

Using a Sediment Quality Triad Approach to Evaluate Benthic Toxicity in the Lower Hackensack River, New Jersey

Mary T. Sorensen,¹ Jason M. Conder,² Phyllis C. Fuchsman,³ Linda B. Martello,⁴ Richard J. Wenning⁴

¹ ENVIRON International Corporation, 1600 Parkwood Circle, Suite 310, Atlanta, Georgia 30339, USA

² ENVIRON International Corporation, 2010 Main Street, Suite 900, Irvine, California 92614, USA

³ ENVIRON International Corporation, 13801 West Center Street, Suite 1, Burton, Ohio 44021, USA

⁴ ENVIRON International Corporation, 6001 Shellmound Street, Suite 700, Emeryville, California 94608, USA

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Abstract. A Sediment Quality Triad (SQT) study consisting of chemical characterization in sediment, sediment toxicity and bioaccumulation testing, and benthic community assessments was performed in the Lower Hackensack River, New Jersey. Chemistry data in sediment and porewater were evaluated based on the equilibrium partitioning approach and other published information to investigate the potential for chemical effects on benthic organisms and communities. Relationships were supported by laboratory toxicity and bioaccumulation experiments to characterize chemical effects and bioavailability. Benthic community results were evaluated using a regional, multimetric benthic index of biotic integrity and four heterogeneity indices. Evidence of slight benthic community impairment was observed in five of nine sediment sample stations. Severe lethal toxicity to amphipods (*Leptocheirus plumulosus*) occurred in four of these five stations. Although elevated total chromium concentrations in sediment (as high as 1900 mg/kg) were the rationale for conducting the investigation, toxicity was strongly associated with concentrations of polycyclic aromatic hydrocarbons (PAHs) rather than total chromium. PAH toxic units (Σ PAH TU) in sediment and Σ PAH concentrations in laboratory organisms from the bioaccumulation experiment showed a clear dose–response relationship with toxicity, with 0% survival observed in sediments in which Σ PAH TU > 1–2 and Σ PAH concentrations in *Macoma nasuta* were >2 μ mol/g, lipid weight. Metals detected in sediment and porewater, with the possible exception of copper, did not correlate with either toxicity or levels in tissue, likely because acid-volatile sulfide levels exceeded concentrations of simultaneous extracted metals at all sample locations. The study reinforces the value of using multiple lines of evidence approaches such as the SQT and the importance of augmenting chemical and biological analyses with modeling and/or other

approaches to evaluate chemical bioavailability and toxicity of sediments.

The Sediment Quality Triad (SQT) approach uses multiple lines of evidence based on the results of benthic community assessments, sediment toxicity, and sediment chemistry to evaluate the relationship between sediment-associated chemicals and biological community quality (Long and Chapman 1985; Chapman 1990; Chapman 1996; Borgmann *et al.* 2001; Hall *et al.* 2005). The combination of potential cause (chemistry) and effect (toxicology and ecology) measurements makes the SQT one of the most effective tools available to establish the extent and significance of pollution-induced degradation.

The Hackensack River, New Jersey is one of two large tributaries that flow into the northern portion of Newark Bay, which is commonly included as part of the larger New York/New Jersey Harbor Estuary. Sediments along the eastern shore of the river near the confluence with Newark Bay are known to contain chromium, which is attributable, in part, to a 0.14-km² former waterfront commercial property that was used for disposal of approximately 800,000 m³ chromate ore processing residue (COPR) from 1905 to 1954. The property is located on Route 440 in Jersey City, New Jersey and designated as “Study Area 7” in the New Jersey Department of Environmental Protection (NJDEP) Hudson County Chromate Project. Sediment sampling conducted along the eastern shore of Droyer’s Point Reach revealed the presence of elevated concentrations of total chromium (as high as 9190 mg/kg in surficial sediments), with approximately one fourth of the sediment samples containing total chromium above the effects range median sediment benchmark value of 370 mg/kg (Long *et al.* 1995).

In addition to total chromium, sediment sampling in Droyer’s Point Reach and adjacent coves along the eastern shore revealed the presence of a wide variety of other metals and organic chemicals. This is not unexpected because combined sewer outfalls (CSOs) and permitted industrial discharges

continue to be a source of chemicals to the entire Newark Bay estuary and are recognized in the literature as significant ongoing sources of chemical contamination (Crawford *et al.* 1995; Shear *et al.* 1996; Huntley *et al.* 1997; Iannuzzi *et al.* 1997; Adams *et al.* 1998). The estuary is surrounded by one of the most heavily urbanized and industrial areas on the eastern U.S. coast, and supports the third largest shipping port in North America. According to Crawford *et al.* (1995), CSOs and storm water runoff may contribute as much as 40% of the total annual metals load to Newark Bay. Connell (1982) estimated that storm water runoff contributes almost 40% of the annual load of total PAHs to the Hudson-Raritan Estuary. The history of urbanization and industrialization of the Passaic River and Newark Bay by Iannuzzi *et al.* (2002) provides considerable evidence of the long history of significant biological, chemical, and physical impacts due to anthropogenic activities for more than two centuries.

This paper presents the results of an SQT analysis conducted using sediments collected offshore in the vicinity of the waterfront property used for COPR disposal located along the eastern shore of Droyers Point Reach in the Lower Hackensack River. The purpose of the investigation was to characterize causal relationships between chemical stressors and local benthic communities. Multiple lines of evidence were evaluated in the SQT framework using co-located biological community surveys, laboratory sediment toxicity tests, and chemical information from analysis of surface sediments. The traditional SQT framework was augmented with lines of evidence that account for chemical bioavailability (Borgmann *et al.* 2001), including chemical analysis of benthic organisms exposed during laboratory bioaccumulation experiments and equilibrium partitioning (EqP) analyses of sediment chemistry data to characterize bioavailability of sediment-associated chemicals. Results also were compared to those reported by Becker *et al.* (2006), who investigated sediment toxicity at a similar COPR-affected site located approximately 7 km upstream of Study Area 7 in the Hackensack River. The information gleaned from this analysis represents an important line of evidence contributing to the evaluation of remediation strategies for sediments in the vicinity of the Study Area 7 site.

Methods

Field Collection of Sediment and Benthic Macroinvertebrates

Sediment and benthic invertebrate samples were collected from nine sample stations in November 2003 (Fig. 1), including six stations (SA7-1 through SA7-6) in the immediate vicinity of Study Area 7 and three reference stations (RF-1 through RF-3) located in comparable sediment environments not influenced by activities at the site. It should be noted that the selection of the locations for reference stations was based on similarity to Study Area 7, both in terms of benthic physical habitat and immediate nearby land uses, with the ultimate goal of characterizing baseline conditions of the local estuary. Reference stations in this study are not intended to represent pristine or ideal benthic habitats. Because Study Area 7 lies within the tidally influenced portion of the New York/New Jersey Estuary, and waters from Newark Bay, Passaic River, and Hackensack River are likely to influence the site, reference stations were selected both upstream and

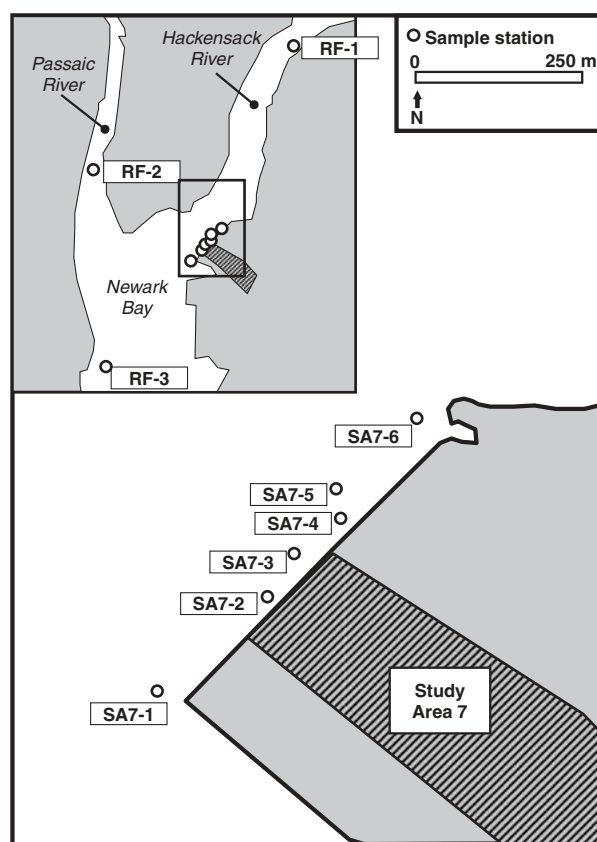


Fig. 1. Location of Study Area 7 (large map) and reference area sample stations (inset) in Upper Newark Bay, New Jersey

downstream of the site. Reference stations were located approximately 2 km south in Newark Bay, 1.5 km northwest in Passaic River, and 2.5 km north in Hackensack River and were offshore of areas with similar land uses (industrial) to Study Area 7. Because benthic communities are affected by depth and salinity in estuaries (Chapman and Wang 2001), sediment and benthic invertebrate samples were collected from similar depths (1.3–4.6 m) and interstitial water salinities (11–16 g/L).

