RH9 had <0.05 TU of Aroclor 1254-OC and ΣPAH-OC (Table 7).

The spatial distribution and relative magnitude of dieldrin and ΣDDT TUs in IW are similar to those for sediment TUs based on OC-normalized sediment LC50s (Table 7). TUs of ΣDDT-IW-total ranged from 0.5 to 2.6 at stations LC1, LC2, and LC3 and 0.0 to 0.25 at the other six stations. There was <0.01 dieldrin-IW-total TU at all stations.

Our TU estimates were obtained by applying LC50s for *H. azteca* or *R. abronius* to *E. estuarius*. This introduces a potential source of error in our TU analysis. Available comparative toxicological data, however, show that the 10-d LC50 for fluoranthene in sediment differs only by a factor of three among these species (15.4, 10.6, and 5.1 mg/kg for *H. azteca*, *E. estuarius*, and *R. abronius*, respectively [28]). Thus, the potential error resulting from our use of LC50s for other species is probably small relative to the major differences in the TUs of ΣDDT compared to TUs of dieldrin, Aroclor 1254, and ΣPAH (Table 7).

By definition, the nominal LC50 for chemical concentrations expressed as toxic units is 1.0 TU. The estimated

Table 7. Toxic unit concentrations of organic-carbon-normalized sediment and interstitial water contaminants at the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations

	Toxic units					
	Sediment				Interstitial water[a]	
Station	PCB-OC[b]	ΣPAH-OC[c]	Dieldrin-OC	ΣDDT-OC[d]	Dieldrin	ΣDDT[d]
LC1			0.018	9.43	0.001	2.55
LC2	0.003	0.284	0.015	7.30	0.001	1.14
LC3			0.013	4.10	0.000	0.525
LC4			0.001	0.509	0.000	0.066
SF5			0.000	0.076	0.000	0.031
SF6	0.008	1.663	0.001	0.243	0.000	0.249
RH7			0.000	0.094	0.000	0.015
RH8			0.000	0.019	0.000	0.004
RH9	0.000	0.047	0.000	0.004	0.000	0.002
LC50[e]			<0.02	6.74	<0.01	1.55

[a]IW-total.
[b]Aroclor 1254.
[c]Σ{Naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz{*a*}anthracene, chrysene, benzo{*b*}fluoranthene, benzo{*k*}fluoranthene, benzo{*a*}pyrene}.
[d]Σ{2,4′-DDE, 4,4′-DDE, 2,4′-DDD, 4,4′-DDD, 2,4′-DDT, 4,4′-DDT}
[e]TU LC50s calculated by probit analysis [26] from mortality (Table 1) and TU concentrations at each station

TU LC50s for ΣDDT-OC and ΣDDT-IW-total were 6.74 and 1.58 TU, respectively (Table 7). The difference from the expected LC50 of 1.0 TU could be explained by interlaboratory differences in the measurement of DDT, interspecies extrapolations used to make the TU LC50 estimates, and/or nonequilibrium of ΣDDT in test sediment. No conclusion should be drawn from these data about the relative importance of sediment vs. interstitial water routes of exposure.

Further evidence that ΣDDT is the primary cause of toxic effects in the Lauritzen Channel is the agreement between ΣDDT-toxicity relations in the Lauritzen Channel, the Palos Verdes Shelf, and the Huntsville streams. ΣDDT is a dominant contaminant at all of these sites. In surveys of surficial sediment on the Palos Verdes Shelf, mortality in toxicity tests with the amphipod *R. abronius* never exceeded 23%, ΣDDT never exceeded 300 μg/g OC, and there was no significant correlation between ΣDDT and toxicity [18,19,21]. However, in a study of 50-cm-deep cores collected on the Shelf (Fig. 3) [20], ΣDDT in some core sections was as great as 2,000 μg/g OC, *R. abronius* mortality in toxicity tests was as high as 90%, and sediment toxicity was significantly correlated with ΣDDT. ΣDDT in sediments from the Huntsville streams ranged from 0 to 7,922 μg/g OC (0–11.27 μg/L IW-total) and was significantly correlated with sediment toxicity to *H. azteca* (Figs. 3 and 4) [16]. The relations between ΣDDT and sediment toxicity to amphipods in this study are similar to those observed on the Palos Verdes Shelf and in the Huntsville streams (Figs. 3 and 4). The field-derived 10-d LC50 for ΣDDT-OC for the Lauritzen (2,500 μg/g OC), Huntsville (2,580 μg/g OC), and Palos Verdes (1,040 μg/g OC) agree within a factor of 2.5, despite differences in test species, habitats, and relative concentrations of DDT metabolites in ΣDDT (Fig. 3). ΣDDT in IW was not analyzed in the Palos Verdes investigations, but the field-derived 10-d LC50s for ΣDDT-IW-total in the Lauritzen Channel (2.88 μg/L) and Huntsville streams (1.57 μg/L) agree within a factor of two (Fig. 4).

