Toxicity assessment of sediments from the Barton Springs watershed, Austin, Texas, USA

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Abstract. The objective of this study was to evaluate the toxicity of sediments from eight sampling locations in the Barton Springs watershed in Austin, Texas, USA. Toxicity tests used in this assessment were a 28-d sediment exposure with the amphipod *Hyalella azteca* evaluating survival and growth. The initial 28-day sediment exposures resulted in no significant declines in either survival or growth. However, subsequent exposure to UV radiation following the sediment toxicity tests resulted in complete mortality of amphipods in two samples. Samples found to be toxic had the highest concentrations of polycyclic aromatic hydrocarbons (PAHs) and metals. A mean PEC quotient of 3.4 (based on concentrations of metals and PAHs) was exceeded in the two sediment samples found to be toxic in the UV tests. A mean quotient of 0.63 was exceeded in one other non-toxic sample from the assessment area. These PEC values are relevant as a 50% incidence of toxicity has been previously reported in a database for sediment tests with *H. azteca* at a mean quotient of 3.4 in 10-d exposures and at a mean quotient of 0.63 in 28-d exposures. However, PEC-Qs described in MacDonald *et al.* (2000a) may over predict the toxicity of samples with PAHs primarily from combustion sources (i.e., with PAHs dominated by high molecular weight PAHs) as was the case with these samples.

Introduction

During the past seven years, elevated concentrations of polycyclic aromatic hydrocarbons (PAHs), organochlorine pesticides, and metals have been documented in bed sediments or in resuspended particles in water in the Barton Springs watershed located in Austin, Texas (Leila Gosselink, City of Austin; personal communication Leila do you have a reference for this that we can use?). The endangered Barton Springs salamander (*Eurycea sosorum*) inhabits the springs. Concern has been expressed that sediments in the springs may be contaminated to concentrations that may be toxic to invertebrates that serve as a food source for these salamanders (Matt Lechner, US Fish and Wildlife Service, Austin, TX; personal communication).

The objective of the current report was to conduct 28-d sediment toxicity tests (ASTM 2001, USEPA 2000b, Ingersoll *et al.* 1998) and subsequent UV exposures with the amphipod *Hyalella azteca* using eight samples collected from Barton Springs Pool. The additional UV exposure was added due to a concern that high molecular weight PAHs in these sediment samples may not be directly toxic, but may become toxic with UV exposure. Consensus-based PECs (MacDonald et al. 2000, Ingersoll et al. 2001) were also used to predict the potential for toxicity in sediments collected from the Barton Springs watershed. Mean PEC quotients (PEC-Q) were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined potential effects of multiple contaminants in sediments collected from the Barton Springs watershed.

Materials and Methods

Sediment collection

Grab samples of surficial sediment used in toxicity tests with *H. azteca* were collected at eight stations in September 2000. Leila do you know how and at what depth these samples were taken? Sediment samples were placed in HDPE buckets and shipped on ice to the Columbia

Environmental Research Center in Columbia, MO. Sediment samples were held at 4°C in the dark for up to six weeks before the start of the toxicity tests (ASTM 2001). Subsamples for the physical and chemical characterization of the sediments were collected from these buckets on Day -1 of the sediment toxicity tests.

Physical and Chemical Characterization of Sediment Samples

Chemical analyses of sediment samples included PAHs, total metals, acid volatile sulfides and simultaneously extracted metals (AVS and SEM), and pore-water metals. Concentrations of PAHs were determined by gas chromatography and mass spectrometry (US EPA 1981, 1984). Concentrations of SEM and AVS were determined using procedures outlined in USEPA (1991). Total concentrations of metals (Ni, Cu, Zn, Cd, and Pb) in pore-waters samples were determined by inductively-coupled plasma mass spectrometry (May *et al.* 1997). Physical characterization of sediment samples included grain size and total organic carbon (TOC) and were determined using procedures outlined in Kemble et al. (1994).

