Environmental Toxicology

EVALUATING THE RISK TO AQUATIC ECOSYSTEMS POSED BY LEACHATE FROM TIRE SHRED FILL IN ROADS USING TOXICITY TESTS, TOXICITY IDENTIFICATION EVALUATIONS, AND GROUNDWATER MODELING

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Abstract—The risk to adjacent aquatic systems posed by leachates from scrap tires used in engineering applications has not been characterized adequately. Toxicity testing, toxicity identification evaluation (TIE), and groundwater modeling were used to determine the circumstances under which tire shreds could be used as roadbed fill with negligible risk to aquatic organisms in adjacent water bodies. Elevated levels of iron, manganese, and several other chemicals were found in tire shred leachates. However, chronic toxicity tests with Ceriodaphnia dubia and fathead minnows (Pimephales promelas) showed no adverse effects caused by leachates collected from tire shreds installed above the water table. Exposure to leachates collected from tire shreds installed below the water table resulted in significant reductions to both survival and reproduction in C. dubia. The TIE results indicated that exposure to soluble metals (likely ferrous iron primarily) and the formation of iron hydroxide precipitates on this invertebrate species likely were the causes of the observed effects. The available chemistry data show that iron concentrations in the affected groundwater decreased substantially within a short distance (0.61 m) downd gradient of tire shred fill. Based on geochemical modeling, the use of tire shreds in applications below the water table is appropriate in settings where dissolved oxygen is greater than 2.0 mg/L, pH is greater than 5.8, and a downd gradient buffer of approximately 3.0 m exists between the fill and the surface water. For settings with lower dissolved oxygen concentrations or lower pH, results of groundwater modeling indicate that a greater buffer distance (~11 m) is needed to dilute the leachate to nontoxic levels under various soil and groundwater conditions solely through advection and dispersion processes.

Keywords—Tire shreds  Leachate  Toxicity testing  Groundwater modeling  Aquatic ecosystems

INTRODUCTION

In 2002, some 300 million scrap tires remained stockpiled as waste in the United States, and an additional 281 million scrap tires were generated that same year. The risk of fires and other environmental problems at scrap tire stockpile sites have prompted efforts during the past several years to develop uses for scrap tires. A substantial market currently exists for scrap tires, with approximately 77% of scrap tires generated in 2001 being reused for a beneficial application [1]. Several states fund legislative initiatives designed to reduce the existing tire stockpiles and to expand markets for scrap tires. Scrap tires are used as fuel supplements for industrial processes, ground rubber products, stamped rubber products, retreads, and civil engineering applications, with the latter experiencing the greatest growth during the last five years [1]. In 2003, 56 million scrap tires were used in this market [2]. This growth has resulted from increasing recognition of the beneficial engineering properties of tire shreds, such as low unit weight, high permeability, low thermal conductivity, and good vibration damping. Civil engineering applications include lightweight fill for highway roadbeds and embankments constructed on weak foundation soils, low-earth-pressure backfill for walls and bridge abutments, insulation to limit frost penetration beneath roads, drainage layers for landfill construction and operation, aggregate for septic field drainage systems, and vibration-damping layers beneath rail lines.

The approved uses for tire shreds vary by state, but uniform construction standards exist and are described in the American Society of Testing and Materials standard practice [3]. Some state agencies have been reluctant to endorse use of tire shreds in a greater variety of engineering applications because of the limited data regarding the risks posed by the leachates to aquatic organisms in adjacent surface-water ecosystems.

Water-quality studies indicate that tire shreds may increase levels of certain metals (particularly iron and manganese) and organic compounds in water but that these levels would remain below primary maximum contaminant levels for drinking water, so human health concerns are minimal ([4–6]; https://www.rma.org; [7]). However, the few available aquatic toxicity studies indicate that tire shred leachate reduces survival, growth, and/or reproductive success of some test organisms at concentrations between 3.1 and 83% by leachate volume [8–14]. The methods used in these studies vary, but the toxicity of leachates extracted from cut tires generally was evaluated in laboratory tests, often under static water conditions and with little consideration for the soil conditions (e.g., geochemistry and groundwater hydrology) where the scrap tires would be used. Such tests do not account for the effects of environmental processes on leachate composition and toxicity. These studies...
are useful in characterizing the chemical composition of leachates and their gross toxicity, but they do not characterize accurately the toxicity of water in contact with tire shreds during actual engineering applications. In addition, these laboratory studies do not take into account the degradation and dilution of tire shred leachate that occur as it migrates with groundwater through soils before entering open bodies of water, despite the likelihood that these processes have significant impact on the chemical composition and associated toxicity.

To our knowledge, a full examination of the effects of the leachate from an engineering application of scrap tire shreds on aquatic organisms has not been published in the scientific literature. This primary data gap precludes a thorough understanding of the ecological risks posed by tire shreds used in engineering applications above and below the water table. The present study was designed to evaluate the toxicity of leachates from tire shreds used as roadbed fill and to define the circumstances under which the use of tire shreds as roadbed fill, both above and below the water table, will pose a negligible hazard to adjacent surface-water ecosystems.

**MATERIALS AND METHODS**

Chronic toxicity tests with two freshwater species were performed using tire shred leachate collected from two previously established study sites in Maine, USA, where tire shreds are used as fill both above and below the water table. Detailed chemical analyses and toxicity identification evaluations (TIEs) were also performed on the tire shred leachates to correlate observed toxicity with specific chemicals. Geochemical modeling, groundwater fate-and-transport modeling, and infiltration modeling were used to estimate the groundwater travel distance (buffer distance) necessary to reduce the levels of toxic constituents in tire shred leachates and the ultimate effects of the leachates on aquatic organisms to negligible levels under different soil and groundwater conditions.

**Site description**

Tire shreds were installed as fill material at two study sites near the towns of North Yarmouth and Orono, Maine, USA [4–7]. The first study site (North Yarmouth) was constructed with tire shreds above the water table and the second site (Orono) with tire shreds at and below the water table. The tire shreds used in these studies have a maximum dimension of approximately 7.6 cm and are composed of a mixture of steel and glass-belted scrap tires. Substantial quantities of steel belts are exposed at the cut edges of the shreds.

