

Sediment Contamination of Residential Streams in the Metropolitan Kansas City Area, USA: Part II. Whole-Sediment Toxicity to the Amphipod *Hyaella azteca*

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Abstract This is the second part of a study that evaluates the influence of nonpoint sources on the sediment quality of five adjacent streams within the metropolitan Kansas City area, central United States. Physical, chemical, and toxicity data (*Hyaella azteca* 28-day whole-sediment toxicity test) for 29 samples collected in 2003 were used for this evaluation, and the potential causes for the toxic effects were explored. The sediments exhibited a low to moderate toxicity, with five samples identified as toxic to *H. azteca*. Metals did not likely cause the toxicity based on low concentrations of metals in the pore water and elevated concentrations of acid volatile sulfide in the sediments.

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Although individual polycyclic aromatic hydrocarbons (PAHs) frequently exceeded effect-based sediment quality guidelines [probable effect concentrations (PECs)], only four of the samples had a PEC quotient (PEC-Q) for total PAHs over 1.0 and only one of these four samples was identified as toxic. For the mean PEC-Q for organochlorine compounds (chlordane, dieldrin, sum DDEs), 4 of the 12 samples with a mean PEC-Q above 1.0 were toxic and 4 of the 8 samples with a mean PEC-Q above 3.0 were toxic. Additionally, four of eight samples were toxic, with a mean PEC-Q above 1.0 based on metals, PAHs, polychlorinated biphenyls (PCBs), and organochlorine pesticides. The increase in the incidence of toxicity with the increase in the mean PEC-Q based on organochlorine pesticides or based on metals, PAHs, PCBs, and organochlorine pesticides suggests that organochlorine pesticides might have contributed to the observed toxicity and that the use of a mean PEC-Q, rather than PEC-Qs for individual compounds, might be more informative in predicting toxic effects. Our study shows that stream sediments subject to predominant nonpoint sources contamination can be toxic and that many factors, including analysis of a full suite of PAHs and pesticides of both past and present urban applications and the origins of these organic compounds, are important to identify the causes of toxicity.

Chemistry, toxicity and bioaccumulation tests are frequently used to evaluate risks of contaminated sediments to environmental receptors (USEPA 2005a). Whereas measurements of chemistry alone provide quantitative information on contaminants, toxicity and bioaccumulation tests provide direct measures of biological impacts (ASTM 2009). A combination of these approaches reveals two dimensions of sediment quality, a result that can be

evaluated by comparisons to effect-based sediment quality guidelines (SQGs; Wenning et al. 2005).

A variety of toxicity tests with aquatic organisms have been developed to provide data for environmental risk assessment and management, natural resource damage assessment, and hazard mitigation related to sediment-bound contaminants (Blaise et al. 1998; USEPA 2000b; ASTM 2009). These procedures discriminate among different test matrixes (e.g., whole sediment, elutriate, or pore water) and use a variety of test organisms. One procedure might be more effective or applicable than others to identify effects of contaminants in sediment (Blaise et al. 1998). Thus, a challenge is how to prescribe the most sensitive or appropriate toxicity test for sediments of interest to evaluate the impact contaminated sediment might pose to sediment-dwelling organisms.

Sediment quality guidelines can also be used to evaluate the severity of sediment contamination (Wenning et al. 2005). Various SQGs have been developed for individual or groups of contaminants. For example, mechanistically based equilibrium partitioning sediment benchmarks (ESBs) have been established for metal mixtures, polycyclic aromatic hydrocarbons (PAH) mixtures, the nonionic organic insecticides endrin and dieldrin, and a mixture of 32 nonionic organic chemicals (USEPA 2000a, 2003, 2005a, 2005b, 2008). Empirically based effects range median (ERM), probable effects levels (PELs), and probable effect concentrations (PECs) have been developed for individual trace metals and PAHs, mixtures of PAHs, and mixtures of chlorinated organic hydrocarbons (Long et al. 1995; MacDonald et al. 2000; Ingersoll et al. 2001).

This investigation is the second part of a study evaluating the potential impact of nonpoint sources on the sediment quality of five adjacent streams within the metropolitan Kansas City area, which straddles the states of Kansas and Missouri, United States (Tao 2005). The first part of the study characterized the distributions and implications of organic contamination in sediment (Tao et al. 2010); this second part presents an evaluation of the sediment toxicity in the study streams. The *Hyaella azteca* 28-day toxicity test was selected with the consideration that limited data on sediment quality were available for this region and that the 28-day test measuring survival and growth is generally more sensitive than the short-term 10-day test with *H. azteca* (Ingersoll et al. 2001). Additionally, the 28-day test provides similar information compared to a 42-day sediment toxicity test with *H. azteca* measuring potential reproductive effects (Ingersoll et al. 1998). Sediment chemistry and toxicity data were used in combination with SQGs to evaluate the quality of the sediments subject to impacts from a limited number of major point sources (e.g., Publicly Owned Treatment Works and a Combined Sewer System) and diffuse and diverse nonpoint

sources from predominant residential land uses and to explore the potential causes for the observed toxic effects. The applicability of the SQGs to identify sediment contamination associated with nonpoint sources was also examined.

Materials and Methods

Sediment Collection, Handling, and Characterization

The study streams—Turkey, Cedar, Brush, Indian, and Tomahawk creeks—are located southeast of the confluence of the Lower Kansas River and Lower Missouri River within the metropolitan Kansas City area (see Fig. 1 in Tao et al. 2010). The adjoining watersheds span a continuum of urban development from residential watersheds to rural agricultural watersheds southwestward, with few major point sources of contamination. Despite the nonindustrial nature of these watersheds, the stream sediments contain elevated concentrations of PAHs and multiple historical-use organochlorine compounds (Lee et al. 2005; Tao 2005; Tao et al. 2010). These contaminants likely originate from a diverse spectrum of nonpoint sources such as traffic-related activities, coal-tar-related applications, and past residential use of pesticides (Tao et al. 2010).

Sediment sampling was conducted at 29 sites in late June 2003 (Tao et al. 2010). Multiple sediment samples (up to 10 cm in depth) collected at each site were composited, homogenized, and subsampled for physical, chemical, and toxicity testing. Procedures for collecting, processing, and analyzing sediment for grain size, total organic carbon (TOC), PAHs, PCBs, and pesticides are described in Tao (2005) and Tao et al. (2010). Oil and grease was determined by using *n*-hexane as the extraction solvent and gravimetric method. Subsamples for acid volatile sulfide (AVS) and simultaneously extracted metals (SEMs) were stored refrigerated in the dark for less than 20 days and analyzed at the departments of Geosciences and Chemistry of the University of Missouri–Kansas City. Subsamples for toxicity testing were held in the dark at about 4°C for less than 2 weeks before shipment to Columbia Environmental Research Center (CERC) of the US Geological Survey (USGS) in Columbia, Missouri. Samples were processed for pore-water extraction and toxicity testing at the USGS laboratory starting within 24 h of sample receipt.

Sediment samples for AVS and SEM analysis were processed using the procedures outlined in Allen et al. (1991). AVS was determined using a Spectronic 20D spectrometer (Milton Roy Company) with the absorbance set at 670 nm. Quality control included establishing the method detection limit (0.038 $\mu\text{mole S}^{2-}/100\text{ ml}$) and linear calibration curves using laboratory reagent blanks

and fortified blanks, and assessing analyte recovery (80–108%). Simultaneously extracted metals (Cd, Cu, Pb, Ni, Zn) were measured using a Varian UltraMass 600 inductively coupled plasma–mass spectrometer (ICP-MS), following US EPA Method 200.8 (USEPA 1994) and are expressed as milligrams per kilogram or micromoles per gram (dry weight).

