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8.0 RISK CHARACTERIZATION

Evaluation of the contaminants of concern in the sediment of Ferry Creek, the Housatonic River near the mouth of Ferry Creek, and the wetlands associated with those areas indicate that they pose a risk to some of the assessment endpoints of this risk assessment, including the benthic community and oyster larvae survival, growth, and reproduction. Risk to benthic invertebrates from CoCs was evaluated using the sediment-quality triad approach (described below). The risk to oyster larvae was measured directly by laboratory toxicity tests, as well as inferred by comparison with benchmark values. These tests are also relevant to interpreting risk to the benthic community as a whole.

8.1 RISK TO THE BENTHIC COMMUNITY

8.1.1 Sediment Toxicity

The sediment-quality triad is a weight-of-evidence approach consisting of synoptically collected measures of bulk sediment chemistry (which are compared with benchmarks), sediment toxicity, and benthic community structure (Chapman et al. 1992). The coincident occurrence of elevated concentrations of CoCs (presented in Section 6.2), greater sediment toxicity (presented in Section 7.1), and benthic community alterations (presented in Section 7.2) act as complementary indicators of adverse impacts to the benthic community.

Under the triad weight-of-evidence approach, a station should not be assumed indicative of unacceptable risk if there is an adverse response in only one of the triad measures. Conversely, the potential for unacceptable risk cannot be dismissed when only one element indicates some potential adverse response. These situations must be interpreted cautiously and according to the site-specific situation.

Indications of adverse response in two of the three triad measures at a station are considered a likely expression of risk. Evidence of toxicity and benthic community alterations, but comparatively low concentrations of CoCs, typically indicate conditions that either the active chemical agent or stressor was not measured by the analytical chemistry; that combinations of contaminants in a mixture acted in synergy; or that environmental conditions exist such that bioavailability of contaminants was altered from the conditions in the field during the sampling and handling process.

Stations with differences in responses of either one or two of the three triad measures indicate some form of stress to biota. These samples require careful consideration and interpretation, however. In some cases, evaluation may involve generating new hypotheses and resampling to determine causative agents, or mitigative agents in the case of high concentrations in sediment chemistry but no apparent toxicity. The easiest interpretation, and clearest demonstration, of unacceptable risk occurs when all three measures in the sediment triad indicate adverse responses.

A tabulation of the results of the sediment-parameter triad used to assess risks to the benthic community is presented in **Table 8-1**. For this table, five key indicator CoCs were selected based on their degree of elevation above either reference samples or sediment quality guidelines, their known association with site-derived waste, and/or their concordance with adverse responses of the bioassessment endpoints noted earlier. These five CoCs are:

- copper
- lead
- total PAHs (tPAH)

- PCB Aroclor 1268
- TCDD toxicity equivalency quotients (TEQ)

Two sediment-quality benchmarks were used, both of which evaluate paired sediment chemistry and toxicity data. An Apparent Effects Threshold (AET) is the concentration of a CoC at which a "probable effect" is observed. The Threshold Effects Level (TEL) is the concentration of a CoC below which an adverse effect is unlikely. Samples with HQ_{AET}s greater than 1 were classified as clearly predictive of unacceptable risk, whereas samples with HQ_{TEL}s less than 1 would indicate a low probability of any risk.

The SEM/AVS ratio is also included in Table 8-1. A ratio less than 1 indicates sufficient AVS to sequester all of the divalent metals measured, while values greater than 1 indicate that a portion of these metals may be bioavailable and may pose potential acute toxicity. Other ligands are known to exist in the sediment, primarily organics, and are known to be influential factors affecting the bioavailability of some of these metals (NOAA 1995). Therefore, values of the AVS ratio slightly greater than 1 are not absolute predictions of acute toxicity. The greater the ratio, however, the more likely that samples could be acutely toxic. The degree to which the SEM/AVS ratio provides predictions of chronic toxicity and bioaccumulation potential is currently a topic of discussion (NOAA 1995).

Samples were considered "toxic" if statistically significant reductions in survival were observed in the laboratory in the amphipod test, relative to the response observed in the control. Statistical comparison was also made to the appropriate reference sample. Optimally, the reference sample replicates all of the characteristics of the test samples (i.e., grain size, TOC, ammonia, sulfides) except the site-related contaminants. Using the reference sample as the comparison response (instead of a laboratory control) is intended to allow for responses due to non-persistent stressors of the sediment matrix. Any response in test sediments beyond that can then be more clearly attributed to stress of site-related contamination. Although the avoidance measurement can be informative, it is given less weight independently as an indication of toxicity (Chapman, pers. commun., 1994).

Samples exhibiting either statistically significantly greater larval abnormality or combined mortality when compared with the control response were considered "toxic" in the oyster larvae bioassay.

Adverse response in benthic community structure was considered present if statistically significant reductions were present at stations when compared with the reference location for any of the following indices of community structure:

- total abundance,
- taxa evenness,
- taxa richness, and
- taxa diversity.

Samples were classified as clearly indicative of unacceptable risk if all three sediment-triad parameters indicated adverse responses. Responses from samples were classified as likely indicators of risk if two of the three parameters indicated adverse responses. Avoidance of sample sediment by amphipods was not given as great a weight as the other measures. Results from either the amphipod or oyster bioassay were used for the sediment-toxicity parameter of the sediment triad. Samples not evaluated by all sediment-triad parameters could be

Table 8-1. Summary of results of sediment quality triad analysis.

Zone	Station/ Sample	Key CoCs versus Guidelines [a]					SEM/AVS ratio [b]	Amphipod Bioassay		Oyster Larvae Bioassay		Altered Benthic Indices [c]				Classification
		Cu	Pb	Total PAHs	PCBs	TEQs		Mor-tality	Avoidance	Abnor-mality	Mor-tality	A	E	R	D	
Boat	HB-23	+	+	+	+	+	+	ns	ns	+	+	+	ns	ns	+++	Unacceptable risk
Club	HB-06	~	•	•	•	•	~	ns	ns	—	—	—	—	—	—	
Wetlands	HB-12	+	~	~	~	•		ns	ns	—	—	—	—	—	—	
Lower	SD-07	~	~	+	+	~		+	ns	—	—	ns	+	ns	++	Unacceptable risk
Ferry	SD-19	~	~	~	+	•	~	ns	+	—	—	ns	ns	ns	++	Potential risk
Creek	SD-10	+	~	~	+	~		ns	+	ns	ns	—	—	—	—	
Upper	SD-13	+	+	+	+	+	~	+	+	+	+	+	ns	+	+++	Unacceptable risk
Ferry	SD-21	+	+	+	+	+	+	+	+	—	—	—	—	—	—	Potential risk
Creek	SD-20	~	~	+	+	+	~	ns	+	—	—	+	+	+	+++	Potential risk

a — + indicates concentration over the AET (*i.e.*, probable effects); ~ indicates value between TEL and AET (*i.e.*, possible effects); and, • indicates below TEL (*i.e.*, improbable effects).

b — + indicates a ratio greater than 5, and ~ indicates a ratio between 1 and 5.

c — A refers to overall abundance; E to evenness; R to richness; and, D to the three diversity numbers of Hill.

—: not tested by this endpoint.

ns: no significant difference was detectable.

categorized only as potentially indicating risk. These classifications, included in Table 8-1, indicate three stations where all sediment-triad parameters clearly indicate significant, unacceptable risk to the benthic community (HB23, SD13, SD07), and three stations which potentially demonstrate conditions of significant risk (SD19, SD20, SD21).