Toxicity Test with Hyalella azteca

Mixed-age amphipods were mass cultured in 80-L glass aquaria containing 50 L of water that received about 6 volume additions/d of well water (hardness 280 mg/L as $CaCO_3$, alkalinity 250 mg/L as $CaCO_3$, and pH 7.80; Ingersoll *et al.* 1998). The amphipods were cultured at a temperature of 23°C and a light intensity of about 500 lux. Amphipods in the cultures were fed

Tetramin® fish food (Ram Fab Aquarium Products, Oak Ridge, TN) and pre-soaked maple leaves *ad libitum*. Each aquarium contained six nylon substrates (20-cm-diameter sections of "Coiled-web material"; 3-M, St. Paul, MN). Amphipods used to start the tests were obtained by collecting organisms from the mixed-aged cultures that passed through a #35 (500-µm opening) U.S. standard size sieve mesh and were stopped by a #40 (425-µm opening) sieve placed under water (ASTM 2001). Amphipods were held in a 2-L beaker for 24 h before the start of a test. Use of this sieving technique resulted in an average length of amphipods of 1.31 mm (0.04 standard error; SE) at the start of the sediment exposures. The sizes of amphipods were comparable with the size of the known-age 7- to 8-d-old amphipods previously used to start sediment tests (Ingersoll *et al.* 1998: 1.2-1.6 mm).

Sediment exposures were conducted for 28 d at 23°C on a 16 light:8 dark photoperiod with a light intensity of about 200 lux. The source of overlying water was well water with two volume additions/d of water to each beaker using an automated system (Zumwalt *et al.* 1994). At the start of the exposure, about 20 amphipods were archived in a solution of 8% sugar formalin for length measurements. A total of 10 amphipods were exposed in each beaker. Sixteen replicates were tested for each sediment sample. Amphipods were fed 1.0 ml of YCT (yeast-cerophyl®-trout chow®,1800 mg/L stock solution; US EPA 2000a) every day.

Hardness, alkalinity, conductivity, dissolved oxygen, conductivity, pH, ammonia, and total residual chlorine were measured in the pore-water samples before the start of the exposures (Day -1) using methods outlined in Kemble *et al.* (1994). Pore-water samples were isolated by centrifugation at 5,200 rpm (7,000 g) for 15 min at 4°C (Kemble *et al.* 1994). About 20 to 50 ml of pore water was used to measure ammonia and a similar volume of pore water was used to measure the other water quality characteristics. A separate 20 ml subsample of pore water was used to prepare the samples for metal analyses (Ingersoll *et al.* 1999). A range in the water quality characteristics of the pore water was observed for hardness (270 to 288 mg/L as $CaCO_3$), alkalinity (240 to 318 mg/L as $CaCO_3$), conductivity (429 to1338 µmho/cm), dissolved oxygen (2.5 to 8.7 mg/L), pH (6.29 to 7.81), total ammonia (1.33 to 50.7 mg/L). See Table 1. The total residual chlorine for all pore water samples was <0.01 ppm. Hardness, alkalinity, conductivity, dissolved oxygen, conductivity, pH, and ammonia were measured in overlying water on Day 0 and 28 of the exposures (Tables 2 and 3). Conductivity and dissolved oxygen in overlying water were also measured weekly during the exposures. Overlying water quality characteristics were generally similar among treatments. Dissolved oxygen levels in overlying water were at or above acceptable levels of 2.5 mg/L in all treatments throughout the study (ASTM 2001, US EPA 2000a).

