Experimental View of the Sub-Tropical Clayey Soils-Acid Mine Drainage Interactions

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Abstract— Issues of acid mine drainage (AMD) from mining activities is a lingering phenomenon to the present day developing South Africa as well as in most developed countries around the globe. However, the consistent rise in environmental contamination in South Africa is drastically attracting massive concerns. Vital concerns of AMD in South Africa still remain the threat to soil, surface, subsurface and ground water reserves among others, which are known to consequentially affect human and environmental health. This persistent challenge has given rise to the need for investigating the buffering efficacy of clayey mineral soils for use as natural contaminant barriers to pollutant species from AMD. Also, increasing climatic changes, growing population and depletion of natural resources from mining and related operations have further increased these environmental challenges. However, the constantly growing challenges gave bearing for this study to experimentally outline series of vital tests on three subtropical clayey soils infused with AMD for the determination of their mineral buffering efficacy towards potential use as natural containment liners in areas affected by AMD. Series of dissolution, diffusion and water sampling analysis were initiated herein on the respective soil systems leading to progressive investigations and analysis for decisive interpretations in subsequent studies. In summary, the general outcome was below the accepted criterion specified for clay liner construction in South Africa. However, the tested samples can be incorporated into the design and construction of composite barriers for acid-generating wastes.

Keywords—Batch Sorption, Contaminants, Acid Mine Drainage (AMD), Clayey Soils, Diffusion

I. INTRODUCTION

Acid mine drainage (AMD) occurs in sulphide-bearing mine waste due to the oxidation of iron sulphides, especially pyrite (FeS₂), pyrrhotite (FeS) and marcasite (FeS₂) causing the creation of several soluble hydrous iron sulphates, the generation of acidity and subsequent leaching of metals.

AMD have several negative human and environmental impacts [1-4] which may be alleviated by lining possible acid-generating waste rock or mine tailings impoundments with clayey or composite clayey liners. For the competent performance of these liners, it is important that their properties are not compromised over time when exposed to AMD. As such, assessment of the chemical interaction and compatibility of clay/clayey liners with aggressive AMD is a pertinent consideration in the proper management of potentially acid generating mine waste [5]. These AMD-soil interaction studies may involve dissolution, diffusion, water sampling, percolation testing, contaminant migration analysis and determination of soil chemical transformations. Although, some studies on the interaction of soils from temperate regions with AMD have been conducted [6-8] there still remains however, insufficient data on the interactive outcomes/compatibility of soils from tropical and subtropical regions with AMD. More to this, sufficient outcomes involving some key tests i.e., dissolution, diffusion etc., were not reported in most of these works which nonetheless, created a path for this study to bridge that knowledge gap. AMD usually has high acidity and is rich in sulphate and potentially toxic elements and heavy metals.

The rate of AMD generation in an environment where sulphidic material is exposed to oxygen and water, will be determined by a number of factors. The reactions will proceed rapidly in warm humid environments, while other environmental factors, such as the specific sulphide mineralogy and grain size, and the presence or absence of acid neutralizing minerals, such as carbonates will also influence the rate of AMD production and its eventual environmental impact. The major source of groundwater contamination by mine tailings have been suggested to be AMD [9]. The environmental impact of AMD as reviewed by [10] include; metal toxicity, sedimentation processes and high acidity, which usually have the potential to cause interrelated negative impacts on terrestrial and aquatic life. Since AMD has potential negative impacts on the environment, efforts are made to either prevent or control its generation and migration. As such, several measures and technologies are available as recorded by [11-14] for mitigating the environmental effects of AMD. These include co-disposal of waste rock and mine tailings and the construction of wetlands, base liners or water covers.

