

Municipal solid waste combustion ash: State-of-the-knowledge

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Abstract

Over the past several years there has been significant controversy concerning the proper management of the residues from combusting municipal solid waste (MSW) and their regulatory classification as hazardous or non-hazardous waste. This controversy and other factors (e.g., lack of federal guidance, heavy metal content, etc.) have resulted in inconsistent management requirements among several states and uncertainty about beneficial utilization of the residues. Heavy metal content and leaching of these metals (especially in the TCLP test) is most often cited as the reasons the material should be managed as a hazardous waste. If not managed properly, contamination of ground water by leaching of soluble salts from the ashes may also be a concern. The United States lags behind some countries in ash utilization. Although research and demonstration projects have indicated that the ashes can be beneficially utilized, less than 5% of the ashes are utilized in the United States. Other countries, including Denmark, the Netherlands, France, Germany, Switzerland, and Japan are further advanced in ash utilization and in establishment of a systematic process for evaluating and selecting disposal and utilization options. This paper discusses ash characteristics, the state of ash management in the United States, federal initiatives, results of laboratory and field characterization of leachates from the ashes, barriers to ash utilization in the United States, and international perspectives.

Keywords: Municipal waste combustion; Ash; Ash characteristics; Ash treatment options; Ash utilization

1. Introduction

In 1993 there were 125 waste-to-energy (WTE) facilities operating in the United States. The combined design capacity of these plants ranges from 99 400 tpy to over 107 000 tpy, with an estimated electrical generating capacity of 1800 to more than 2900 MW. [1, 2]. The combined estimated quantities of ash produced by these facilities is approximately 8.5–9 Mt/yr. This will increase to as much as 17 Mt/yr or more in the future as more WTE facilities are placed into service. This ash must be managed on a daily basis.

Municipal waste combustion (MWC) residues are generated at several points in the process of burning municipal solid waste (MSW) for energy recovery. Solids retained on furnace grates following combustion and solids passing through the grates (siftings) are generally referred to as bottom ash. Entrained particulates that are trapped and residues generated by acid gas scrubbers and subsequently removed by fabric filters and/or electrostatic precipitators (ESPs) are normally referred to as air pollution control (APC) residue. In some cases, especially in Europe, ESPs are used to remove particulates before wet scrubbers. This stream may be considered as an APC residue or as a fly ash. Entrained particulates and condensed vaporized metals trapped in heat exchangers generate a small quantity of ash, referred to as heat recovery ash. Heat recovery ash is combined with either the APC residue or the bottom ash. Approximately 80% of the residues generated are bottom ash. In the United States, these fractions are normally collected together as a combined ash.

The physical characteristics of bottom ash resembles an aggregate while the APC residue is much finer. The major elements in bottom ash are O, Si, Fe, Ca, Al, Na, K and C. Major elements in the APC residues are O, Si, Ca, Al, Cl, Na, K, S and Fe. Although many metals are present as oxides, there are also significant metal chlorides, metal sulfates, and metal carbonates. The APC residue contains significantly higher concentrations of cadmium, lead and zinc than does bottom ash. The APC fraction also contains higher concentrations of soluble salts.

Leaching of cadmium and lead from the residues in laboratory tests, particularly the Toxicity Characteristics Leaching Procedure (TCLP) [3], has been the key issue regarding the classification and management of these materials. Concentrations of heavy metals in leachates from laboratory tests are normally much higher than the metal concentrations in the leachates from ash monofills. There is disagreement among the technical community, regulators, environmental groups and others concerning the use of laboratory tests to predict the ultimate fate of metals when the ashes are placed into the environment, either for utilization or disposal. Rather than a single test, one can learn much more from several tests designed to determine concentrations in the ashes, the amounts available for release under worst case conditions, and the amount expected to be released over time under the actual conditions of utilization and disposal. Only a small fraction of the metals present in the ashes are normally leached. Soluble salts, on the other hand, are almost all released and should be properly managed [4].

