TITLE: Leaching of Aluminum and Chromium from Steel Slag Embankments into Surface Waters

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ABSTRACT

Large quantities of by-products are produced annually during the process of steel/iron production. Steel slag is one of these by-products. There is a growing interest in reusing steel slag in highway structural fills although concerns have been raised for its reuse due to its high pH and metal leaching behavior under alkaline conditions. In case steel slag material is used in a highway embankment fill, the percolating rainwater would either leach through the underlying subgrade and merge with groundwater, or it would come out of the embankment and leach into the nearby surface waters such as rivers and streams. In order to evaluate the surface water impacts of trace metal leaching from both virgin and treated steel slag samples, an experimental study was conducted through a series of sequential water leach tests and sequential column leach tests. In addition, the extent of surface water contamination in typical highways was simulated with a recently developed numerical model. The results indicate that encapsulating the pure or treated steel slag material within a clay soil decreases the metal concentrations within the effluent leachates significantly. Moreover, the results of the numerical model analysis showed that once the slag leachate merges with the surface waters, the concentrations decrease further with increasing distance from embankment corner.

INTRODUCTION

The American Society of Civil Engineers estimates that nearly $3 trillion is needed over a five-year period to bring the nation’s infrastructure to a good condition (2011). As a result, there is a policy shift at the state level aimed at substantially increasing the use of waste materials in geotechnical construction. Steel slag generation in the U.S. is estimated to be in 13 million metric tons, and nearly 40% of the total slag is currently being disposed in landfills or lagoons, occupying valuable land space (USGS 2015, Yildirim and Prezzi 2011).

Use of steel slag in highway structural fills (i.e., embankments) is an efficient way to recycle large amounts of material in engineering applications. Steel slag has favorable mechanical properties, such as good compatibility and high friction angle, however it is prone to volumetric expansion when it gets into contact with water (Dayioglu et al. 2014). In addition, another key issue that has prevented its large-volume use in highway construction is the surface water impacts caused by metals in steel slag (SS) leachate. The SS is mainly composed of environmentally benign oxides of calcium, silicon and iron, but it may also contain low concentrations of potential toxic elements, including chromium and aluminum. The high calcium and iron levels create an environment that is generally conducive to sequestering these pollutants; however, as rainwater percolates through an embankment profile, trace elements leach from SS and may reach rivers and streams.

Limited information exists on the pollutant characteristics of steel slag in surface waters during construction of highway embankments. Boyer (1994) has conducted a case study in Baltimore, Maryland where a slag bed 3 to 7 m thick sandwiched within an embankment was resulting in effluent leachates with a pH of 12.5 to 13, which resulted in a necessity of a fenced enclosure and containment pond construction. Banks et al. (2006) studied the use of engineered constructed wetlands to mitigate the impacts of the slag leachate. A detailed hydrogeochemical analysis of different steel slag surface discharges in United Kingdom indicated that the CO$_2$ in the atmosphere affects the pH of the surface water-leachate mixture significantly and rapid calcite precipitation is observed in all slag leachates upon emergence (Mayes et al. 2008).
pH conditions and pollutant leaching potential are highly influenced by the intended slag application and the possible use of simple additives to mitigate adverse properties. Understanding these issues, and finding ways to reduce environmental risks are necessary to promote the safe and beneficial reuse of steel slag in construction in Maryland. Use of steel slag has several benefits, including reduction in solid waste disposal costs incurred by industry, reduction in landfill requirements, minimization of damage to natural resources caused by excavating earthen materials for construction, obtaining added value from waste materials, conservation of production energy, and ultimately providing sustainable construction and economic growth.

In this study, leaching of Aluminum and Chromium from pure or treated steel slag embankments into surface waters was studied in case the percolating rainwater leaves the corner of the embankment and merges with the nearby rivers and streams. For this purpose, a series of laboratory leaching tests are conducted. In addition, to obtain the concentration within the surface waters with respect to distance from embankment corner, a numerical model called UMDSurf was utilized.

MATERIALS

The steel slag (SS) material used in this study was produced via blast oxygen furnace (BOF). Only 2-year-old (2YS) steel slag was used in laboratory tests to control the effects of aging. Debris and foreign materials were removed by hand before testing. Mechanical properties of the material are provided in Dayioglu et al. (2014).