Sediment was collected from the top 15 cm of sediment using a 0.1-m² Van Veen grab sampler. Subsamples of each grab were preserved in commercial laboratory-supplied glass sample containers and shipped at 4°C for chemical and physical analyses. Remaining sediment was placed in food-grade polypropylene bags, shipped at 4°C to a commercial laboratory, stored for 4–5 weeks, and sieved (2000- μ m) to remove large debris before use in laboratory experiments. Epibenthic and infaunal invertebrates at each station were collected from three additional grabs, which were sieved (500- μ m sieve) in the field to collect organisms. Organisms were preserved in a 10% (w/w) buffered formalin solution followed by a 70% (w/w) ethanol solution and identified by taxonomists to the lowest practical taxonomic level (typically genus or species) and enumerated. Invertebrates from the three grabs were pooled and weighed to determine total biomass on a wet weight (ww) basis, which was converted to a dry weight (dw) basis, assuming a moisture content of 75% (USEPA 1993c).

Sediment Chemical and Physical Analyses

Sediments were analyzed for chemical and physical parameters by certified analytical laboratories according to standard protocols (*e.g.*,

Table 1. Analytical methods used for chemical analyses of sediment, sediment porewater, and tissue.

Constituent ^a	Analysis Method	Sediment	Sediment Porewater	Tissue
Total Metals	USEPA 6010B; 7471 (Hg)	•	—	•
Dissolved Metals	USEPA E200.7; 6010 (Cr); C245.1 (Hg)	—	•	—
Hexavalent Chromium	USEPA 7199	•	—	—
AVS/SEM	Allen <i>et al.</i> (1993)/USEPA 6010B	•	—	—
Organotins	OR 560, NOAA Technical Memo 130	•	—	•
SVOCs	USEPA 8270C	•	—	•
Coplanar PCBs	USEPA 1668	•	—	•
PCB Homologues	USEPA 1668	•	—	—
PCDDs/PCDFs	USEPA 8290.	•	—	—
PBDEs	USEPA 1614	•	—	—
Pesticides	USEPA 8081	•	—	•
pH	USEPA 9045	•	—	—
TOC	ASTM D2579, modified	•	—	—
Ammonia	USEPA 350.1–350.2	•	•	—
Grain size	ASTM D422	•	—	—
Lipid content	NOAA (1998), modified	—	—	•

^a AVS = acid volatile sulfide, SEM = simultaneously extracted metal, SVOC = semi-volatile organic compound, PCB = polychlorinated biphenyl, PCDD = polychlorinated dibenzo(*p*)dioxin, PCDF = polychlorinated dibenzofuran, PBDE = polybrominated diphenyl ether, TOC = total organic carbon

USEPA 2003a; Table 1). Sediment porewater was extracted by centrifugation (ASTM 2001) and analyzed for 24 total and dissolved metals (including total chromium). Chemical and physical testing of sediments and biota tissues was performed by Columbia Analytical Services (Rochester, NY) and Severn Trent Laboratories (Edison, NJ).

Laboratory Toxicity and Bioaccumulation Testing

The study design included four laboratory experiments with whole sediment: a 10-d survival experiment with amphipods (*Leptocheirus plumulosus*), a 28-d survival and growth experiment with polychaetes (*Neanthes arenaceodentata*), a 28-d survival and bioaccumulation experiment with clams (*Macoma nasuta*), and a 28-d survival and bioaccumulation experiment with polychaetes (*Nereis virens*). Experimental conditions (Table 2) were in accordance with standard procedures for toxicity and bioaccumulation experiments (USEPA and USACE 1991; USEPA 1993a; USACE 1998; USEPA and USACE 1998; ASTM 2003a, 2003b). Laboratory testing was conducted by MEC Analytical Laboratory Systems (Carlsbad, CA). Ancillary overlying water quality (temperature, salinity, O₂, pH, and ammonia) was monitored in all replicates at the initiation of exposure (Day 0) and daily in one replicate per treatment for the remainder of the experiments. All parameters were within acceptable ranges as specified in protocols, except for salinity, which exceeded acceptable ranges by approximately 5 g/L in the *N. arenaceodentata* toxicity test. Organism responses in this experiment showed no evidence of effects related to the higher salinity. Overlying water and sediment porewater ammonia concentrations were measured at the experiment initiation in all replicates and weekly in one replicate per treatment in *N. arenaceodentata* and *L. plumulosus* tests and were within acceptable ranges as specified in protocols. Sediments obtained from Booth Bay Harbor, MA (*N. arenaceodentata*, *L. plumulosus*, and *M. nasuta* experiments) and Discovery Bay, WA (*N. virens* experiment) were known to provide organisms with adequate substrate for survival and growth and served as control sediments.

Upon termination of each experiment, sediments were sieved to enable collection and enumeration of survivors. *N. arenaceodentata* survivors were rinsed, blotted dry, placed in a pre-weighed container, dried for 24 h, and weighed to 0.01 mg, dw. Growth during the experiment was calculated using final weights and average weight of

30 juvenile polychaetes measured at the initiation of the test. *M. nasuta* and *N. virens* survivors were placed in sediment-free containers for 24 h to allow organisms to purge gut contents. After gut purging, tissues were placed in clean glass jars with Teflon-lined lids, frozen, homogenized, and analyzed for 13 metals (including total chromium), 4 organotins, 21 pesticides, 12 polychlorinated biphenyls (PCBs), 16 polycyclic aromatic hydrocarbons (PAHs), and percent lipids (determined by hexane extraction) by a certified analytical laboratory (Severn Trent Laboratories–Burlington, Colchester, VT) according to standard protocols (NOAA 1998; USEPA 2003a; Table 1).

Data Analysis

All chemicals detected in at least one of the sediment or porewater samples were evaluated using a cause–effect approach to identify chemicals or chemical classes likely to contribute to toxicity in laboratory experiments. Aquatic toxicity data (USEPA 2002a, 2002b, 2003b, 2003c) were used to derive site-specific sediment quality benchmarks for most organic compounds and metals using the EqP approach (Di Toro *et al.* 1991; USEPA 1993b; Di Toro *et al.* 2000a, 2000b; USEPA 2003b, 2003c, 2005). Using USEPA (2003b) methodology, PAH mixtures in sediment were assessed using an additive model in which porewater concentrations of each PAH (estimated by EqP) were divided by its corresponding EqP benchmark to yield PAH toxic units (TU). Toxic units for each sample were summed to yield ΣPAH TU. A site-specific adjustment factor of 1.6 (i.e., ΣPAH TU × 1.6) was applied to estimate ΣPAH TU derived from alkylated PAHs, which were not analyzed as part of this study. The site-specific adjustment factor was estimated from the ratio of alkylated ΣPAH TU to unsubstituted ΣPAH TU in sediment collected from the same sample stations during July 2005. A modified EqP approach (Fuchsman 2003) was used to evaluate organic compounds with Log K_{ow} < 3. Sediment toxicity test results from previously published laboratory and field studies and published EqP models were used to screen chlorinated benzenes (Fuchsman *et al.* 1999), phthalate esters (Call *et al.* 2001a, 2001b), dibenzofuran (Di Toro *et al.* 2000a), DDT and its metabolites (Swartz *et al.* 1994; Ferraro and Cole 1997), PCB mixtures (Fuchsman *et al.* 2006), dioxin (Barber *et al.* 1998), and tributyltin (Meador *et al.* 2002).