Most MWC residues generated in the United States are disposed into monofills lined with either clay soil liners or synthetic liners or both. The designs usually include provisions for leak detection and leachate collection. While some other countries utilize a significant portion of the residues, probably less than 5% are used in the United States. The debate over classification and concern about the release of heavy metals combined with a lack of federal guidance and consideration of related issues have impeded utilization in the United States [4-7].

There are options for using ash residues and for treating them prior to use or as a requirement for disposal. Treatment options include processing to remove ferrous metals, compaction aging during storage, solidification/stabilization, vitrification and chemical extraction. Major utilization options include aggregate for road base,

embankments, asphalt pavements and aggregate in portland cement for construction. Research and field projects to demonstrate these various uses have been conducted or are underway or are planned [5–9]. In order for utilization to proceed, results from these projects must demonstrate no adverse effect on the environment or human health. Additionally, criteria and technical guidance to assure safe use must be developed.

2. Definition of MSW combustion ashes and residues

The ashes from combustion of MSW are routinely classified into three categories: bottom ash, APC residues, and combined (i.e., the combination of bottom ash and APC residues). In the United States these streams are normally combined for disposal, while in most European countries and in Canada they are separated into APC residues and bottom ash fractions [10].

The terms ash and residue are often used interchangeably. Interchangeable use, however, can be misleading. Ash refers to material remaining after complete combustion of materials while residue refers to unburnt material, scrubber sludge, reaction products and similar materials which end up in the ash/residue streams. Fig. 1 depicts points in a combustion facility from which ash/residue streams are generated [10]. Table 1 provides a technically correct description of the various fractions of the ashes and residues.