Water treatment residual (WTR) was amended to steel slag to decrease the effluent pH and trace metal concentrations within the effluent leachate. In addition, Dayioglu et al. (2014) showed that WTR addition decreases the ultimate swelling ratio of the steel slag significantly. WTR is a by-product of drinking water treatment, and does not contain any surface or groundwater pollutants. The water treatment residual (WTR) was collected from the City of Rockville Water Treatment Plant, and air-dried and sieved through U.S. No. 40 sieve before being mixed with steel slag. The measurements conducted, according to EPA Method SW-846 Method 9045, showed that the pH of SS and WTR materials were 12.4 and 6.9, respectively. WTR was added at various percentages by weight to the SS material.

A clayey, or borrow material, commonly used by SHA was utilized for simulating an encapsulation layer around pure or treated SS materials in embankments. The borrow material was classified as low plasticity clay (CL), according to Unified Soil Classification System (USCS). All mixtures prepared in the current study were compacted at their optimum moisture contents \( (w_{\text{opt}}) \) and maximum dry unit weights \( (\gamma_{\text{dry-max}}) \) using the standard Proctor effort (ASTM D698). Total elemental analyses (TEA) were performed using an inductively coupled plasma optical emission spectrometer (ICP-EOS) by the University of Wisconsin Soil and Plant Analysis Laboratory on all materials to aid in selecting the metal analytes of interest. Table 1 provides a summary of the physical properties as well as the Al and Cr composition of the materials used in this study.

SS material was mixed with different percentages by weight WTR. All mixtures prepared in the current study were compacted at 2% dry side of the optimum moisture contents \( (w_{\text{opt}}) \) and maximum dry unit weights \( (\gamma_{\text{dry-max}}) \) using the standard Proctor effort (ASTM D 698). SS was
sieved through a 9.5-mm sieve before it was mixed with WTR in accordance with ASTM D4874.

Table 1. Physical properties and Al and Cr composition of the steel slag, water treatment residual and encapsulation soil

<table>
<thead>
<tr>
<th>Material</th>
<th>Gs</th>
<th>Ip (%)</th>
<th>w_{opt} (%)</th>
<th>γ_{dry-max} (kN/m^3)</th>
<th>pH</th>
<th>Al (mg/L)</th>
<th>Cr (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Steel Slag (SS)</td>
<td>3.45</td>
<td>NP</td>
<td>13.5</td>
<td>22.2</td>
<td>12.4</td>
<td>10,600</td>
<td>1300</td>
</tr>
<tr>
<td>Water Treatment Residual (WTR)</td>
<td>1.88</td>
<td>NP</td>
<td>42</td>
<td>10.3</td>
<td>6.9</td>
<td>159,700</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>Encapsulation Soil (ENC)</td>
<td>2.66</td>
<td>18</td>
<td>26</td>
<td>16.4</td>
<td>5.9</td>
<td>47,700</td>
<td>37.1</td>
</tr>
</tbody>
</table>

Note: Ip: plasticity index, Gs: specific gravity NP: non-plastic, w_{opt}: optimum moisture content, γ_{dry-max}: maximum dry unit weight.

METHODS

Sequential Water Leach Tests (SWLT)

Sequential batch water leach tests (SWLT) were conducted on pure SS and SS-WTR mixtures in accordance with ASTM D3987 with slight modifications mentioned below. Pure SS and pure WTR were also tested as control materials. The specimens were prepared at a liquid-to-solid ratio (L:S) of 20:1. All materials were air-dried and sieved through the No. 10 (2.0 mm) sieve before use. A 0.02 M NaCl solution was added to 2.5 grams of dry mixtures in 50 mL high-density polyethylene (HDPE) bottles. Next, the solutions were rotated at a rate of 29 rpm at room temperature (24 Cº) for 18 hours, in accordance with ASTM D3987. After rotation, the samples were allowed to sit for five minutes and centrifuged at 6000 rpm for 15 minutes. Upon centrifugation, the suspended solids were filtered through the 0.2-μm pore size, 25 mm diameter membrane disk filters fitted in a 25-mm Easy Pressure syringe filter holder by using a 60-mL plastic syringe. The pH and electrical conductivity (EC) measurements were conducted, and the volume of the filtrated solution was measured.