The first site was constructed in August 1993 in a manner that prevents the tire shred layer from coming in contact with groundwater. Three sample collection basins are located beneath the edge of the roadway at this site (Fig. 1). Precipitation infiltrates through the road embankment and into these collection basins for subsequent sampling. The first sample collection basin was located under a portion of the roadway that does not have a tire shred layer and served as a reference (designated A-Control). The other two sample collection basins were located directly below a portion of the roadway where tire shreds are used as subgrade fill. For the present study, samples were taken from these two basins and composited to provide a single tire shred leachate sample for testing (designated A-1).

The second site was constructed in January 1994, in a configuration such that tire shreds in a trench come into direct contact with groundwater. This site represents a likely scenario for tire shreds to affect groundwater and, subsequently, the aquatic environment to which the groundwater flows. The soil type at this location is fibrous peat. Water samples were collected from three sample wells at this site (Fig. 1). The first well (designated B-Control) was located upgradient of the tire trench. The second well (designated B-1) was located within the tire shred–filled trench so that samples of groundwater in direct contact with the tire shreds could be collected. The third well (designated B-2) was located approximately 0.61 m downgradient of the trench.

**Sample collection**

Reference water (e.g., A-Control and B-Control) and leachate-affected water from each site were sampled twice. Phase 1 sampling occurred on October 9, 2000 (below water table), and on November 8, 2000 (above water table). Phase 2 sampling occurred on November 11, 2001 (below water table), and on January 1, 2002 (above water table). Samples were collected in precleaned, high-density polyethylene containers; placed in ice in insulated coolers; and shipped overnight to the testing laboratory.

**Chemical analyses**

Based on the results of previous analyses of tire shred leachates [4–14], aliquots of tire shred leachate from the two study sites were analyzed for metals (Methods 6010B, 7060A, 7470A, 7740, and 7841), volatile organic compounds (VOCs; Method 8260B), and semivolatile VOCs (SVOCs; Method 8270C) using the identified U.S. Environmental Protection Agency (U.S. EPA) preparation (metals, Methods 3010A and 3020A; organic compounds, Methods 5030 and 3510) and analytical methods [15]. Leachate sample analyses were accompanied by the analysis of method blank samples and spike recovery samples to evaluate compliance with quality-assurance/quality-control objectives, and the objectives were met in all cases.

**Toxicity testing**

On receipt at the toxicity testing laboratory, the water samples were stored in the dark at 4°C, except when being used to prepare test solutions. The toxicity of the reference water (roadbed soil leachate water or upgradient groundwater) and leachate samples was evaluated using short-term chronic toxicity tests with two representative freshwater species based on U.S. EPA methods [16]. The three-brood (6–8 d) survival and reproduction test with the crustacean *Ceriodaphnia dubia*, and the 7-d survival and growth test with larval fathead minnows (*Pimephales promelas*).

In the short-term chronic *C. dubia* test, individual females were exposed to a series of reference water and leachate dilutions (6.25, 12.5, 25, 50, and 100%) prepared with laboratory control and dilution water that consisted of a mixture of commercial spring waters. Fresh test solutions were prepared daily and were amended with appropriate amounts of the green alga *Selenastrum capricornutum* and yeast-Cerophyll-trout chow (YCT; Aquatic BioSystems, Ft. Collins, CO, USA) to provide food for the test organisms. Each test treatment had 10 replicates, with each replicate consisting of 15 ml of test solution in a 30-ml plastic cup. The tests were initiated with the random allocation of one *C. dubia* neonate (age, <24 h) into each of the replicate cups. The replicate cups were placed into a tem-
temperature-controlled water bath at 25°C. On each day of the tests, each replicate cup was examined, and surviving original individual organisms were transferred to new replicate cups containing fresh test solution, after which the number of neonate offspring produced by each original organism was determined by visual examination of the media remaining in the old replicate cup. On determination that 60% of the *C. dubia* in the control treatment had produced their third brood of offspring, the corresponding test was terminated.

In the short-term chronic fathead minnow test, larval fish were exposed to a similar series of leachate dilutions prepared with laboratory control and dilution water that consisted of synthetic U.S. EPA moderately hard water, with fresh test solutions being prepared daily. Each test treatment had four replicates, with each replicate consisting of 250 ml of test media in a 600-ml glass beaker. The tests were initiated with the random allocation of 10 larval fathead minnows (age, <24 h) into each of the replicates. These replicate beakers were placed in a temperature-controlled water bath at 25°C. The test fish were fed brine shrimp nauplii twice daily.

Each replicate was examined daily, and the number of live fish was determined, after which approximately 80% of the old water was carefully poured out and replaced with fresh test solution. After 7 d of exposure, the tests were terminated, and the number of live fish in each replicate beaker was recorded. The fish from each replicate were then killed in methanol, rinsed in deionized water, and transferred to a predried and prepared drying pan. These fish were then dried at 100°C for 24 h and reweighed to determine the total weight of fish in each replicate. Next, the total weight was divided by the
initial number of fish per replicate \((n = 10)\) to determine the biomass value.

For both tests, the resulting reference water and leachate sample response data were compared with the laboratory control water sample data and analyzed to determine the no-observed-effect concentration (NOEC), lowest-observed-effect concentration, lethal concentration for 50% of the test population, and concentration inducing a 25% reduction in reproduction or growth (IC25). All statistical analyses were performed using the ToxCalc® (TidePool Scientific, McKinleyville, CA, USA) statistical package. Reference water (field control) and leachate sample data also were compared to each other to assess the contribution to effects associated with the tire shred leachate. The survival and growth/production responses for test organisms in the laboratory control water treatments met U.S. EPA test acceptability criteria (survival of \(\geq80\%\) mean dry wt per individual larval fish at the end of the test of \(\geq0.25\) mg; 60% of \(C.\ dabia\) females must produce three broods with 15 or more offspring per female). As an additional quality-assurance/quality-control measure, reference toxicant tests were performed concurrently with the leachate tests. The reference toxicant tests were performed similarly to the leachate water toxicity tests, except that test solutions consisting of laboratory control water spiked with copper (as \(CuSO_4\)) over appropriate concentration ranges were used in place of the leachate waters. The resulting test endpoint estimates (e.g., median lethal concentration and IC25) for these reference toxicant tests fell within the typical response ranges established by the mean + two standard deviations from the 20 most recently performed tests, confirming that the test organisms used in the present study responded to toxicant stress in a typical fashion.