Samples from stations HB23 at the Housatonic Boat Club, plus stations SD13 and SD07 in Ferry Creek, were all classified as adversely impacted. There were indications at those stations of statistically significant mortality following exposure to sediment; exceedance of sediment-quality guidelines in the samples; and impacted benthic community composition at the stations where those samples were collected. These samples had substantially elevated concentrations of the five indicator CoCs (copper, lead, PCBs, total PAHs, and TCDD TEQ). HQ_{AET}s for all five indicator CoCs in samples from stations SD13 and HB23 were above 1 and reached a maximum of 47. The sample from station SD07 contained PCBs and PAHs above their respective AETs, while HQ_{AET}s for TCDD TEQs, copper, and lead were all less than 1 (~0.75). These samples also had detectable levels of other CoCs, including a variety of chlorinated pesticides and chromium. The sample from station HB23 also had the highest SEM/AVS ratio of 37, indicating a fair potential for bioavailable, toxic, divalent metals. The sample from station SD-13 from Upper Ferry Creek had the clearest demonstration of adverse impacts since all sediment-triad parameters were in clear agreement; i.e., sediment analytical chemistry indicated contamination above AET benchmarks; both the amphipod and oyster toxicity bioassays indicated risk; and the benthic community was severely altered. These three stations all present significant, unacceptable risk to benthic organisms. These organisms are likely stressed by chronic lethality, reduced scope for growth, and reproductive impairment.

Samples from stations SD19, SD20, and SD21 were all classified as potentially exhibiting risk to the benthos. These samples either lacked at least one of the bioassessment measures (benthic community at SD21) or provided mixed indications. This situation somewhat limits the certainty with which definitive conclusions regarding risk can be made for these stations. However, in each case the measures available suggest the presence of significant risk. Discussions of these indications at each station follow.

Samples from station SD19 exhibited significant benthic community alterations. These alterations were characterized by reduced number of species, increased dominance by abundant species, and the near-total absence of amphipods. Amphipods are considered sensitive species (Lamberson et al. 1992), and their absence often indicates adverse impacts from chemical contamination. The amphipod toxicity test showed no significant reduction in survival, although test organisms avoided the sample. Avoidance may interfere with the survival endpoint since it tends to reduce exposure levels. However, chemical analysis of the sediment samples, when compared with sediment-quality guidelines (TELs and AETs), did not suggest substantial risk. Concentrations for all five indicator CoCs were between TELs and AETs. The maximum HQ_{AET} calculated was 0.9 for chromium. Hazard quotients were, in fact, intermediate to the two high-salinity reference stations. The oyster-larvae toxicity test was not conducted at this location. This station lies in a side channel, or inlet, on the west side of Lower Ferry Creek. Possibly, the benthic community is responding to stressors other than the CoCs associated with the site-related waste material. It is also likely that the amphipod bioassay may not provide a comprehensive, acute response to the organic contaminants present at this location (e.g., PCBs, dioxins).

The benthic community at SD20 was characterized as having reduced abundance, taxa richness, and a near absence of amphipods. Because this was the station closest to the head of the creek, the benthic community structure may partially reflect the influence of tidal fluctuations, although this would not fully explain the severely reduced abundance of insects, the depressed diversity of species, and reduced overall density. The benthic community at this station may be responding to the toxic stress of organic CoCs. This sample was not identified as toxic by the amphipod bioassay; however, the lack of statistically significant difference in responses of amphipods may be explained by the fact that some of these organic CoCs (especially the chlorinated compounds) would not have come to steady-state during a ten-day amphipod test. Therefore, the acute lethality results may not reflect the impacts to which the benthic community is responding under chronic exposures. Also, the amphipods avoided test sediments which would diminish exposure levels. Mean amphipod mortality in this sample was 23%, just beyond the rejection level for statistically significant differences from the reference value (ρ=0.069). These results certainly seem to indicate toxicity, and there clearly is some form of stress to the benthic community at this station in concordance with general contamination trends. The sediment sample from SD20 exhibited elevated concentrations of PAHs and PCBs, relative to AET sediment-quality guidelines (HQ_{AET}s of 4.7 and 6.8, respectively), plus the second highest concentration of endrin measured. This sample also had the second highest concentration of cadmium (HQ_{AET} of 2.3). The overall HI for this sample was approximately twice that of the reference station. Although the exact nature of the stress evident in the benthic community structure, and the portion of risk posed by chemical contamination, cannot be definitively determined from the available data, these data certainly suggest that the benthic community at this station is potentially at risk from exposure to CoCs.

There were four stations at which samples were analyzed for sediment chemistry and amphipod toxicity, although no survey of the benthic community was conducted. SD21, in Upper Ferry Creek, was one of these four stations. The sample from this station had statistically significant reductions in amphipod survival—the second greatest reduction observed in all the samples where the test was performed. This sample contained the highest concentrations of Cu, Pb, and TCDD TEQs, plus the second highest concentration of PCBs (HQ_{AET}s ranging from 2 to 44). The overall HI for this sample was the highest among all samples and an order of magnitude greater than those for reference samples. Based on the toxicity to amphipods and elevated concentrations of CoCs, this station was considered indicative of unacceptable risk to the benthic community, despite the lack of direct benthic community observations.

There were three additional samples not classified as adversely or potentially affected—those from stations HB06 and HB12 at the Housatonic Boat Club wetland and SD10 in Lower Ferry Creek. The sample from station HB12 was not toxic to amphipods, but did contain moderate levels of copper (HQ_{AET} of 1.5). However, the SEM/AVS ratio for this sample indicated that the copper measured would not be biologically available. Therefore, an acute response in the toxicity test would not be expected from copper. The sample from station HB06 showed neither elevated concentrations of the indicator CoCs nor statistically significant reductions in mean survival in the amphipod toxicity test. Aside from the copper in the sample from station HB12, concentrations of indicator CoCs in these two samples were generally lower than TELs, although occasionally between TELs and AETs. In contrast, the sample from station SD10 was not toxic either by the amphipod or oyster toxicity test, although it exceeded some sediment-quality guidelines. Copper, PAHs, and PCBs exceeded their respective AETs (HQ_{AET} from 1.5 to 5.8). However, the SEM/AVS ratio would suggest that divalent metals (including copper) were not biologically available in this sample.

The absence of adverse biological responses in general accordance with the sediment chemistry further substantiates the integrity of the sediment triad approach, and thereby the conclusions regarding the risk factors applied to other stations using this methodology. The sediment triad analysis indicates that chemical contaminants found in the sediment of Ferry

Creek and the wetland adjacent to the Housatonic Boat Club pose an unacceptable and significant risk to the benthic community. Stations throughout the sampling area had elevated concentrations of CoCs and adverse responses in a variety of indicators of benthic community health. The likelihood of risk was confirmed by the measurement of sediment toxicity in laboratory tests and *in situ* biological effects as measured by alterations to the benthic community structure. Samples with the greatest impacts observed in the bioassessment measures also had the largest number of CoCs present and generally the highest observed concentrations.

To further investigate the association between the biological responses observed and bulk sediment chemistry, the mean concentration of all CoCs in "toxic" samples were compared against those categorized as "non-toxic." Ratios of these means were then calculated: A ratio substantially greater than 1 would indicate a generally greater contribution to the overall contamination by that CoC. Mean toxic concentrations were also compared with AET sediment-quality guidelines. These upper thresholds of toxicity represent the level above which adverse biological responses would *always* be predicted, based on the concentration of just one CoC, as indicated by any one of the biological endpoints included in the AET database. Adverse biological responses also occur when sediment contamination is below the AET value, especially in situations of multiple, cumulative exposure to several CoCs. Results of these analyses are presented in **Table 8-2**. These analyses confirm that PCBs, dioxin TEQs, copper, and lead are the CoCs elevated to the greater degree in toxic samples. These analyses also suggest that cadmium, chlordanes, endrin, and heptachlor epoxide may appear to be potential secondary contributors to risk, according to their relative concentrations in toxic samples. The CoCs that apparently present the greatest proportion of risk, by comparison with AET guidelines, in order, are copper, PAHs, lead, PCBs, and dioxins. Hazard quotients for these mean concentrations of CoCs in the "toxic" samples, relative to AETs, ranged from 56.9 to 8.8. This analysis supports conclusions from the sediment triad that biological impacts observed are driven by exposure.