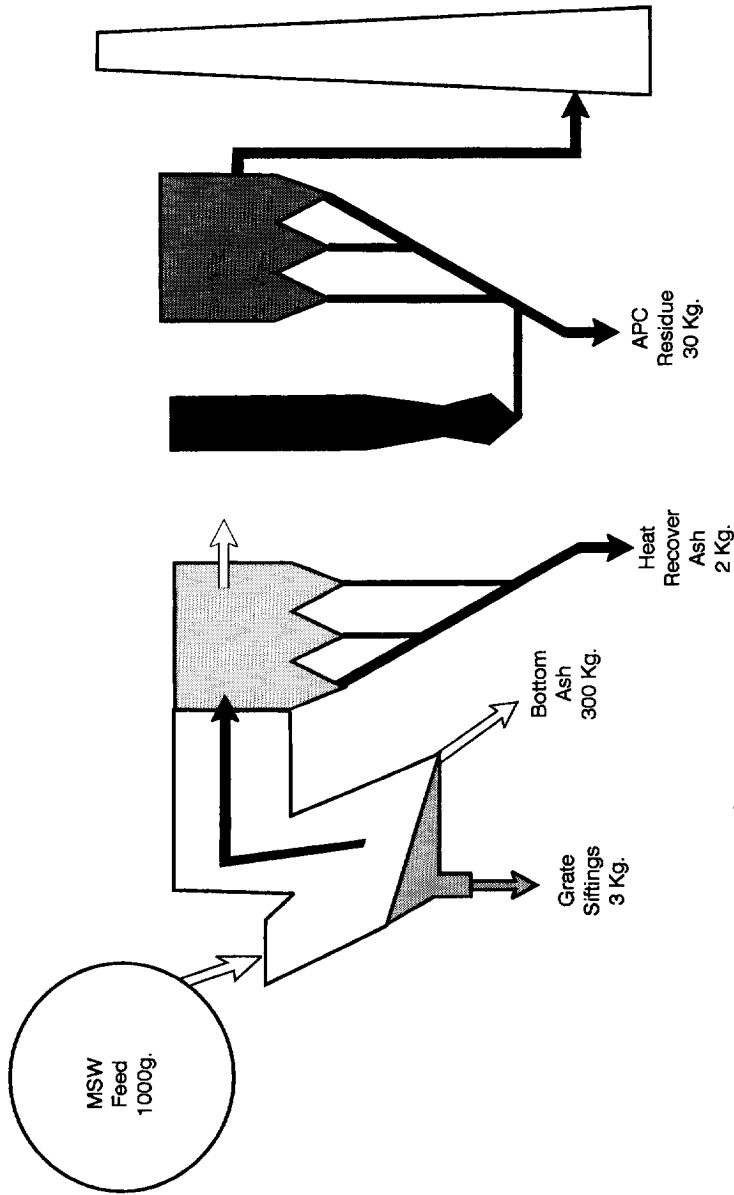
2.1. Physical characteristics

Bottom ash: Bottom ash is a heterogeneous mixture of slag, ferrous and non-ferrous metals, ceramics, glass, other non-combustibles, and uncombusted organics. Up to 20% of the bottom ash has a particle size of >10 cm, consisting of ferrous and non-ferrous metals, slags, and construction-type materials. The fraction <10 cm is somewhat uniform with up to 10% fines <200 mm. This latter fraction resembles well graded sand and gravel. Bottom ash is a very porous lightweight aggregate with high specific surface areas. It has dry densities of 950 kg/cm^3 or higher, with specific gravities of 1.5–2 for the fine fraction and 1.8–2.4 for the coarse fraction.

Bottom ash contains varying amounts of moisture as a result of quenching. In modern WTE facilities, the geotechnical water content normally ranges from 15% to 25% (wet weight/dry weight). Higher levels may be found in older systems, and lower levels in mass burn facilities equipped with ram dischargers.

This moisture content is important for fugitive dust control and as an aid in compaction. With optimum moisture content of approximately 16%, the bottom ash can be compacted to proctor densities of 1600 kg/m^3 resulting in hydraulic conductivities of 10^{-6} cm/s or less. Such hydraulic conductivities can play an important role in the management and utilization of these residues.

APC Residues: The residues from dry/semidry APC systems are fine particulate mixtures of fly ash consisting of reaction products of primarily calcium chlorides



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Fig. 1. Sources of residues in the MSW combustion process.

Source: International Ash Working Group

Table 1
Description of ash fractions generated from the combustion of MSW

Ash fraction	Description	Comments
Bottom ^a	Material discharged from the bottom of the furnace, primarily the grate	Normally the term 'bottom ash' also includes grate siftings. May be referred to as grate ash or clinker in Europe
Grate siftings or riddlings	Material falling through the furnace grates	Generally combined with bottom ash in the quench system
Heat recovery ash (HRA)	Particulate matter collected from the heat recovery system	May be further subdivided into boiler, ash economizer ash, or superheater ash, etc. depending on the area of the heat recovery system from which it was collected. May be combined with either bottom ash or APC residue depending on facility design
Fly ash	Particulate matter carried over from the furnace and removed from the flue gas prior to injection of sorbents	Includes volatiles condensed during flue gas cooling. Excludes ashes from the heat recovery system
APC residue	Combined material collected in the air pollution control devices, including fly ash, injected sorbents, and flue gas condensate	
Combined ash	Mixture of bottom ash, grate siftings, and APC residues	Waste-to-energy facilities in the United States routinely manage combined ash. Bottom ash and APC residues are collected and managed separately in Canada and Europe

^aBottom ash comprises the major portion of the residues generated, depending on the combustion facility design, operating conditions, and characteristics of the waste being combusted.

and unreacted lime used for acid gas emission controls. The fly ash has the largest particle size, followed by the residues from the electrostatic precipitators and the fabric filters. APC residues are usually highly soluble in water (25%–85% by weight) due to their high concentrations of soluble salts.

2.2. Chemical characteristics

The chemical characteristics of ashes and residues are the major reason for the concern regarding their classification as hazardous or non-hazardous wastes and their ultimate management requirements. Ash and residue fractions contain varying amounts of trace metals and soluble salts which potentially could result in adverse environmental effects if improperly managed. Cadmium and lead have represented major areas of concern; more recently some concern has also been expressed regarding the aquatic toxicity of copper; and mercury, due to more stringent controls on mercury emissions.

Table 2 provides the ranges of constituents found in bottom ash, fly ash, and wet and dry APC system residues from many facilities [10].