**Toxicity identification evaluation**

A targeted TIE, focusing primarily on nonpolar organic compounds and metals, was performed on samples that exhibited significant toxicity in the short-term chronic evaluations. The TIE consisted of a series of physical and chemical manipulations performed on separate aliquots of the leachate sample that were designed to isolate and remove a particular group of chemicals or a specific chemical from the leachate. These treated samples were then retested, as before, to evaluate changes in toxicity compared to the baseline (i.e., untreated leachate) test results. Thus, the TIE facilitated identification of the chemicals or processes responsible for the toxicity observed in the initial toxicity tests of the leachate. Several techniques were used in these targeted TIEs [17].

**TIE fractionation method blank.** As part of the TIE process, a method blank is used for each fractionation treatment to determine whether any of the fractionation procedures contributes artifactual toxicity to the manipulated sample. The fractionation method blank for this test consisted of aliquots of the laboratory control water that were subjected to each of the TIE fractionation treatments.

**Baseline test procedures.** The baseline toxicity test is performed concurrently with the TIE fractionation tests to confirm the persistence of toxicity in the untreated leachate and to provide a baseline reference value against which the toxicity removal exhibited by the TIE treatments can be compared.

**Centrifugation test procedures.** Centrifugation of the sample can affect sample toxicity through the removal of toxicants associated with suspended particulates. Two-liter aliquots of leachate were centrifuged at 4,500 g for 30 min, with 1 L of the supernatant set aside for direct testing and the remaining 1 L used in subsequent C18 solid-phase extraction (SPE) manipulation treatments. Aliquots of laboratory control water were similarly treated to serve as the corresponding method blanks.

**C18 SPE fractionation procedures.** The C18 SPE test was used to identify leachate toxicity resulting from compounds that are removed or sorbed onto chromatogram resin specific for nonpolar organic compounds (and some relatively nonpolar metal chelates). One liter of each centrifuged leachate sample was passed over a C18 SPE column. Aliquots of laboratory control water were similarly treated to serve as the corresponding method blanks.

**Ethylendiaminetetra-acetic acid test procedures.** The addition of ethylendiaminetetra-acetic acid (EDTA) to the sample can produce nontoxic complexes (via chelation) with selected cationic metals. Therefore, loss of toxicity in EDTA-treated samples indicates that cationic metals are a potential cause of toxicity. The EDTA additions were made to aliquots of each leachate sample at concentrations of 0.5, 3, and 8 mg/L. Aliquots of laboratory control water were treated similarly to serve as the corresponding method blanks.

**Sodium thiosulfate (oxidant-reduction) test procedures.** Sodium thiosulfate (STS) also can produce nontoxic complexes with some cationic metals and can be used to characterize sample toxicity caused by some metals as well as toxicity caused by residual chlorine and other oxidants in the sample. Additions of STS were made to aliquots of each leachate sample at concentrations of 1, 5, and 10 mg/L. Aliquots of laboratory control water were treated similarly to serve as the corresponding method blanks.

**Geochemical modeling**

The U.S. Geological Survey geochemical model PHREEQC Version 2 (a computer program for speciation, reaction-path, advective-transport, and inverse geochemical calculations) was used to investigate the precipitation of metals (especially iron) from tire shred leachate and the resulting leachate/groundwater mixture [18]. A set of chemical concentrations and reactions was specified to mimic water characteristics at the field sites. This model was used to confirm the results from the TIE regarding geochemical reactions controlling toxic leachate constituents and to make predictions about the relative attenuation and reduction in toxicity of tire shred leachate as it migrates with groundwater through subsurface soils to adjacent aquatic systems based solely on chemical reactions expected to occur in the groundwater environment.

**Fate-and-transport modeling**

An analytical solution of the advection-dispersion equation presented by Domenico [19] was used to predict solute concentrations in adjacent leachate-affected groundwater downgradient of tire shred fill. This model is capable of predicting the concentration of chemical species at any time and location downgradient of a contaminant source area of known size and strength based solely on dilution from advection and dispersion processes. In this model, the two parameters with the greatest impact on leachate dispersion are hydraulic gradient and hydraulic conductivity, because they relate directly to the velocity of groundwater flow and, thus, dilution. These parameters were varied independently over their range of expected values; all other parameters were held constant. The effect of rainfall
Toxicity tests and chemical analyses

Samples from above the water table. No adverse effects were observed on *P. promelas* survival or growth in either the A-Control (reference soil leachate water) or A-1 treatment groups (Table 1). Substantial reductions were observed in *C. dubia* survival in the phase 2 A-Control relative to the laboratory control water; however, no impairment of *C. dubia* survival was observed in the phase 1 A-Control or in either of the A-1 leachates (Table 1). The conductivity of the phase 2 A-Control sample was measured at 2,100 µS/cm, which was markedly higher than that in any of the other leachate samples and which may have contributed to the observed mortalities.

Elevated levels of some VOCs and metals (especially iron and manganese) were detected in samples B-1 and B-2, indicative of water quality different than the natural groundwater at this site.

Table 1. Effects of leachate-affected waters from above-the-water-table (A) collection basins and below-the-water-table (B) sample wells on the survival and growth/reproduction of *Pimephales promelas* and *Ceriodaphnia dubia*

<table>
<thead>
<tr>
<th></th>
<th><em>P. promelas</em></th>
<th>C. dubia</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Survival (% leachate)</td>
<td>Growth (% leachate)</td>
</tr>
<tr>
<td></td>
<td>LC50 [95% CL]</td>
<td>IC50 [95% CL]</td>
</tr>
<tr>
<td>Phase 1 samples</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A-Control</td>
<td>&gt;100% NA</td>
<td>&gt;100% NA</td>
</tr>
<tr>
<td>A-1</td>
<td>&gt;100% NA</td>
<td>&gt;100% NA</td>
</tr>
<tr>
<td>B-Control</td>
<td>&gt;100% NA</td>
<td>69% [57–75]</td>
</tr>
<tr>
<td>B-1</td>
<td>&gt;100% NA</td>
<td>77% [9–95]</td>
</tr>
<tr>
<td>B-2</td>
<td>&gt;100% NA</td>
<td>89% NA</td>
</tr>
</tbody>
</table>

Phase 2 samples

|                      |               |           |     |               |                       |     |
| A-Control            | >100% NA     | >100% NA | >100% | 49% NA | 28% [15–30] | 6.25% |
| A-1                  | >100% NA     | >100% NA | >100% | >100% NA | 100% NA | 100% |
| B-Control            | >100% [85–100] | 70% [58–81] | 50% | >100% NA | 70% [64–75] | 50% |
| B-1                  | >100% NA     | 91% NA | 12.5% | >100% NA | 28% [11–38] | 25% |
| B-2                  | >100% NA     | 78% NA | >100% | >100% NA | 54% [47–88] | 50% |

* NOEC = no-observable-effect concentration.
* LC50 = median lethal concentration.
* IC25 = 25% inhibitory concentration.
* NA = not applicable. Because of an absence of mortality or significant reduction in growth or reproduction (or limited number of test treatments exhibiting partial responses), the 95% CL could not be calculated.