8.1.2 Potential Risk to Oyster Larvae

Oyster larval toxicity tests were conducted on sediment samples from one station each in Upper Ferry Creek (SD13), Lower Ferry Creek (SD10), and the Housatonic Boat Club wetland (HB23). Test results showed statistically significant increases in the percent of abnormally developed larvae at Stations HB23 and SD13 as well as increases in combined mortality (i.e., percent abnormality plus percent mortality). Stations HB23 and SD13 showed highly elevated concentrations of the five indicator CoCs (HQ_{AETS} from 2 to 47)—most notably Cu, Pb, and TCDD TEQs (see Tables 6-3, 6-4, and 8-2). The presence of elevated CoCs and measurable toxicity in the oyster larval toxicity test indicate that sediment from the Housatonic Boat Club wetland and Upper Ferry Creek pose an unacceptable risk to recruitment in oyster spat beds if sediment from the sample areas is transported to the beds.

8.2 BIOACCUMULATIVE RISK

Several of the CoCs are known to bioaccumulate or biomagnify. These types of CoCs pose the greatest risk to higher-trophic-level organisms through food-web exposures. Risks to fish and to avian species were evaluated primarily by comparing bioaccumulation of CoCs measured in tissues collected from the study areas to benchmark body-burden values associated with known or predicted toxic impacts. This sort of HQ approach identifies samples where toxic benchmarks are exceeded and adverse effects are possible. However, this approach does not define the actual occurrence or magnitude of the corresponding risk.

Table 8-2 Comparison of mean CoC concentrations in toxic samples with mean of non-toxic samples, contrasted with sediment-quality guideline values.

Analyte	Mean of non-toxic samples (n=7±SD)	Mean of toxic [a] samples (n=5±SD)	Ratio of means (toxic/ nontoxic)	TEL	AET	Ratio of toxic/ AET
Arsenic	9.2 ± 3.1	*11.8 ± 3.3*	1.3	7.2	57	0.2
Cadmium	3.0 ± 2.5	**12.5 ± 8.3**	4.2	0.68	2.7	4.6
Chromium	195 ± 88	**245 ± 158**	1.3	52	96	2.6
Copper	606 ± 332	**6030 ± 7478**	10	19	390	15.5
Lead	196 ± 128	**4566 ± 5528**	23	30	430	10.6
Mercury	0.65 ± 0.41	**0.77 ± 0.34**	1.2	0.13	0.41	1.9
Nickel	52 ± 21	**157 ± 75**	3.1	16	110	1.4
Silver	2.26 ± 0.69	**2.43 ± 0.74**	1.1	0.73	0.56	4.3
Zinc	427 ± 151	**1300 ± 567**	3.0	124	410	3.2
Total PAH	30878 ± 55007	**79339 ± 45664**	2.6	1684	~9000	8.8
Total PCB	250 ± 302	**7397 ± 5391**	30	22	130	56.9
TCDD-TEQs [b]	7.8 ± 7.7	**837 ± 1104**	108	5	25	33.5
DDE,4-4	4.8 ± 6.1	*6.3 ± 5.8*	1.3	2.1	16	0.4
DDD,4-4	6.9 ± 4.4	**24 ± 19**	3.5	1.2	16	1.5
DDT,4-4	7.1 ± 12.0	*9.0 ± 6.3*	1.3	1.2	12	0.8
Total DDT	19 ± 21	**39 ± 23**	2.1	3.9	37	1.1
Aldrin	2.0 ± 2.2	3.7 ± 2.2	1.8			
α BHC	3.2 ± 4.0	3.2 ± 2.7	1.0			
β BHC	3.1 ± 3.3	8.8 ± 3.4	2.8			
γ BHC	1.3 ± 1.4	**3.0 ± 2.4**	2.4	0.32	0.99 [c]	3.0
γ Chlordane	4.7 ± 4.4	18 ± 16	3.8			
α Chlordane	3.4 ± 2.5	14 ± 18	4.3			
Total Chlordane	8.1 ± 5.7	**32 ± 33**	4.0	2.3	4.8 [c]	6.7
Dieldrin	2.9 ± 3.4	**9.9 ± 8.0**	3.5	0.72	4.3	2.3
Endrin-A	8.1 ± 7.3	92 ± 83	11			
Heptachlor Epoxide	1.5 ± 1.1	8.0 ± 8.2	5.5			

a *Italic* entries lie between the TEL and AET, indicating possible toxicity. Entries in **bold** lie above the AET, indicating probable toxicity.

b Guidelines from Ianuzzi et al. (1995) and EPA (1993) used for TEL and AET, respectively.

c AET value not available; PEL value from MacDonald et al. (1996).

8.2.1 Potential Risk to Fish

To estimate risk to fish species within the study area, fish tissue body burdens of CoCs were compared to available MATCs. Also, measured water concentrations were compared to AWQCs.

As shown in Table 7-11, three CoCs were observed in mummichog tissues at levels that exceeded their respective MATCs—PCBs, Cd, and PAHs. This evaluation suggests that mummichog in Upper Ferry Creek, nearest the facility, could be at risk due to exposure to cadmium and PAHs. The body burdens of Cd in mummichog samples from Upper Ferry Creek (UF-04 and UF-03) resulted in HQs of 4.4 and 3.8. The MATC for total PAHs was exceeded in mummichog at UF-04 and UF-03 by factors of 3.5 and 3.4, respectively. The HQ calculated for one sample from the reference area RF02 for PCBs was almost 3. As discussed earlier, there were difficulties in the analysis of this sample, and this value may be inaccurate.

Given the magnitude of the HQs (i.e., less than 5), plus the differences in tissues analyzed from the areas of interest versus those represented by the MATC values (i.e., whole vs. eggs or liver), it cannot be stated definitively whether these HQs represent an unacceptable risk to the population of mummichog in Ferry Creek. Fish enzyme systems are quite efficient at metabolizing PAHs. Therefore, the presence of PAHs in whole-animal samples is surprising. However, comparing whole-fish concentrations to an organ-specific MATC (such as liver concentrations) would result in an HQ that underestimates the risk, due to the likelihood that the concentrations in liver in sampled fish would be proportionally higher than the whole-body concentration reported. These considerations would support the conclusion that stocks of mummichog in Upper Ferry Creek might be at risk of reproductive impairment, but the risk to the population throughout the creek cannot be stated with certainty.

A full assessment of potential impact to predatory fish, as indicated by the existing white perch data, could not be completed because of the lack of requisite information. Therefore, no complete estimates of risk to predatory fish are possible.