Approximately 80–90% by weight of the bottom ash consists of O, Si, Fe, Ca, Al, Na, K, and C. Minor elements (1000 mg/kg–10 000 mg/kg) are Mg, Ti, Cl, Mn, Ba, Zn, Cu, Pb and Cr. Trace elements (i.e., <1000 mg/kg) consist of Sn, Sb, V, Mo, As, Se, Sr, Ni, Co, Ce, Ag, Hg, B, Br, F and I. The composition of major and most minor elements is similar to basaltic and other geologic materials. Some of the minor elements and many of the trace elements (e.g., Pb, Cu, F, Cd, and Hg) are enriched in the bottom ash. Grate siftings comprise approximately 1%–3% by weight of the bottom ash. This fraction, however, contributes a significant fraction of the elemental

Table 2

Elemental composition of bottom ash from all types of incinerators and fly ash, dry/semidry, and wet APC system residues from mass burn incinerators

Element	Range for bottom ash (mg/kg)	Range for fly ash (mg/kg)	Range for dry/semidry APC system residues (mg/kg)	Range for wet APC system residue without fly ash (mg/kg)
Ag	0.29–37	2.3–100	0.9–60	—
Al	22 000–73 000	49 000–90 000	12 000–83 000	21 000–39 000
As	0.12–190	37–320	18–530	41–210
B	38–310	—	—	—
Ba	400–3000	330–3100	51–14 000	55–1600
C	10 000–60 000	—	—	—
Ca	37 000–120 000	74 000–130 000	110 000–350 000	87 000–200 000
Cd	0.3–71	50–450	140–300	150–1400
Cl	800–4200	29 000–210 000	62 000–380 000	17 000–51 000
Co	6–350	13–87	4–300	0.5–20
Cr	23–3200	140–1100	73–570	80–560
Cu	190–8200	600–3200	16–1700	440–2400
Fe	4100–150 000	12 000–44 000	2600–71 000	20 000–97 000
Hg	0.02–7.8	0.7–30	0.1–51	2.2–2300
K	750–16 000	22 000–62 000	5900–40 000	810–8600
Mg	400–26 000	11 000–19 000	5100–14 000	19 000–170 000
Mn	83–2400	800–1900	200–900	5000–12 000
Mo	2.5–280	15–150	9.3–29	1.8–44
N	110–900	—	—	1600
Na	2900–42 000	15 000–57 000	7600–29 000	720–3400
Ni	7–4300	60–260	19–710	20–310
O	400 000–500 000	—	—	—
P	1400–6400	4800–9600	1700–4600	—
Pb	98–14 000	5300–26 000	2500–10 000	3300–22 000
S	1000–5000	11 000–45 000	1400–25 000	2700–6000
Sb	10–430	260–1100	300–1100	80–200
Se	0.05–10	0.4–31	0.7–29	—
Si	91 000–310 000	95 000–210 000	36 000–120 000	78 000
Sn	2–380	550–2000	620–1400	340–450
Sr	85–1000	40–640	400–500	5–300
Ti	2600–9500	6800–14 000	700–5700	1400–4300
V	20–120	29–150	8–62	25–86
Zn	610–7800	9000–70 000	7000–20 000	8100–53 000

Source: Modified from the IAWG [10].

Pb, Al, Cu, and Zn to the bottom ash of mass burn systems. The ranges of these elements in the grate siftings of mass burn systems has been reported as Pb (5600–34 000 mg/kg), Al (38 000–63 000 mg/kg), Cu (2400–25 000 mg/kg) and Zn (2450–5300 mg/kg). These metals are largely present in their elemental form and may enter into redox reactions, generating H₂ and causing swelling. Investigations have indicated that the grate siftings contain almost 50% of the lead in the residues considered available for leaching [10].

Total dissolved solids content (e.g., from the dissolution of NaCl, CaCl₂, CaSO₄) is fairly low in bottom ash, ranging from 3 to 14 wt% in modern mass burn facilities.

The major elemental constituents in APC residues are O, Si, Ca, Al, Cl, Na, K, S and Fe. Zn, Mg and Pb are often found in concentrations exceeding 10 000 mg/kg, while Hg is usually found in concentrations below 10 mg/kg. Other trace elements (i.e., Cu, Sb, Cd, Sr, Ni, As, Ag, Co, V, Mo and Se) are present in concentrations below 1000 mg/kg.