RESULTS

Infiltration on the dilution of groundwater was incorporated using a spreadsheet model with varying precipitation rates, groundwater travel times, and soil types. The ultimate goal of the fate-and-transport modeling was to estimate the groundwater travel distance necessary to reduce leachate constituents to nontoxic levels under different soil and groundwater conditions.

Greater than 80% mortality of *C. dubia* was observed in both phase 1 100% leachate samples, with additional significant reductions in reproduction found in the 25 and 50% leachate treatments. The magnitude of the toxicity decreased in the downgradient B-2 leachate. No significant reductions in survival were observed in the phase 2 leachates; however, significant reductions in reproduction (>80% relative to the laboratory water control samples) also were found in both leachates, with the magnitude of the toxicity again being less in the B-2 leachate. These test results indicate that when below the water table, tire shred leachates can be significantly toxic to *C. dubia*, but that this toxicity is reduced as the leachate travels through as little as 0.61 m of soil. Moreover, comparing the reproduction IC25 and NOEC of B-Control to those of B-1 (Table 1) showed that a two- to fourfold dilution of the B-1 leachate would reduce the toxic effects to the same level as those of the groundwater at the site.

Elevated levels of some VOCs and metals (especially iron and manganese) were detected in samples B-1 and B-2, in-
Table 2. Chemical concentrations measured in reference and leachate-affected water collected from above-the-water-table (A) collection basins and below-the-water-table (B) sample wells.

<table>
<thead>
<tr>
<th>Chemical</th>
<th>Collection basins</th>
<th>Sample wells</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Phase 1 samples</td>
<td>Phase 2 samples</td>
</tr>
<tr>
<td></td>
<td>A-Control (mg/L)</td>
<td>A-1 (mg/L)</td>
</tr>
<tr>
<td>Inorganic</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Arsenic</td>
<td>0.009</td>
<td>&lt;0.005</td>
</tr>
<tr>
<td>Barium</td>
<td>0.08</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>Chromium</td>
<td>0.03</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Copper</td>
<td>0.03</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Iron</td>
<td>18</td>
<td>0.3</td>
</tr>
<tr>
<td>Lead</td>
<td>&lt;0.05</td>
<td>&lt;0.05</td>
</tr>
<tr>
<td>Manganese</td>
<td>0.19</td>
<td>0.91</td>
</tr>
<tr>
<td>Vanadium</td>
<td>0.03</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Zinc</td>
<td>0.26</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>Organic</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Acetone</td>
<td>&lt;0.01</td>
<td>0.014</td>
</tr>
<tr>
<td>Aniline</td>
<td>&lt;0.005</td>
<td>0.005</td>
</tr>
<tr>
<td>Benzene</td>
<td>&lt;0.0005</td>
<td>&lt;0.0005</td>
</tr>
<tr>
<td>Chloroethane</td>
<td>&lt;0.001</td>
<td>0.001</td>
</tr>
<tr>
<td>1,1-Dichloroethane</td>
<td>&lt;0.0005</td>
<td>0.0003</td>
</tr>
<tr>
<td>cis-1,2-Dichloroethene</td>
<td>&lt;0.0005</td>
<td>0.0015</td>
</tr>
<tr>
<td>Bis-(2-ethylhexyl)phthalate</td>
<td>&lt;0.0005</td>
<td>0.0005</td>
</tr>
<tr>
<td>N-nitrosodiphenylamine</td>
<td>&lt;0.0005</td>
<td>&lt;0.0005</td>
</tr>
<tr>
<td>Toluene</td>
<td>&lt;0.0005</td>
<td>&lt;0.0005</td>
</tr>
<tr>
<td>Trichloroethene</td>
<td>&lt;0.0005</td>
<td>&lt;0.0005</td>
</tr>
<tr>
<td>Trichlorofluoromethane</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>
During the TIE testing (particularly during the C18 SPE treatment), no toxicity was removed. No toxicity was removed by either the EDTA or STS treatments. This pattern of toxicity removal suggests that the precipitate and/or contaminants associated with the precipitate are contributing to the observed toxicity. The precipitation of iron on organism and cell surfaces and its behavior in pore waters of suboxic estuarine sediments [20–22]. The formation of ferric EDTA is ethylenediaminetetraacetic acid. STS is sodium thiosulfate.

**Table 3. Summary of results for chronic toxicity identification evaluation with Ceriodaphnia dubia performed on phase 1 B-1 and B-2 leachate samples**