Polycyclic aromatic hydrocarbons analytes included two- to three-ring low molecular weight (LMW) PAHs (i.e., naphthalene, acenaphthene, acenaphthylene, fluorene, phenanthrene, and anthracene), four- to six-ring high molecular weight (HMW) PAHs (i.e., fluoranthene, pyrene, benzo[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, indeno[1,2,3-*cd*]pyrene, and benzo[*g,h,i*]perylene), along with 2-methylnaphthalene. Pesticide-related analytes included 2,4-D, 2,4,5-T, alachlor, atrazine, B-/G-HCH, bentazon, benefin, chlorpyrifos, *cis*-/*trans*-chlordane, technical chlordane (based on three isomer peaks: *trans*-chlordane, *cis*-chlordane, and *trans*-nonachlor), diazinon, dicamba, dieldrin, endrin, heptachlor, heptachlor epoxide, metribuzin, *o,p'*/*p,p'*-DDT, *o,p'*/*p,p'*-DDD, *o,p'*/*p,p'*-DDE, pendimethalin, pentachlorophenol, picloram, and trifluralin. Aroclor mixtures of polychlorinated biphenyls (PCBs)—Aroclor 1248, Aroclor 1254, and Aroclor 1260—were also measured from the pesticide extract (Tao et al. 2010). Concentrations of PAHs and pesticides are expressed as micrograms per kilogram (dry weight). Quality assurance and quality control measures are described in Tao et al. (2010).

Sediment Toxicity Testing

The sediment samples were evaluated using a 28-day whole-sediment exposure with *H. azteca* following USEPA (2000b) and ASTM (2009). Ten amphipods were exposed in 100 mL of sediment in 300-mL beakers with 175 mL of overlying water receiving 2 volume additions daily of overlying water at 23°C, with four replicate beakers tested for each treatment. Two control sediments, each having eight replicates per treatment, were used in anticipation of a broad range of grain size and TOC in the field collected samples: Florissant soil, a fine silt–clay material with about 1.2% total organic carbon (Ingersoll and Nelson 1990), and West Bearskin sediment, a sandy loam material with about 9.6% total organic carbon (Ingersoll et al. 1998, 2002). Both control sediments were used to evaluate test acceptability and to establish toxicity of the field-collected sediment samples. The source of overlying water was well water (hardness: 280 mg/L as CaCO₃; alkalinity: 250 mg/L as CaCO₃). The amphipods were fed 1.0 ml YCT/day (1,800 mg/l stock; YCT: yeast, Cerophyl[®], and Trout Chow) during the toxicity test. End points measured at the end of the exposure included survival and growth (body

length). The length of amphipods was measured using a Zeiss[®] Interactive Digital Analysis System with a Zeiss SV8 stereomicroscope at a magnification of 25× (Kemble et al. 1994). The average length of amphipods at the start of the toxicity test was 1.39 (±0.38) mm, which corresponds to the size of about 7-day-old amphipods (Ingersoll et al. 1998). On days 0 and 28 of the exposure, measurement was taken of overlying water for temperature, dissolved oxygen, conductivity, pH, alkalinity, total ammonia, and hardness. The first four parameters were also determined on days 7, 14, and 21. In general, the characteristics of the overlying water quality were similar among the samples (Tao 2005). Mean values (ranges in parentheses) of water quality were dissolved oxygen 6.6 (4.2–7.4) mg/l, conductivity 650 (634–675) μS/cm, pH 8.2 (8.0–8.3), alkalinity 245 (226–261) mg/l, total ammonia 0.3 (0.1–2.5) mg/l, un-ionized ammonia 0.002 (<0.012) mg/l, and hardness 283 (274–300) mg/l.

Pore-water quality was measured at the start of the exposures. About 100 mL pore water was isolated from each sediment sample using a refrigerated, large-capacity centrifuge (Model PR-7000, International Equipment Company) with centrifugation set at 4°C for 15 min at a speed of 5200 rpm (7,000g). About half of each pore-water sample was used to measure basic water quality parameters following Kemble et al. (1994). Mean values (ranges in parentheses) of pore-water quality were dissolved oxygen 5.8 (2.6–8.7) mg/l, conductivity 1031 (233–3280) μS/cm at 19°C (16–21), pH 8.1 (6.7–9.6), alkalinity 235 (120–480) mg/l, total ammonia 4.13 (0.41–48.8) mg/l, un-ionized ammonia 0.0239 (0.0006–0.2938) mg/l, and hardness 306 (190–740) mg/l. The remaining portion of pore-water samples were filtered through 0.45-μm Millipore paper and acidified with high-purity concentrated nitric acid to obtain a 1% nitric solution. The solution was then used to measure the trace metals following USEPA (1994): Cr, Mn, Co, Ni, Cu, Zn, Cd, and Pb using ICP-MS.

Data Analysis

Statistical analyses for the amphipod 28-day exposure tests were conducted following the guidance in ASTM (2009) using PASW (Predictive Analytics Software) Statistics 17.0 (SPSS Inc. 2009). The end points—percent survival and length—were subject to Shapiro–Wilk’s test of normality. Survival in 13 of the 31 treatments and length in 2 of the 31 treatments (including two control treatments—West Bearskin and Florissant soil) were not normally distributed. Hence, percent survival data were arcsin-square-root-transformed and length data were log-transformed and reanalyzed for normality. Transformation of the data did not improve the normality of the data distribution. The nonparametric Mann–Whitney *U*-test (Wilcoxon rank sum)

was then used to compare each sample with both control samples. Sediment samples were designated as toxic if the response of the amphipod was significantly lower than the response observed in both control sediments (Ingersoll et al. 2002; ASTM 2009).

Spearman rank correlation was used to examine the relations between toxicity and the physical or chemical characteristics of the sediments. Toxicity data were further compared to SQGs for individual contaminant or groups of contaminants. The purpose of this evaluation was to explore the potential causes for the observed toxicity. The SQGs used in this evaluation include the ESBs for metal mixtures and PAH mixtures (USEPA 2003, 2005a, 2005b), consensus-based SQGs for PAHs and pesticides (MacDonald et al. 2000), and other empirical threshold values for general sediment parameters (e.g., ammonia; Ingersoll et al. 2009).

Metal concentrations in interstitial water were evaluated using toxic units (IWTUs). The IWTUs were calculated as: $IWTU = [Md]/LC50s$, where [Md] is the dissolved metal concentration in the pore water and the LC50s is the concentration of the metal causing 50% mortality of the test species in a water-only test (USEPA 2005a). This evaluation used 10-day LC50s for *H. azteca* in a soft water (40 mg/l hardness as $CaCO_3$). Cadmium (Cd) was not included in the IWTU calculation because it was below the practical quantitation limit (estimated at 0.9 μg Cd/L) for all of the samples. Because metal toxicity can be inhibited by sediment phases capable of binding metals, such as AVS and organic carbon (Di Toro et al. 1992; Hansen et al. 1996; USEPA 2005b), values of $(\Sigma SEM-AVS)/f_{oc}$ (where ΣSEM is the sum of SEMs and f_{oc} is the fraction of organic carbon) were calculated to derive an organic carbon (OC)—normalized excess SEM that is theoretically bioavailable and can be causally linked to toxicity. A sediment sample with a $(\Sigma SEM-AVS)/f_{oc} < 130 \mu mol/g_{oc}$ was predicted to be not toxic due to metals, whereas a sample with a $(\Sigma SEM-AVS)/f_{oc} > 3,000 \mu mol/g_{oc}$ was predicted to be toxic due to metals, and the toxicity of a sample due to metals with $(\Sigma SEM-AVS)/f_{oc} \mu mole/g_{oc}$ between 130 and 3,000 was uncertain (USEPA 2005a).