The total organic content (TOC) is generally below 10 000 mg/kg for all types of APC residues. The pH for dry/semidry APC system residues typically exceeds 12, while wet scrubber sludges have pH values around 10.5. The excess lime in the dry/semidry system results in higher buffering capacities compared to electrostatic precipitator (ESP) fly ash and wet sludge.

MWC residues and ashes contain small amounts of polychlorinated dibenzo-*p*-dioxins/polychlorinated dibenzofurans (PCDDs/PCDFs). In most studies, the levels detected in ashes from modern facilities have been well below levels of concern. In the US EPA/Coalition on Resource Recovery and the Environment (CORRE) MWC Ash Study, ashes sampled from five modern mass burn facilities contained PCDD/PCDFs at levels below the Center for Disease Control recommended level of 2,3,7,8-tetrachlorodibenzodioxin (TCDD) Toxicity Equivalency of 1 part per billion in residential soil [11, 12]. Note that this level is based on health risk assessments of ingesting soil daily over a lifetime.

3. Leaching of the residues

Perhaps, the most controversial issue associated with MWC residues is that of leaching or perceived leaching of potentially toxic contaminants, particularly Pb and Cd. Over the years this controversy focused on the fact that at times some of the residues failed the EPA's Extraction Procedure for Toxicity (EP_{TOX}) and later the Toxicity Characteristics Leaching Procedure (TCLP) test used to determine if a waste was hazardous based on 'toxicity' characteristics. Even so, these residues were not routinely being managed as hazardous waste because they were considered by many to be exempt from the Resource Conservation and Recovery Act (RCRA) subtitle C hazardous waste regulations [13]. Some states, however, have required disposal of the residues into subtitle C-like facilities. This issue eventually led to litigation that was recently decided by the US Supreme Court, which ruled that the residues are not exempt from testing [14].

Technical arguments have been made that laboratory tests such as the TCLP grossly over estimate the actual release of contaminants and do not measure the release rates that one would expect under field conditions. Significant field data supports this argument.

In general, laboratory leaching tests are designed either to simulate a field leaching scenario to measure a specific fundamental leaching property of the material being tested. Specific leaching properties to be measured are availability, solubility, and release rate. Availability is defined as the potential quantity of a species which may leach under the specified conditions over a prolonged period of time (e.g., 1000-10 000 yr). Results of availability tests are reported in units of mass leached/mass of initial material (typ.: mg Pb/kg ash) and are not indicative of the time frame over which release may occur. Availability tests are characterized by high liquid-to-solid ratios (e.g., 100:1), small particle sizes of material to be tested (e.g., <300 μm), extreme conditions (e.g. relative to pH), and contact times which permit dissolution to be achieved. Solubility tests are designed to determine the concentration of specific species in solution under saturated conditions. Results of solubility studies are reported in concentration units (typ.: mg/l). Solubility tests are characterized by low liquid-to-solid ratios (e.g., 6:1), small particles sizes of material to be tested, varied conditions (e.g., relative to pH), and contact times which permit chemical equilibrium to be achieved between the solid phase and solution. Release rate tests are used to estimate the rate of release of specific species from a material. Results are usually reported as a rate per mass or surface area basis (typ.: mg released/sq material/day) or mg released/sq m/day or for diffusion-controlled release rates as an effective diffusion coefficient. Release rate tests typically are carried out using column flow studies for granular materials and tank leaching studies for monolithic materials. Serial batch tests can also be used to determine release rates over time. Determination of specific leaching properties permits estimation of releases under varied environmental conditions while employment of leaching tests which simulate a specific disposal scenario do not readily allow transferability of results.

Considerable confusion and debate has resulted from the misinterpretation of leaching test results. For example, the TCLP was designed to simulate codisposal of a material with MSW in a landfill. Thus, results from TCLP testing are not indicative of potential releases under different environmental exposure scenarios. Results of extensive laboratory leaching tests have indicated that the variables which impact release potential of heavy metals the most significantly are solution final pH and liquid-to-solid ratio. Typically, cadmium solubility, and hence release, increases substantially with decreasing pH at pH less than 8. Lead release increases substantially with decreasing pH at pH less than 6 and also increases substantially with increasing pH at pH greater than 10. Thus, with a solution pH between 8 and 10 a minimum release of both lead and cadmium occurs.