<table>
<thead>
<tr>
<th>Fractionation</th>
<th>Laboratory control</th>
<th>Leachate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Survival (%)</td>
<td>Mean no. of offspring</td>
</tr>
<tr>
<td><strong>B-1</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Baseline sample</td>
<td>90</td>
<td>28.4</td>
</tr>
<tr>
<td>Centrifugation</td>
<td>100</td>
<td>29.9</td>
</tr>
<tr>
<td>C18 SPE</td>
<td>90</td>
<td>26.3</td>
</tr>
<tr>
<td>EDTA (0.5 mg/L)</td>
<td>100</td>
<td>21.6</td>
</tr>
<tr>
<td>EDTA (3.0 mg/L)</td>
<td>100</td>
<td>30.5</td>
</tr>
<tr>
<td>EDTA (5.0 mg/L)</td>
<td>100</td>
<td>21.0</td>
</tr>
<tr>
<td>STS (1 mg/L)</td>
<td>100</td>
<td>28.6</td>
</tr>
<tr>
<td>STS (5 mg/L)</td>
<td>100</td>
<td>25.9</td>
</tr>
<tr>
<td>STS (20 mg/L)</td>
<td>90</td>
<td>29.1</td>
</tr>
<tr>
<td><strong>B-2</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Baseline sample</td>
<td>100</td>
<td>20.0</td>
</tr>
<tr>
<td>Centrifugation</td>
<td>90</td>
<td>21.4</td>
</tr>
<tr>
<td>C18 SPE</td>
<td>90</td>
<td>14.3</td>
</tr>
<tr>
<td>EDTA (0.5 mg/L)</td>
<td>100</td>
<td>18.0</td>
</tr>
<tr>
<td>EDTA (3.0 mg/L)</td>
<td>100</td>
<td>15.7</td>
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<tr>
<td>EDTA (5.0 mg/L)</td>
<td>100</td>
<td>16.8</td>
</tr>
<tr>
<td>STS (1 mg/L)</td>
<td>100</td>
<td>16.0</td>
</tr>
<tr>
<td>STS (5 mg/L)</td>
<td>100</td>
<td>14.1</td>
</tr>
<tr>
<td>STS (20 mg/L)</td>
<td>100</td>
<td>17.6</td>
</tr>
</tbody>
</table>

*a* The letter B refers to the below-the-water-table samples.

*b* SPE = solid-phase extraction.

*c* EDTA = ethylenediaminetetraacetic acid.

*d* STS = sodium thiosulfate.

dicating that these chemicals do leach from tire shred fill below the water table (Table 2). Also, the general decrease in the number of chemicals detected and in the concentration of detected chemicals, particularly iron, from B-1 to B-2 indicates that attenuation and degradation mechanisms are acting on the tire shred leachate as it migrates in the groundwater dendrin to the leachate. These changes in the chemistry of the leachate-affected groundwater indicate that these mechanisms are acting on the groundwater within a short distance (~0.61 m) of the fill material.

**Toxicity identification evaluation.** Because reduced survival and reproduction were observed for *C. dubia* exposed to the phase 1 B-1 and B-2 leachate samples, a TIE was performed on these samples. As preliminary information that will be useful in interpreting the results of TIEs, it is important to note certain characteristics of samples before and during testing. In the present study, the B-1 and B-2 samples exhibited low pH and low dissolved oxygen levels, and they were characterized by a very-fine-grained, rust-colored precipitate at the time of sample receipt. Additional rust-colored precipitate was formed in the 100% leachate samples during the toxicity tests and during the TIE testing (particularly during the C18 SPE treatment), with noticeably more precipitate formed in the B-1 leachate than in the B-2 leachate. In addition to observations about sample characteristics, a coating of the very-fine-grained, rust-colored precipitate was observed on the external surfaces of *C. dubia* and appeared to be associated with both survival and reproductive toxicity.

A summary of the results of the chronic TIE performed on the B-1 and B-2 samples is presented in Table 3. No single process successfully removed all toxicity from sample B-1, implying that more than one toxicant was responsible for the effects observed. Centrifugation and STS both removed essentially all the survival toxicity, and EDTA removed most (but not all) of the effects on survival. However, additional reproductive toxicity (with a virtual absence of any reproduction at any of the test treatments) occurred, and none of the TIE treatments was fully effective in ameliorating this toxicity. The pattern of TIE treatments on reducing mortality suggests the following: The precipitate and/or contaminants associated with the precipitate are contributing to the observed toxicity to *C. dubia* survival; multiple metals/metalloids are present, one of which is removable by both EDTA and STS and one of which is removable only by STS, that contribute to the observed toxicity; and one of the previously mentioned stressors, in the absence of the other(s), or another, unidentified contaminant is still causing the reproductive toxicity.

The effects on survival that were observed previously in the initial test of the 100% B-2 leachate (i.e., 80% mortality) were diminished (reduced to 30% mortality) by the time of the TIE; however, reproduction was still completely absent in the leachate. Centrifugation was able to remove all the observed toxicity (effects on both survival and reproduction), although some effects on reproduction were observed during the subsequent C18 SPE treatment. No toxicity was removed by either the EDTA or STS treatments. This pattern of toxicity removal suggests that the precipitate and/or contaminants associated with the precipitate are contributing to the observed effects on *C. dubia*.

The results of the TIE performed on the B-1 leachate, along with the chemical analyses of leachate waters, implicate iron precipitates and soluble metals (primarily ferrous iron) as likely causes of the observed toxicity to *C. dubia*. Similarly, the TIE of the B-2 leachate implicates iron precipitates as a cause of toxicity, although the magnitude of the toxicity has been greatly diminished, reflecting the changes in the chemical matrix of the B-2 leachate. These results are consistent with those reported in the literature regarding the precipitation of iron on organism and cell surfaces and its behavior in pore waters of suboxic estuarine sediments [20–22]. The formation of ferric
by applying principles of chemical thermodynamics and mass distribution of chemical elements among aqueous species, sur-

The NOEC for bioavailable and toxic form of iron to aquatic organisms [23].

iron precipitates also reduces the available concentrations of ionic ferrous iron in solution, and ferrous iron is the most bioavailable and toxic form of iron to aquatic organisms [23]. The NOEC for \textit{C. dubia} reproduction is reported to be approximately 0.4 mg/L in waters where ferrous iron constituted 10 to 20% of the total iron measured [23].

Identifying a mechanism of toxicity is helpful in determining the potential for natural attenuation in the environment. Tire shreds used as roadbed fill likely will be placed very near the surface of the water table, and the leachate-affected groundwater, after migrating out of the fill material, likely will be exposed to significant amounts of dissolved oxygen. The mixing of the iron-rich suboxic leachate with oxygenated groundwater will oxidize some of the aqueous iron and produce iron hydroxide precipitates, with a concomitant adsorption of other metals to the iron hydroxide matrix [24]. The addition of oxygen to that already dissolved in site groundwater through a minimal amount of mixing, diffusion, and infiltration was sufficient to precipitate and filter out a substantial amount of ferrous iron within a short distance of the tire shreds. The effects of the oxidation-reduction and sorption reactions are illustrated by the significant drop in concentrations of iron and other metals and recovery of dissolved oxygen concentrations in the B-2 samples collected just 0.61 m downgradient of the fill material relative to concentrations in the samples from the tire shred fill (see Tables 2 and 4). Similar trends were noted in other water-quality parameters in these samples, with pH increasing in B-1 and declining in B-2 and hardness and alkalinity decreasing in B-1 and rebounding in B-2.