On Day 28 of the exposures, sediments in each beaker (n=16/treatment) were sieved through a #50 sieve (300-µm opening). The debris and amphipods remaining in the sieve were rinsed into a glass tray and searched for up to 20 minutes for organisms, and surviving amphipods were counted. Half of the replicate (n=8/treatment) were then preserved in 8% sugar formalin for length measurements. Length of amphipods was measured along the dorsal surface from the base of the first antenna to the tip of the third uropod along the curve of the dorsal surface using a microscope and digitizing system (Kemble *et al.* 1994). The remaining replicates (n=8) were then placed in 300 ml beakers with about 15 ml of clean sand for the UV exposures. For each treatment 4 replicates were exposed to two levels of UV exposure. Level 1 was a control with UVB- 0 $\mu W/cm^2$, UVA- 3.2 $\mu W/cm^2$, and visible light 250 $\mu W/cm^2$. Level 2 was an elevated dose with UVB- 4 $\mu W/cm^2$ - 4/h-d, UVA- 57 $\mu W/cm^2$, and visible light 1800 $\mu W/cm^2$. This elevated UV exposure dose has been previously described as a medium dose by Little et al. (2000). No data are available on the irradiance of UV that penetrate the water column in the Barton Springs watershed. Therefore, the elevated UV treatment used in these exposures was selected to represent about 1% of incidence light at water depth of 10 cm at other sites in North America (Hurtubise *et al.* 1998, Barron *et al.* 2000). Amphipods were fed 1.0 ml of YCT every day. The UV toxicity tests were performed in a solar simulator equipped with cool-white, UVB (280-320 nm) and UVA 320-400 nm) flourescent lamps and halogen flood lamps as described by Little et al. (2000).

Exposures were static renewal with a renewal at 24 hours. Exposures were conducted for 96 hours. However, after 72 hours there was a significant reduction in survival of the control amphipods. Therefore we considered only the first 48 hours of the testing.

Data Analyses

Statistical analyses for the amphipod exposures were conducted using one-way analysis of variance (ANOVA) at $\alpha = 0.05$ for all endpoints except length which was analyzed using a one-way nested ANOVA at $\alpha = 0.05$ (amphipods nested within a beaker). If the results of the ANOVA were significant, mean separation was performed relative to each of the control sediments by Fisher's protected least-significant difference test at $\alpha = 0.05$. Percent survival and length data were log transformed before analysis. All statistical analyses were performed with

Statistical Analyses System programs (SAS 1998).

Probable effect concentrations (PECs) were used to assess the relationship between sediment chemistry and toxicity. The PECs are effect-based sediment quality guidelines that were established as concentrations of individual chemicals above which adverse effects in sediments are expected to frequently occur in field-collected sediments (MacDonald *et al.* 2000a). Mean quotients based on PECs were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined effects of multiple contaminants in sediments (MacDonald *et al.* 2000a, US EPA 2000b, Ingersoll *et al.* 2001). A PEC quotient (PEC-Q) was calculated for each chemical in each sediment sample by dividing the dry-weight concentration of a chemical by the PEC for that chemical. A mean PEC-Q was then calculated for each sample by summing the individual quotient for each chemical and dividing this sum by the number of PECs evaluated. In calculating concentrations of total PAHs, half the detection limit was used for compounds reported below the detection limit (Ingersoll et al. 2001). If the concentration was below the detection limit but above the PEC, this value was excluded from the calculation of the PEC-Q.

We were interested in equally weighting the contribution of metals and PAHs in the evaluation of sediment chemistry and toxicity (assuming these diverse groups of chemicals exert some form of collective toxic action). For this reason, we first calculated an average PEC-Q for up to the metals in a sample based on dry weight concentrations of cadmium, copper, lead, nickel, and zinc. A mean quotient was then calculated for each sample by summing the average quotient for metals, the quotient for total PAHs, and the quotient for total PCBs, and then dividing this sum by three (n = 2 quotients/sample; see Ingersoll *et al.* (2001) for additional details on the procedure used to calculate mean PEC-Q). Use of this approach for calculating the quotients was selected to avoid over weighting the influence of an individual chemical (i.e., a single metal) on the combined mean quotient (USEPA 2000b).