The results of synthetic acid rain extractions have not been significantly different from extractions with distilled water because of the high alkalinity of the residuals. Liquid-to-solid ratio impacts most leaching test results because most leaching tests are carried out at a liquid-to-solid ratio where a saturated solution (solubility-con-

trolled) exists at the end of the contacting interval. Thus, greater liquid-to-solid ratios result in release of a greater mass of heavy metal. However, it is important to note that even at extreme laboratory leaching conditions, tests have shown that only a small fraction (typically much less than 10%) of the total heavy metal element present in the residual is released. Conversely, most of the alkali salts present (i.e., Na, K, Cu, Cl and SO₄) in the residuals are readily available for leaching and are released independent of solution pH.

Field studies of leaching of combined MWC residuals in monofills have indicated that leachate concentrations of heavy metals are overestimated by laboratory leaching tests. Typical leachate concentrations observed for these metals are below or near drinking water standards. However, total dissolved solids (soluble salts) concentrations are several orders of magnitude greater than drinking water standards and, at low liquid-to-solids ratios, are comparable to sea water.

The EPA/CORRE MWC-Ash Study sampled and analyzed leachate from the ash disposal units of five mass-burn MWC facilities. The data from this study indicated that all the metals were below their EP_{TOX} maximum allowable limit. Also indicated by the data was that some of the metals concentrations in the leachate met primary or secondary drinking water standards and the leachate did not contain significant quantities of PCDDs/PCDFs. Those that are most often found are the highly chlorinated homologs, which are the ones with the relatively lower toxicity equivalency factors (TEFs) [11].

This study also investigated leaching of ashes collected from the facilities with six different extraction procedures, including:

- (1) acid 1 (EP_{TOX}),
- (2) acid 2 (TCLP Fluid 1),
- (3) acid 3 (TCLP Fluid 2),
- (4) deionized water (method SW-924, also the Monofill Waste Extraction Procedure or MWEP),
- (5) CO₂ saturated deionized water, and
- (6) simulated acid rain (SAR).

Analysis showed that the EP_{TOX}, TCLP₁ and TCLP₂ extracts contained higher levels of the metals than did extracts from the other procedures. While the extracts from the EP_{TOX} and TCLP procedures occasionally exceeded the EP toxicity allowable limit for some metals, extracts from the other three tests did not. The deionized H₂O, the CO₂ and the SAR extractions procedures more closely simulated the concentrations of Pb and Cd in the field leachates.

Some results from 5 years of leachate sampling and analyses from the North Marion County (Oregon) Disposal Facility (Woodburn Monofill) are summarized below [12].

The major leachate constituent is salts and the main salt constituents are chloride, sulfate, calcium, and sodium.

- All metal concentrations in the leachates were below the EP_{TOX} and TCLP maximum allowable limits.
- Iron, manganese and lead exceeded federal safe drinking standards in leachate from the closed cell; several metals (Al, Ba, Cd, Fe, Pb, Mg and Hg) exceeded

those standards in the leachate from the active cell. (The closed cell is no longer used and has been covered to keep it dry according to regulatory/permit requirements. The active cell is still in use and does not have a final cover.)

- Salt concentrations in the leachates have generally decreased over the 5-year period. In one exception the sulfate in the closed cell increased during one sampling year, but then decreased during the last sampling period.
- Levels of pH in the closed cell have remained fairly constant between 6.7 and 7.0. In the active cell, the pH has ranged from 5.7–6.1. (At this pH, $\text{CO}_2(\text{g}) \rightleftharpoons \text{CO}_2(\text{ag})$ has been reached. This does not happen in fresh leachates or in laboratory tests where CO_2 uptake is controlled.)

Results from these and similar studies tend to verify technical arguments that the TCLP does over estimate the actual release of metals that one would expect to find in the field.