Risk to fish was also evaluated by comparing surface-water sample concentrations of CoCs with AWQC for the protection of aquatic life. Results of this analysis are presented in **Table 8-3**. AWQC were exceeded for a number of trace elements and PCBs. Samples from SD13 in Upper Ferry Creek contained the largest number of analytes exceeding their respective AWQC, including PCBs, copper, chromium, lead, mercury, and zinc. The only other sample with copper above AWQC was from HB12. Samples from this station also exceeded AWQC for chromium, lead, and mercury. Elevated surface-water concentrations of mercury were measured at many of the other stations sampled. The AWQC for mercury is based on risks from bioaccumulation of mercury, and does not indicate risk from direct exposure for aquatic species. Also, the toxicity of chromium varies considerably depending on the speciation, which was not measured in any samples. The samples with values above AWQC may indicate potential risk depending on the form present. Freshwater AWQC for lead and zinc are a function of the water hardness. A value of 100 mg/L calcium carbonate has been assumed. However, the levels observed may be close enough to the criteria that if the exact hardness of the sample were known, these values may not exceed the hardness-based criteria. The only clear indication of risk is likely associated with the sample from SD13, due to the number and magnitude of exceedances.

Table 8-3. Comparison of AWQC for CoCs with measured water concentrations (µg/L) exceeding criteria. D.L. = detection limit.

COC	Chronic AWQC[a] Freshwater	Chronic AWQC[a] Marine	D.L. (mg/L)	Concentration in Surface Water	Qualifier	Station[b]
Copper	12+	2.9 acute	3.8–45	121	J	SD13
				138	J	HB12
Chromium	11	50 (CrVI)	3.2	20.5	J	SD13
	210	10300 (CrIII) acute		11.1		SD21
				59.2		HB12
Lead	3.2 +	8.5	2.1 & 42	3.7		SD20
				147	J	SD13
				6.1	J	SD21
				5.9		SD21
				37.2		HB12
Mercury	0.012	0.025	0.2	0.57	J	HB23
				2.2	J	HB06
				0.29	J	SD10
				0.37	J	SD21
				0.22		SD21
				6.0	J	RF02
				2.2	J	RF03
				3.5		HB12
				1.2		SD25
				1.9		SD29
				0.80	J	SD28
				0.29	J	SD30
				0.39	J	SD01
				0.57	J	SD22
				3.3	J	SD12
				0.83	J	SD14
				0.78	J	SD06
				0.41	J	SD06
				1.2	J	SD23
				0.31	J	SD37
				0.27	J	SD32
				1.0	J	SD36
				0.47	J	SD36
Zinc	110+	86	2.6–62	127	J	SD13
Total PCBs	0.014	0.03	0.5	0.072	J	SD13

Only detected concentrations are presented.
J = estimated
[a] All AWQC are in µg/L.A; + indicates that the AWQC is hardness-dependent; the value at 100 mg/L $CaCO_3$ shown.
[b] Stations compared with freshwater criteria include 13, 20, and 21.

8.2.2 Potential Risk to Birds

Potential risk to avian receptor species was evaluated using an HQ approach, based on doses derived from a food-web model. Total daily ingestion by each receptor species and CoC was estimated for Ferry Creek, the Housatonic Boat Club wetlands, and Milford Point reference areas. The total daily dose for each CoC was compared with its RTV to calculate an HQ (total daily dose/RTV). If the HQ exceeds 1.0, that CoC is considered to pose some level of risk. The magnitude of the HQ provides an approximate, qualitative indication of the potential risk to the receptor. However, the relationship between the HQ ratio and risk is not linear, and therefore the magnitude of risk is uncertain.

Exposure of black-crowned night heron was evaluated by considering consumption of fish, crabs, terrestrial insects, and sediment. To estimate dietary exposure, fiddler crabs were collected from all sampling areas, fish were collected from Ferry Creek, and terrestrial insects were collected from upper Ferry Creek only. Dietary exposure through fish ingestion was not estimated for the Housatonic Boat Club wetlands because fish were not collected at this area since the wetlands drain completely during low tide. It was assumed that the birds spent 100% of their time feeding at each area (i.e., Ferry Creek, Housatonic Boat Club wetlands, and Milford Pond reference area), therefore a home range exposure factor of 1 was used in the food-web model.

Results of the food-web model indicate that adverse effects to the black-crowned night heron colony at Charles Island (~3.5 miles east of Ferry Creek) will not result from consumption of fish, crab, terrestrial insect, and sediment from Ferry Creek or the boat club wetlands (Tables 7-14a-c). Lead was the only CoC whose HQ exceeded 1.0 at the site-related areas but not at the reference location. The maximum HQ for lead was 3.45 for Ferry Creek, with 60% of the lead exposure coming from an assumed incidental sediment ingestion equal to 5% of the herons' dietary ingestion rate. Moreover, this assessment was based on conservative assumptions for some factors within the food-web model. For instance, despite their feeding-site fidelity, considering that this area is urbanized with houses close to Ferry Creek, it is probably not a preferred foraging area for herons that attracts large numbers of birds. Because there are several other good foraging sites near Charles Island, herons may not feed exclusively near the Raymark facility. Considering the magnitude of the HQs, plus the distance from the heron colony and the other feeding grounds within that distance, exposure to CoCs is not likely to pose substantial risk to the herons.

Exposure of red-winged blackbirds was evaluated by considering consumption of terrestrial insects that may have emerged from an aquatic life stage completed in the Ferry Creek wetlands. The assumptions employed were that red-winged blackbirds spend 90% of their time feeding in the wetlands and 10% feeding in upland areas; also, they feed their nestlings only insects. Based on the results of this assessment, the red-winged blackbird is not at risk of adverse effects from exposure to CoCs from consumption of terrestrial insects present in the wetlands along Ferry Creek. None of the HQs exceeded 1 (see Table 7-15).

9.0 UNCERTAINTY ASSESSMENT

There are many uncertainties associated with an ecological risk assessment. What traditionally is referred to as "uncertainty" actually may be classified as one of two conditions: natural variability and true uncertainty. Natural variability arises from circumstances such as the heterogeneity of responses, test individuals, or ambient conditions. This form of variation in the measurements can be mathematically described. On occasion, sources of variation can be identified and controlled or minimized. True uncertainty, however, represents gaps in knowledge that cannot be mathematically described. In some cases, it may be possible to describe the direction of influence this sort of uncertainty may have on risk estimates; the magnitude of influence may even be discussed. But usually, this form of uncertainty is described qualitatively.

The overall impact of uncertainty in a risk assessment is to introduce a range of confidence about the estimates of risk ultimately derived. This confidence band can be discussed in terms of over- or underestimation of risk and its magnitude. Optimally, risk estimates should be phrased in terms of probabilities. There are circumstances in which probabilistic modeling can be used to estimate the bounds of either the variability or uncertainties. This would require numerical inputs for all aspects of the uncertainty, a situation that is not common and is resource-intensive. The following are typical categories of uncertainty factors that may have major influence in ecological risk assessments:

- extent of the chemical database used to characterize the facility;
- mathematical approximation or distribution used for exposure point concentrations;
- appropriateness of reference areas;
- strength of association between assessment and measurement endpoints;
- use of surrogate species; and
- assumptions of models, including any extrapolations required.

For the benthic community assessment endpoint, the information on chemical nature and extent of contaminants in the sediment was considered reasonable. There were sufficient data to determine an appropriate distribution function for this data. Replication of sediment grabs per station for analyzing benthic community structure was also reasonable (i.e., n=4 each). However, these stations had to be distributed among the four areas of interest. Since it was known that the pattern of contamination within Ferry Creek and the boat club wetlands is extremely heterogeneous, a greater number of stations within each area would have been preferable. The strength of associations between contaminant concentrations and biological measurements within a given area was diminished by the limited number of sampling stations available to characterize both the locations known to have a high degree of contamination (based on previous sampling efforts) and those known to be relatively less impacted by contamination within each of the four areas of interest.