Results and Discussion

Physical and Chemical Characteristics of Sediment Samples

A range in grain size and TOC was observed in the sediment samples collected from the eight sample stations (range of 54 to 78% sand and 5.5 to 13% TOC; Table 4). There was a range of metals among the porewater from the sediment samples; Ni (2.5 to 8.1 ng/ml), Cu (<2.5 to 25 ng/ml), Zn (<1.7 to 62 ng/ml) Cd (<0.08 to 0.34) and Pb (<0.34 to 3.6) (Table 5). Cu, Zn, Cd, and Pb concentrations (µg/g dry weight) were highest from sediment in site 1266 (Table 6). Results of the SEM and AVS analysis indicated a lack of positive SEM-AVS differences for any of the sediments suggesting a low potential for toxicity from metal contaminants. Concentrations of AVS have been demonstrated to influence pore-water concentrations and bioavailability of divalent metals in sediment toxicity and bioaccumulation tests (Ankley *et al.* 1996). An excess molar concentration of SEM relative to AVS and elevated concentrations of metals in pore water indicate that metals may be contribute to the toxicity of a sediment sample (Ankley *et al.* 1996).

Concentrations of total PAHs exceeded the PEC of 22.8 µg/g in 3 of the 8 samples from the assessment area (Table 6). A mean PEC quotient of 3.4 was exceeded in 2 of the sediment samples and a mean quotient of 0.63 was exceeded in 3 sediment samples from the assessment area. A 50% incidence of toxicity was reported in a database for sediment tests with *H. azteca* at a mean quotient of 3.4 in 10-d exposures and at a mean quotient of 0.63 in 28-d exposures (Ingersoll *et al.* 2001). Results of these chemical analyses of the mean PEC-Q indicate a that samples from the assessment area would be expected to be toxic to sediment-dwelling organisms.

Toxicity Test with *Hyalella azteca*

Survival of amphipods in the control was ≥ 94% at the end of both the 28-d sediment exposures and the 48-h UV exposures. Results of the 28-d sediment exposures are reported in Table 7. Survival rates of all treatments were ≥ 84% indicating no significant mortality of amphipods relative to the control. This relationship also held for the growth measurement. No significant reduction in length was seen in any treatment relative to the control. To the contrary, seven of the eight sites exhibited growth significantly greater than that seen in the control (Table 7). This enhanced growth may have been due to those sediments having a greater percentage of nutrients assessable to the amphipods.

Results of the replicates which received UV exposures following the 28-d sediment toxicity test are described in Table 8. Significant reductions in survival were observed in two of the samples in the elevated UV exposure. Samples from sites 1266 and 1234 had 0% survival after the first 24-h of the 48-h UV exposure. At the conclusion of the 48-h exposure, survival for the remaining 5 samples exposed to the dosed UV exposure ranged from 84 to 94% which was not significantly different from the control. In the control (low) UV exposure there were no significant reductions in survival in any of the treatments compared to the control.

The importance of evaluating photoinduced toxicity of PAHs and other compounds by UV has been recognized in aquatic environments (Oris and Giesy 1985, Davenport and Spacie 1991, Ankley *et al.* 1994, Cleveland *et al.* 1997). Moreover, standard toxicity testing methods provide guidance to support the design of experiments to assess photoinduced toxicity (ASTM 2001, USEPA 2000a). Therefore, the influence of UV should be considered when evaluating the toxicity of compounds expressed by this mode of action. Photoinduced toxicity is particularly important when evaluating sediments in clear, shallow water systems.