There are many factors which affect the leaching of constituents from ash. Data compiled by the International Ash Working Group (IWAG) of results from regulatory leaching tests from several countries indicate that concentrations of metals in most of the extracts are solubility-controlled [10]. These results can be modified by changes in ash alkalinity when the final pH of the extraction procedure is not specified. The California WET test is an exception because the metal complexation by citric acid results in availability-controlled leaching.

The approach (proposed by the IAWG and which is a modification of that proposed for Netherlands) to determine how the ash can ultimately be managed involves several tests. First, total concentrations of ash constituents are determined using appropriate analytical methods such as neutron activation analysis, X-ray fluorescence, and total digestion/digestate quantification. This is followed by an assessment of the amount of these constituents available for leaching under a presumed worst-case condition. This test is referred to as the availability leach test; it determines the maximum quantity of a specific element or species that can be expected to leach over a geologic time frame (e.g., 1000–10 000 yr). The test uses a high liquid-to-solid ratio (e.g., equal to or greater than 100), acidic final pH 4 for cation leaching and pH 7 for oxyanion leaching. Availability-controlled leaching results should be presented as the measured release (mg element leached/kg of material extracted). The composition of the initial extraction solution, liquid-to-solid ratio, and final pH also should be indicated.

The observed release of specific elements or species of interest under availability-controlled leaching conditions is independent of the conditions used to reach availability-controlled conditions. The following are several experimental conditions which result in availability-controlled release:

- Liquid-to-solid ratios greater than 10, concurrent with pH less than or equal to 4, results in availability-controlled leaching for most elements and species of interest. This is similar to the Dutch availability test.
- Liquid-to-solid ratios greater than 10, concurrent with near neutral pH and 0.2 molar citric acid, results in availability-controlled leaching for most elements and species of interest. These are conditions met by the California wet test.

- Liquid-to-solid ratios greater than 10, concurrent with near neutral pH and 0.016 molar EDTA, results in availability-controlled leaching for most elements and species of interest.

All three of the above availability-controlled leaching conditions result quantitatively in the same release of a specific element or species. The exception is Pb from APC residues, which often shows a maximum availability at pH greater than 11.

The availability of specific elements or species of interest is a fraction of the total concentration present in the ash and may be significantly different for different ash types. Grate siftings are significantly higher in Cu and Pb availability compared to other ash types. APC residues are significantly higher in Cd, Pb, Zn, Cl, Br, and SO₄ availability compared to other types, with the possible exception of Pb in grate siftings. The total soluble mass fraction of APC residues is much greater than that of other ash types.

After determining the amount available for leaching, column or serial batch leaching tests can be used to assess the release of elements or species as a function of time. Tests such as the Monolith Leach Test, a modification of the ANSI 16.1 test, and the ANSI 16.1 test can also be used to assess the release of the selected species over time, but do so under different scenarios (diffusion compared to percolation). Tests used, however, must consider the nature of the material to be tested [e.g., granular (solubility controlled) or monolithic (diffusion controlled)].

Based on the information gained from these tests, one can develop estimates of expected releases over time for species of concern when placed in various disposal or utilization options. Cumulative release in the field can be estimated on knowledge of the unified pH curve, availability, anticipated field pH and the anticipated liquid-to-solid rates. If field redox conditions are also known, estimates can be further refined.

Aging or weathering of ash normally results in a decrease of leachate pH towards neutral. One aging reaction results from uptake of CO₂ and self-neutralization of the ash. Other aging reactions that promote metal immobilization include (i) hydrolysis of oxides to hydroxides, (ii) the weathering of glassy slags to form illite-like clays, and (iii) the oxidation of elemental metal (like Fe) to form oxyhydroxide surface coatings. These changes result in decreased solubility of many elements and consequently decreased release.

Once a class or type of residues has been completely characterized with respect to leaching behavior and chemical/physical characteristics then it should only be necessary to apply a good QA/QC program to certify that the residue is the same as others in the class and acceptable for a given utilization option. This or a similar process is not being used in the United States, except in the case of some utilization demonstrations.

