



Waste to energy ash monofill mining: An environmental characterization of recovered material



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HIGHLIGHTS

- Mixed ash samples excavated from U.S. ash monofill representing 17 year age range.
- Ash did exhibit natural weathering and corresponding pH decrease while monofilled.
- Evidence elements were washed from ash in greater magnitude at monofill surface.
- Trend of decreasing pH with increasing bore depth seen in ash samples.

GRAPHICAL ABSTRACT



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ABSTRACT

Samples of combined bottom and fly ash produced at a U.S. waste-to-energy facility were collected from an ash monofill. These samples represented ash monofilled between 1991 and 2008. The ash samples were characterized for total element content and leachability; trends in these parameters were evaluated as a function of sample depth and ash age. Comparison to risk thresholds was used to assess the relative magnitude of the total and leachable mass of elements in the monofilled ash. Natural carbonation was found to have occurred in the monofilled ash, reducing the pH and leachability of Al and Pb. Sb was the element with the highest leachable concentration when compared to risk thresholds, driven primarily by the pH of the ash (9.8). The release of Mo, Sr, Ba, Na and K (all readily soluble elements in ash) was higher (48–122%) when comparing the samples taken from the 0 to 1.5 m bore to the 6.1–7.62 m bore; total concentration analysis also demonstrated that more of these elements were present in the deeper samples (25–53%). These data support the hypothesis that as infiltrating rainwater moves through an ash monofill leached concentrations are depleted from the upper layers of the ash first.

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1. Introduction

Preserving and recovering resources from the municipal waste stream is paramount in moving society's environmental steward-

ship forward into the next generation. A substantial portion of the world's municipal solid waste (MSW) is managed through combustion for energy recovery, or waste to energy (WTE) [1]. European nations, with a more stringent focus on reducing landfilling and promoting resource recovery have led in efforts related to waste combustion, recovery of ferrous and non-ferrous metals from the combustion ash, and environmentally responsible beneficial use of these residues [2–5]. These practices result in a carbon offset, a

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reduction in mined aggregate demand, and provide a cost savings to ash generators and end users (through landfill diversion and the generation of saleable commodities) [6,7].

In the United States, efforts related to WTE ash reuse have been attempted multiple times over the past several decades [8,9]. Some of these attempts have seen initial success, but efforts have been limited, in part due to the U.S. regulatory framework and hazardous waste characterization under the Toxicity Characteristic Leaching Procedure (TCLP) [5]. The TCLP has been widely criticized by scientists and some within the U.S. regulatory community, yet it remains the test which dictates the management outcomes for many wastes (including WTE ash) [10]. Historically in the U.S., WTE bottom and fly ash have been comingled to produce a “mixed” ash with the goal of preventing fly ash from being classified as a hazardous waste under the TCLP [5]. This practice of producing a mixed ash steam differs dramatically from European and Asian WTE ash management policies where fly ash and bottom ash are separated, allowing the bottom ash (lower in concentrations of many heavy metals) to be reused [3,11].

At present, beneficial use of waste and industrial byproducts has become an issue of substantial focus within the U.S., due to both a need to recover critical materials in a cost effective manner, as well as an increased emphasis on developing methods to reduce greenhouse gas (GHG) emissions [2]. Advanced metals recovery technologies (for WTE ashes) have been demonstrated to yield significant quantities of saleable non-ferrous metals; interested parties within the U.S. and abroad have begun to focus on the potential of mining monofilled ash to recover these resources [12–15]. Quantification of the lifecycle impacts of MSW landfill mining has found that the most significant GHG benefit of this practice is metals recovery, while adverse impacts are mostly attributed to the emissions from mining and operational equipment [16]. As WTE ash monofills become an a source for metals recovery, beneficially using the post-mined material could be a viable approach to increase the GHG offset (as the beneficial use of WTE ashes has been found to have a positive GHG impact), while also creating an avenue for cost savings [14].

While reuse of processed mixed ash could provide multiple benefits, it is imperative that an appropriate environmental characterization of the material is conducted. Such examination would quantify the leaching of constituents of potential concern (COPC) from the ash to water supplies (in a beneficial use scenario) and examine the magnitude of the total mass of COPC in the material (to assess direct human exposure risk). The fly ash present in the material would be likely to influence the total elemental concentration and element release into the environment for certain contaminants (such as As and Cd) [17]. Limited environmental data are available on monofilled mixed ash residues. Saffarzadeh et al. evaluated mixed ash recovered from a monofill; however, the focus of the study was related to ash mineralogy and geochemical changes to the material over time. No data related to monofill ash leachability or beneficial use were presented [18,19].