Soil covers and liners may also be used as hydraulic barriers for containing AMD leakage at mine sites. When used for this purpose, it is usually pertinent to determine whether the soil liner can buffer contaminants or is compatible with AMD. Consequently, several studies have been conducted on the compatibility of some soils from temperate regions with AMD.
Johannesburg (CoJ), South Africa, where waste disposal
three compacted clayey soils sampled around the City of
prerequisite approaches in the forms of dissolution, diffusion
clayey soils-AMD interaction with respect to vital or
of this study, the chemical and geotechnical properties, and the
the storage of mine waste being generated from mining
and acid water containment ponds are being constructed for
sulphidic ore exploitation in South Africa, mine tailings dams
African mine environment is imperative. Currently, with the
mitigating the effects of AMD contamination of the South
environment as the water, in accordance with well-known and
surface flow or decant of mine water is also of concern to the
lead to the widespread generation of AMD in and around areas
where such operations occur in South Africa.

AMD is known to occur in a number of areas in South
Africa, mostly in the Witwatersrand Gold Fields, the country’s
various Coal Fields and the O’Kiep Copper District. Thus,
potable sources of water in mining areas, some of which are
densely populated, are severely at risk due to AMD
contamination. The key factors which differentiate the
developing problem in South Africa from international
scenarios is the degree of interconnection of large voids, the
sheer scale of the Witwatersrand operations and the fact that
many of the problem areas are located in or close to major
urban centres. Most mining centres are also covered by the
natural habitats of some rare and endangered species.
Bioaccumulation of metals from AMD adversely affects flora
and fauna in South Africa and this has led to the loss of food
sources and elimination of some sensitive species [22-25]. The
surface flow or decant of mine water is also of concern to the
environment as the water, in accordance with well-known and
researched chemical and geochemical reactions between the
mine rock strata, wastes and oxygen, readily becomes acidic,
characterized by elevated concentrations of salts, heavy metals
and radionuclides. The immediate concern as recorded by [26]
is the threat posed by decanting mine water in the
Witwatersrand area, which poses potentially severe
environmental and safety impacts on the receiving water
environment and associated surface areas. In this light,
mitigating the effects of AMD contamination of the South
African mine environment is imperative. Currently, with the
sulphidic ore exploitation in South Africa, mine tailings dams
and acid water containment ponds are being constructed for
the storage of mine waste being generated from mining
operations, and assessment of the suitability of local
construction materials is therefore essential. In an earlier stage
of this study, the chemical and geotechnical properties, and the
mineralogical composition of three sampled soils, prior and
subsequent to AMD percolation were determined and the
results were presented.

However, the study herein experimentally outlines the
clayey soils-AMD interaction with respect to vital or
prerequisite approaches in the forms of dissolution, diffusion
and water sampling analysis on AMD permeation through
three compacted clayey soils sampled around the City of
Johannesburg (CoJ), South Africa, where waste disposal
activities occur. To assess the impact of these contaminant
migration processes on the buffering efficacy and performance
of the respective soils as lining materials, these key tests were
prone and are subsequently discussed herein. The outcomes
forms part of a larger progressive study towards very decisive
analysis and interpretations of the soils-AMD interaction for
the determination of their buffering efficacy for potential use
as natural earthen lining materials (locally available clayey
mineral liners) for the containment of mine waste and
associated pollutants.

II. RELATED BACKGROUND

The contaminant mass transport mechanisms in natural and
compacted clayey soils have been extensively investigated by
several researchers over time [27-32]. Compacted clays
usually have low permeability and as such, groundwater flow
through them is usually slow. Consequently, the mass of
solute transport by flowing contaminating solutions
(advection) is likely minute, whereas it is usually significant in
solute transport due to concentration gradients (diffusion). As
such, dissolution, diffusion and percolation tests are
commonly conducted to determine retardation factors,
effective diffusion coefficients and hydraulic conductivities
respectively. Hence, the study found the need to experimentally assess the principles involved in dissolution,
diffusion and water sampling analysis of the transport
mechanisms towards determining the buffering efficacy of the
respective sampled subtropical clayey soils.