Table 2. Experimental design and conditions for toxicity and bioaccumulation experiments

Parameter	<i>L. plumulosus</i> survival	<i>N. arenaceodentata</i> survival and growth	<i>N. virens</i> survival and bioaccumulation	<i>M. nasuta</i> survival and bioaccumulation
Exposure media	2-cm sediment layer (~150 ml) plus 900 ml gently-aerated overlying water	1-cm sediment layer (~50 ml) plus 150 ml gently-aerated overlying water	4-cm sediment layer (~5 L) plus 10 L gently-aerated overlying water	5-cm sediment layer (~5 L) plus 10 L gently-aerated overlying water
Water renewal	None	Static renewal: 60% every 7 d	Flow through: 20.0–25.0 ml/min	Flow through: 21.4–25.0 ml/min
Test chamber	1-L glass jars	250-ml glass jars	22-L fiberglass trays	22-L fiberglass trays
Replicates/treatment	5	10	5	5
Organisms/replicate	20	1	10	25
Feeding	None	Slurry containing 2 mg fish food flakes and 1 mg alfalfa added every 3.5 d	None	None
Organism age	30 d	Juveniles	Adults	Adults
Test photoperiod (hours light:dark)	24:0	12:12	16:8	16:8
Method reference	USEPA and USACE 1998; ASTM 2003b	USACE 1998; ASTM 2003a	USEPA and USACE 1991; USEPA 1993a	USEPA and USACE 1991; USEPA 1993a

In addition to estimating standard heterogeneity indices (Shannon-Wiener [H'], Margalef Species Richness [SR], Pielou's Evenness [J'], and Swartz's Dominance Index [SDI; minimum number of taxa comprising 75% of abundance]), benthic community data were interpreted using a Benthic Invertebrate Index of Biotic Integrity (B-IBI) developed for the New York/New Jersey Estuary by USEPA (Adams *et al.* 1998). The B-IBI is a multimetric index based on the IBI originally developed by Karr (1981) to examine the ecological health of fish communities. Benthic community data from reference and Study Area 7 stations were expressed as five metrics (number of taxa, biomass, total abundance, abundance of pollution-indicative taxa, and abundance of pollution-sensitive taxa), each of which was scored according to the scoring scheme given in Adams *et al.* (1998). Classification of pollution-indicative and pollution-tolerant taxa were based on local criteria developed in USEPA Environmental Monitoring and Assessment Program for estuaries (Adams *et al.* 1998). Scores of 1 (impaired), 3 (slightly impaired), or 5 (unimpaired) were assigned to the five metrics at each sample station, based on deviation of the metric from values indicative of ecologically healthy reference sites (Adams *et al.* 1998). The five individual metric scores were averaged to calculate composite B-IBI scores, which were interpreted according to Adams *et al.* (1998): <2 indicated an impaired benthic community; ≥ 2 and <3 indicated a slightly impaired benthic community; and ≥ 3 indicated an unimpaired benthic community. Mean composite B-IBI scores and heterogeneity indices for the nine sample stations were compared statistically using Tukey-Kramer Honestly Significant Difference (HSD) ($\alpha = 0.05$).

Laboratory toxicological data generated during each of the four experiments were compared statistically using Tukey-Kramer HSD ($\alpha = 0.05$; Newman 1995), to compare results among all samples. Dunnett's test was used to confirm results of the Tukey-Kramer HSD specifically for comparisons to control results (Newman 1995). Prior to statistical tests, mortality data were arcsine square root transformed (Newman 1995), and *N. arenaceodentata* growth data were \log_{10} transformed. Sediments were identified as toxic if mean responses were significantly less than mean control response. The results of these statistical tests are also compared to minimum significant difference criteria identified for survival endpoints (80% of control; Thursby *et al.* 1997) and *N. arenaceodentata* growth (44% of control; Phillips *et al.* 2001). Median lethal dose (LC₅₀) estimates were calculated via the Trimmed Spearman-Kärber method (Hamilton *et al.* 1977) using ΣPAH TU as the measure of exposure. A dose-response

relationship was confirmed using a nonlinear Gompertz model (Newman 1995). Relationships between toxicity and chemical concentrations were also explored using parametric and nonparametric correlation, partial correlation, and multiple regression analyses.

Results and Discussion

Benthic Community

Benthic community indices and B-IBI scoring for the reference and Study Area 7 sample stations were typical of the values observed in a 2003 Newark Bay study (Table 3; Adams *et al.* 1998). The high abundance of pollution-indicative species (*e.g.*, *Streblospio benedicti*, *Oligochaetes*, *Mulinia lateralis*, and *Polydora cornuta*) and low abundance of sensitive species (*e.g.*, *Acteocina canaliculata*, *Tellina* sp., *Ampharetidae*, and *Nephtys* sp.) indicated impairment at all reference and Study Area 7 stations. The number of taxa, total abundance, and total biomass of invertebrates indicated less evidence of impairment and were more variable among stations. Composite B-IBI scores (Table 3) suggested that benthic communities of two of the three reference stations (RF-1 and RF-2) were slightly impaired. Benthic communities at all Study Area 7 stations were scored as slightly impaired, except for stations SA7-1 and SA7-5, which were scored as unimpaired.

By comparison, several stations measured by Adams *et al.* (1998) were within approximately 1–2 km of Study Area 7 (in Passaic River, Hackensack River, and Upper Newark Bay), with composite B-IBI scores of 1.0–2.6, similar to or lower than most of the composite B-IBI values estimated in this study. One of the Adams *et al.* (1998) stations was within approximately 0.5 km of RF-3, but indicated a higher level of benthic community impairment (composite B-IBI of 1.8).

The use of abundance and biomass metrics in the B-IBI developed by Adams *et al.* (1998) is a source of uncertainty in this analysis. Some benthic invertebrate community indices exclude the use of abundance and biomass metrics (DeShon

Table 3. Mean (SD) benthic community values and Benthic Index of Biotic Integrity (B-IBI) scores (Adams *et al.*, 1998) for reference and Study Area 7 stations

Station	Number of Taxa (number)		Abundance (number/m ²)		Biomass (g dw/m ² , pooled replicates)		Abundance of Pollution-indicative Taxa (%)		Abundance of Pollution-sensitive Taxa (%)		Composite B-IBI Score
	Value	Score	Value	Score	Value	Score	Value	Score	Value	Score	
Reference Stations											
RF-1 (n = 3)	15 (0.6)	3.0 (0)	6,703 (1,914)	5.0 (0)	1.1 (0.93)	3 (1)	77 (7.9)	1.0 (0)	0.1 (0.2)	1.0 (0)	2.6 ^{a,b} (0)
RF-2 (n = 3)	20 (2.5)	3.7 (1.2)	4,977 (2,589)	4.3 (1.2)	0.93 (0.93)	1 (1)	39 (3.1)	2.3 (1.2)	2.8 (2.6)	1.7 (1.2)	2.6 ^{a,b} (0.4)
RF-3 (n = 3)	28 (4.4)	5.0 (0)	16,640 (10,820)	1.7 (1.2)	3.7 (1.2)	5 (1.2)	40 (8.1)	2.3 (1.2)	0.0 (0.1)	1.00 (0)	3.0 ^a (0.4)
Study Area 7 Stations											
SA7-1 (n = 3)	21 (3.2)	3.7 (1.2)	7,577 (3,453)	4.3 (1.2)	4.7 (1.2)	5 (1.2)	69 (2.1)	1.0 (0)	1.4 (1.2)	1.0 (0)	3.0 ^a (0.4)
SA7-2 (n = 3)	26 (3.1)	5.0 (0)	10,983 (2,552)	3.7 (1.2)	27 (1.2)	3 (1.2)	81 (7.6)	1.0 (0)	0.4 (0.3)	1.0 (0)	2.7 ^{a,b} (0.2)
SA7-3 (n = 3)	18 (1.0)	3.0 (0)	8,580 (970)	5.0 (0)	1.4 (0)	3 (0)	87 (4.8)	1.0 (0)	0.2 (0.3)	1.0 (0)	2.6 ^{a,b} (0)
SA7-4 (n = 3)	21 (2.1)	4.3 (1.2)	12,757 (2,179)	3.0 (0)	4.9 (0)	5 (0)	66 (3.2)	1.0 (0)	1.5 (0.4)	1.0 (0)	2.9 ^{a,b} (0.2)
SA7-5 (n = 3)	23 (3.1)	4.3 (1.2)	4,443 (1,630)	5.0 (0)	5.9 (0)	5 (0)	64 (10.6)	1.0 (0)	1.3 (0.7)	1.0 (0)	3.3 ^a (0.2)
SA7-6 (n = 2)	11 (2.8)	1.0 (0)	4,420 (2,489)	4.0 (1.4)	1.4 (1.4)	3 (1.4)	72 (13.0)	1.0 (0)	0.2 (0.2)	1.0 (0)	2.4 ^b (0.3)
Newark Bay (Adams <i>et al.</i> 1998)											
Various (n = 28)	14 (2.6)	1 (0)	11,000 (47,000)	3 (0)	5.4 (2)	5 (2)	65 (7.1)	1 (0)	0.3 (0.3)	1 (0)	2.2 (—)