If the beneficial use of mixed ash is to occur, data are needed to evaluate the feasibility of mining and recycling projects. This manuscript presents results from an environmental characterization study conducted on mixed WTE ash excavated from 8 different locations within an ash monofill. Total concentration and leach testing were conducted on the ash samples to determine: the relative magnitude of elements (total and leachable) and the trends in elemental concentration with respect to depth and age. These trends are presented and used to provide information on the changes to the ash over time. Ash carbonation, a result of exposure to natural CO₂, is one of the primary factors which controls ash pH and element release [20–23]. The degree of carbonation was examined by measuring the pH of the ash as a function of location and depth.

2. Materials and methods

2.1. Site description and ash sampling

Ash samples were collected from an ash monofill in Florida, U.S. This monofill was the sole disposal source for a 1,000 ton per day, mass burn WTE facility. This facility utilizes the MSW from the surrounding county as its feedstock. The WTE combustion system at the facility employs: a rotary grate stoker boiler, injected quicklime for acid gas control, and utilizes a bag house for scrubber residue and particulate collection. A satellite image of the monofill cells and bore locations as well as a date range for each of the ash bores are provided in Fig. S1. Once a monofill cell was filled to an intermediate height it was capped with a high density polyethylene (HDPE) cover. Samples were taken using a 10.16 cm (4”) diameter solid shaft continuous flight auger and a truck mounted drill ring. Samples were collected by removing the HDPE liner and any cover soil that was over the placed waste. The drill rig then bored down in 1.52 m (5’) increments and the auger cuttings were collected and stored in sealed 19 L (5-gal) buckets until time of testing.

The ash monofill cells tipping area and the exposed surfaces of the ash (slide slopes and top of the monofill) were open to the atmosphere for a period ranging from 5 to 8 years, dependent on the waste filling rate. Eight samples were retrieved from three different monofill cells (Cells A1–A3).

For the ash samples collected from monofill cells A1 and A2 the depth of the bores was set to 6.1 m (20 ft.); bores in cell A3 were drilled to a depth of 7.62 m (25 ft.), due to a slightly higher maximum cell height (approximately 7.62 m depth to the leachate collection system in cells A1 and A2 and 9.15 m in cell A3). This resulted in the generation of 34 discrete samples from the ash monofill; each bore either had four (0–1.54 m, 1.54–3.05 m, 3.05–4.57 m, 4.57–6.1 m) or five (for cell A3: 0–1.54 m, 1.54–3.05 m, 3.05–4.57 m, 4.57–6.1 m, 6.1–7.62 m) samples collected. Using historic waste tonnage data and topographic surveys of the cells during filling, the date of waste placement at each of the bore locations was estimated. Bores were taken from three locations in cells A1 and A2 the: eastern side of the cell, center of the cell, and western side of the cell. Samples from cell A3 were collected on the eastern and western sides of the cell; all of the samples were collected from the north-south center point of the cells at each location. A photo showing the location of the collected samples is available in the supplemental information (SI) section. The bores are referred to by their locations (e.g., A1E—the furthest east bore in the A1 cell) and each sample is designated by depth (e.g., A1E 0–1.5 is equal to the 0–1.54 m sample from the A1E bore).

2.2. Laboratory tests and elemental analysis

All samples were tested for moisture content in accordance with ASTM D2216. Leaching of the monofill bores was evaluated using the synthetic precipitation leaching procedure (SPLP) conducted in triplicate (EPA Method 1312) [24]. The pH of the SPLP test was used as an indicator of each ash samples pH. The pH data was collected for each of the triplicate SPLP samples following the 18-h extraction. Additional details on leaching and total extraction tests are provided in the SI section. Five replicates of EPA method 3050b, a total environmentally available digestion, were conducted on each ash sample [24]. Following leaching and total concentration extraction, all samples were analyzed for Al, As, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, Pb, Sb, Se, Sn, Sr, V, and Zn using inductively coupled plasma atomic emission spectrometry (ICP-AES) in accordance with EPA method 6010c [24].

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