During phase 1, total iron concentrations declined from 80 mg/L in B-1 to 6.1 mg/L in B-2, whereas dissolved oxygen increased from 0.8 to 3.1 mg/L. During phase 2, total iron concentrations in groundwater samples dropped from 5.1 mg/L in B-1 to 2.4 mg/L in B-2, and dissolved oxygen increased from 1.6 to 3.7 mg/L. One would expect, therefore, that after some additional travel distance, the concentrations of iron and metals in leachate-affected groundwater would be reduced below levels that would produce harmful precipitates or ferrous iron concentrations in surface-water systems. Toxicity test results suggest that a reduction in total iron concentrations to approximately 1 mg/L was necessary to achieve no-effect levels.

\textit{Geochemical modeling}

Equilibrium geochemical models attempt to explain the distribution of chemical elements among aqueous species, surface-complexed species, and mineral phases in water samples by applying principles of chemical thermodynamics and mass balance. Such models have been applied successfully to solve problems in a variety of disciplines, including contaminant hydrology, ore geochemistry, and soil science. The PHREEQC model was used to verify the hypothesis that the formation of iron precipitates and the removal of a large fraction of aqueous iron from leachate-affected groundwater were likely to occur in most geochemical settings.

Selected water-quality parameters measured during the sampling events are shown in Table 4. Hardness reflects the combined concentrations of Ca\textsuperscript{2+} and Mg\textsuperscript{2+} ions. Although many compounds can be associated with alkalinity, in this system alkalinity reflects the combined concentrations of HCO\textsubscript{3}\textsuperscript{−} and CO\textsubscript{3}\textsuperscript{2−} species [25]. During both sampling events, the following changes in water chemistry were observed: Depletion of dissolved oxygen between the B-Control and B-1 samples, followed by a subsequent recovery in B-2; a slight rise in pH between the B-Control and B-1 samples, followed by subsequent decline in B-2; and a precipitate of iron forming, with subsequent reduction in aqueous iron concentrations.

Qualitatively, these observations are explained by the following reactions [26]. First, within groundwater that is in contact with steel belts in the tire shreds, the metallic iron is converted to ferrous iron, whereas both H\textsuperscript{+} and O\textsubscript{2} are consumed:

\[
2\text{Fe}(s) + 4\text{H}^+ + \text{O}_2 \rightarrow 2\text{Fe}^{2+} + 2\text{H}_2\text{O} \tag{reaction 1}
\]

The soluble ferrous iron likely will be carried in groundwater out of the tire shred fill until conditions become oxidizing, at which time it will convert to ferric iron and precipitate as a hydroxide:

\[
4\text{Fe}^{2+} + 10\text{H}_2\text{O} + \text{O}_2 \rightarrow 4\text{Fe(OH)}_3(s) + 8\text{H}^+ \tag{reaction 2}
\]

Because of the limited solubility of atmospheric oxygen in water, the oxidation of significant quantities of metallic iron (>20 mg/L) will exhaust the oxygen in solution, which will allow the remaining soluble ferrous iron to migrate until sufficient oxygen is available from infiltration or mixing to convert it to ferric hydroxide. Reaction 1 will decrease the oxygen concentration and increase the pH by consuming H\textsuperscript{+} ions, whereas reaction 2 will also consume oxygen but lower the pH by producing H\textsuperscript{+} ions. This analysis is supported by the actual field data for site B in Table 4. As the iron concentration increases at B-1, the dissolved oxygen levels decrease, and the pH increases. However, as oxygen increases at B-2, the iron levels decrease along with the pH.

Three comparable chemical systems were simulated and evaluated using the PHREEQC model: One assuming parameters similar to those measured, one representing a carbonate-

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Fe (mg/L)</th>
<th>Dissolved O\textsubscript{2} (mg/L)</th>
<th>pH</th>
<th>Alkalinity\textsuperscript{a} (mg/L)</th>
<th>Hardness\textsuperscript{a} (mg/L)</th>
<th>Conductivity (\textmu S/cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phase 1 samples</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B-Control\textsuperscript{b}</td>
<td>1.4</td>
<td>4.9</td>
<td>6.41</td>
<td>145</td>
<td>95</td>
<td>277</td>
</tr>
<tr>
<td>B-1</td>
<td>80</td>
<td>0.8</td>
<td>6.63</td>
<td>79</td>
<td>38</td>
<td>262</td>
</tr>
<tr>
<td>B-2</td>
<td>6.1</td>
<td>3.1</td>
<td>6.25</td>
<td>108</td>
<td>70</td>
<td>238</td>
</tr>
<tr>
<td>Phase 2 samples</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B-Control</td>
<td>1.2</td>
<td>4.7</td>
<td>6.75</td>
<td>142</td>
<td>114</td>
<td>279</td>
</tr>
<tr>
<td>B-1</td>
<td>51</td>
<td>1.6</td>
<td>6.79</td>
<td>26</td>
<td>56</td>
<td>178</td>
</tr>
<tr>
<td>B-2</td>
<td>2.4</td>
<td>3.7</td>
<td>6.46</td>
<td>87</td>
<td>78</td>
<td>183</td>
</tr>
</tbody>
</table>

\textsuperscript{a} As CaCO\textsubscript{3} equivalent.

\textsuperscript{b} B-Control samples collected just 0.61 m downgradient of the fill material relative to concentrations in samples from the tire shred fill.
rich system, and one representing an organic-rich system. In all three systems, iron tended to precipitate rapidly as Fe(OH)$_3$, FeCO$_3$, and/or FeS, resulting in lower aqueous concentrations of iron a short distance downgradient. This precipitation of iron was accompanied by an increase in pH and a temporary reduction in dissolved oxygen. These results are consistent with the chemical reactions expected in these types of systems, and they reinforce the hypothesis that iron precipitates likely would form as a result of geochemical reactions of tire shred leachates with groundwater. In addition, the formation of these precipitates implies a reduction in the amount of aqueous iron in groundwater as it travels away from the emplaced tire shreds and an associated reduction in the toxicity of the leachate-affected water. From the geochemical modeling results, it can be generalized that the use of tire shreds in applications below the water table is appropriate in settings with dissolved oxygen greater than 2.0 mg/L, pH greater than 5.8, and a downgradient buffer of approximately 3.0 m between the fill and the surface water.