The sum equilibrium sediment benchmark toxic unit for 34 PAHs ($\Sigma ESBTU_{FCV,34}$), defined as total PAHs, was calculated for different confidence levels (e.g., 50 and 95%) based on 13 of the quantified PAHs for each sediment sample (USEPA 2003). The $\Sigma ESBTU_{FCV,34}$ estimates were compared with a threshold of 1.0. The $\Sigma ESBTU_{FCV,34}$ threshold of 1.0 was established based on an estimated critical body burden of about 2 μmol total PAHs/g lipid for 28-day growth of *H. azteca* in water-only exposures and a final chronic value of about 2.2 μmol total PAHs/ g_{oc} in sediment (Dave Mount, personal communication, March 17, 2008). A sediment sample with a $\Sigma ESBTU_{FCV,34} < 1.0$

was predicted to be not toxic due to PAHs, whereas a sample with a $\Sigma ESBTU_{FCV,34} > 1.0$ was predicted to be toxic to *H. azteca* in 28-day whole-sediment exposures.

The PEC quotients for multiple chemicals were calculated following the procedure outlined in Ingersoll et al. (2001). Table 1 lists PECs of individual chemicals or compound groups considered in this study. A mean PEC-Q

Table 1 Interstitial-water and sediment quality guidelines for calculating IWTU, PEC-Q, and exceedance of PECs

Interstitial water		
Metals ($\mu g/L$)	LC50 ^a	Exceedance
Cu	35	1
Ni	780	0
Pb	16	0
Zn	73	0
Sediment		
Metals (mg/kg, dry weight)	PEC ^b	Exceedance
Cd	4.98	0
Cu	149	1
Ni	48.6	0
Pb	128	1
Zn	459	1
LMW PAHs (2- to 3-ring members, $\mu g/kg$ dry weight)		
Naphthalene	561	0
Fluorene	536	2
Phenanthrene	1,170	13
Anthracene	845	2
HMW PAHs (4- to 6-ring members, $\mu g/kg$ dry weight)		
Fluoranthene	2,230	13
Pyrene	1,520	11
Benzo[<i>a</i>]anthracene	1,050	11
Chrysene	1,290	11
Benzo[<i>a</i>]pyrene	1,450	8
Total PAHs	22,800	4
Polychlorinated biphenyls ($\mu g/kg$ dry weight)		
PCBs	676	0
Organochlorine pesticides ($\mu g/kg$ dry weight)		
Chlordane	17.6	18
Dieldrin	61.8	0
Sum DDDs	28.0	5
Sum DDEs	31.3	7
Sum DDTs	62.9	0
Total DDTs	572	0
Heptachlor epoxide	16	0
Lindane (γ -BHC)	4.99	0

^a The 10-day LC50s for *H. azteca* in soft water (hardness: 45 mg/L as $CaCO_3$ at pH 7.8–8.2) (USEPA 2000b)

^b MacDonald et al. (2000)

was calculated by averaging the PEC-Qs for up to four chemical classes (e.g., metals: Cd, Cu, Ni, Pb, and Zn; total PAHs; PCBs; or pesticides: chlordane, dieldrin, and sum DDEs; Ingersoll et al. 2001). The average PEC-Q for metals was based on SEM concentrations of Cd, Cu, Pb, Ni, and Zn rather than total metal concentrations, which might underestimate total metal concentrations by about 30–40% (Ingersoll et al. 2009). Half of the reporting limit is used as a proxy concentration for a sample that had the concentration of a chemical less than its reporting limit, which, in turn, is less than the corresponding PEC. If the reporting limit was greater than the PEC, that chemical was not included in the calculation of PEC-Qs (Ingersoll et al. 2001). This occurred in only about 3% of the 29 samples for individual LMW PAHs and less than 25% of the samples for individual HMW PAHs listed in Table 1. LMW PAHs were calculated as a sum concentration of 2- to 3-ring members, which were also included in PEC-Q calculation with the exception of acenaphthene, acenaphthylene, and 2-methylnaphthalene, because no PECs were available for these compounds (MacDonald et al. 2000). HMW PAHs were calculated as a sum of 4- to 6-ring members, which, similar to LMW PAHs, were included for calculating PEC-Qs with the exception of benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, indeno[1,2,3-*cd*]pyrene, and benzo[*g,h,i*]perylene, the compounds with no established PECs (MacDonald et al. 2000).

Results

Sediment Physical and Chemical Characteristics

The sediment samples exhibited relatively broad ranges in physical and chemical characteristics (Table 2). The percentage of sand (>62.5 μm) ranged from 0 to 70% and the percentage of clay (<4 μm) varied between 16 and 53%. Twenty-five samples contained <2% TOC; the remaining four samples ranging from 2.1 to 5.1% TOC were collected from the sites located in the lower reaches of Brush and Turkey creeks. Similarly, 23 samples contained ≤ 2 mg/kg oil and grease; the samples from the lower reaches of Brush and Turkey creeks had >2.0 mg/kg oil and grease (Table 2). Concentrations of AVS ranged from 0.33 to 44.2 $\mu\text{mol/g}$, with 25 samples containing less than 10 $\mu\text{mol/g}$ AVS. About two-thirds (20) of the samples, including all of the samples from Brush Creek, had an excess (range 0.57–24.9 $\mu\text{mol/g}$) of AVS relative to the sum of SEMs (Cd, Cu, Pb, Ni, Zn).

Total concentrations of PAHs ranged from 290 to 82,150 $\mu\text{g/kg}$, with a median of 14,990 $\mu\text{g/kg}$. Four- to six-ring HMW PAHs were predominant in the PAH composition profiles representing more than 80% of total PAH

concentration. The four-ring components, fluoranthene and pyrene, were especially abundant. Concentrations of PCB aroclors were detected in only 3 of the 29 samples, with a range of 37–320 $\mu\text{g/kg}$. Historically used organochlorine pesticides had elevated concentrations (i.e., 77–235 $\mu\text{g/kg}$ for total DDTs, 51–610 $\mu\text{g/kg}$ for chlordane, and 3–26 $\mu\text{g/kg}$ for dieldrin) at all the sites at Brush Creek (Table 2). Chlordane and dieldrin were also commonly detected in samples from Turkey and Indian creeks.

Toxicity Testing

Mean survival of amphipods in both control sediments was above 90% at the end of the 28-day sediment exposures, which met acceptability requirements outlined in USEPA (2000b) and in ASTM (2009; Table 3). Survival of amphipods was significantly reduced in 5 of the 29 samples relative to the two control sediments: site 7 on Indian Creek, site 17 on Turkey Creek, and sites 20, 22, and 26 on Brush Creek. Length of amphipods was similar in both control sediments after the 28-day exposure and was not significantly reduced in any of the treatments relative to the control sediments (Table 3).