The lack of toxicity of sediments collected from Barton Springs Pool to *H. azteca* at the end of the 28-d exposure was unexpected (i.e., before the UV exposure). Based on calculated mean PEC-Qs, a probability of toxicity exceeding 0.5 was predicted for three of these samples (1279, 1234, and 1266; Table 4). The source of PAHs in these sediments is thought to include asphalt from older parking areas deposited during storm events (Leila Gosselink, City of Austin, personal communication). The percentage of lower molecular weight (LMW) PAHs relative to total PAH in these samples was quite low in the samples from Barton Springs Pool (only about 10 to 13%; Table 6) as might be expected for PAHs from combustion sources such as asphalt. In contrast, the proportion of LMW PAHs tended to be much higher (i.e., 15 to 95%) in the samples that were previously evaluated in the assessment of PECs (USEPA 2000a) suggesting that petroleum-based sources of PAHs were predominant (Van Meter *et al.* 2000). Van Meter *et al.* (2000) described an approach that can be used to distinguish between combustion- and petroleum-based PAHs in sediments (a ratio of the sum two- and three-ring PAHs to the sum combustion PAHs). These ratios were $\leq$0.12 for all of the samples listed in Table 4 and were less than or equal to the lowest ratios reported in Van Meter *et al.* (2000) for a range of sediment samples collected from across the United States. Results of these analyses support the conclusion that the PAHs in Barton Springs Pool are primarily from combustion sources.

Preliminary analyses of the USEPA (2000a) database indicate a trend of lower correct classification of the toxicity attributed to PAHs in samples with a lower percentage of LMW PAHs (<40%) compared to samples with a higher percentage of LMW PAHs (>40%). Therefore, PEC-Qs described in MacDonald *et al.* (2000a) may over predict the toxicity of samples with PAHs primarily from combustion sources (i.e., with PAHs dominated by HMW PAHs). However, the samples evaluated from Barton Springs Pools with a low percentage of LMW PAHs were toxic to *H. azteca* following subsequent exposure of amphipods to UV. Additional analyses of the USEPA (2000a) database are ongoing to evaluate the toxicity associated with LMW and HMW PAHs to sediment-dwelling organisms (i.e., using the approach used by Van Meter *et al*. 2000 for classifying combustion- vs. petroleum based sources of PAHs in sediment).

Conclusions

Elevated concentrations of polycyclic aromatic hydrocarbons (PAHs) and metals have been documented in bed sediments or in resuspended particles in water in the Barton Springs watershed located in Austin, Texas. The endangered Barton Springs salamander (*Eurycea sosorum*) inhabits the springs and concern has been expressed that sediments in the springs may be contaminated to concentrations that may be toxic to invertebrates that serve as a food source for these salamanders. Consensus-based probable effect concentrations (PECs) were used to predict the potential for toxicity in sediments collected from the springs. These PECs were developed to determine the concentration of contaminants above which adverse effects are likely to be observed to sediment-dwelling organisms. Mean PEC quotients (PEC-Q) were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined potential effects of multiple contaminants in sediments collected from the Barton Springs watershed. In addition, 28-d sediment toxicity tests were conducted with the amphipod *Hyalella azteca* with samples collected from Barton Springs Watershed. Mean PEC-Q in sediments from the watershed were frequently elevated to levels that would be predicted to be toxic to sediment-dwelling organisms and elevated concentrations of PAHs contributed to these elevated quotients. Sediments from the Barton Springs watershed were toxic to *H. azteca* following exposure of amphipods to UV. Results of these evaluations indicate that sediments in watershed are contaminated with PAHs to concentrations that are likely toxic to invertebrates that serve as a food source for the endangered salamanders. Additional studies are recommended to determine: (1) the depth of UV penetration into the water column within the watershed during different times of the year and (2) the contribution of UV to the toxicity of contaminants to organisms inhabiting the watershed.

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Table 2: Overlying Water Quality Characteristics Measured at the Start of the Amphipod Toxicity Tests with Samples from Austin, TX.

Exposures	Hardness (mg/L as $CaCO_3$)	Alkalinity (mg/L as $CaCO_3$)	Conductivity (µmho/cm)	Dissolved oxygen (mg/L)	pH	Ammonia (mg/L)
Control	274	240	605	8.1	8.25	0.364
879-1	292	276	681	6.5	8.12	1.13
879-2	292	278	684	7.5	8.12	0.753
219	320	280	658	7.2	8.07	1.08
1279	330	280	657	6.9	8.11	<0.1
1234	290	262	649	7.2	8.25	<0.1
35	274	260	636	7.9	8.28	<0.1
463	282	266	643	7.2	8.15	0.598
1266	300	276	649	7.9	7.88	<0.1

Table 3: Overlying Water Quality Characteristics Measured at the End of the Amphipod Toxicity Tests with Samples from Austin, TX.