**Fate-and-transport modeling**

*Three-dimensional analytical model.* The movement of tire shred leachate in the subsurface was investigated using dispersion and infiltration modeling to determine the level of dilution that could be expected under typical groundwater flow scenarios and to estimate the groundwater travel distance over which the leachate is diluted to nontoxic levels. An analytical solution of the advection-dispersion equation presented by Domenico [19] was used to predict solute concentrations adjacent to an installed tire shred layer. Hydraulic gradient and hydraulic conductivity were varied independently; all other parameters were held constant. The hydraulic conductivity values chosen are representative of the range of values that might be expected in aquifers that consist of materials ranging from coarse sand to fine silt. Cases with aquifers consisting of tightly packed tills or clays were not considered, because flow velocities in these types of soils are very slow and appreciable amounts of leachate are not expected to migrate as groundwater from the tire shred fill.

To produce conservative estimates of leachate concentrations, retardation effects and chemical reactions were not allowed to slow leachate migration; thus, the first-order decay constant was set to zero. The three-dimensional groundwater model produces chemical concentration estimates at multiple, user-specified distances and depths from the source. Only leachate concentrations at the aquifer surface along the centerline of the leachate source at 1.5-m intervals were considered in this analysis. These leachate concentrations represented the maximum estimated values in the three-dimensional model space.

*Toxicity criterion.* The toxicity testing described previously indicates that the leachate is rendered nontoxic as dilution increases. The dilution of leachate reduces toxicity by reducing concentrations of chemical constituents and facilitating the precipitation of iron and other metals. The most sensitive endpoint was *C. dubia* reproduction. In the most conservative case, if the leachate was diluted to constitute less than 12.5% (Table 1) of the water matrix, the reproductive success of the test population was unaffected by the exposure (i.e., no statistically significant difference from the control population response was observed). As noted previously, the natural groundwater at the site affected reproductive success; thus, the 12.5% dilution criteria is, indeed, conservative for the field site below the water table. The goal of this modeling exercise was to estimate the groundwater travel distance from the tire shred–filled roadbed to where leachate constitutes less than 12.5% of total groundwater volume (i.e., the dilution threshold).

*Modeling estimates.* Nine scenarios were evaluated: Coarse sand with low, medium, and high hydraulic gradients; fine sand with low, medium, and high hydraulic gradients; and silt with low, medium, and high hydraulic gradients. The hydraulic conductivity, gradient, and distances at which the leachate concentration decreases below the 12.5% threshold for each scenario are presented in Table 5. It is apparent from the modeling results that leachate concentrations generally are below the dilution threshold of 12.5% within a relatively short distance of the tire shreds. Four of the nine scenarios indicated that sufficient dilution would occur within 1.5 m of the tire shred–filled roadbed to render the affected groundwater nontoxic. For four of the remaining five scenarios in which the dilution threshold was exceeded, the threshold was met after the groundwater had traveled approximately 12 m from the fill. Scenario 5, a fine sand soil with medium to low hydraulic gradient, represented the one case in which groundwater must travel a considerable distance (~50 m) before meeting the leachate dilution threshold.

*Source thickness.* During initial model runs, it was evident that the depth below the water table to which tire shreds are placed (i.e., source thickness) has an effect on the subsequent downstream leachate concentrations. For most projects, as much of the tire shred fill as possible will be installed above the water table to improve slope stability and reduce embankment settlement. A majority of applications likely will require

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### Table 5. Distances at which leachate dilution threshold is met for nine scenarios based on groundwater modeling

<table>
<thead>
<tr>
<th>Scenario(^a)</th>
<th>Aquifer material</th>
<th>Hydraulic conductivity (m/d)</th>
<th>Hydraulic gradient (m/m)</th>
<th>Distance (m)</th>
<th>Leachate concentration (% leachate)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Coarse sand, high gradient</td>
<td>85</td>
<td>0.1</td>
<td>1.5</td>
<td>0.76</td>
</tr>
<tr>
<td>2</td>
<td>Coarse sand, medium gradient</td>
<td>85</td>
<td>0.01</td>
<td>1.5</td>
<td>6.59</td>
</tr>
<tr>
<td>3</td>
<td>Coarse sand, low gradient</td>
<td>85</td>
<td>0.001</td>
<td>7.6</td>
<td>11.51</td>
</tr>
<tr>
<td>4</td>
<td>Fine sand, high gradient</td>
<td>0.85</td>
<td>0.1</td>
<td>7.6</td>
<td>11.51</td>
</tr>
<tr>
<td>5</td>
<td>Fine sand, medium gradient</td>
<td>0.85</td>
<td>0.01</td>
<td>50</td>
<td>12.38</td>
</tr>
<tr>
<td>6</td>
<td>Fine sand, low gradient</td>
<td>0.85</td>
<td>0.001</td>
<td>12</td>
<td>9.25</td>
</tr>
<tr>
<td>7</td>
<td>Silt, high gradient</td>
<td>0.0085</td>
<td>0.1</td>
<td>12</td>
<td>9.25</td>
</tr>
<tr>
<td>8</td>
<td>Silt, medium gradient</td>
<td>0.0085</td>
<td>0.01</td>
<td>1.5</td>
<td>1.84</td>
</tr>
<tr>
<td>9</td>
<td>Silt, low gradient</td>
<td>0.0085</td>
<td>0.001</td>
<td>1.5</td>
<td>0.00</td>
</tr>
</tbody>
</table>

\(^a\) Constant model parameters are as follows: Initial source concentration, 1,000 mg/L; porosity, 0.4; bulk soil density, 1.7 g/cm\(^3\); first-order decay constant, 0.0 d\(^{-1}\); width of source area, 30 m; depth of source area, 0.61 m; duration of groundwater flow, 3,650 d.
Infiltration parameters are as follows: Annual rainfall, 0.64 m/year; aquifer thickness, 5 m; aquifer porosity, 0.4; groundwater velocity, 0.22 m/d; and groundwater travel distance, the ratio varies between 1 and 2 for the 1.2-m case. The ratio is highest in scenarios with coarse to fine sandy soils, and hydrologic soil group 3 is representative of silty soils. The results of modeling a conservative, nonreactive tracer indicate that the most conservative modeling assumptions, dispersion and infiltration will reduce leachate concentrations to below levels that may be detrimental to C. dubia reproduction within 11 m of a tire shred layer installed in contact with groundwater for eight of the nine scenarios analyzed. In scenario 5 (groundwater velocity, 0.022 m/d), the travel distance necessary to reduce leachate concentration to less than 12.5% was 32 m. Aquifers with higher groundwater velocities will produce sufficient dispersion to dilute the leachate to nontoxic levels within 7.6 m of the roadbed. Settings with lower groundwater velocities provide a longer time for infiltration, and tire shred leachate is diluted below the threshold within 11 m of the tire sheds. Settings with extremely low hydraulic conductivities (e.g., glacial tills with significant fines or clays) are expected to provide a very-low-permeability barrier that precludes significant migration of the leachate-affected groundwater.