The small number of samples tested with the amphipod test limited the ability to interpret the results. Large numbers of samples, better representing the environmental conditons, would have allowed greater confidence, or greater specificity, in statements regarding the toxicity of individual stations or even entire areas. Additionally, there was a large degree of variability associated with the toxicity measured in the laboratory replicates for two samples—SD07 and SD21. Large variation in laboratory replicates is often an indication of poor laboratory procedures. Well homogenized samples, treated equally, should in theory provide consistent results. The impact of this variability is to widen the confidence intervals about the data. The fact that it was possible to categorize these samples as toxic, given the wider confidence intervals, would tend to strengthen any estimates regarding risk. The fact that these toxicity tests, by design, incorporate factors such as cumulative impacts of multiple chemicals and bioavailability, and are a direct biological measure of effects from exposure to contaminated sediments, also strengthens the conclusions regarding risk. However, the exposure period involved with this bioassay (10 days) is generally too short to reach steady-state for many hydrophobic, organic contaminants, thereby introducing uncertainty regarding the observation of effects from these organic CoCs. Moreover, the amphipod toxicity test relies on acute lethality as its measurement parameter (as opposed to a sublethal, chronic measure of impact). Together, these two factors would tend to underestimate the potential risk when organic contaminants are involved. This is especially true for those CoCs, such as dioxins, whose primary impact is one of latent, reproductive impairment.

Data to support the measurement endpoint associated with the assessment endpoint evaluating the impacts to oyster spat are the most limited. The only direct measure was the oyster developmental test. Resource limitations made it impossible to collect replicate samples in each area of interest for this test. Again, because of the extremely heterogeneous nature of contamination within areas of interest, a single sample per area would tend to increase uncertainty (no estimate of variability can be calculated) and thereby make it more difficult to arrive at conclusions of risk. Also, seasonal difficulties made it impossible to perform the test with the eastern oyster, thus western oyster spat were used as a surrogate. While introducing additional uncertainty in interpreting the results, studies suggest that both species are expected to have similar responses (Dinnel, pers. commun., 1995). Despite any uncertainties, the oyster larvae test still provided indications of risk associated with the CoCs. Similar to the amphipod test, the fact that this endpoint provides direct, biological measures of effects from exposure to sediments contaminated by CoCs also tends to strengthen conclusions of risk. The fact that predictive approaches to estimating risk (i.e., HQs) agreed well with the responses observed with the oyster larvae test also tends to corroborate and strengthen conclusions of risk.

The assessment endpoints for fish involved comparing tissue body burdens of CoCs with benchmark values (MATCs). For predatory fish, such as the white perch, there were insufficient data (including MATCs) to derive any acceptable, complete estimate of risk. For lower-trophic-level fish, as represented by the mummichog, uncertainties in the risk estimate come from three primary sources: (1) a limited number of composite samples, (2) differences between tissues and species as represented by the mummichog and species represented by the MATCs, and (3) the potential for cumulative toxicity from multiple CoCs.

The samples of mummichog were composites of numerous individuals, but there were only four samples per area. Composite samples tend to mask the range of variability in tissue concentrations by averaging the body burdens of individual fish. This tends to broaden the confidence bands with respect to individual fish, but is more representative of the overall population conditions. Fish were collected over a range that is likely smaller than their home range. There is disagreement on the exact home range of mummichog, but 36 m was presumed for this assessment. Although this introduces some level of uncertainty about the exact sediment exposure represented by these composite samples (in the form of increased variation), this factor would result in a reasonable representation of the exposure at the population level.

There were substantial differences in the tissues represented by the MATCs. Many MATCs were tissue burdens in eggs, while the mummichog data were the whole body. Many of the organic CoCs listed in Table 7-11 are highly lipophilic and tend to accumulate in lipid-rich tissues such as eggs and liver. Concentrations for whole-animal body burdens would be less than the value for such lipid-rich tissues due to dilution by other tissues (e.g., muscle). Data presented by Stout et al. (1981) and NOAA (Mearns et al. 1988) suggest that extrapolation factors for DDT between these tissues are an order of magnitude or less. Data presented in Wiener & Spry (1994) for mercury suggest extrapolation factors between brain, liver, muscle, and whole-body concentrations in fish are approximately two- to threefold or less (as total mercury). This use of whole-body burdens in mummichog to derive HQs for CoCs whose MATCs were for eggs may derive HQs that are lower by about an order of magnitude, thus underestimating risk. However, even if levels of TCDD TEQs and DDTs in mummichog were an order of magnitude greater than those represented by whole-body burdens, they would still be less than the MATCs for these CoCs. Therefore, it is unlikely that removing the uncertainty would result in a change in conclusions of risk to the mummichog for these CoCs.

The maximum body burden of PCBs observed in a composite sample of mummichog was in a sample from the reference area. This concentration was flagged as an estimate during quality checks of the data, and was noted as having problems associated with the laboratory analysis. No other analytes were elevated in this sample, and the next-highest body burden observed in reference samples was almost an order of magnitude lower. The nature of this anomalously high PCB level in a reference sample represents another uncertainty factor.

The impact of joint-action toxicity that may occur in circumstances with multiple CoCs is an uncertainty that cannot be addressed in detail. There is very little information that describes the joint-action toxicity of multiple contaminants from broad chemical classes with different modes of toxic action in fish. The common assumption is that toxicity is additive. Although it is known that this is a poor model for general joint-action toxicity, the state of knowledge in wildlife toxicology does not provide a better alternative.

In terms of breadth and possibly magnitude, the greatest degree of uncertainty connected with this ecological risk assessment is associated with the avian food-web models. There were numerous inputs to these models for which assumptions or estimates had to be made. For each of these unknowns, conservative estimates or assumptions were used, which would generally tend to overestimate risk.

There is disagreement among sources referenced about the amount of feeding by blackbirds in a wetland once nesting has started (90% was assumed). Also, it was assumed that the insects fed to nestlings were the same species and the same relative proportions as those caught by net and analyzed for CoC content.

There was no site-specific information on the degree to which heron from the Charles Island colony feed exclusively within the areas sampled (100% was assumed). Black-crowned night heron are opportunistic, general predators; therefore their diet can change dramatically (US EPA 1995). One study of birds on the coastline indicates a diet of 80% fish with the remainder composed primarily of annelids (chiefly *Nereis virens*), crustaceans, and a few insects. Yet another study in an inland marsh indicates a diet of only 30% fish, composed mostly of young birds (primarily gull chicks), beetles, and other terrestrial prey (US EPA 1995). Diet is apparently dependent on local availability of prey. These feeding studies are also based on small sample sizes. Factors such as these obviously lead to higher uncertainties in estimates of doses.

Very limited data on assimilation efficiency of contaminants were available. The maximum value encountered, 85%, was applied to all CoCs (except copper, for which a maximum of 65% was available). Compared with assimilation-efficiency factors reviewed for other taxa (e.g., fish), these assumptions appear to be high and thus may be overly conservative. Assimilation values observed in fish and other taxa are apparently on the order of 55% to 65% for hydrophobic organic contaminants, and lower for super-hydrophobics such as dioxins and some PCBs (Gobas et al. 1988; Barber et al. 1991; Nichols, pers. commun., 1997).

The only RTVs available were for species other than those species of concern (the lowest values encountered in the literature were used). Some RTVs required extrapolation factors to arrive at NOEL levels. Extrapolation factors for species-to-species comparisons generally fall within an order of magnitude (US EPA unpubl.). This would correspond to, at most, an order of magnitude uncertainty in the effect estimate, as expressed by HQs. Because HQs estimate effects at the level of individuals, the ultimate risk to the population would not necessarily correspond to an order-of-magnitude range. For instance, if only a small percentage of individuals from the Charles Island colony received their entire diet from within the study area, those individuals may be at risk, whereas the colony as a whole would not.