Relationships Between Toxicity and Sediment Physical or Chemical Characteristics

Spearman rank correlation analysis indicated that there were significant correlations (ρ : -0.44 to -0.50 ; $P < 0.01$) between amphipod survival and the mean PEC-Q based on sum DDEs, chlordane, and dieldrin, and the mean PEC-Q based on metals, PAHs, PCBs, and organochlorine pesticides (Table 4). Survival of amphipods was not significantly correlated to any of the pore-water parameters, sediment grain size, oil and grease, ($\Sigma\text{SEM-AVS}$)/ f_{oc} , or parameters related to PAHs only. Length of amphipods was not significantly correlated to any of the water or sediment characteristics (Table 4).

Comparisons of Sediment Physical or Chemical Characteristics to Toxicity Thresholds

Conventional parameters of pore water and whole sediment were generally within recommended ranges or established toxicity thresholds (Ingersoll et al. 1996, 2002, 2009; USEPA 2000b; ASTM 2009). For example, concentrations of dissolved oxygen in pore water at the start of the exposures were all above the recommended minimum concentration of 2.5 mg/l in overlying water (ASTM 2009). Sediment from site 26 on Brush Creek contained the highest unionized ammonia in pore water of 0.294 mg/l (at pH = 9.6), below a toxicity threshold of 1.5 mg/l for *H. azteca* in sediment (Ingersoll et al. 2009). Grain size and

Table 2 Physical and chemical characteristics of sediment samples from streams in Metropolitan Kansas City area

Stream	Site	Grain size (% clay)	TOC (%)	Oil and grease (mg/kg)	SEM-AVS ($\mu\text{mol/g}$)	LMW PAHs ($\mu\text{g/kg}$)	HMW PAHs ($\mu\text{g/kg}$)	Total PAHs ($\mu\text{g/kg}$)	Total aroclor ($\mu\text{g/kg}$)	Technical chlordane ($\mu\text{g/kg}$)	Total DDTs ($\mu\text{g/kg}$)	Dieldrin ($\mu\text{g/kg}$)
Indian creek	1	44.2	0.6	0.8	2.5	550	4,890	5,440	69	39	0	0
	2	ND	0.9	1.6	0.1	3,910	33,000	36,910	0	46	4	3
	3	34.7	0.8	0.2	2.4	0	0	0	0	0	0	0
	4	33.3	1.2	1.4	-0.6	8,200	36,000	44,200	0	46	0	2
	5	33.5	1.3	1.2	-3.1	2,300	21,950	24,250	0	36	0	0
Tomahawk creek	6	30.0	0.7	0.2	-1.9	0	0	0	0	0	0	0
	7	50.5	1.3	1.0	-4.6	300	1,880	2,180	0	0	0	0
	8	33.7	0.9	0.4	-1.7	0	0	0	0	0	0	0
	9	36.0	0.8	0.6	-2.7	0	310	310	0	0	0	0
	10	33.3	1.2	0.4	-5.2	0	0	0	0	0	0	0
	11	52.6	0.9	0.4	2.4	0	0	0	0	0	0	0
	12	39.5	0.7	1.8	4.3	1,840	13,150	14,990	0	28	0	0
	13	33.1	0.9	1.4	-5.3	1,200	11,780	12,980	0	57	0	2
	14	42.0	0.9	1.9	-8.2	3,010	24,000	27,010	0	52	0	0
	15	40.5	1.4	4.1	0.3	810	5,160	5,970	0	31	0	2
	16	30.7	0.9	0.3	1.3	0	0	0	0	0	0	0
Brush creek	17	36.7	1.9	2.7	0.9	2,700	21,200	23,900	0	250	0	21
	18	24.0	2.1	2.4	-3.5	2,450	16,960	19,410	0	55	0	11
	19	46.8	3.7	12.2	-23.8	1,200	6,140	7,340	320	580	235	26
	20	39.7	5.1	3.9	-13.1	1,500	19,090	20,590	0	490	179	21
	21	36.8	2.1	5.4	-24.9	0	13,210	13,210	0	610	168	19
	22	20.6	0.2	0.9	-1.2	1,000	7,770	8,770	0	140	82	7
	23	25.7	0.3	1.6	-3.9	2,690	17,740	20,430	0	240	114	8
Cedar creek	24	17.5	1.1	1.0	-1.0	4,700	27,000	31,700	0	240	81	10
	25	30.4	0.4	2.0	-2.8	650	5,710	6,360	0	51	77	3
	26	16.1	0.5	1.5	-2.1	12,650	69,500	82,150	0	250	77	9
	27	43.3	1.0	0.7	-4.8	0	0	0	0	0	0	0
	28	27.4	0.6	0.2	0.4	0	290	290	0	0	0	0
	29	33.5	1.0	0.2	-3.0	0	0	0	0	0	0	0

Note: Concentrations below detection limits were treated as zero value; ND: not determined due to either insufficient sample volume or difficulty in making the measurement. LMW PAHs measured in this study: naphthalene, 2-methylnaphthalene, acenaphthene, acenaphthylene, fluorene, phenanthrene, and anthracene. HMW PAHs measured in this study: fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[k]fluoranthene, benzo[e]pyrene, indeno[1,2,3-*cd*]pyrene, and benzo[ghi]perylene. PCBs: sum of the three aroclors (1248, 1254, and 1260). Total DDTs (ΣDDTs): a total of sum DDTs (*o,p'*- and *p,p'*-DDT), sum DDEs (*o,p'*- and *p,p'*-DDE), and sum DDDs (*o,p'*- and *p,p'*-DDD)

Table 3 Response of *H. azteca* in 28-day exposures to sediment samples from streams in Metropolitan Kansas City

	Site	<i>H. azteca</i>	
		Survival (%)	Length (mm)
	Control-WB	94 (1.8)	3.63 (0.06)
	Control-FL	91 (2.3)	3.44 (0.07)
Indian creek	1	98 (4.8)	4.36 (0.09)
	2	93 (3.3)	4.83 (0.12)
	3	95 (5.0)	3.60 (0.18)
	4	95 (5.0)	3.51 (0.06)
	5	90 (5.8)	3.84 (0.03)
Tomahawk creek	6	98 (2.5)	3.86 (0.16)
	7	78 (6.3)*	3.97 (0.08)
	8	93 (4.8)	4.00 (0.09)
	9	93 (2.5)	4.30 (0.13)
	10	98 (2.5)	4.67 (0.08)
Turkey creek	11	80 (12)	3.80 (0.12)
	12	90 (0.0)	4.10 (0.10)
	13	90 (7.1)	4.12 (0.03)
	14	95 (5.0)	3.96 (0.06)
	15	88 (7.5)	4.38 (0.06)
	16	93 (4.8)	4.23 (0.20)
	17	0 (0.0)*	ND
	18	83 (8.5)	3.66 (0.15)
Brush creek	19	100 (0)	4.41 (0.11)
	20	55 (5.0)*	3.42 (0.11)
	21	88 (2.5)	3.88 (0.15)
	22	5 (2.9)*	ND
	23	88 (2.5)	3.76 (0.09)
Cedar creek	24	88 (6.3)	4.10 (0.10)
	25	85 (6.5)	3.81 (0.08)
	26	75 (2.9)*	4.50 (0.13)
	27	98 (2.5)	3.47 (0.05)
	28	100 (0.0)	3.72 (0.20)
	29	95 (5.0)	4.03 (0.16)

ND not determined due to survival <40%. For the *H. azteca* exposures, means ($n = 4$; SE in parenthesis) with an asterisk within a column and within a set of exposures are significantly lower than both of the control treatments (West Bearskin sediment and Florissant soil) at $\alpha = 0.05$

TOC of the sediments (Table 2) were also within the tolerance ranges for *H. azteca* (Ingersoll et al. 1996, 2002, 2009; USEPA 2000b; ASTM 2009).