DISCUSSION

The present study reinforces the value of integrating toxicity testing with geochemical and groundwater fate-and-transport modeling to evaluate the risks of leachates from scrap tire shreds used as engineering fill to aquatic organisms in adjacent waterbodies. The short-term chronic tests used here to measure leachate effects are relatively sensitive and the same as those used to evaluate effluent toxicity. Because both the tests with

![Graph](image)

**Fig. 2.** The effects of doubling (from 0.6 to 1.2 m) the thickness of the tire shred fill located below the water table on the ratio of leachate concentrations expected in groundwater downgradient of the fill.

<table>
<thead>
<tr>
<th>Original leachate concentration (% leachate)</th>
<th>HSG 3 and 2 dilution ratio (infiltration volume:groundwater volume)</th>
<th>Infiltration-augmented leachate concentration of HSG 3 and 2 (% leachate)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenarios 3 and 4 at 6.1 m</td>
<td>13.5</td>
<td>0.010:1 to 0.024:1</td>
</tr>
<tr>
<td>Scenario 5 at 32 m</td>
<td>18.8</td>
<td>0.56:1 to 1.25:1</td>
</tr>
<tr>
<td>Scenarios 6 and 7 at 11 m</td>
<td>18.5</td>
<td>1.37:1 to 3.08:1</td>
</tr>
</tbody>
</table>

*Infiltration parameters are as follows: Annual rainfall, 0.64 m/year; aquifer thickness, 5 m; aquifer porosity, 0.4; groundwater velocity, 0.22 m/d (scenarios 3 and 4), 0.022 m/d (scenario 5), and 0.0022 m/d (scenarios 6 and 7). HSG = hydrologic soil group.
effluents and leachates (as in this case) are designed to evaluate the toxicity of mixtures entering surface-water ecosystems, these tests provide representative estimates of the potential effects on fish and invertebrates and of a dilution threshold below which the likelihood of effects is negligible. In the present study, the tests showed no significant effects from exposures to the above-the-water-table leachates but significant effects on C. dubia from exposures to the below-the-water-table leachates. A NOEC protective against effects on survival and reproduction of 12.5% was identified as a dilution threshold for these leachates.

The TIE results suggest that the observed toxicity likely was related to metals (particularly ferrous concentrations and iron hydroxide precipitates), and the water chemistry data, specifically the reduction in aqueous iron concentrations (and other potentially toxic metals) immediately downgradient of the fill material, indicate that natural processes are precipitating iron complexes (and other sorbed metals) and reducing the toxicity of the leachate-affected groundwater as it migrates. Geochemical and groundwater fate-and-transport modeling, in conjunction with the dilution threshold, allowed estimates to be made of the groundwater travel distance necessary in various soil conditions to mitigate the toxicity of the leachate-affected water.

Undiluted leachate produced by tire shreds placed below the water table is toxic to C. dubia, likely because of the release of iron from steel belts exposed at cut edges of the tire shreds resulting in high aqueous iron concentrations. If conditions are highly anaerobic and acidic, such that high iron levels remain in groundwater entering adjacent surface-water ecosystems, the toxicity test results indicate that the formation of iron oxide complexes on zooplankton species and/or high ferrous iron concentrations in the affected water could compromise their survival and reproduction. However, this unique effect is unlikely to occur under most conditions, because the iron in leachate-affected groundwater quickly oxidizes and forms insoluble, immobile particles in subsurface soils within a short distance of the tire shreds.

In general, the use of tire shreds under saturated conditions will result in only a highly localized area of high ferrous iron concentrations. Natural dilution effects and geochemical reactions will promote the formation of stable iron hydroxides, reducing ferrous iron concentrations within a short distance of the tire shreds. Only aquifer conditions where the native groundwater is highly anaerobic and acidic would result in a significant migration of aqueous iron from the leaching of the tire shreds in below-the-water-table applications.

From the geochemical modeling results, it can be generalized that the use of tire shreds in below-the-water-table applications is appropriate in settings where dissolved oxygen is greater than 2.0 mg/L, pH is greater than 5.8, and a downgradient buffer of at least 3 m exists between the fill and the surface water. The below-the-water-table applications of tire shreds in lower dissolved oxygen concentration and pH conditions should be evaluated on a case-by-case basis using dilution and infiltration modeling results. In the absence of geochemical conditions necessary to precipitate iron out of solution, dispersion and dilution of tire shred leachate remain the primary attenuation processes available to reduce leachate toxicity. In all but one of the soil and groundwater scenarios evaluated, sufficient dispersion and dilution would be expected to occur within 11 m of the emplaced tire shreds to render leachate nontoxic to exposed aquatic organisms. In the worst-case fate-and-transport scenario modeled (scenario 5 [fine sand, medium gradient]), dispersion and dilution would likely render tire shred leachate nontoxic to aquatic organisms within 32 m of the tire shred fill.

This evaluation provided a quantitative assessment of the potential ecological risks posed by an engineering application of scrap tires. The approach taken was to integrate the results of toxicity tests, TIEs, chemical analyses, and modeling to identify the environmental conditions under which scrap tires could be used with negligible risk to fish and invertebrates in adjacent aquatic ecosystems. This approach appears to be applicable to the evaluation of various man-made materials that leach and may be used in engineering applications near aquatic ecosystems.

Acknowledgement—This research was funded, in part, by the California Integrated Waste Management Board.

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