To simulate the flow of leachate through an encapsulating clay layer, the leachate solution from the first water leach test was put into a centrifuge tube and dry clay was added to maintain an L:S of 20:1. The samples were rotated for another 18 hours and the steps mentioned above were repeated. After the pH and EC measurements were taken, the samples were acidified to pH <2 with 2% HNO_3. Before use, all equipment (centrifuge tubes, filter holders syringe, etc.) were washed with 2% HNO_3 acid solutions and rinsed with deionized (DI) water. All samples were stored at 4 Cº for chemical analysis. Duplicate SWLTs were conducted on all mixtures. Test setup for SWLT is shown in Figure 1.
Sequential Column Leach Tests (SCLT)

In order to simulate the encapsulation of pure or treated SS materials in a clayey soil, sequential column leach tests (SCLT) were conducted. In these tests, the effluent tubing of the first column, which contains pure or treated SS materials, is connected to the second column that houses the encapsulating clay layer. Effluent samples were collected from both columns in order to compare the pH, EC and metal concentrations. The SCLT setup is shown in Figure 2.

The SCLTs were conducted on pure SS and SS-WTR mixtures. All specimens were compacted at their optimum moisture contents in a PVC mold with a 101.6-mm diameter and 116.4-mm height using standard Proctor effort (ASTM D698). In order to increase the total solid surface area in the solid matrix, air-dried steel slag was sieved over a 9.5 mm sieve to remove larger particles. PVC molds were preferred because it minimizes the outside effects on effluent metal concentrations.

The columns were operated in an upflow mode using a peristaltic pump on the influent line. The polypropylene (PP) influent lines were connected to a polyethylene reservoir tank, which was filled with the 0.02 M NaCl solution. On the effluent end of the column, polytetrafluoroethylene (PTFE) tubing transferred the effluent solution into the collection bottle.

An inflow rate of 15 mL/hr was used due to the low permeability of the encapsulating clay layer (K= 1.1 x 10⁻⁶ cm/s).
Figure 2. Sequential Column Leach Test (SCLT) setup

Modeling of Contaminant Transport in Surface Waters

In the case of steel slag-amended embankments, the contamination of surface waters like rivers and streams is of concern. Usually, computer programs designed to model the flow of groundwater through multiple soil layers ignore the surface runoff that may occur at the embankment surface and instead assume that the entire precipitated water infiltrates through the pavement structure and soil vadose zone.

Analytical solutions of the advection-dispersion equation (ADE) and related models are necessary to simulate contaminant transport processes in streams and rivers. The ADE distinguishes two transport modes: advective transport as a result of passive movement along with water, and dispersive/diffusive transport to account for diffusion and small-scale variations in the flow velocity, as well as any other processes that contribute to solute spreading.

Van Genuchten (2013a, 2013b) developed one- and multi-dimensional solutions for the advection-dispersion equation to define advective, dispersive, longitudinal transports and lateral dispersion. These solutions were utilized to develop a numerical model, named UMDSurf herein, to estimate the concentration distributions as a function of distance.

For one-dimensional transport, the solute flux $J_s$ can be written as:

$$ J_s = uC - D_x \frac{\partial C}{\partial x} $$

where $u$ is the longitudinal fluid flow velocity, $C$ is the solute concentration expressed as mass per unit volume of water, $D_x$ is the longitudinal dispersion coefficient accounting for the combined effects of ionic or molecular diffusion and hydrodynamic dispersion, and $x$ is the longitudinal coordinate.
The mass balance equation is then formulated by considering the accumulation of the solute in a control volume over time as a result of the divergence of the flux (i.e., net inflow or outflow):

\[
\frac{\partial C}{\partial t} = -\nabla \times J_s - R_s + R_w C_e
\]  

(2)

where \( t \) is time and \( R_s \) represents arbitrary sinks or sources of solute; i.e. \( R_s < 0 \) means consumption, whereas \( R_s > 0 \) means the feeding of solute, while the last term denotes injection (\( > 0 \)) or pumping (\( < 0 \)) of water with constituent concentration \( C_e \) at a rate \( R_w \). If these two last terms are ignored, the generally used ADE is obtained using the following equation:

\[
\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - u \frac{\partial C}{\partial x}.
\]  