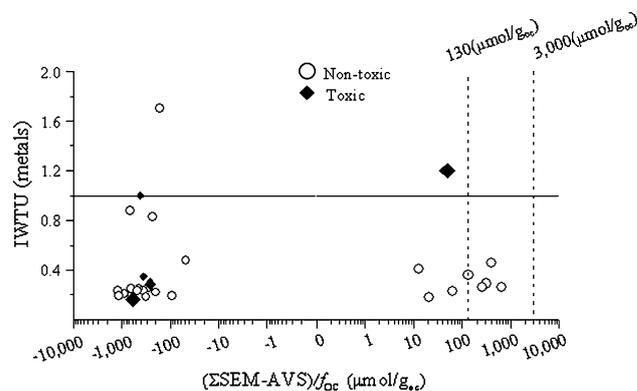
Survival of amphipods was reduced at higher concentrations of oil and grease in the sediment; however, oil and grease is a broadly based measurement of sediment chemistry that reflects natural and anthropogenic hydrocarbons and fat-soluble materials such as DDT and PCBs (DiSalvo et al. 1977). Hence, it is unclear which individual contaminants or specific chemical group might have

Table 4 Spearman rank correlation between toxicity data and the physical or chemistry data

Variable		Survival ($n = 29$)	Length ($n = 26$)
Pore water	Dissolved oxygen	0.25	-0.12
	Total ammonia	-0.26	0.04
	Un-ionized ammonia	-0.21	-0.02
	Toxic units of metals	-0.20	0.04
Sediment	% Sand	-0.16 ^a	0.17
	% Clay	0.13	-0.14
	Oil & grease	-0.45	0.01
	(Σ SEM-AVS)/ f_{oc}	0.07	0.17
	No. of PAH detected	-0.44	0.07
	LMW PAHs	-0.44	0.13
	HMW PAHs	-0.33	0.23
	Total PAHs	-0.33	0.25
	Total PAHs/ g_{oc}	-0.37	0.30
	No. of pesticide-related compounds detected	-0.40	0.10
	PEC quotient (PAHs)	-0.44	0.05
	Mean PEC quotient (sum DDEs, chlordane, and dieldrin)	-0.50*	0.06
Mean PEC quotient (Σ SEMs, PCBs, PAHs, sum DDEs, chlordane, and dieldrin)	-0.47*	-0.04	

^a $n = 28$

* Correlation is significant at the 0.01 level (two-tailed)

**Fig. 1** Toxicity versus interstitial water toxic units of metals (Cu, Zn, Ni, Pb) or (Σ SEM-AVS)/ f_{oc} . The size of the diamonds is proportional to the degree of mortality to *H. azteca*

contributed to the correlation between survival and oil and grease, and no effect-based sediment threshold has been established for oil and grease in sediment.

Thresholds were rarely exceeded for individual metals either in pore water or in whole sediment (Table 1). In addition, 24 samples had a (Σ SEM-AVS)/ $f_{oc} < 130 \mu\text{mol}/g_{oc}$, a threshold below which metal toxicity is not predicted (Fig. 1; USEPA 2005b). Five of these samples with

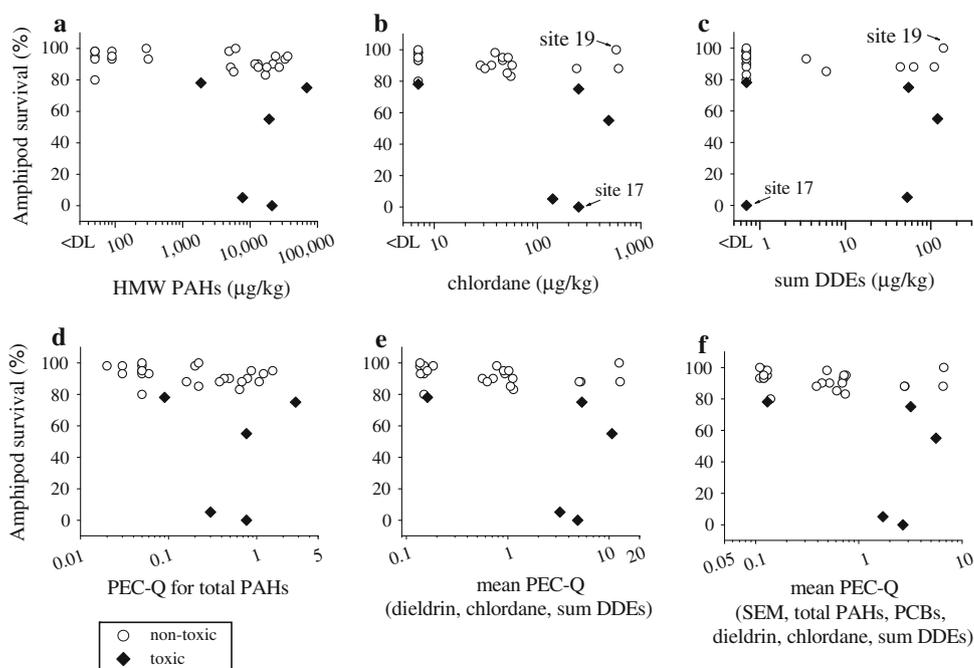
$(\Sigma\text{SEM-AVS})/f_{oc} < 130 \mu\text{mol/g}_{oc}$ were toxic to amphipods, which indicates that metals were not likely contributing to the toxicity. Five of the 29 samples had a $(\Sigma\text{SEM-AVS})/f_{oc}$ value between 130 and 3,000 $\mu\text{mol/g}_{oc}$, a range in which toxic effects due to metals are uncertain (USEPA 2005b); none of these samples were toxic to amphipods (Fig. 1).

Further comparison between the $\Sigma\text{SEM-AVS}$ and interstitial water toxic units (IWTUs) for metals indicated that 9 of the 29 samples contained a slight excess of ΣSEM s relative to AVS (0.11–4.26 $\mu\text{mol/g}$, Table 2; the dots representing these samples are located on the right side of Fig. 1), with one of these samples having an IWTU for metals over 1.0 (Fig. 1). All of the 20 samples that had an excess of molar AVS content relative to ΣSEM s level had one or more metals (Cu, Ni, Zn, and Pb) measurable in pore water (Table 2 and Fig. 1; dots representing these samples are located on the left side of Fig. 1). Four of those samples had an IWTU for metals greater than 0.5, with two samples greater than 1.0 (Fig. 1). Among the five toxic samples, three of these samples had an IWTU < 0.5 ; two had an IWTU slightly over 1.0, which supports the conclusion that the toxicity observed in these five sediment samples was not likely due to metals.