Composite B-IBI scores with the same superscript are not significantly different (Tukey-Kramer Honestly Significant Difference, $\alpha = 0.05$); mean scores <3 and ≥ 3 indicated slightly impaired and unimpaired communities, respectively. Mean (90% CI) benthic community values for Newark Bay (Adams *et al.*, 1998) were scored and provided for reference.

1995; USEPA 1999) because they are difficult to interpret. A typical model of abundance/biomass response to sediment toxicity assumes one of four conditions: (1) nontoxic conditions support intermediate abundance/biomass; (2) slightly toxic conditions may support unusually high abundance/biomass of tolerant species; (3) intermediate toxicity can cause a return to intermediate abundance/biomass; and (4) severely toxic conditions support only low abundance/biomass (Rakocinski *et al.* 2000). The uncertainty lies, for example, in distinguishing condition 1 from condition 3, because both are associated with intermediate abundance and biomass. Also, biomass can be strongly affected by the presence of a few large individuals (*e.g.*, large bivalves), as occurred at station SA7-2.

In addition to examining composite B-IBI scores and constituent metrics, heterogeneity indices (diversity, species richness, dominance, and evenness) were used as additional lines of evidence to support conclusions regarding benthic community health. Although low values for heterogeneity indices are often a consequence of pollution and suggest impairment (Newman and Unger 2003), the absolute values of the indices themselves cannot be used to rate benthic communities at each station as impaired, slightly impaired, or unimpaired. For each index, stations were separated into three groups based on statistical significance (Fig. 2). Although relative statistically significant differences do not necessarily imply ecologically significant differences, conclusions from this approach provided an addi-

tional line of evidence to support the B-IBI conclusions. In general, the results of this approach agreed with the results of the B-IBI scoring. For example, stations RF-1, SA7-3, and SA7-6 were consistently in the lowest groupings for two to four of the heterogeneity indices (Fig. 2), suggesting possible relative impairment. This concurred with the B-IBI score of slightly impaired for these stations (Table 3). Stations RF-2, SA7-2, and SA7-4 were ranked in the moderate grouping for three of the four heterogeneity indices, providing some additional evidence to support the B-IBI scores of slightly impaired. Heterogeneity indices for stations RF-3, SA7-1, and SA7-5 were consistently within the high groupings for two to four of the indices, suggesting that relative impairment was minor, concurring with the B-IBI scoring as unimpaired.

Toxicity

Distinct differences in toxic responses were observed among test species, with the estuarine amphipod *L. plumulosus* exhibiting the greatest sensitivity and polychaetes (*N. arenaceodentata* and *N. virens*) exhibiting the least sensitivity (Fig. 3). Survival of *L. plumulosus* was significantly reduced in all sediments compared to controls, with the exception of SA7-4 (81% mean survival; Figure 3a). Severe toxicity (0–5% mean survival) was observed for *L. plumulosus* exposed to sediment

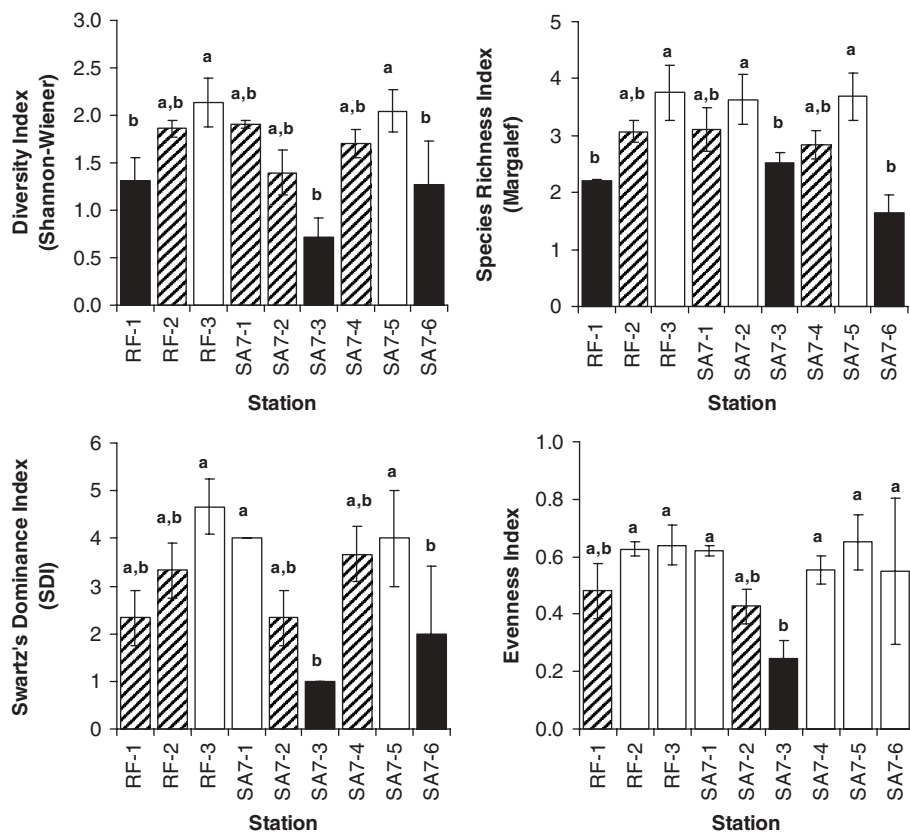


Fig. 2. Mean benthic heterogeneity indices for reference (RF-1, RF-2, RF-3) and Study Area 7 stations (SA7-1 through SA7-6). Error bars represent one standard deviation. Columns with the same letter are not significantly different (Tukey-Kramer Honestly Significant Difference, $\alpha = 0.05$). Heterogeneity indices represented by unshaded columns indicate the stations with the highest values. Striped and solid columns represent moderate and lower values, respectively, suggestive of higher relative levels of impairment

from four sample stations (RF-1, SA7-2, SA7-3, and SA7-6) and moderate toxicity (32–74% mean survival) was observed at three sample stations (RF-2, RF-3, SA7-1, and SA7-5). Significant reductions in *M. nasuta* survival were observed at two stations (RF-1 and SA7-6; Figure 3d), although the magnitude of effect was less than that observed in the amphipod tests at the same stations. Although no significant differences were observed in survival among sample stations for either polychaete species (Fig. 3b and 3c), growth of *N. arenaceodentata* in SA7-5 sediment (52% of control growth) was significantly lower than controls ($p = 0.012$, Figure 3e). The biological significance of this effect is uncertain and may be slight because of the high inherent variability in *N. arenaceodentata* growth data. In many bioassays involving field-collected sediment, significant sublethal toxic effects are only clearly evident for this test endpoint when means are less than 44% of control growth (Phillips *et al.* 2001). For the survival endpoint in all four test species, the results of statistical comparisons were consistent with the minimum significant difference (80% of control) identified by Thursby *et al.* (1997).