(3)

However, natural processes such as biodegradation or inactivation, radioactive decay and production may affect the concentration of contaminants, and can all be included in the sink/source term, \( R_s \), in Eq. (2). As long as this term is described in terms of linear processes, the transport problem can still be solved analytically. For a case involving one or several sets of zero- and first-order rate expressions, the governing equation can be represented in the following form, where \( \mu \) is a general first-order decay rate, and \( \gamma \) is a zero-order production term:

\[
\frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - u \frac{\partial C}{\partial x} - \mu C + \gamma.
\]  

(4)

In this study, a numerical one-dimensional solution of ADE with a third type inlet condition is used as \( (\omega=1) \). The initial and boundary conditions are given below:

\[ C(x, 0) = f(x) \]  

(5)

\[ (uC - \omega D_x \frac{\partial C}{\partial x})_{x=0^+} = u g(t) \]  

(6)

\[ \frac{\partial C}{\partial x} (\infty, t) = 0 \text{ or } \frac{\partial C}{\partial x} (L, t) = 0 \]  

(7)

There is a semi-infinite domain with uniform initial concentration, \( f(x) = C_i \), and no production or decay is assumed to exist. The inlet concentration function, \( g(t) \), is of the pulse type (a Heaviside step function) with the constant concentration \( C_0 \), and is written as follows:

\[ g(t) = \begin{cases} C_0, & 0 < t \leq t_0 \\ 0, & t \geq t_0 \end{cases} \]  

(8)

The solution of Equation 4 is given below:
\[ C(x, t) = \begin{cases} C_i + (C_0 - C_i)A(x, t), & 0 < t \leq t_0 \\ C_i + (C_0 - C_i)A(x, t) - C_0 A(x, t - t_0), & t \geq t_0 \end{cases} \] (9)

where

\[ A(x, t) = \frac{1}{2} \text{erfc} \left[ \frac{x - ut}{\sqrt{4D_x t}} \right] + \frac{u^2 t}{\pi D_x} \exp \left[ -\frac{(x - ut)^2}{4D_x t} \right] - \frac{1}{2} \left( 1 + \frac{ux}{D_x} + \frac{u^2 t}{D_x} \right) \exp \left( \frac{ux}{D_x} \right) \text{erfc} \left[ \frac{x + ut}{\sqrt{4D_x t}} \right] \] (10)

RESULTS

Sequential Water Leach Tests (SWLT)

Duplicate sequential batch water leach tests (SWLT) were conducted on steel slag (SS), water treatment residual (WTR) and their mixtures. The leachates collected from the specimens were introduced into the encapsulation clay, and the tests were run for another 18 hours to simulate the flow of leachate through the encapsulation clay.

Figure 3 shows that the addition of WTR decreases pH significantly; however, the rate of decrease drops with increasing WTR amounts. It is speculated that the WTR addition decreases the release of free lime (CaO), hydrated calcium silicate (C-S-H) and portlandite Ca(OH)\(_2\) from the steel slag, resulting in a decrease in pH (Ozkok et al. 2015). The WTR used in this study is an aluminum-based material and, thus, the reduction in Ca released from the WTR-treated steel slag might be due to the formation of less soluble secondary mineral phases as a result of excess aluminum in the solution. Juenger et al. (2006) studied the hydration products of ground granulated blast furnace slag (GGBFS), which is a by-product of iron industry and has a similar elemental composition as steel slag via X-ray transmission microscopy and related the slow hydration rate of GGBFS to the formation of an aluminosilicate layer on the slag surface. Also Ozkok et al. (2015) has performed an experimental study on mitigation of alkalinity of steel slag and claimed that WTR mixing may result in formation of an aluminosilicate layer may have encapsulated the slag particles and hindered CaO hydration.

The data in Table 2 show that the presence of an encapsulation layer can significantly influence the pH due to high adsorption and the buffering capacity of the clayey soils. The relatively large surface area of clay particles increases the contact time and influences the buffering effect, which ultimately reduces the pH and the concentration of the most elements, including Cr and Se (Sauer et al. 2012). Guozhen et al. (1992) also stated that buffering capacity of a soil is directly related to its particle size.