Although there were frequent exceedances of PECs for individual PAHs (especially HMW PAHs), chlordane, and DDT degradates (DDE and DDD; Table 1), most of these individual exceedances were not associated with toxic samples. A comparison of concentrations of organic contaminants in sediment to the toxicity observed to amphipods exhibited several trends. First, toxicity was observed across a wide range of concentrations of individual

compounds or mixtures of compounds. For instance, the samples identified as toxic had concentrations of HMW PAHs ranging from less than 2,000 to nearly 70,000 $\mu\text{g/kg}$ (Fig. 2a). Second, samples with lower survival did not always coincide with samples with the highest concentrations of some individual compounds, or toxicity did not always correspond to samples with the highest PEC-Q for total PAHs (Fig. 2d), mean PEC-Q for organochlorine pesticides (Fig. 2e), or mean PEC-Q based on all chemical classes analyzed (Fig. 2f). For example, site 17 had 100% mortality, but its concentrations of organic contaminants were relatively low in comparison with those of other samples, whereas site 19 had 100% survival and concentrations of chlordane and sum DDEs were among the highest measured in this study (Fig. 2b, c). Third, the frequency of toxicity increased with increasing concentrations of individual organochlorine pesticides (Fig. 2b or c), with increasing mean PEC-Q for organochlorine pesticides (Fig. 2e), or with increasing mean PEC-Qs based on all chemical classes analyzed (Fig. 2f). Only 1 of the 17 samples with the mean PEC-Q for organochlorine pesticides < 1.0 was toxic; 4 of the 12 samples with a mean PEC-Q for organochlorine pesticides > 1.0 and 4 of the 8 samples with a mean PEC-Q for organochlorine pesticides > 3.0 were toxic (Fig. 2e). When all the chemical classes (i.e., metals, PAHs, PCBs, dieldrin, chlordane, sum DDEs) were included into a mean PEC-Q calculation, only 1 of the 21 samples (5%) with the PEC-Q < 1.0 was toxic, and 4 of the 8 samples (50%) were toxic when the PEC-Q was > 1.0 (Fig. 2f). This trend was less evident for HMW PAHs (Fig. 2a) and for the PEC-Q for total PAHs (Fig. 2d).

Fig. 2 The relation between the selected chemical parameters (concentrations of HMW PAHs, chlordane, and DDE; PEC-Q for PAHs; mean PEC-Qs for organochlorine pesticides, and mean PEC-Qs for various chemical groups) and the survival of *H. azteca* in the 28-day whole-sediment toxicity test. In the upper graphs, samples with chemical concentrations lower than the reporting limits are displayed at $< \text{DL}$ (detection limit) along the x-axis



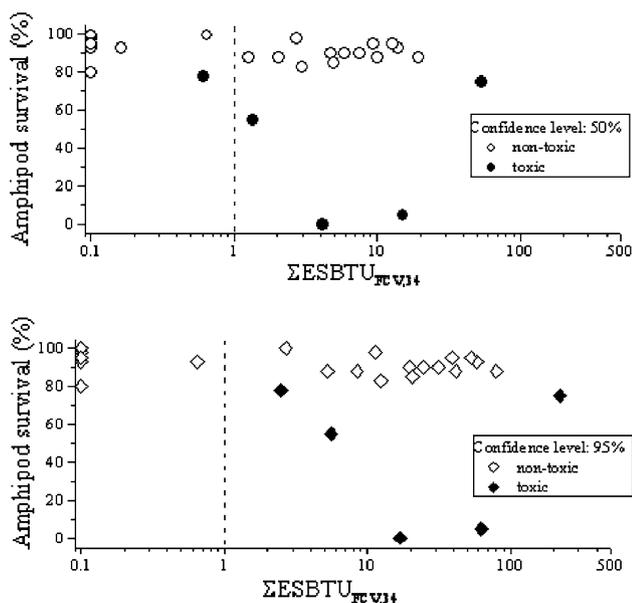


Fig. 3 Survival of *H. azteca* versus $\Sigma\text{ESBTU}_{\text{FCV}}$ of 34 PAHs at the confidence levels of 50% and 95%

The relationship between amphipod survival and ESB for PAH mixtures is illustrated in Fig. 3. At the confidence level of 50%, 17 samples contained more than 1.0 $\Sigma\text{ESBTU}_{\text{FCV}}$ of the mixture of the 34 PAHs, a level above which effects would be predicted on sensitive benthic organisms such as *H. azteca* (USEPA 2003). However, only 4 of these 17 samples were toxic to amphipods. At the confidence level of 95%, 19 samples contained more than 1.0 $\Sigma\text{ESBTU}_{\text{FCV},34}$ of PAHs, and only 5 of these 19 samples were toxic to amphipods.

Discussion

Sediments in the study streams were subject to the influences of a limited number of major point sources (POTWs and a CSS) and diffuse and diverse nonpoint sources from a combination of predominant residential land uses with commercial and light industrial land uses (Tao et al. 2010), and they exhibited low to moderate toxicity and contamination based on mean PEC-Q for metals, total PAHs, PCBs, and sum DDEs (Fig. 4). In comparison, sediments influenced by industrial sources at other locations across North America had a wider range of contamination based on the same groups of chemicals and exhibited a relatively higher incidence of toxicity (Fig. 4; Ingersoll et al. 2001; MacDonald et al. 2002). Additionally, note that toxicity testing in the present study was conducted with *H. azteca*, instead of multiple test organisms (e.g., midge or oligochaetes), which could have revealed more information on potential biological effects (Ingersoll et al. 2001, 2002).

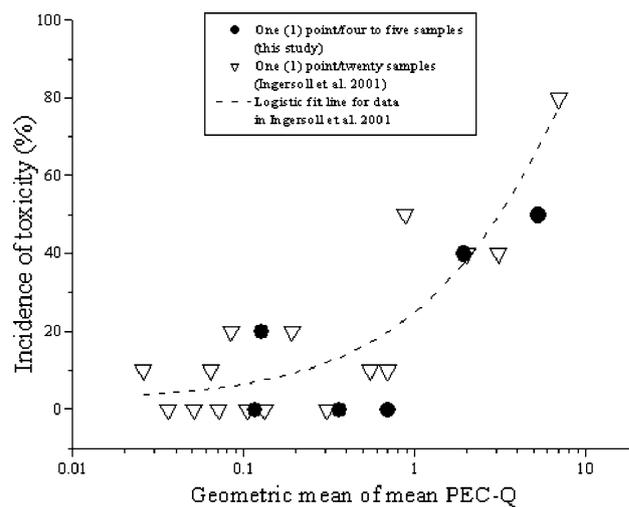


Fig. 4 Comparison of toxicity observed in the present study to other *H. azteca* 28-day sediment data compiled by Ingersoll et al. (2001). Mean quotients were calculated based on metals, total PAHs, PCBs, and sum DDEs

The observed toxicity was not likely caused by conventional parameters (e.g., dissolved oxygen, ammonia) or physical characteristics of the sediments (grain size or TOC), because *H. azteca* reportedly tolerates a wide range of physical characteristics of sediments (ASTM 2009). Similarly, concentrations of metals in the pore-water or in whole-sediment samples were below concentrations expected to cause toxicity. This conclusion is consistent with the lack of industrial point sources discharging into the streams and with the local, predominantly residential urban landscape.