There is considerable difference in the toxicity between different states of chromium. Cr^{+6} generally has an order-of-magnitude lower thresholds of toxicity than those for Cr^{+3}. The benchmark for Cr in the avian food-web model was for Cr^{+3}. Comparison of total Cr concentration with this benchmark may underestimate toxicity from Cr exposure. There is added uncertainty to this comparison, however, in that all parties involved agreed not to expend limited resources on speciation of Cr in samples. Since the actual ratio of Cr^{+6} to Cr^{+3} is unknown, there is uncertainty in the dose. An order-of-magnitude decrease in the RTV for Cr would result in HQs exceeding 1 for both the heron and the blackbird. However, because the HQ for Cr for heron was driven by sediment ingestion, which itself was estimated, and the largest HQ for Cr was observed at the reference site, interpretations of risk would still be uncertain. Likewise, the greater HQ for Cr was observed for the reference site.

There is variability and uncertainty associated with all of the analytical results associated with this risk assessment. This is particularly illustrated by the analysis of PCB in crab tissues. Analytical labs may use different techniques to quantify the results of chromatography analysis. For instance, the peak height of a response curve versus the area under the curve might be used to quantify the response. Different peaks and a different number of peaks in a chromatogram may be selected to compare against pure standards to determine which Aroclor mixture is present and at what quantity. These factors and more lead to discrepancies in which value is finally reported for a concentration. In the case of the crab tissues, the original lab reported total PCB concentrations which were on average 70% greater than EPA's interpretation of the same chromatograms. The EPA calculations were the values used in the avian food-web model calculations.

The crab samples were the only tissue samples to be analyzed so as to allow quantification of Aroclor 1268. Omission of Aroclor 1268 in other tissue samples would tend to underestimate the concentration of total PCBs. However, not all PCBs are equally reactive in biological systems (Zabel et al. 1995). Since Aroclor 1268 is dominated by nonaclors, which may have very low biological activity, there is not necessarily a corresponding underestimation of risk.

While all of the factors discussed above add uncertainty to the assessment of risk, any conclusions of risk made in this assessment are substantiated by the fact that evaluations which have taken different approaches to arrive at the same conclusion. This convergence of results and accordance among measurement endpoints from a variety of perspectives reinforces the conclusions that have been made.

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11.0 ACRONYMS

AET	Apparent Effects Threshold
Ag	silver
ANOVA	analysis of variance
As	arsenic
ASTM	American Society for Testing and Materials
AVS	acid volatile sulfide
AWQC	ambient water quality criterion
BJ	bioavailability factor
BSAF	biota sediment accumulation factor
Cd	Cadmium
cm	centimeter
CoC	contaminant of concern
Cr	Chromium
Cu	Copper
DDD	dichloro-diphenyl-dichloro-ethane
DDE	
DDT	dichloro-diphenyl-trichloro-ethane
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
EVS	EVS Environment Consultants, Inc.
Fe	iron
g	gram
Hg	mercury
HQ	Hazard Quotient
HR	home range exposure factor
in	inch
km	kilometer
K_{oc}	organic carbon partitioning coefficient
L	liter
lb	pound
LOAEL	lowest observed adverse effects level
LOEL	lowest observed effects level
m	meter
MATC	maximum acceptable tissue concentrations
mm	millimeter
Ni	nickel
NOEL	no observed effects level
oz	ounce

PAH	polynuclear aromatic hydrocarbon
Pb	lead
PCB	polychlorinated biphenyl
PCDD	polychlorodizenso-p-dioxins
PCDF	polychlorodibenzo-p-furans
PSDDA	Puget Sound Dredged Disposal Analysis
PEL	permissible exposure levels
PLSD	possible least significant difference
ppm	parts per million
ppt	parts per thousand
QA/QC	quality assurance/quality control
QQ	Quantile-Quantile
RA	risk assessment
RI	remedial investigation
SAP	sampling and analysis plan
SEM	simultaneously extracted metals
SEM/AVS	simultaneously extracted metals/acid volatile sulfide
TEF	toxic equivalency factor
TEL	Threshold Effect Level
TEQ	toxic equivalency quotient
TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
TEQ	toxicity equivalency quotient
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence level
VOC	volatile organic compound
Zn	zinc

APPENDIX A
10-d *Leptocheirus plumulosus* Sediment Toxicity Test Raw Data

EVS CONSULTANTS

Amphipod Survival and Emergence Data

Client:	Raymark	Test Species:	*Leptocheirus plumulosus*
Project #:	9/575-29.1	Date Initiated:	August 29, 1995
Work Order:	9500632	Date Terminated:	September 8, 1995
Test Type:	10-d static	Number of Test Organisms:	20

Sample ID	Rep	No. Survivors	No. Emerged Days 1-10	Survival Mean (out of 20)	Survival S.D.[1]	Mean Survival (%)	Emergence (#/jar/day) Mean	Emergence (#/jar/day) S.D.[1]
RM-HB-06-AM	A	19	4	19.6	0.5	98.0	0.3	0.1
	B	20	2					
	C	19	2					
	D	20	3					
	E	20	2					
RM-HB-12-AM	A	17	1	17.6	0.5	88.0	0.2	0.3 CdM
	B	18	7					
	C	17	1					
	D	18	0					
	E	18	3					
RM-HB-23-AM	A	14	2	15.6	2.4	78.0	0.1	0.1
	B	19	1					
	C	13	1					
	D	17	2					
	E	15	1					
RM-RF-02-AM	A	18	0	16.6	0.9	83.0	0.0	0.0
	B	16	1					
	C	17	0					
	D	16	0					
	E	16	0					
RM-RF-03-AM	A	17	0	15.6	2.1	78.0	0.1	0.1
	B	16	0					
	C	14	1					
	D	18	2					
	E	13	0					
RM-SD-07-AM	A	1	7	6.0	6.1	30.0	0.4	0.2 CdM
	B	0	1					
	C	13	4					
	D	4	4					
	E	12	2					

[1]S.D. = Standard Deviation.

C. McPherson
Oct 10/95

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EVS CONSULTANTS

Amphipod Survival and Emergence Data

Client:	Raymark	Test Species:	*Leptocheirus plumulosus*
Project #:	9/575-29.1	Date Initiated:	August 29, 1995
Work Order:	9500632	Date Terminated:	September 8, 1995
Test Type:	10-d static	Number of Test Organisms:	20

Sample ID	Rep	No. Survivors	No. Emerged Days 1-10	Survival Mean (out of 20)	Survival S.D.[1]	Mean Survival (%)	Emergence (#/jar/day) Mean	Emergence (#/jar/day) S.D.[1]
RM-SD-10-AM	A	18	1	18.4	0.9	92.0	0.2	0.1
	B	19	2					
	C	19	1					
	D	17	3					
	E	19	3					
SM-SD-13-AM	A	9	2	11.6	3.4	58.0	0.3	0.2 RM
	B	17	3					
	C	13	4					
	D	10	0					
	E	9	5					
RM-SD-19-AM	A	19	1	15.8	1.9	79.0	0.2	0.1
	B	15	1					
	C	14	4					
	D	15	2					
	E	16	3					
RM-SD-20-AM	A	15	0	15.4	1.7	77.0	0.2	0.2
	B	13	4					
	C	17	0					
	D	17	4					
	E	15	4					
RM-SD-21-AM	A	10	4	6.2	5.0	31.0	0.6	0.3 OK
	B	2	6					
	C	3	11					
	D	13	5					
	E	3	3					
RM-SD-RF-01-AM	A	20	4	19.8	0.4	99.0	0.4	0.2
	B	19	7					
	C	20	4					
	D	20	6					
	E	20	1					

[1] S.D. = Standard Deviation.