III. DISCUSSION OF FINDINGS

A. Dissolution Tests

To determine the dissolution degree, $S_o$, or distribution
coefficient, $K_{ds}$, of potassium, sodium, zinc, nickel, cobalt,
chloride and sulphate the batch sorption test was conducted in
conformance to [33]. The dissolution degree, $S_o$, is a measure
of the extent to which the AMD permeant dissolves
contaminant species from the respective soils. However, for
species that were sorbed on the soils from the AMD permeant,
the distribution coefficient $K_{ds}$ was then determined from the
batch sorption tests. Before the sorption tests were conducted,
representative soil samples were air-dried, disaggregated and
passed through the No. 40 sieve. 20 mm of AMD was added
to 5 g of sieved soil in plastic centrifuge bottles to obtain a
soil: AMD ratio of 1:4. The use of AMD in place of single ion
solutions was to allow for potential competitive sorption
between ions for available soil exchange sites.

The tests were conducted at laboratory temperature of 22 ±
2°C and with soil-AMD interaction times of 5, 10 and 15 days
before being agitated in a ball bearing electric laboratory
shaker. For each 5 day quota of the interactive period, the
mixture was agitated for 16 hours, meaning that, for batch
sorption tests lasting over 10 and 15 days, the mixtures were
agitated for periods of 32 hours and 48 hours respectively.
Interaction times of 3, 7 and 11 days were reported by [34]
however for the purpose of this study, longer soil-AMD
interaction times were pursued. Agitation was terminated a
day prior the expiration of the interaction/contact time to allow
the mixtures to settle. For instance, a test with interaction time
of 10 days meant the mixture was agitated for a total period of 32 hours, 16 hours within the initial 5 days and another 16 hours of agitation from day 6 to day 9. Agitation did not take place on day 10 of the interaction time to allow for settlement of the mixture. On expiration of the interaction times, each mixture was centrifuged at 10,000 g for 20 minutes in a high speed centrifuge. The supernatant was then decanted and sampled for chemical analysis. The contamination coefficient, \( K_d \), of each nominal contaminant species was determined using the relationship in (1).

\[
K_d = \frac{\frac{C_f - C_i}{R}}{\frac{C_f}{M_a}} \times \frac{V_f}{M_a} \tag{1}
\]

Where: \( C_i \) = Initial concentration of species in leachate (mg/L); \( C_f \) = Concentration of species in decanted solution at the expiration of test (mg/L); \( V_f \) = Volume of leachate added to the soil (mL); and \( M_a \) = Mass of air-dried soil (g).

The contaminant species retardation factor expression as discussed in a previous study was then used to calculate the retardation factors, \( R \), of the various species. More to this, the relationship expressed in (2) was used to calculate the dissolved amount of a species (dissolution degree, \( S_o \)) per unit area of air-dry mass of soil.

\[
S_o = \frac{C_f - C_i}{C_i} \times \frac{V_f}{M_a} \tag{2}
\]

Where: \( S_o \) = Dissolution degree (mL/g).

### B. Diffusion Tests

For the soil porewater concentration profiles of the chemical species diffusing through the saturated soils to be obtained, the diffusion test was performed. The diffusion cell used for the study conformed to the bespoke device used by [34] which comprised of a cylindrical stainless steel mould with an internal diameter of 54 mm and a height of 70 mm. It had a stainless steel base plate and top cap fitted tightly together by steel rods and screws. Grooves were created in the top and bottom of the cylinder for placement of O-rings and silicon sealant to keep the top and bottom caps tightly sealed to the cylindrical mould. Two ports were provided in the top cap. The first port held a 9 mm diameter Teflon stirring rod with paddle connected to a low-speed stirrer. While the second port meant for periodic sampling of 1 mL of permeate from the permeant chamber was continuously stirred at 5 rev/min throughout the test. Table I summarizes the properties of the compacted soil used for the diffusion tests.