Historical-use organochlorine pesticides, rather than PAHs, might have contributed to the toxicity observed in this study. The frequency of toxicity generally increased with the increase in the concentrations of organochlorine compounds or with the increase in mean PEC-Qs for organochlorine chemicals (Fig. 2b, c, e) but not with the increase in concentrations of PAHs (Fig. 2a, d). A similar pattern was also noted for the combined mean PEC-Q based on metals, PAHs, PCBs, and organochlorine pesticides (Fig. 2f), which appears to be driven by the PEC-Qs for the organochlorine pesticides that can be several orders of magnitude higher than the PEC-Qs for the other chemicals. Nevertheless, it is uncertain whether organochlorine pesticides were the sole cause of the observed toxicity. Toxicity identification evaluation (TIE) procedures (USEPA 2007) would be useful in helping to identify groups of compounds potentially contributing to the observed sediment toxicity.

Although the frequency of toxicity tended to increase with increasing concentrations of organochlorine pesticides, there were samples with elevated concentrations of organic contaminants (including PAHs) that were not always toxic (Fig. 2). The cause of this discrepancy might

be related to the short list of analytes that might have omitted contaminants that have been the primary cause for the toxicity. For example, none of the PCB congeners was measured; chlordane contains more than 140 components, 3 of which were quantified in this study. Similarly, pyrethroid insecticides, a group of insecticides widely used in urban environments across the United States (Amweg et al. 2006), were not evaluated in the present study. Other possible causes might be associated with the bioavailability of the contaminants in the sediment. Coal-tar products and traffic-related particles were likely primary sources of the PAHs in the stream sediments in the present study (Tao et al. 2010). The bioaccumulation of PAHs can be attenuated by soot (black carbon) due to physical occlusion within the solid phase (Sundelin et al. 2004). As a result, PAHs in a soot matrix of traffic-related origins present a reduced potential for toxicity (Jonker and Koelmans 2002). This might partially explain the low toxicity observed in some of our samples with the PEC-Q for total PAHs greater than 1.0. In comparison, a higher incidence of toxicity was observed by Ingersoll et al. (2001) for the sediments containing total PAHs at a concentration range comparable to that in this study. The sediments evaluated by Ingersoll et al. (2001) were collected from sites affected by industrial sources and had a similar proportion of LMW to HMW PAHs (i.e., similar proportion of petrogenic to pyrogenic PAHs), whereas the ratio of LMW to HMW PAHs in the current study was typically less than 0.25 (Tao et al. 2010). Analyses are ongoing with the data summarized by Ingersoll et al. (2001) and with additional data to determine if sediments with a higher proportion of petrogenic to pyrogenic PAHs are more toxic than sediment with a lower proportion of petrogenic to pyrogenic PAHs (Ingersoll, unpublished data).

There might have been other confounding factors that influenced the toxicity of organic contaminants in the sediments or potentially contributed to the lack of toxicity of some samples with high PEC-Qs (Fig. 2f). These factors could include the chiral molecular configuration of organochlorine compounds (e.g., dieldrin, *o,p'*-DDT), the enantiomers of which might exhibit different enantiomer-selective toxicity (Lewis et al. 1999; Garrison 2006), or the sequestration or aging of some persistent organic contaminants (e.g., DDT and chlordane), a process that could cause these compounds to show a marked decline in their mortality to certain exposed species over time (Robertson and Alexander 1998). Such chiral properties of some compounds and their property alteration in the postdeposition environment have not been addressed in the current available SQGs, which treat individual organic compounds as a single substance, an approach that could compromise an estimate of the toxicity due to organic contaminants, such as dieldrin and chlordane residues.

Length of amphipods did not identify any samples as toxic in the present study. Ingersoll et al. (2001) reported growth of amphipods was frequently a sensitive indicator of toxicity of sediment in 28-day exposure with *H. azteca*. It is unclear whether any antagonistic effect between sediment contaminants or any stimulatory effect resulting from exposure to detected or unmeasured contaminants influenced the growth of amphipods in this study. Further analyses are underway with the data summarized in Ingersoll et al. (2001) and with additional data to evaluate use of total biomass of *H. azteca* as an end point to integrate effects on both survival and growth of amphipods.

A sediment sample with an $\Sigma\text{ESBTU}_{\text{FCV},34} > 1.0$ was predicted to be toxic to *H. azteca* due to PAHs. Nevertheless, only a small fraction of the samples (4 of the 17 samples at a confidence level of 50% and 5 of the 19 samples at the confidence level of 95%) with an estimated $\Sigma\text{ESBTU}_{\text{FCV},34} > 1.0$ were toxic to amphipods (Fig. 3). The sediments contained PAHs that were likely associated with traffic or coal-tar applications (Tao et al. 2010). These PAHs might partition less to interstitial water in sediments containing soot or coal particles than expected with typical organic carbon—the partitioning phase based on which the $\Sigma\text{ESBTU}_{\text{FCV},34}$ procedure was developed (USEPA 2003). As a result, the ESB approach for PAH mixtures might not be effective for evaluating these types of contaminated sediments. Additionally, only 13 PAHs were evaluated in the present study, whereas quantification of a minimum of 34 specified PAHs is recommended in calculating ΣESBTUs for total PAHs (USEPA 2003). Incorporating uncertainty factors in the calculation did not improve the relationships between sediment toxicity and PAHs. Hence, the incomplete characterization of PAHs in the present study likely compromised the effectiveness of the ΣESBTUs approach for evaluating the bioavailability of PAHs in sediment.

Conclusions

Stream sediments subject to the influence of residential nonpoint sources in the metropolitan Kansas City area exhibited low to moderate levels of contamination and toxicity, with 5 of the 29 samples identified as toxic to *H. azteca*. Neither pore-water quality, nor physical characteristics of the sediment (e.g., dissolved oxygen, ammonia, grain size, TOC), nor metals likely caused the toxicity. Use of PEC-Qs helped to identify historical-use organochlorine pesticides rather than PAHs, to be associated with the observed toxicity. Use of mean PEC-Qs incorporating different classes of chemicals was more informative in predicting *H. azteca* toxicity compared to exceedances of individual PECs. Different origins of PAHs (e.g.,

petrogenic vs. pyrogenic PAHs in a soot matrix) or different enantiomeric composition for chiral compounds might have contributed to the uncertainty in relationships between sediment chemistry and toxicity. Similarly, the ESB approach should be applied cautiously when the Σ ESBTU for total PAHs is estimated from a limited number of PAH analytes. In the present study, toxicity was evaluated with a single organism, *H. azteca*, which might not be as sensitive to some chemicals as other test organisms. Future studies of stream sediment in the study area should expand the list of analytes, including pyrethroid insecticides, employ multiple test organisms to evaluate toxicity, and apply TIE procedures to help better identify groups of contaminants that might be causing the observed toxicity.