Sediment and Tissue Chemistry

Maximum sediment concentrations of six chemical constituents exceeded cause-effect screening sediment benchmarks (Table 4). PAHs appear to have the strongest link with observed toxicity. Σ PAH concentrations in sediment were as high as 7000 $\mu\text{g/g}$ organic carbon, with Σ PAH TU ranging from 0.5 to 16. Stations in which Σ PAH TU were >2 were lethally toxic to *L. plumulosus*, with a clear dose-response relationship evident

in the data (Fig. 4a). The LC_{50} (95% CI) based on Σ PAH TU was 1.2 (1.07–1.24) and did not differ greatly from the theoretical chronic threshold of 1 TU (USEPA 2003b). Sediments in which *M. nasuta* accumulated tissue Σ PAH concentrations of $\geq 2 \mu\text{mol/g}$ lipid were lethally toxic to *L. plumulosus* (Fig. 4b), a value similar to the tissue Σ PAH concentration threshold of 2.24 $\mu\text{mol/g}$ lipid associated with chronic toxicity (USEPA 2003b). The measured tissue Σ PAH concentrations are a lower limit of the actual concentrations, because only detected PAHs were considered, and concentrations were not corrected for unmeasured alkylated PAHs. Given the strong dose-response relationship between modeled estimates of PAH exposure and measured accumulation of PAHs in tissue to toxic thresholds, it is likely that PAHs accounted for a significant portion of the observed toxicity. PAHs are a widespread chemical contaminant of industrial and urban waterways (van Metre *et al.* 2000; USEPA 2004), and the presence of bioavailable PAHs at toxic concentrations is not surprising given the numerous current and historical sources of PAHs and uses of the Lower Hackensack River and its watershed. In *N. virens*, Σ PAH concentrations were below detection limits (approximately 0.1 $\mu\text{mol/g}$ lipid) for all stations except for RF-1.

Of the remaining chemicals implicated in Table 4, only heptachlor epoxide is likely to have contributed significantly to toxicity, though only at one location. Heptachlor epoxide was detected at station SA7-6 in sediment at a concentration 4–5 times higher than its cause-effect sediment benchmark; this sediment was severely toxic to *L. plumulosus* and significantly toxic to *M. nasuta*. Heptachlor epoxide was also detected in *N. virens* exposed to sediment from the same location. Thus, heptachlor epoxide was evidently bioavailable

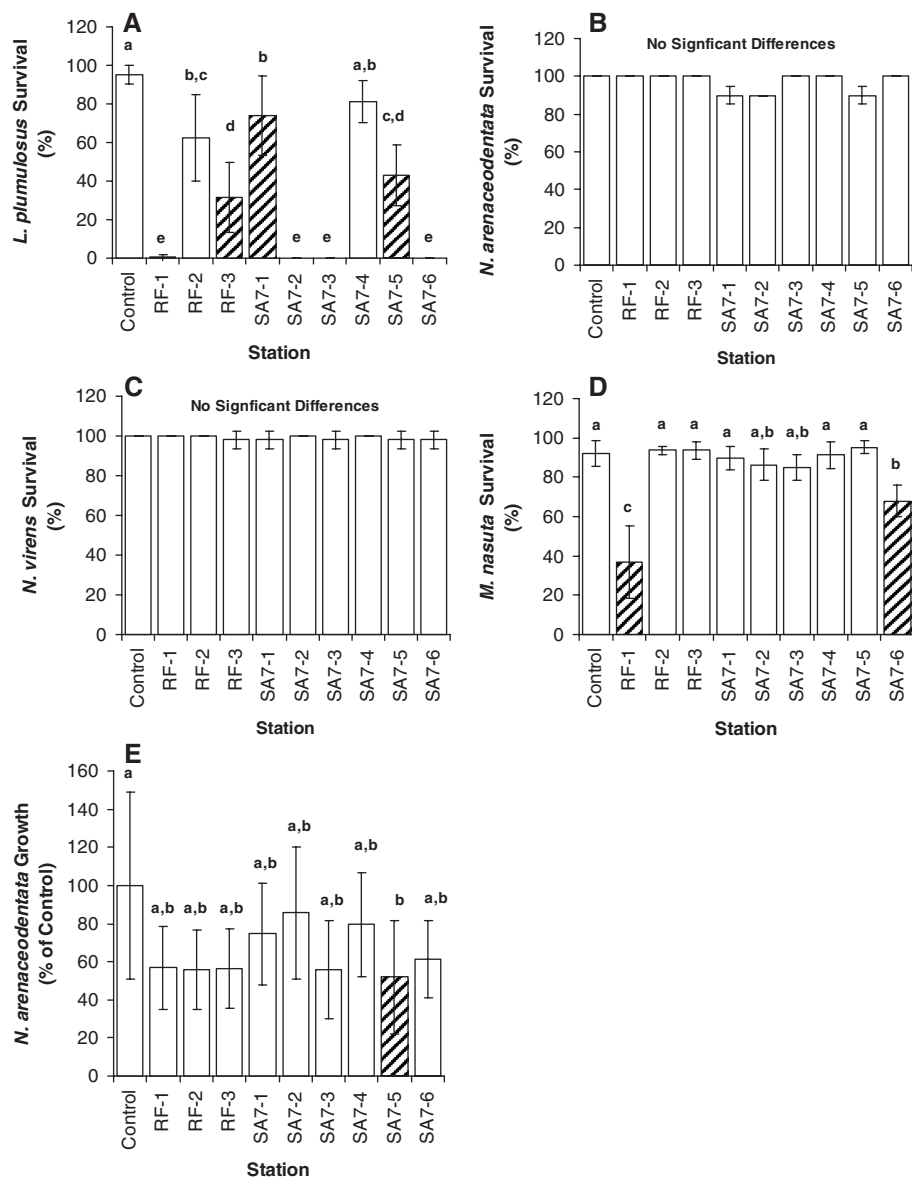


Fig. 3. Mean percent survival for *Leptocheirus plumulosus* (a), *Neanthes arenaceodentata* (b), *N. virens* (c), and *Macoma nasuta* (d) and growth for *N. arenaceodentata* (e) exposed to laboratory control sediment and sediments collected from reference (RF-1, RF-2, RF-3) and Study Area 7 stations (SA7-1 through SA7-6). Error bars represent one standard deviation. Columns with the same letter are not significantly different (Tukey-Kramer Honestly Significant Difference, $\alpha = 0.05$). Survival values represented by unshaded columns indicate no significant toxicity, and striped columns indicate intermediate toxicity

and may have contributed to the toxicity observed in sediment from this station.

Cationic metals detected in sediment porewater are unlikely to have contributed to toxicity, because acid volatile sulfide (AVS) was found to exceed concentrations of simultaneously extracted metals (SEM) at each of the triad sampling locations (Di Toro *et al.* 2000b; USEPA 2005). Although copper and lead concentrations in sediment porewater exceeded saltwater ambient water quality criteria (USEPA 2002a) at station SA7-3 (lead) and all stations except for RF-2 and RF-3 (copper), the maximum SEM/AVS ratio was 0.21 (stations SA7-2 and RF-3), suggesting that toxicity was not likely due to metals. Additionally, metals in porewater can overstate risks due to overextraction during centrifugation (USEPA 2001) or inability to distinguish bioavailable metals from unavailable metals sorbed to dissolved organic carbon or bound to inorganic ligands such as hydroxides and carbonates (Di Toro *et al.* 2000b; Evangelou 1998; USEPA 2005).