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Oct 10/95

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EVS CONSULTANTS

Amphipod Survival and Emergence Data

Client:	Raymark	Test Species:	*Leptocheirus plumulosus*
Project #:	9/575-29.1	Date Initiated:	August 29, 1995
Work Order:	9500632	Date Terminated:	September 8, 1995
Test Type:	10-d static	Number of Test Organisms:	20

Sample ID	Rep	No. Survivors	No. Emerged Days 1-10	Survival Mean (out of 20)	Survival S.D.[1]	Mean Survival (%)	Emergence (#/jar/day) Mean	Emergence (#/jar/day) S.D.[1]
Negative Control[2]	A	20	1	18.5	1.7	92.5	0.1	0.1
	B	19	0					
	C	16	0					
	D	19	1					

[1]S.D. = Standard Deviation.
[2]Replicate E was accidentally dropped prior to test termination.

C. McPherson
Oct 10/95

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EVS CONSULTANTS - 10-d SEDIMENT TOXICITY TESTS
SEDIMENT DESCRIPTION AND CHARACTERIZATION

Client: Raymark
EVS Project No.: 9/575-~~09.1~~ 29.1
EVS W.O. No.: 9500632

Day 0: aug 29, 1995
Day 10: September 8, 1995
Test Species: Leptocheirus plumulosus

SAMPLE I.D.	COLOUR	GRAIN SIZE	SMELL	SHELLS/ DEBRIS	OTHER OBSERVATIONS	TECH. INITIAL
RM-HB-23-Am	Black	silt	Hydrogen Sulfide	grass, Leaves		RAL
RM-SD-07-Am	Black	mud	strong Hydrogen Sulfide	Leaves, Twigs		RAL
RM-SD-19-Am	Brown/ Black	mud	Hydrogen sulfide	grass rocks		ART
RM-SD-21-Am	Black with tan streaks	sand/mud	slight Hydrogen Sulfide	Leaves, grass		RAL
RM-RF-03Am	Black	silt	strong Hydrogen Sulfide	some grass		RAL
RM-SD-13-AM	Black	Mud	strong Hydrogen sulfide	grass, twigs		ART
RM-SD-10Am	Black	fine silt	slight Hydrogen	grass, Twigs Leaves		RAL
RM-SD-20-AM	Brown Black	Mud	Slight Petroleum	grass, twigs		ART
RM-HB-12Am	Black	Heavy mud	strong Hydrogen Sulfide	twigs leaves		RAL

Be descriptive when you characterize the sediments. Colour and grain size information must be complete. If the sediment has an odour, describe the type of smell. Note any shells or debris that are present. Be sure to record anything else in the Observations section.

C. [illegible] Oct 5/95

EVS CONSULTANTS - 10-d SEDIMENT TOXICITY TESTS
SEDIMENT DESCRIPTION AND CHARACTERIZATION

Client: Raymark
EVS Project No.: 9/575-29.1
EVS W.O. No.: 9500632

Day 0: Aug 29, 1995
Day 10: September 8, 1995
Test Species: Leptocheirus plumulosus

SAMPLE I.D.	COLOUR	GRAIN SIZE	SMELL	SHELLS/ DEBRIS	OTHER OBSERVATIONS	TECH INITIAL
RM-SD-RF-01-AM	black	mud	slight petroleum	twigs (small amount)		ART
RM-HB 06AM	Grey	Sand	none	Small Rocks		RA
RM-RF-02-AM	grey-black	mud	slight petroleum	none		ART
Negative Sediment	grey-black	silt	none	none		ART

Be descriptive when you characterize the sediments. Colour and grain size information must be complete. If the sediment has an odour, describe the type of smell. Note any shells or debris that are present. Be sure to record anything else in the Observations section.

C. McPherson
Oct 5/95

sediment.des

RESULTS OF ANALYSIS - Water

File No. F3564

	Ammonia Nitrogen N	Sulphide S
Negative Control Day 10 1995 Sep 8	4.51	<0.02
RM-HB-06 AM Day 10 1995 Sep 8	<0.02	<0.02
RM-HB-12 AM Day 10 1995 Sep 8	1.95	<0.02
RM-HB-23 AM Day 10 1995 Sep 8	0.06	<0.02
RM-RF-~~20~~ 32 AM Day 10 1995 Sep 8	0.98	<0.02
RM-RF-03 AM Day 10 1995 Sep 8	0.86	<0.02
RM-SD-21 AM Day-10 1995 Sep 8	2.44	<0.02
RM-SD-~~07~~ 35 AM Day 10 1995 Sep 8	0.20	<0.02
RM-SD-13 AM Day 10 1995 Sep 8	4.55	<0.02
RM-SD-RF 01AM Day 10 1995 Sep 8	3.21	<0.02
RM-SD-19 AM Day 10 1995 Sep 8	0.33	<0.02
RM-SD-07 AM Day 10 1995 Sep 8	4.59	0.02
RM-SD-10 AM Day 10 1995 Sep 8	5.70	<0.02

< = Less than the detection limit indicated.
Results are expressed as milligrams per litre.

RESULTS OF ANALYSIS - Water

File No. F3911

	Ammonia Nitrogen N	Sulphide S
Neg Control Day 0	0.03	<0.02
Control Sediment Day 0	0.04	<0.02
RM-SD-13 -OY Day 0	0.37	0.04
RM-SD-10 -OY Day 0	0.45	0.05
RM-RF-02 -OY Day 0	0.14	0.03
RM-~~M~~HB-23 -OY Day 0 AT	0.21	<0.02

Results are expressed as milligrams per litre.
< = Less than the detection limit indicated.

APPENDIX B
48-h *Crassostrea gigas* Larval Development Test Raw Data

EVS CONSULTANTS
SEDIMENT DESCRIPTION AND CHARACTERIZATION

Client: Ray Marks
EVS Project No.: 9/575-29.2
EVS W.O. No.: 9500633

Test Species: C. gigas
Test Type/Duration: 48 hr.
Day 0: Sept 20/95

Sample I.D.	Colour	Grain Size	Smell	Shells/ Debris	Other Observations	Tech. Initial
RM-SD-10-04	Black	Fine.	Sulpher	Organics	→ grass & twigs & leaves	J
RM-HB-23-04	Brown to Black	Fine clay-silt	NA	Organics	→ grass & small roots	J
RM-RF-02-~~04~~ OY AT	Dark Grey	silt and fine sand	sulphur	None	Fine layer of what looks like a greyish mould on top of sed.	J
RM-SD-13-~~04~~ OY AT	Black	very fine	sulphur	Lots of organics leaf litter	Roots. Rootlets.	J

Be descriptive when you characterize the sediments. Colour and grain size information must be complete. If the sediment has an odour, describe the type of smell. Note any shells or debris that are present. Be sure to record anything else in the Observations section.