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References

- Allen HE, Fu G, Boothman WD, DiToro M, Mahony JD (1991) Determination of acid volatile sulfide and selected simultaneously extractable metals in sediment. Draft method, EPA-821-R-91-100. US Environmental Protection Agency, Washington, DC
- Amweg EL, Weston DP, You J, Lydy MJ (2006) Pyrethroid insecticides and sediment toxicity in urban creeks from California and Tennessee. *Environ Sci Technol* 40:1700–1706
- ASTM (American Society for Testing and Materials) (2009) Standard test methods for measuring the toxicity of sediment-associated contaminants with freshwater invertebrates. In: Annual book of ASTM standards, E1706-05, vol 11.06. ASTM, West Conshohocken, PA, pp 947–1063
- Blaise C, Wells PG, Lee K (1998) Microscale testing in aquatic toxicity: introduction, historical perspective, and context. In: Wells PG, Lee K, Blaise C (eds) *Microscale testing in aquatic toxicology*. CRC Press, Boca Raton, FL, pp 1–12
- Di Toro DM, Mahony JD, Hansen DJ, Scott KJ, Carlson AR, Ankley GT (1992) Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. *Environ Sci Technol* 26:96–101
- DiSalvo LH, Guard HE, Hirsch ND, Ng J (1977) Assessment and significance of sediment-associated oil and grease in aquatic environments. US Army Corps of Engineers, Washington, DC
- Garrison AW (2006) Probing the enantioselectivity of chiral pesticides. *Environ Sci Technol* 40:16–23
- Hansen DJ, Berry WJ, Mahony JD, Boothman WS, DiToro DM, Robson DL, Ankley GT, Ma D, Yan Q, Pesch DE (1996) Predicting the toxicity of metal-contaminated field sediments using interstitial concentrations of metals and acid-volatile sulfide normalizations. *Environ Toxicol Chem* 15:2080–2094
- Ingersoll CG, Nelson MK (1990) Testing sediment toxicity with *Hyalella azteca* (Amphipoda) and *Chironomus riparius* (Diptera). In: Landis WG, van der Schalie WH (eds) *Aquatic toxicology and risk assessment*, vol 13, STP 1096. American Society for Testing and Materials, Philadelphia, PA, pp 93–109
- Ingersoll CG, Haverland PS, Brunson EL, Canfield TJ, Dwyer FJ, Henke CE, Kemble NE, Mount DR, Fox RG (1996) Calculation and evaluation of sediment effect concentrations for the amphipod *Hyalella azteca* and the Midge *Chironomus riparius*. *J Great Lakes Res* 22:602–623
- Ingersoll CG, Brunson EL, Dwyer FJ, Hardesty DK, Kemble NE (1998) Use of sublethal endpoints in sediment toxicity tests with the amphipod *Hyalella azteca*. *Environ Toxicol Chem* 17:1508–1523
- Ingersoll CG, MacDonald DD, Wang N, Crane JL, Field LJ, Haverland PS, Kemble NE, Lindskoog RA, Severn CG, Smorong DE (2001) Predictions of sediment toxicity using consensus-based freshwater sediment quality guidelines. *Arch Environ Contam Toxicol* 41:8–21
- Ingersoll CG, MacDonald DD, Brumbaugh WG, Johnson BT, Kemble NE, Kunz JL, May TW, Wang N, Smith JR, Sparks DW, Ireland DS (2002) Toxicity assessment of sediments from the Grand Calumet River and Indiana Harbor Canal in Northwestern Indiana, USA. *Arch Environ Contam Toxicol* 43:156–167
- Ingersoll CG, Kemble NE, Kunz JL, Brumbaugh WG, MacDonald DD, Smorong D (2009) Toxicity of sediment cores collected from the Ashtabula River in northeastern Ohio USA to the amphipod *Hyalella azteca*. *Arch Environ Contam Toxicol* 57:315–329 (Erratum: *Arch Environ Contam Toxicol* 57:826–827)
- Jonker MTO, Koelmans AA (2002) Sorption of polycyclic aromatic hydrocarbons and polychlorinated biphenyls to soot and soot-like materials in the aqueous environment: mechanistic considerations. *Environ Sci Technol* 36:3725–3734
- Kemble NE, Besser JM, Brumbaugh WG, Brunson EL, Dwyer FJ, Ingersoll CG, Monda DP, Woodward DF (1994) Toxicity of metal-contaminated sediments from the Upper Clark Fork River, MT to aquatic invertebrates in laboratory exposures. *Environ Toxicol Chem* 13:1985–1997
- Lee CJ, Mau DP, Rasmussen TJ (2005) Effects of nonpoint and selected point contaminant sources on stream water quality and relation to land use in Johnson County, Northeastern Kansas, October 2002 through June 2004. US Geological Survey Scientific Investigations Report 2005-5144
- Lewis DL, Garrison AW, Wommack KE, Whittemore A, Steudier P, Melillo J (1999) Influence of environmental changes on degradation of chiral pollutants in soils. *Nature* 401:898–901
- Long ER, MacDonald DD, Smith SL, Calder FD (1995) Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environ Manage* 19:81–97
- MacDonald DD, Ingersoll CG, Berger T (2000) Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Arch Environ Contam Toxicol* 39:20–31
- MacDonald DD, Ingersoll CG, Smorong DE, Lindskoog RA, Sparks DW, Smith JR, Simaon TP (2002) An assessment of injury to sediments and sediment-dwelling organisms in the Grand Calumet River and Indiana Harbour area of concern, USA. *Arch Environ Contam Toxicol* 43:141–155
- Robertson BK, Alexander M (1998) Sequestration of DDT and dieldrin in soil: disappearance of acute toxicity but not the compounds. *Environ Toxicol Chem* 17:1034–1038

- SPSS Inc (2009) PASW (predictive analytics software) 17.0. SPSS Inc., Chicago
- Sundelin B, Wiklund AKE, Lithner G, Gustafsson Ö (2004) Evaluation of the role of black carbon in attenuating bioaccumulation of polycyclic aromatic hydrocarbons from field-contaminated sediments. *Environ Toxicol Chem* 23:2611–2617
- Tao J (2005) Identification of landscape and socioeconomic factors contributing to sediment contamination in urban streams and assessment of potential ecotoxicological effects: a case study in Metropolitan Kansas City area. PhD dissertation, University of Missouri-Kansas City, MO
- Tao J, Huggins D, Welker G, Dias JR, Ingersoll CG, Murowchick JB (2010) Sediment contamination for residential streams in the metropolitan Kansas City Area, USA: Part I. Distribution of polycyclic aromatic hydrocarbon and pesticide-related compounds. *Arch Environ Contam Toxicol*. doi:10.1007/s00244-010-9497-2
- USEPA (United States Environmental Protection Agency) (1994) Determination of trace elements in waters and wastes by inductively coupled plasma-mass spectrometry. Method 200.8, Rev. 5.4. Environmental Monitoring Systems Laboratory, EPA Office of Research and Development, Cincinnati, OH
- USEPA (2000a) Methods of the derivation of site-specific equilibrium partitioning sediment guidelines (ESGs) for the protection of benthic organisms: nonionic organics. EPA/822/R-00/002. Office of Research and Development, Washington, DC
- USEPA (2000b) Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates, 2nd edn. EPA/600/R-99/064. US EPA, Washington, DC
- USEPA (2003) Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: PAH mixtures. EPA/600/R-02/013. Office of Research and Development, Washington, DC
- USEPA (2005a) Predicting toxicity to amphipods from sediment chemistry. EPA/600/R-04/030. Office of Research and Development, Washington, DC
- USEPA (2005b) Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Metal mixtures (cadmium, copper, lead, nickel, silver and zinc). EPA/600/R-02/011. Office of Research and Development, Washington, DC
- USEPA (2007) Sediment toxicity identification evaluation (TIE): phases I, II, and III Guidance Document. EPA/600/R-07/080. US EPA, Washington, DC
- USEPA (2008) Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: Compendium of Tier 2 values for nonionic organics. EPA/600-R-02–016. Office of Research and Development, Washington, DC
- Wenning RJ, Batley G, Ingersoll CG, Moore DW (eds) (2005) Use of sediment quality guidelines and related tools for the assessment of contaminated sediments. SETAC Press, Pensacola, FL