Exposures	Hardness (mg/L as $CaCO_3$)	Alkalinity (mg/L as $CaCO_3$)	Conductivity (µmho/cm)	Dissolved oxygen (mg/L)	pH	Ammonia (mg/L)
Control	310	240	629	8.2	8.26	0.143
879-1	268	236	627	7.4	8.24	0.497
879-2	280	240	638	7.6	8.28	0.103
219	270	230	754	7.5	8.36	<0.1
1279	284	260	790	7.8	8.39	<0.1
1234	298	270	795	7.8	8.35	<0.1
35	282	260	784	7.8	8.28	<0.1
463	296	242	640	7.6	8.25	0.141
1266	288	250	786	7.8	8.36	<0.1

Table 1: Porewater Quality Characteristics Measured from the Bulk Sediment Used for the the 28-day Amphipod Toxicity Tests with Samples from Austin, Tx.

Exposures	Hardness (mg/L as $CaCO_3$)	Alkalinity (mg/L as $CaCO_3$)	Conductivity (µmho/cm)	Dissolved oxygen (mg/L)	pH	Ammonia (mg/L)
Control	270	240	712	4.5	6.29	10.9
879-1	282	260	1234	3.1	7.51	4.05
879-2	288	318	1338	2.8	7.58	4.60
219	284	284	911	2.8	7.51	17.3
1279	298	260	965	7.0	7.74	1.45
1234	286	246	429	6.9	7.81	1.33
35	288	248	744	2.9	7.45	1.44
463	280	270	803	2.5	7.20	50.7
1266	284	242	1232	8.7	7.44	5.81

Table 5: **Concentrations (ng/ml) of Metals in Sediment Porewater from the Bulk Sediment Used for the the 28-day Amphipod Toxicity Tests with Samples from Austin, Tx.**

Exposures	Ni	Cu	Zn	Cd	Pb
Control	67	5.3	51	<0.08	0.73
879-1	5.1	4.1	2.8	<0.08	0.37
879-2	8.1	4.2	8.8	<0.08	2.6
219	3.8	<2.5	<1.7	<0.08	0.52
1279	7.2	14	23	0.34	3.6
1234	2.8	3.3	10	0.31	<0.34
35	2.5	6.0	<1.7	<0.08	<0.34
463	5.6	5.3	4.8	<0.08	1.2
1266	4.0	25	62	0.11	0.92

Table 4: Sediment Characteristics (Particle Size and Total Organic Carbon) of 8 Field-collected Sediment Samples from Austin, Tx.

		Particle Size (%)		Total Organic Carbon (%)
Exposures	sand	clay	silt	carbon
879-1	67	25	8.0	12
879-2	60	36	4.0	9.0
219	63	24	13	11
1279	54	30	16	10
1234	72	18	10	7.5
35	49	21	30	13
463	78	18	4.0	12
1266	74	19	7.0	5.5

Table 7: Survival and Growth (Reported as Mean Lengths with Standard Deviations in Parenthesis) Results from a *Hyalella azteca* 28-d Sediment Toxicity Test Using West Bearskin Sediment as the Control to Evaluate 8 Field-collected Sediment Samples from Austin, Texas.

Exposures	Survival (%; n=16)	Length (mm; n=8)
Control	94 (0.8) A	3.85 (0.46) A
879-1	95 (2.3) A	4.70 (0.59) B
879-2	91 (0.4) A	5.11 (0.41) B
219	99 (0) A	4.43 (0.43) B
1279	93 (1.5) A	4.78 (0.53) B
1234	84 (1.5) A	4.66 (0.45) B
35	93 (1.5) A	4.00 (0.39) A
463	97 0.4) A	4.29 (0.40) B
1266	98 (0.8) A	4.15 (0.46) B

Mean values within a column followed by a similar letter are not significantly different ($p \geq 0.05$).