It is possible that AVS may have decreased in shallow sediment layers because of continual aeration of the overlying water during the laboratory toxicity tests, thereby releasing labile metals (*e.g.*, copper, lead, and zinc) into porewater and overlying water (Carbonaro *et al.* 2005). However, experiments performed to investigate the potential for release of metals during extreme physical disturbances of Study Area 7 sediments (for example by dredging, ship propellers, or by river currents and tides generated during severe weather events) indicated that this does not occur (ENVIRON 2006; unpublished data). Sediments collected from Study Area 7 and vigorously mixed with overlying water and aerated to saturated dissolved oxygen conditions for up to 24 h did not result in significantly elevated dissolved metal concentrations. Despite small decreases in AVS concentrations (<50%), only slight increases in concentrations of dissolved copper and lead in the sediment elutriate were observed. In contrast, hexavalent chromium was not detected in elutriate either before or after aggressive mixing and aeration of the sediment.

Table 4. Cause–effect evaluation of chemicals detected in Study Area 7 and reference sediment and porewater (N = 9)

Constituent	Units	Range	Cause–effect screening benchmark	Number of samples \geq benchmark	Benchmark source
Metals (sediment)					
Cadmium, copper, lead, nickel, zinc, mercury	$\mu\text{mol excess SEM/g OC}$	<0	130	0	USEPA (2005)
Chromium	NA	AVS detected in all samples	AVS presence/absence ^a	0	USEPA (2005)
Metals (porewater)					
Arsenic	$\mu\text{g/L}$	<7 – 52.5	36	3	USEPA (2002a)
Total Chromium	$\mu\text{g/L}$	<3 – 17.1	50	0	USEPA (2002a; hexavalent Cr)
Copper	$\mu\text{g/L}$	<2 – 21.2	3.1	2	USEPA (2002a)
Lead	$\mu\text{g/L}$	<2 – 20.9	8.1	1	USEPA (2002a)
Mercury	$\mu\text{g/L}$	<0.1 – 0.34	0.94	0	USEPA (2002a)
Zinc	$\mu\text{g/L}$	12 – 73	81	0	USEPA (2002a)
SVOCs					
PAHs	ΣTU	0.5 – 16	1	6	USEPA (2003b)
Chlorinated benzenes	ΣTU	<0.001 – 0.012	1	0	Fuchsman <i>et al.</i> (1999)
Bis(2-chloroethyl)ether	mg/kg	<0.05 – 0.31	29	0	Di Toro <i>et al.</i> (1991); USEPA (2002b); Fuchsman (2003)
Bis(2-ethylhexyl)phthalate	$\mu\text{g/g OC}$	<13 – 310	> 65,000	0	Call <i>et al.</i> (2001a); Call <i>et al.</i> (2001b)
Carbazole	$\mu\text{g/g OC}$	1.2 – 28	190	0	Di Toro <i>et al.</i> (1991); USEPA (2002b)
4-Chloroaniline	mg/kg	0.02 – 0.11	0.012	4	Di Toro <i>et al.</i> (1991); USEPA (2002b); Fuchsman (2003)
Dibenzofuran	$\mu\text{g/g OC}$	1.7 – 134	1,650	0	Di Toro <i>et al.</i> (2000a)
4-Methylphenol	mg/kg	<0.02 – 0.11	6.2	0	Di Toro <i>et al.</i> (1991); USEPA (2002b); Fuchsman (2003)
Nitrobenzene	mg/kg	<0.05 – 0.36	7.7	0	Di Toro <i>et al.</i> (1991); USEPA (2002b); Fuchsman (2003)
Pesticides					
Total DDT	$\mu\text{g/g OC}$	<1 – 7.1	30 – 300	0	Ferraro and Cole (1997); Swartz <i>et al.</i> (1994)
Endrin ketone	$\mu\text{g/g OC}$	<0.3 – 0.58	5.4	0	USEPA (2003c)
Heptachlor epoxide	$\mu\text{g/g OC}$	<0.4 – 0.48	0.11	1	Di Toro <i>et al.</i> (1991); USEPA (1980)
Other Constituents					
Total PCBs	$\mu\text{g/g OC}$	0.1 – 22	400 – 1,100	0	Fuchsman <i>et al.</i> (2006)
Total PCDD/PCDFs	$\mu\text{g/g OC}$	0.04 – 1.1	>1.4	0	Barber <i>et al.</i> (1998; 2,3,7,8-TCDD)
Total PBDEs	$\mu\text{g/g OC}$	0.06 – 24	80	0	Di Toro <i>et al.</i> (1991); Wollenberger <i>et al.</i> (2002)
Total Organotins	$\mu\text{g/g OC}$	<0.1 – 0.27	0.6 – 6	0	Meador <i>et al.</i> (2002)

Maximum concentrations in bold indicate values greater than cause-effect screening benchmark.

^a Chromium toxicity is not expected if AVS is detectable.

In the sediment triad study, some indication of bioavailable lead was noted in the *M. nasuta* bioaccumulation test; lead concentrations were significantly different among the sediments, ranging from 0.2 mg/kg, ww in controls to 1.1 mg/kg in clams exposed to RF-2 sediment. However, mean tissue concentrations of *M. nasuta* exposed to the four lethally toxic sediments (0.4–0.7 mg/kg, ww) were not statistically different from concentrations of *M. nasuta* exposed to only sediment with no toxicity (SA7-4, 0.8 mg/kg, ww). Thus, although bioavailable lead was present, elevated levels of bioavailable lead were not related to toxicity. There were no significant differences in the concentration of lead in *N. virens* (0.4–0.6 mg/kg, ww). For copper, no significant differences were found among *M. nasuta* (1.9–2.6 mg/kg) or *N. virens* (1.9–3.2 mg/kg, ww) tissues. However, it is unclear whether tissue concentrations in these organisms would indicate increases in copper bioavailability, because there is conflicting evidence regarding

the potential for physiological regulation in bivalves exposed to elevated copper concentrations (Cain and Luoma 1990; Mersch *et al.* 1996). Two sediment samples, SA7-2 and SA7-3, exceeded the cause–effect screening benchmark for copper in porewater (Table 4) and were both lethally toxic to *L. plumulosus* (Fig. 3a). Thus, although the contribution of lead to toxicity in the *L. plumulosus* experiment can be discounted, it is possible that copper may have contributed to toxicity in two of the sediments in this study.

Although elevated total chromium concentrations in sediment (as high as 1900 mg/kg in SA7-6) were the rationale for conducting the investigation, dissolved total chromium was detected in pore water at trace concentrations that were well below the chronic saltwater ambient water quality criterion for hexavalent chromium (50 $\mu\text{g/L}$). The hexavalent chromium criterion was used because USEPA has not designated a trivalent chromium saltwater criterion. Therefore, total chro-

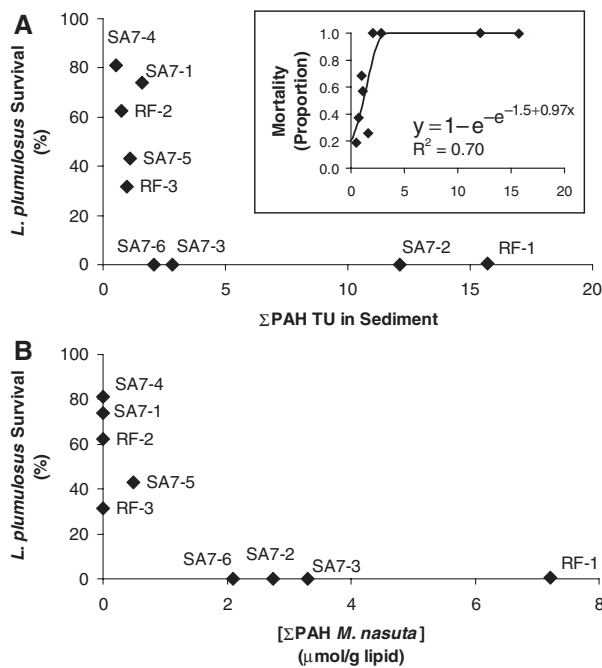


Fig. 4. Relationship between *Leptocheirus plumulosus* survival and Σ PAH Toxic Units (TU) in sediment (a) and Σ PAH concentrations in *Macoma nasuta* exposed to sediment (b). Labels indicate reference (RF-1, RF-2, and RF-3) and Study Area 7 (SA7-1 through SA7-6) stations. The inset figure in (a) illustrates a nonlinear dose–response model (Gompertz) fit of the mortality data, using Σ PAH TU as a measure of dose

mium was unlikely to contribute to toxicity to benthic organisms in these laboratory experiments. The presence of AVS and additional indicators of reducing conditions (e.g., ferrous iron, divalent manganese) in the sediment (Martello et al. 2007) are strong indicators of reducing conditions, which according to several studies precludes the presence hexavalent chromium in favor of the less toxic, less soluble trivalent chromium species (Berry et al. 2004; Besser et al. 2004; Becker et al. 2006).