Data Certified By: C. McPherson

Date Certified: Oct 6/95

RESULTS OF ANALYSIS - Water

File No. F3306

	Ammonia Nitrogen N	Sulphide S
Control Sediment Day 0 1995 Aug 29 11:00	3.02	0.02
RM-SD-RF -01-AM Day 0 1995 Aug 29 11:00	2.51	0.03
RM-RF-02 -AM Day 0 1995 Aug 29 11:00	0.42	0.08
RM-RF-03 -AM Day 0 1995 Aug 29 11:00	0.93	0.03
RM-HB-06 -AM Day 0 1995 Aug 29 11:00	0.80	0.06
RM-HB-12 -AM Day 0 1995 Aug 29 11:00	2.28	0.04
RM-HB-23 -AM Day 0 1995 Aug 29	0.74	0.04
RM-SD-07 -AM Day 0 1995 Aug 29 11:00	3.15	0.07
RM-SD-10 -AM Day 0 1995 Aug 29 11:00	3.30	0.07
RM-SD-13 -AM Day 0 1995 Aug 29 11:00	2.31	0.04
RM-SD-19 -AM Day 0 1995 Aug 29 11:00	1.81	<0.02
RM-SD-20 -AM Day 0 1995 Aug 29 11:00	0.61	0.03
RM-SD-21 -AM Day 0 1995 Aug 29 11:00	1.83	0.06

Results are expressed as milligrams per litre.
< = Less than the detection limit indicated.

File No. F4029r

RESULTS OF ANALYSIS - Water

		Negative Control 95 09 22	Control Sediment 95 09 22	RM-RF-02-0Y 95 09 22	RM-HB-23-0Y 95 09 22	RM-SD-10-0Y 95 09 22
Nutrients						
Ammonia Nitrogen	N	0.02	0.02	0.09	0.04	0.28
Inorganic Parameters						
Sulphide	S	<0.02	<0.02	0.06	0.05	<0.02

Results are expressed as milligrams per litre except where noted.
< = Less than the detection limit indicated.

RESULTS OF ANALYSIS - Water

File No. F4029r

		RM-SD-13 -0Y 95 09 22
Nutrients		
Ammonia Nitrogen	N	0.28
Inorganic Parameters		
Sulphide	S	<0.02

Results are expressed as milligrams per litre except where noted.
< = Less than the detection limit indicated.

BIVALVE LARVAL DEVELOPMENT TOXICITY TEST RAW DATA RECORD

Client: Raymark
Project Number: 9/575-29.2
Work Order Number: 9500633
Test Species: *Crassostrea gigas*
Book: 7 Page: 71-78

Date Initiated: Sept. 20, 1995
Date Terminated: Sept. 22, 1995

Initial Density: 30000 embryos/L
Aliquot Size:(mL) 10
Test Volume:(mL) 1000

Sample ID	Rep/ Conc.	Normal Larvae	Abnormal Larvae	Total Larvae	% Abnormal Larvae	Mean % Abnormal	Mean Net % Abnormal	% Combined Mortality	Mean % Combined Mortality	Mean % Net Combined Mortality
Control Sediment	A	306	11	317	3.5	4.1	NA	-2.0	3.1	-1.3
	B	283	14	297	4.7			5.7		
	C	273	12	285	4.2			9.0		
	D	299	15	314	4.8			0.3		
	E	292	10	302	3.3			2.7		
Control Seawater[1]	A	239	7	246	2.8	2.7	NA	20.3	4.3	0.0
	B	291	12	303	3.8			3.0		
	C	323	4	327	1.1			-7.7		
	D	295	9	303	2.8			1.8		
	E	288	10	297	3.2			4.2		

NA = Not Applicable

[1] Due to the variability between replicates, the backup vials were counted to confirm the original counts. Therefore, the normal and abnormal larvae values consist of the average of the original and backup counts.

C. McPherson
Oct 4/95

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BIVALVE LARVAL DEVELOPMENT TOXICITY TEST RAW DATA RECORD

Client: Raymark
Project Number: 9/575-29.2
Work Order Number: 9500633
Test Species: *Crassostrea gigas*
Book: 7 Page: 71-78

Date Initiated: Sept. 20, 1995
Date Terminated: Sept. 22, 1995

Initial Density: 30000 embryos/L
Aliquot Size:(mL) 10
Test Volume:(mL) 1000

Sample ID	Rep/ Conc.	Normal Larvae	Abnormal Larvae	Total Larvae	% Abnormal Larvae	Mean % Abnormal	Mean Net % Abnormal	% Combined Mortality	Mean % Combined Mortality	Mean % Net Combined Mortality
Reference Toxicant (SDS in mg/L)	10.0A	0	1	1	100.0	100.0	100.0	100.0	100.0	100.0
	B	0	3	3	100.0			100.0		
	5.6 A	0	24	24	100.0	100.0	100.0	100.0	100.0	100.0
	B	0	23	23	100.0			100.0		
	3.2 A	11	67	78	85.9	89.6	89.3	96.3	97.7	97.6
	B	3	53	56	94.6			99.0		
	1.8 A	202	34	236	14.4	15.0	12.6	32.7	31.0	27.9
	B	212	39	251	15.5			29.3		
	1.0 A	246	11	257	4.3	4.6	1.9	18.0	9.5	5.4
	B	297	15	312	4.8			1.0		

C. McPherson
Oct 4/95

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BIVALVE LARVAL DEVELOPMENT TOXICITY TEST RAW DATA RECORD

Client: Raymark
Project Number: 9/575-29.2
Work Order Number: 9500633
Test Species: *Crassostrea gigas*
Book: 7 Page: 71-78

Date Initiated: Sept. 20, 1995
Date Terminated: Sept. 22, 1995

Initial Density: 30000 embryos/L
Aliquot Size:(mL) 10
Test Volume:(mL) 1000

Sample ID	Rep/ Conc.	Normal Larvae	Abnormal Larvae	Total Larvae	% Abnormal Larvae	Mean % Abnormal	Mean Net % Abnormal	% Combined Mortality	Mean % Combined Mortality	Mean % Net Combined Mortality
RM-SD-10-OY	A	206	22	228	9.6	12.2 CdM	NA	31.3	34.7	31.8
	B	193	25	218	11.5			35.7		
	C	194	33	227	14.5			35.3		
	D	192	33	225	14.7			36.0		
	E	194	23	217	10.6			35.3		
RM-RF-02-OY	A	186	20	206	9.7	11.5	NA	38.0	43.7	41.1
	B	148	23	171	13.5			50.7		
	C	180	20	200	10.0			40.0		
	D	143	23	166	13.9			52.3		
	E	188	24	212	11.3			37.3		
RM-HB-23-OY	A	45	10	55	18.2	20.2	NA	85.0	83.7	82.9
	B	45	12	57	21.1			85.0		
	C	58	17	75	22.7			80.7		
	D	58	11	69	15.9			80.7		
	E	39	12	51	23.5			87.0		

NA = Not Applicable

C McPherson
Oct 4/95

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BIVALVE LARVAL DEVELOPMENT TOXICITY TEST RAW DATA RECORD

Client: Raymark
Project Number: 9/575-29.2
Work Order Number: 9500633
Test Species: *Crassostrea gigas*
Book: 7 Page: 71-78

Date Initiated: Sept. 20, 1995
Date Terminated: Sept. 22, 1995

Initial Density: 30000 embryos/L
Aliquot Size:(mL) 10
Test Volume:(mL) 1000

Sample ID	Rep/ Conc.	Normal Larvae	Abnormal Larvae	Total Larvae	% Abnormal Larvae	Mean % Abnormal	Mean Net % Abnormal	% Combined Mortality	Mean % Combined Mortality	Mean % Net Combined Mortality
RM-SD-13-OY	A	61	61	122	50.0	47.4	NA	79.7	79.3	78.4
	B	70	51	121	42.1			76.7		
	C	52	58	110	52.7			82.7		
	D	61	52	113	46.0			79.7		
	E	66	57	123	46.3			78.0		

NA = Not Applicable

C. McPherson
Oct 4/95

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