Table 8: Survival (%, N=4) Results From 48-hour UV Water-only Exposures of *Hyalella azteca* to a low and elevated UV exposure. These Tests Were Performed Following a 28-d Sediment Exposure to Evaluate 8 Field-collected Sediment Samples from Austin, Tx.

	24-hr		**48-hr**	
Exposures	low UV	elevated UV	low UV	elevated UV
Control	98 (0.4) A	97 (1.0) A	98 (0.4) A	95 (0.4) A
879-1	100 (0) A	98 (0.8) A	100 (0) A	98 (0.8) A
879-2	100 (0) A	100 (0) A	100 (0) A	95 (0.5) A
219	100 (0) A	100 (0) A	100 (0) A	100 (0) A
1279	100 (0) A	100 (0) A	100 (0) A	85 (2.3) A
1234	100 (0) A	0 (0) B	100 (0) A	0 (0) B
35	100 (0) A	95 (1.0) A	100 (0) A	95 (1.0) A
463	100 (0) A	97 (0.4)A	100 (0) A	89 (1.3) A
1266	100 (0) A	0 (0) B	98 (1.9) A	0 (0) B

Mean values within a column followed by a similar letter are not significantly different

Table 6: Measured Chemistry and Predicted Probability of Toxicity Based on the Analytical Chemistry Results (Dry Weight) of the Eight Sediment Samples Collected from Austin, Texas

Analytes	Control	879-1	879-2	219	1279	1234	35	463	1266
Metals (mg/kg)									
Cadmium	0.07	0.18	0.18	0.09	0.28	0.18	0.18	0.20	0.45
Copper	0.08	<0.19	0.56	<0.19	5.16	6.23	1.25	0.34	33.2
Lead	0.06	13.20	12.10	10.20	20.10	17.50	12.70	9.58	40.1
Nickel	0.20	6.33	5.51	3.29	6.81	4.73	5.90	6.70	5.24
Zinc	0.09	36.2	38.8	20.10	64.40	53.20	23.90	30.30	93.3
PAH (ug/kg)									
Acenaphthene	<100	<309	<309	<426	375	726	<282	<18	1042
Acenaphthylene	<100	<309	<309	<426	<198	178	<282	<18	305
Anthracene	<100	<309	<309	<426	355	2166	<282	<18	4320
2-methylnaphthalene	<100	<309	<309	<426	<198	<128	<282	<18	<128
Benz(a)anthracene	<100	2099	1692	<426	5128	20382	621	48	49555
Benzo(a)pyrene	<100	4938	5231	596	8679	25478	1469	78	69886
Chrysene	<100	2716	2985	723	8087	24204	537	104	80051
Dibenz(a,h)anthracene	<100	1667	1815	<426	2959	8535	621	45	25413
Fluoranthene	<100	4630	4923	894	10651	58599	1158	108	132147
Fluorene	<100	<309	<309	<426	375	1006	<282	<18	1525
Naphthalene	<100	<309	<309	<426	<198	<128	<282	<18	<128
Phenanthrene	<100	1173	1200	<426	4734	22930	311	39	43202
Pyrene	<100	3395	4000	468	8087	44586	904	91	108005
LMW PAHs	350	2099	2123	1489	6134	27134	1158	95	50521
HMW PAHs	300	19444	20646	3106	43590	181783	5311	473	465057
Total PAHs	650	21543	22769	4596	49724	208917	6469	568	515578
% LMW of Total PAHs	54	10	9	32	12	13	18	17	10
Mean PEC-Q	0.06	0.51	0.53	0.12	1.14	4.62	0.17	0.04	11.4
Predicted probability of toxicity	0.05	0.43	0.44	0.11	0.70	1.0	0.16	0.03	1.0
Observed survival (%) after UV exposure	95	98	95	100	84	0	95	90	0