Table 2 summarizes the aluminum (Al) and chromium (Cr) concentrations as well as the effluent pH values in sequential water leach tests. The variation in concentrations of Al and Cr is plotted against WTR content in Figure 3. The results show that, the effluent concentrations of aluminum (Al) are lower for the pure SS than the SS-WTR mixtures. This is due to very high aluminum content of WTR and a high pH. The solubility of Al is minimum at a pH of 6.5-7.0
and increases under alkaline conditions (Lim et al. 2004, Komonweeraket et al. 2015). The data in Figure 3 shows that the Al concentrations decrease even if the WTR amount is increased, since effluent pH decreases with increasing WTR content and Al remains insoluble.

Chromium (Cr) concentrations within the leachate decrease with the addition of WTR. As seen in Table 1, SS contains a significant amount of Cr, whereas the amount in WTR is negligible (1,300 mg/L versus <0.1 mg/L). However, the decrease in Cr concentrations is not linear due to a variation in pH. The solubility of Cr is highly dependent on the pH of the aqueous solution. Cr mobility is very low at a neutral pH, but increases significantly at both acidic and basic conditions (Sparks, 2003).

Table 2. Aqueous metal concentrations in SWLTs. Concentrations exceeding EPA Water Quality Limits (WQLs) in bold.

<table>
<thead>
<tr>
<th>Sample</th>
<th>WTR Content (%)</th>
<th>pH</th>
<th>Al (mg/L)</th>
<th>Cr (μg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SS-ENC</td>
<td>0</td>
<td>11.9</td>
<td><strong>3.66</strong></td>
<td>51.95</td>
</tr>
<tr>
<td>SS10WTR-ENC</td>
<td>10</td>
<td>9.94</td>
<td><strong>14.7</strong></td>
<td>17.15</td>
</tr>
<tr>
<td>SS20WTR-ENC</td>
<td>20</td>
<td>9.75</td>
<td><strong>9.08</strong></td>
<td>15.8</td>
</tr>
<tr>
<td>SS30WTR-ENC</td>
<td>30</td>
<td>8.68</td>
<td><strong>0.91</strong></td>
<td>13</td>
</tr>
<tr>
<td>SS60WTR-ENC</td>
<td>60</td>
<td>8.15</td>
<td><strong>1.42</strong></td>
<td>15</td>
</tr>
<tr>
<td>SS80WTR-ENC</td>
<td>80</td>
<td>7.22</td>
<td>&lt;0.05</td>
<td>12</td>
</tr>
<tr>
<td>WTR-ENC</td>
<td>100</td>
<td>6.01</td>
<td>&lt;0.05</td>
<td>&lt;1</td>
</tr>
<tr>
<td>EPA WQL</td>
<td>-</td>
<td>-</td>
<td>0.75</td>
<td>570</td>
</tr>
</tbody>
</table>

Note: Ex: SS10WTR-ENC designates specimens that consists of 90% steel slag and 10% WTR, whose effluent leachate then was mixed with encapsulation clay.

EPA WQL: Water Quality Limit for Aquatic Life

Sequential Column Leach Tests (SCLT)
Figure 4 shows the temporal characteristics of effluent pH of the pure SS and the SS-WTR mixtures, as well as the effluent pH after leaching through the encapsulation clay material. Considering the low hydraulic conductivity of the clay material (K= 1.1 x 10^{-6} cm/s) compared with that of the pure SS (K= 9.3 x 10^{-4} cm/s) and SS-WTR mixtures (K= 0.7-3 x 10^{-6} cm/s), which were measured via falling head test method at the end of SCLT’s, all tests were continued until a minimum of 20 pore volumes (PV) of flow were obtained to examine the behavior and persistency of pH. Even though the pH levels of the influent solutions were kept between 6 and 6.5, the stabilized pH levels of the effluent solutions of pure or treated SS were significantly high (pH>11). Pure steel slag has the highest pH (~12.6) and decreases with increasing WTR amounts (Figure 4). Similar to the observations made in SWLTs, the addition of WTR appears to have a significant effect on pH, and further decreases in pH can be obtained when the leachate passes through the encapsulating clay layer. These results also show that the clay reduces the system pH, regardless of the WTR percentage in the mixture, mainly due to the high buffering capacity of the clayey soils. It is speculated that the addition of WTR might have resulted in a decrease in pH, due to the formation of ettringite by coating the particles and preventing Ca^{2+} leaching.