Becker et al. (2006) also confirmed the low bioavailability of trivalent chromium and limited presence of hexavalent chromium in sediments containing AVS collected further upstream on the Hackensack River at locations potentially impacted by a different COPR site. In that study, *L. plumulosus* exhibited no significant mortality at the highest chromium concentration tested (3970 mg/kg), and the amphipod *Ampelisca abdita* exhibited no significant mortality at chromium concentrations up to 1490 mg/kg in sediment (Becker et al. 2006).

Furthermore, the relative sensitivity of benthic species to sediments in the present study is not consistent with the known relative sensitivity to chromium. Specifically, several polychaetes (including *N. virens* and *N. arenaceodentata*) are among the most sensitive species to hexavalent chromium, whereas various amphipods exhibit intermediate sensitivity, and the clam *Macoma balthica* (closely related to *M. nasuta*) is the least-sensitive species known (USEPA 1985, 2002b).

N. virens and *M. nasuta* tissue analysis results following the bioaccumulation study showed the presence of total chromium, as would be expected for this essential nutrient. While

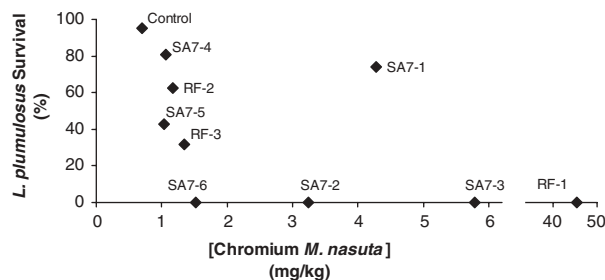


Fig. 5. Relationship between *Leptocheirus plumulosus* survival and chromium concentrations in *Macoma nasuta* exposed to sediment. Labels indicate laboratory control, reference (RF-1, RF-2, and RF-3), and Study Area 7 (SA7-1 through SA7-6) stations

there was variability in chromium tissue concentrations between locations, the variability was not explained by concentrations of total or hexavalent chromium in sediment or total chromium in porewater ($r^2 = <0.0001-0.05$). For example, location SA7-6 had the maximum detected total chromium concentration in sediment (1900 mg/kg), but had *M. nasuta* mean (SD) total chromium in biological tissues of 1.5 (0.3) mg/kg, ww, and *N. virens* mean (SD) total chromium in biological tissues of 2.8 (1.4) mg/kg, ww. Both values were moderate in comparison to tissue concentrations observed at other locations. In contrast, location RF-1 with the third highest concentration of total chromium in sediment (238 mg/kg), had the highest mean (SD) concentrations of total chromium in *M. nasuta* tissue, 45 (38) mg/kg, ww. RF-1 *N. virens* total chromium tissue concentrations 1.8 (1.6) mg/kg, ww were comparable to those seen at SA7-6, but both RF-1 and SA7-6 were less than the *N. virens* concentrations seen at RF-2 (4.2 (4.2) mg/kg, ww). Differences among chromium concentrations in *N. virens* are minor, as concentrations did not differ significantly ($p = 0.24$) among controls, reference stations, and Study Area 7 stations. Not including RF-1 results, mean chromium concentrations in *M. nasuta* ranged from 1.2 (RF-2) to 1.3 (RF-3) mg/kg, ww in reference stations, from 1.0 (SA7-5) to 5.8 (SA7-3) mg/kg, ww in Study Area 7 stations, and 0.7 mg/kg, ww in controls. Mean concentrations were statistically different among sample stations. Five groups could be distinguished using Tukey-Kramer HSD (from highest to lowest): SA7-3 > SA7-1 \geq SA7-2 \geq SA7-6 \geq RF-3, RF-2, SA7-4, SA7-5, and controls. However, these relationships show no statistically significant relationship to total chromium concentrations in sediment, as location SA7-6 is within the center of these groups (with total chromium concentrations in mg/kg noted in parentheses after each sample location): RF-1 (238), SA7-3 (223), SA7-1 (171), SA7-2 (329), SA7-6 (1900), RF-3 (148), RF-2 (157), SA7-4 (192), SA7-5 (139).

N. virens sediment toxicity testing results showed 98 to 100% survival at each location, but *L. plumulosus* and *M. nasuta* showed some degree of toxic response at a variety of locations. Although four of the five stations (including RF-1) with the highest levels of total chromium concentrations in *M. nasuta* were severely toxic to *L. plumulosus* (0% survival), there was no relationship between toxicity and bioavailability (Fig. 5). It is unlikely that total chromium concentrations in whole tissue approached toxic levels, because concentrations were similar to those measured in the mussel *Mytilus edulis* (5–9 μ g/g, ww) exposed to very low

(nontoxic) levels of chromium (5–10 µg/L hexavalent chromium in seawater) (Zarogian and Johnson 1983). The majority of chromium in marine mussels is thought to be derived from the ingestion of trivalent chromium associated with particulate matter (Wang *et al.* 1997). In this study, however, chromium concentrations in the clam *M. nasuta* were not associated with concentrations of total chromium in sediment, hexavalent chromium in sediment, or total chromium in porewater ($r^2 = < 0.0001$ –0.05). Physiological stress related to PAH exposure may have disrupted internal regulation of chromium concentrations in *M. nasuta*, because chromium concentrations in *M. nasuta* tissue were positively correlated with ΣPAH TU ($r^2 = 0.62$, $p = 0.0119$). Mean (SD) concentrations of total chromium in *N. virens* did not differ significantly ($p = 0.24$) among controls, reference stations, and Study Area 7 stations, with tissue concentrations ranging from 0.6 (0.09) mg/kg, ww in sample RF-3 to 9.3 (10.0) mg/kg, ww in the controls.

Arsenic concentrations in sediment porewater also exceeded the USEPA's (2002a) water quality criterion at stations SA7-3 and SA7-6, two of the four sample stations exhibiting severe toxicity to *L. plumulosus* (Fig. 3). SEM/AVS relationships cannot be used to confirm or refute the porewater arsenic data, because in contrast to copper and lead, arsenic does not form insoluble sulfide complexes. However, the bioavailability of arsenic in marine sediments under anoxic conditions is very low, similar to chromium (Neff 1997). Not surprisingly, tissue concentrations of arsenic in the *M. nasuta* and *N. virens* bioaccumulation tests did not suggest significant levels of bioavailable arsenic. Although mean (SD) arsenic concentrations in *M. nasuta* controls (1.8 (0.33) mg/kg, ww) were significantly lower than concentrations observed in animals exposed to sediment from reference and Study Area 7 stations, which ranged from 2.2 (0.40) mg/kg, ww in SA7-5 to 3.0 (0.49) mg/kg, ww in SA7-3, arsenic concentrations were not significantly different among reference and Study Area 7 stations. Mean (SD) arsenic concentrations in *N. virens* did not differ significantly among control sediments (1.7 (0.12) mg/kg) or Study Area 7 and reference sediments, which ranged from 1.4 (0.12) mg/kg in RF-3 to 1.8 (0.12) mg/kg in SA7-5. Thus, there was no evidence for elevated levels of bioavailable arsenic sufficient to explain the pattern of toxicity among reference and Study Area 7 stations.

Although 4-chloroaniline was detected in sediment at concentrations significantly above the cause–effect screening sediment benchmark in four of the reference and Study Area 7 sediments (RF-2, RF-3, SA7-4, and SA7-5), these sediments were moderately or insignificantly toxic (Fig. 3). The aquatic toxicity data identified from the USEPA's ECOTOX database for 4-chloroaniline were highly variable (USEPA 2002b), and the screening value provided in Table 4 is based on data for a freshwater species. Saltwater organisms may be much less sensitive to 4-chloroaniline than freshwater species (WHO 2003).