There is also good agreement between ΣDDT-OC and the abundance and distribution of amphipods in this study and on the Palos Verdes Shelf (Fig. 5) [18,19,21,29]. In this study few or no amphipods (except *G. japonica*) were collected at ΣDDT greater than about 100 μg/g OC (>0.4 μg/L IW-free), whereas on the Palos Verdes Shelf few or no amphipods were collected at ΣDDT greater than about 200 μg/g OC.

In summary, there are experimental and corroborative field data that indicate ΣDDT is the major factor causing sediment toxicity and a depression in amphipod populations in the Lauritzen Channel. First, the metal-to-AVS molar ratio and the TUs of dieldrin, Aroclor 1254, and ΣPAH are too low, whereas the TUs of ΣDDT are sufficient to exert acute toxicity (Table 7). Second, the patterns of ΣDDT concentration response (sediment toxicity and amphipod abundance), the thresholds of response, and the field-derived LC50s for ΣDDT are similar at three sites of ΣDDT contamination (Figs. 3 and 4). Together, these data provide good evidence that ΣDDT is the dominant ecotoxicological factor in the Lauritzen Channel.

Partitioning of ΣDDT

Partition coefficients are usually estimated for individual compounds [8]. However, because ΣDDT was found to be predictive of toxicological and biological responses (Tables 1–3, 6, 7; Figs. 3–5), we calculated apparent ΣDDT partition coefficients based on OC, IW, and sediment ΣDDT concentrations for each sample in the Lauritzen and Huntsville studies (Table 3) [16]. The mean ΣDDT-IW-total log K_{oc} values for the Lauritzen and Huntsville sediments (6.07 and 6.14, respectively) were not significantly different. The mean ΣDDT-IW-free log K_{oc} for the Lauritzen sediment was 6.15. These field-derived K_{oc} estimates should be applied to other sediments with caution, especially when 4,4′-DDT and 4,4′-DDD are not dominant DDT metabolites, as they were at Huntsville [16] and the Lauritzen Channel [22].

Comparison of sediment toxicity to other coastal sites

Few data are available on the toxicity of sediment from contaminated field sites to *E. estuarius*. However, the present bioassay data can be compared with sediment toxicity surveys conducted with *R. abronius*, as *E. estuarius* is typically only slightly less sensitive than *R. abronius* [28]. Station LC1, with a mean mortality of 66.2%, is a severe case of sediment toxicity. It ranks among the worst cases of sediment toxicity reported by Swartz et al. [30] (Table 8). The mean mortality in sediments from all stations in the Lauritzen Channel/Santa Fe Channel/Richmond Harbor (33.2%) is typical of highly urbanized embayments that contain some sites of severe sediment contamination (e.g., San Diego Bay, CA; Table 8). The mean mortality of *E. estuarius* in the present study is similar to the mean mortality of this species at another Superfund site in the Hylebos Waterway, Washington (38.3%) [28].

Pinza et al. [9] conducted sediment toxicity tests with *R. abronius* on composites of sediment samples from each of three groups of five stations in Richmond Inner Harbor. They found no significant difference from their reference sediment at two sets of stations within the Richmond Inner Harbor Channel (mean percentage mortality 17 and 13%). There was a statistically significant increase in mortality of amphipods exposed to a composite sediment sample from five stations on the southeastern bank of the Richmond Inner Harbor Channel (mean percentage mortality 42%). The mean percentage mortality of *R. abronius* for all of the Pinza et al. [9] toxicity tests of Richmond Harbor sediment (24.0%) is virtually identical to the mean percentage mortality of *E. estuarius* in the present toxicity tests of Richmond Inner Harbor sediment (23.7%; Table 1).

Sediment contamination and toxicity can be very patchy, even on small spatial scales [30,31]. Patchiness is shown at station LC1, where mortality varied from 35 to 100% in sediment samples taken, at most, only meters apart (Tables 1 and 8). Mortality was 100% in sediment from two of the eight grabs at station LC1, indicating the occurrence of small patches of sediment that cause extremely high toxicity.