Table 3 and Figure 5 show that concentrations of Al and Cr that leach from pure SS or SS-WTR mixtures exceed the groundwater quality limits set by EPA, but the concentrations decrease significantly after leachate percolates through the encapsulating clay layer. Al exhibits an amphoteric pattern, i.e., higher leaching concentrations at extreme pH levels and lower concentrations at a near-neutral pH level (Langmuir 1997, Kenkel 2003). A mixed trend in Al leaching is observed with the WTR amendment to SS due to relatively higher amounts of Al in the WTR (159,700 mg/L versus 10, 600 mg/L, Table 1). The WTR addition decreases the effluent pH below 12; however, large amounts of WTR (about 80% of total composition based on total elemental analysis) contribute to higher leached Al concentrations. The data in Table 3 show that the Al concentrations are elevated with an increase in WTR content; however, the concentrations decrease below detection limits when the effluent solution passes through the encapsulation clay layer and the pH drops below 7.5. This agrees with the results of other studies that showed Al leaching is the lowest at a neutral pH and is the highest under very alkaline conditions (Lim et al. 2004, Komonweerak et al. 2015).
Figure 5 shows that a decrease in the initial Cr metal concentrations from 73 to 3 μg/L occurs with an increasing WTR content. The solubility of Cr is highly dependent on the pH of the aqueous solution; i.e., Cr concentration is very low at a neutral pH, but increases significantly under very acidic and basic conditions. As seen in Table 3, the high pH of the SS leachate (pH ~12.6) causes an increase in peak Cr concentrations. Fallman (2000) performed a study to determine the controlling mechanism for chromium leaching from SS and showed that there are interdependencies between concentrations of barium, chromium, sulphur and calcium. Fruchter et al. (1990) indicated that aqueous concentrations of chromate (Cr$^{6+}$) is controlled by solid solutions of barium sulphate (BaSO$_4$) and solid solutions of Ba(S,Cr)O$_4$. However, none of these are reported to exist as primary minerals in steel slag. No testing was conducted to identify the oxidation state of Cr species in the SCLT leachates collected in the current study; however, Chaurand et al. (2007) claimed that Cr generally appears as Cr$^{3+}$ in steel slag leachates due to the absence of pre-edge peak in the spectra, or an indicator of hexavalent chromium, in the X-ray absorption near edge structure spectroscopy analysis (XANES). Pourbaix diagrams for the Cr-O-H system indicate that the Cr measured in SWLT and SCLT leachates is likely to exist as Cr(OH)$_3$ or Cr(OH)$_4$ for the pH 7-12.5 present in the current study (Stumm and Morgan, 1996). This oxyanionic species cannot be adsorbed and is soluble; thus, Cr concentrations of the pure SS leachate are higher than the SS-WTR mixtures. Increasing WTR content also decreases the pH; thus, Cr$^{3+}$ exists in an amorphous Cr(OH)$_3$ form, and the Cr concentration for both the effluent of slag-WTR mixture and the effluent passing through the encapsulation clay are below the detection limits.

<table>
<thead>
<tr>
<th>Sample</th>
<th>WTR Content</th>
<th>pH</th>
<th>Al-peak (mg/L)</th>
<th>Al-stab (mg/L)</th>
<th>Cr- peak (μg/L)</th>
<th>Cr- stab (μg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>S</td>
<td>0</td>
<td>12.59</td>
<td>0.6</td>
<td>0.43</td>
<td>73</td>
<td>15</td>
</tr>
<tr>
<td>SS10WTR</td>
<td>10</td>
<td>12.23</td>
<td>173</td>
<td>89</td>
<td>18</td>
<td>5</td>
</tr>
<tr>
<td>SS20WTR</td>
<td>20</td>
<td>11.51</td>
<td>213</td>
<td>116</td>
<td>15</td>
<td>5</td>
</tr>
<tr>
<td>SS30WTR</td>
<td>30</td>
<td>11.10</td>
<td>240</td>
<td>188</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>SS10WTR-ENC</td>
<td>10</td>
<td>7.30</td>
<td>0.70</td>
<td>&lt;0.05</td>
<td>&lt;1</td>
<td>&lt;1</td>
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<tr>
<td>SS20WTR-ENC</td>
<td>20</td>
<td>7.28</td>
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<td>&lt;1</td>
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</tr>
<tr>
<td>SS30WTR-ENC</td>
<td>30</td>
<td>7.23</td>
<td>0.81</td>
<td>&lt;0.05</td>
<td>&lt;1</td>
<td>&lt;1</td>
</tr>
</tbody>
</table>