Correlation analyses were performed to further investigate the relationship between *L. plumulosus* toxicity and chemicals that exceeded sediment-screening benchmarks (Table 4). Relationships between heptachlor epoxide and 4-chloroaniline could not be elucidated because of the low number of detections (1 and 4, respectively). Among the remaining chemicals

(arsenic, copper, lead, and PAHs), all were significantly negatively correlated with survival ($p < 0.05$), based on both parametric and nonparametric tests. However, chemical concentrations were also correlated with one another (cross correlation), preventing a clear understanding of the relationships between toxicity and chemical concentrations. A partial correlation analysis was conducted to analyze the relationships between each variable and survival while holding the other variables constant. The only significant negative partial correlation between survival and chemical concentration was for ΣPAH TUs ($p < 0.0001$), suggesting that among the variables in this analysis, PAHs were the single chemical variable most strongly associated with toxicity. However, this approach does not account for the possibility of mixture effects among the chemicals.

Stepwise multiple regression analysis conducted to identify models involving more than one chemical variable indicated that the optimal multiple regression model was a two-variable model utilizing ΣPAH TUs and arsenic, where survival = $(-3.1 \times \Sigma\text{PAH TUs}) + (-0.48 \times \text{arsenic}) + 56$ (adjusted $R^2 = 0.47$, $p < 0.0001$). The addition of arsenic to the model represented only a modest improvement in fit (R^2 of 0.30 based on a single-variable, linear model using ΣPAH TUs to an adjusted R^2 of 0.47 using the two-variable model). This modest improvement suggests that arsenic may be only a minor contributor to toxicity. As discussed above, the bioaccumulation experiments did not suggest that arsenic was bioavailable. In contrast, the evidence provided by this correlation analysis complements multiple lines of evidence that identify PAHs as a primary source of toxicity, including the presence of bioavailable PAHs (Fig. 4b) and observations of significant toxicity at the theoretical threshold of ΣPAH TU = 1 (Fig. 4a). It is important to note, however, that partial correlation and stepwise regression analyses require the use of parametric methods, which assume linearity, whereas a nonlinear model provided a better fit for understanding the contribution of ΣPAH TU to sediment toxicity (Fig. 4a).

Sediment Quality Triad Assessment

The benthic community, laboratory toxicity and bioaccumulation tests, and chemical lines of evidences were combined in a SQT approach (Chapman 1990, 1996) to yield inferences regarding chemical impairment to benthic communities at six locations in the Lower Hackensack River and three reference locations elsewhere in Newark Bay (Table 5). Strong or possible evidence for benthic community impairment was observed in four of the six Study Area 7 stations (SA7-2, SA7-3, SA7-4, and SA7-6), and one of the three reference stations (RF-1). With the exception of SA7-4, significant toxicity was observed in toxicity tests with *L. plumulosus*. Toxicity was strongly associated with elevated concentrations of toxic and bioavailable PAHs, with possible influence of heptachlor epoxide in sediment from station SA7-6 and copper in SA7-2 and SA7-3. The strong association between benthic community impairment and chemical impacts was also found in Newark Bay by Adams *et al.* (1998), who concluded that the bay exhibited the most degraded benthos and highest levels

Table 5. Sediment quality triad inferences for the reference and Study Area 7 lines of evidence

Station	Line of evidence ^a			Triad inference ^c
	Toxic and bioavailable chemicals	Laboratory toxicity	Community impairment ^b	
<i>Reference stations</i>				
RF-1	+ Sediment ΣPAH TU > 2; [<i>M. nasuta</i> ΣPAH] > 2 μmol/g lipid	+ Severe (<i>L. plumulosus</i>); Intermediate (<i>M. nasuta</i>)	+ Slightly impaired (B-IBI); heterogeneity indices suggest highest relative impairment	Toxic chemicals (PAHs) are stressing the system and may be sufficient to significantly impair the community
RF-2	- Sediment ΣPAH TU < 1	+/- Intermediate (<i>L. plumulosus</i>)	+/- Slightly impaired (B-IBI); heterogeneity indices suggest moderate relative impairment	Evidence that there may be pollution-induced impairment, potentially due to unmeasured chemicals or conditions
RF-3	- Sediment ΣPAH TU < 1	+/- Intermediate (<i>L. plumulosus</i>)	- Unimpaired (B-IBI); heterogeneity indices suggest lowest relative impairment	Evidence that there is little or no pollution-induced impairment, although unmeasured chemicals or conditions may exist with the potential to cause impairment
<i>Study Area 7 Stations</i>				
SA7-1	+/- Sediment ΣPAH TU 1-2	+/- Intermediate (<i>L. plumulosus</i>)	- Unimpaired (B-IBI); heterogeneity indices suggest lowest relative impairment	Evidence that there is no pollution-induced impairment; however, PAHs, unmeasured conditions, or unmeasured chemicals may have the potential to cause impairment
SA7-2	+ Sediment ΣPAH TU > 2; [<i>M. nasuta</i> ΣPAH] > 2 μmol/g lipid	+ Severe (<i>L. plumulosus</i>)	+/- Slightly impaired (B-IBI); heterogeneity indices suggest moderate relative impairment	Possible evidence of pollution-induced impairment due to presence of PAHs and possibly copper
SA7-3	+ Sediment ΣPAH TU > 2; [<i>M. nasuta</i> ΣPAH] > 2 μmol/g lipid	+ Severe (<i>L. plumulosus</i>)	+ Slightly impaired (B-IBI); heterogeneity indices suggest highest relative impairment	Toxic chemicals (PAHs and possibly copper) are stressing the system and may be sufficient to significantly impair the community
SA7-4	- Sediment ΣPAH TU < 1	- Insignificant	+/- Slightly impaired (B-IBI); heterogeneity indices suggest moderate relative impairment	Possible evidence that there is impairment; however, alteration of benthic community is not due to chemicals
SA7-5	+/- Sediment ΣPAH TU 1-2	+/- Intermediate (<i>L. plumulosus</i>); Slight to Intermediate (<i>N. arenaceodentata</i> growth)	- Unimpaired (B-IBI); heterogeneity indices suggest lowest relative impairment	Evidence that there is no pollution-induced impairment; however, PAHs, unmeasured conditions, or unmeasured chemicals may have the potential to cause impairment
SA7-6	+ ΣPAH TU > 2; [<i>M. nasuta</i> ΣPAH] > 2 Σmol/g lipid; heptachlor epoxide present	+ Severe (<i>L. plumulosus</i>); Intermediate (<i>M. nasuta</i>)	+ Slightly impaired (B-IBI); heterogeneity indices suggest highest relative impairment	Toxic chemicals (PAHs and heptachlor epoxide) are stressing the system and may be sufficient to significantly impair the community

^a Line of evidence present (+), absent (-), or inconclusive (+/-)

^b Benthic-Invertebrate Index Of Biotic Integrity scores (Table 3) and four heterogeneity indices (Fig. 2) were used to provide evidence of benthic community impairment. For each of the four heterogeneity indices, stations were placed into 3 groups (low, moderate, high) according to statistical significance, with stations with the lowest values hypothesized to exhibit possible impairment

^c Based on Chapman (1990) and Chapman (1996).

of elevated chemical and incidences of toxicity anywhere in the New York/New Jersey Estuary. In only one case in which benthic impairment was suggested (station SA7-4),

impairment could not be linked to chemical impacts because of the absence of both toxicity and chemicals at concentrations in sediment or porewater indicative of toxicity.

Although an important motivation for this investigation was the presence of total chromium associated with historical industrial activities, chemical-induced impairment of biological communities and the observed sediment toxicity was most strongly associated with PAHs. There was no indication using multiple lines of evidence that total chromium contributed to toxicity to benthic organisms in the four laboratory experiments or to biological community impairment. Additional work is needed to more fully understand the sources of PAHs to the Lower Hackensack River.

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