In the absence of chemical contamination, *E. estuarius* tolerates a very broad range of sediment conditions. Mortal-

Table 8. Comparison of mortality of *Eohaustorius estuarius* in sediment from the Lauritzen Channel/Santa Fe Channel/Richmond Harbor stations[a] with the mortality of *Rhepoxynius abronius* at other U.S coastal sites [30]

Sediment source	Percent mortality		Comment
	mean	range	
Yaquina Bay, OR	2	0–5	Amphipod collection site
NW Santa Monica Bay, CA	7	0–5	Reference site
Palos Verdes Shelf, CA	16	0–40	Near sewage outfall
San Diego Bay, CA	29	0–100	56 stations
[a]Mean LC1–4, SF5–6, RH7–9	33.2	5–100	Mean for present study
Near-shore Commencement Bay, WA	38	5–95	9 stations
South San Francisco Bay, CA	45	20–100	26 stations
Hylebos Waterway, WA	52	5–100	26 stations, Superfund site
Houston Ship Channel, TX	60	45–80	5 stations
[a]Lauritzen Channel LC1	66.2	35–100	Worst case, present study
Elizabeth River, VA	78	30–100	Creosote contamination
New York Bight, NY	88	80–95	Sludge dump site
San Diego, CA station 28	97.5	90–100	Worst case, San Diego
Eagle Harbor, WA station 8	100		Creosote contamination

[a]Data, this study.

ity of *E. estuarius* in apparently uncontaminated sediment collected in Puget Sound and the central Oregon coast averaged 5.6% and rarely exceeded 20% [28]. In contrast, even at stations with low ΣDDT, dieldrin, Aroclor 1254, and ΣPAH concentrations in Richmond Inner Harbor, mean mortality of *E. estuarius* was ≥23% (Tables 1 and 3; Figs. 3 and 4). Apparently some unknown factor(s) is causing a higher background level of mortality in the present study than in the Puget Sound/Oregon reference stations. The grain size and salinity of our sediment samples are within the tolerance range of *E. estuarius* [28]. ΣDDT, dieldrin, Aroclor 1254, and ΣPAH probably do not contribute substantially to the background toxicity level because those contaminants collectively contribute only about 0.05 TU to sediment at station RH9, where mean mortality of *E. estuarius* was 24% (Tables 1 and 7). In addition, metals are probably not involved because the metal-to-AVS molar ratio was <0.5 at all stations. The unknown factor(s) causing the high background toxicity may be unmeasured contaminants, natural sediment features, or interactions between several parameters. A toxicity identification evaluation [32] of the sediment in Richmond Inner Harbor might help identify the factor(s) responsible for the background toxicity.

Field validation of sediment toxicity tests

The ecological relevance of a sediment toxicity test can be evaluated by comparing test results with gradients of sediment contamination and biological effects in the field. Responses of valid tests should be negatively related to the field distribution of pollution-sensitive species and positively related to chemical contamination [18]. Positive correlations have been demonstrated between ΣDDT concentrations in field sediment and the percentage mortality of *E. estuarius*, *R. abronius*, and *H. azteca* in acute sediment toxicity tests (Fig. 3 and Table 6) [16,18–21]. Negative correlations have been demonstrated between abundance of sensitive amphipods in the field sediment and percentage mortality of *E. estuarius* and *R. abronius* in acute sediment toxicity tests (Fig. 4 and Table 6) [18–21]. These results indicate that significant acute sediment toxicity to these three amphipod species in lab tests provides reliable evidence of biologically adverse sediment contamination in the field.

Grandidierella japonica has been recommended as a sediment toxicity test species [3,33–35]. Toxicological comparisons have shown that the sensitivity of *G. japonica* to contaminated field sediments and reference toxicants is comparable to that of other amphipods [36–38]. In the present study, however, the occurrence of *G. japonica* was negatively related to the occurrence of other amphipods (Fig. 2 and Table 6), and the abundance of *G. japonica* was positively correlated with sediment toxicity and ΣDDT contamination (Fig. 6 and Table 6). Further research is needed to resolve the apparent paradox of the tolerance of *G. japonica* to ΣDDT-contaminated sediment in the Lauritzen Channel and the use of this species to test sediment toxicity.

Summary of effect concentrations for ΣDDT

The mean 10-d LC50 for ΣDDT-OC for amphipods exposed to field-collected sediments from the Lauritzen Channel, Huntsville, and Palos Verdes studies was 2,040 μg/g OC (Fig. 3). The threshold for 10-d sediment toxicity occurred at about 300 μg ΣDDT/g OC. The mean 10-d LC50 for ΣDDT-IW-total in the Lauritzen Channel and Huntsville studies was 2.22 μg/L, with a toxicity threshold at about 0.5 μg/L (Fig. 4). Chronic effects of ΣDDT on amphipod populations in the field were evident at concentrations lower than those that caused acute toxicity in the lab. Amphipod abundance sharply declined at ΣDDT concentrations between 100 and 200 μg/g OC on the Palos Verdes Shelf and in the Lauritzen Channel (Fig. 5). The minimum ecotoxicological effect concentration appears to be about 100 μg ΣDDT/g OC.

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