**Note:** Ex: SS10WTR-ENC designates specimens that consists of 90% steel slag and 10% WTR in the first column, whose effluent leachate then goes into the second column filled with encapsulation clay.
Figure 5. CLT elution curves for aluminum and chromium

UMDSurp Results

Figure 6 presents the UMDSurp-predicted concentrations of aluminum and chromium in the stream after 20, 50, 100, 200, 500 and 1000 seconds. The results are obtained based on the assumptions of an instantaneous injection (t= 10 s) of 1 kg solute in the main channel of a river having a cross-section of 10 m², an average flow velocity of 1 m/s and a dispersion coefficient of 5 m²/s per recommendations of De Smedt et al. (2005) and van Genuchten (2013). The maximum concentrations obtained by the column tests are used as the input concentration at t=0 sec. As expected, metal concentrations decrease significantly with time and distance from the surface of the SS-WTR-encapsulation clay system. For all cases, at 20 m horizontal distance from the corner of the embankment, the concentrations decrease to approximately 50% of the initial concentration. Moreover, concentrations of all metals are lower than the EPA WQLs at all locations.

For pure SS, Al concentrations at 1000 m away from the corner of the embankment decrease to 13 μg/L, which is significantly lower than the EPA Maximum Concentration Limit (MCL) of 200 μg/L. For steel slag leachate, EPA MCL can be achieved at 50 m, whereas the leachate of S30WTR percolating through encapsulation layer decreases to the EPA MCL at 80 m from the corner of the embankment. The data in Figure 6 also suggests that the first-flush effect is minimized by the addition of WTR to SS, with the exception of aluminum due to very high Al content of the WTR material itself. Encapsulation of the embankment with a clayey soil (borrow material) decreases the concentrations to acceptable limits in surface waters. Similar trends can be observed for the Cr concentrations. At 20 m from the corner of the embankment, both the EPA MCL and WQL are satisfied. It should be noted that the rate of decrease increases if the initial concentration is higher. When S30WTR instead of 100% slag is used, the concentrations in surface waters are below the EPA MCL and WQL. The presence of an encapsulation clay layer further decreases the Cr concentrations.
CONCLUSIONS

A laboratory study was conducted to investigate the leaching characteristics of pure or WTR-amended steel slag in surface waters near highway embankments. The effects of WTR addition to steel slag, and presence of an encapsulating clay layer on pH and metal concentrations were studied through laboratory leaching tests. The observations from the current study are as follows:

1) The results of sequential water leach tests showed that increasing WTR percentage results in a decrease in pH. Flow of this leachate passing through an encapsulating clay layer decreases the pH further.

2) Aluminum concentrations increased when WTR amount increased from 0 to 10% due to high alum content of WTR, however, then decreased with increasing WTR percentage due to its amphoteric nature. Chromium concentrations decreased with increasing WTR content, whereas Al concentrations remained below EPA limits.

3) The results of sequential column leach tests indicated that an increase in WTR decreased the pH; however, WTR percentage did not have a significant effect on final pH values once the leachate percolated through the encapsulation layer.

4) Al concentrations increased dramatically with addition of alum-rich WTR but decreased significantly once the leachate passed through the encapsulation clay layer. It should be noted that Al is on the EPA list of secondary drinking water regulations, and there are no limits for Al specified in Maryland groundwater protection guidelines. Cr concentrations were below EPA limits after percolating through the encapsulation layer.

5) The results of chemical transport model showed that approximately 40 m away from the corner of the embankment, metal concentrations decreased by 50%. At that distance, Al concentrations also remained below the EPA limits.
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REFERENCES