![Fig. 1. Schematic of the bespoke diffusion cell](image)

<table>
<thead>
<tr>
<th>Properties</th>
<th>ASTM Design.</th>
<th>Soil A</th>
<th>Soil H</th>
<th>Soil K</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural Water Content (%)</td>
<td>D 2216</td>
<td>20</td>
<td>32</td>
<td>27</td>
</tr>
<tr>
<td>Water Content (Air-dry soil) (%)</td>
<td>D 2216</td>
<td>4.7</td>
<td>1.3</td>
<td>1.4</td>
</tr>
<tr>
<td>Optimum Water Content (%)</td>
<td>D 698</td>
<td>16.2</td>
<td>15.4</td>
<td>15.7</td>
</tr>
<tr>
<td>Bulk Unit Weight (kNm⁻¹)</td>
<td>-</td>
<td>18.52</td>
<td>19.11</td>
<td>19.97</td>
</tr>
<tr>
<td>Compaction Water Content (%)</td>
<td>-</td>
<td>15.91</td>
<td>15.15</td>
<td>15.58</td>
</tr>
<tr>
<td>Initial Degree of Saturation</td>
<td>-</td>
<td>81.9</td>
<td>79.8</td>
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</tr>
<tr>
<td>Final Degree of Saturation</td>
<td>-</td>
<td>99.5</td>
<td>99.7</td>
<td>99.4</td>
</tr>
<tr>
<td>Degree of Compaction (%)</td>
<td>-</td>
<td>99.1</td>
<td>98.9</td>
<td>99.3</td>
</tr>
<tr>
<td>Dry Unit Weight (kNm⁻¹)</td>
<td>-</td>
<td>15.15</td>
<td>16.37</td>
<td>17.33</td>
</tr>
<tr>
<td>Moulded Dry Unit Weight (kNm⁻¹)</td>
<td>-</td>
<td>15.13</td>
<td>16.35</td>
<td>17.31</td>
</tr>
<tr>
<td>Wet Unit Weight (kNm⁻¹)</td>
<td>-</td>
<td>17.7</td>
<td>18.9</td>
<td>20.1</td>
</tr>
<tr>
<td>Void Ratio</td>
<td>-</td>
<td>0.76</td>
<td>0.77</td>
<td>0.74</td>
</tr>
<tr>
<td>Porosity</td>
<td>-</td>
<td>0.45</td>
<td>0.44</td>
<td>0.43</td>
</tr>
</tbody>
</table>

On termination of the diffusion test, the permeant chamber was carefully emptied out of the cell and sampled for ion analysis. The soil sample was then extruded from the mould and sectioned into nine approximately equal thicknesses. The moisture content was determined for each sectioned soil sample. Other portions were then squeezed with a pneumatic porewater squeezer to obtain soil porewater for chemical analysis. In a case where the post-diffusion water content was insufficient to generate enough volume of porewater for chemical analysis, a saturation extract was obtained and the measured concentrations were adjusted for dilution.

### C. Analysis of Water Sample

As already discussed in an earlier study, while measuring the hydraulic conductivity of compacted specimens of the soils using a constant-head rigid wall bespoke permeameter device, effluents were also collected for chemical analysis in a 500 mL graduated cylinder over the test duration. Soil porewater, influent permeant and the effluents obtained during hydraulic conductivity testing were acidified by high-purity nitric acid and analyzed for cationic or heavy metal concentrations using inductively coupled plasma-atomic emission spectroscopy (ICP-AES). To accommodate for any
The study depicts an experimental view of and assessment of three subtropical clayey soils interacted with AMD towards determining their mineral buffering efficacy for use as natural containment liners in areas under AMD attack. From sequential prolonged testing on the respective soil systems, pertinent tests leading to more decisive interpretations and analysis in succeeding studies were achieved. The species concentrations obtained from either the effluents or the solution and sections of soil samples obtained from dissolution and diffusion tests suggested that the soils–AMD interaction is a complex process. The results of the study suggest that interactions between AMD and the soils plausibly resulted in dissolution of metals from soil grains, desorption of adsorbed contaminant species at the exchangeable sites of the soils and, alteration and dissolution of soil minerals. Hence, in consideration of the general outcomes of the study with further interpretations and analysis in succeeding studies, the three sampled natural subtropical soils were found to be incompatible with AMD which invariably may not be utilized solely as natural clayey mineral lining materials towards AMD containment disposal. However, may be considered for use in the construction of composite barriers for containing acid-forming waste bodies.

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References


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