4. Treatment options

There are several options available for treating MSW combustion residues. The need for treatment is not clear and depends on site-specific conditions, including

regulatory requirements and utilization or disposal objectives. Treatment options include:

Ferrous metal recovery: Magnetic separation and screening of ferrous metal from the bottom ash is routinely practised at many WTE facilities. Ferrous metal accounts for approximately 15% of the bottom ash (from mass burn facilities) and is normally sold as scrap. The ferrous metal content of the bottom ash from RDF combustion facilities may be less due to more extensive processing of the MSW prior to combustion. In some cases, separation of non-ferrous metals by eddy current separations may also be practiced.

Compaction: The physical properties of the residues are such that compaction at optimum moisture content can be effective in significantly reducing the permeability of the residues when placed in a monofill or other disposal/utilization options. This reduces infiltration of water and, therefore, the potential for release of contaminants.

Tests have shown that bottom ash can be compacted to Proctor densities of 1600 kg/m³ or more. At optimum moisture content of approximately 16% and optimum compaction effort, saturated hydraulic conductivities are around 10⁻⁶ cm/s. Depending on ash characteristics, optimum moisture contents and compactive effort may change. Laboratory and field permeabilities of less than 10⁻⁸ cm/s have been reported. Compacted bottom ash exhibits good bearing capacities as measured by the California bearing ratio test.

Classification: Screening or separation of ashes into oversized and undersized fractions can be beneficial and is in some cases necessary for utilization purposes. Screening coarser material (3/4-in or perhaps 3/8-in) from bottom ash improves the characteristics as a coarse aggregate. Bottom ash used in the Laconia, New Hampshire asphalt paving project was screened to less than 3/4-in. This ash also did not contain grate siftings or heat recovery ash and had been aged for 5 months [9]. From a physical perspective, fly ash can be screened to meet specifications for fine cement aggregate (ASTM C33) [15].

Solidification/stabilization (S/S): S/S refers to technologies or processes that use additives (or binders) to physically and/or chemically immobilize hazardous constituents in wastes, soils and sludges [16, 17]. Solidification normally refers to the conversion of a liquid or semiliquid to a solid form. While solidification does not necessarily involve a chemical interaction of the constituent of concern with the solidifying agent, the process can restrict contaminant mobility by encapsulating the contaminant in a treated product of reduced surface area, lower permeability and better handling characteristics. Stabilization converts the contaminants into less soluble or less toxic forms without necessarily achieving solidification. The best approach to S/S normally is to chemically stabilize, then solidify the waste. The most common binders used are inorganic systems based on cement and pozzolanic materials, although some hazardous waste S/S processes have used thermoplastic binders, synthetic polymers and organophylic clays. The process involves mixing binders with the waste and water, sometimes incorporating additives such as sodium silicate, cement kiln dust, coal fly ash, bentonite or proprietary materials. The ratio of waste, binder, other additives and water should be tested to optimize the final product

integrity and performance (contaminant leachability, strength, curing rate, etc.) to meet required treatment objectives. Depending on the S/S process, the final product may be either monolithic or granular in nature.

S/S processes using portland cement and similar binders will usually result in a treated product with more weight and increased volume. This could significantly affect transportation and disposal costs. The sequence of binder additives and mixing may significantly affect the quality of the product and its ultimate performance. Adequate mixing is also important.

S/S processes are not effective for treating soluble salts. Therefore, since APC residues contain high levels of salts, leaching of these salts from the S/S matrix will likely result in poor performance. The subsequent loss of physical properties and durability of the treated product may result in increased release of the metals.

The addition of portland cement and other additives is being practiced by several facilities in the United States. One example is the Commerce Refuse-to-Energy Facility (CREF) located in Commerce, CA [18]. The treatment facility consists of bottom ash conveying, separation and storage equipment; a fly ash conveyor and silo; and treatment equipment similar to a concrete batching plant. The treatment process mixes screened bottom ash with fly ash, Type II portland cement and water. A standard concrete mixer truck is used to mix the batch. The slurry is discharged into roll-off-bins. After setting, the resulting approximate 16-ton monolith is transported to a non-hazardous waste landfill where it is crushed and used as road sub-base aggregate. The bottom ash larger than 2" in size is removed by screening prior to treatment.

The WES-PHix Ash Immobilization Process, patented by Wheelabrator Environmental Systems Inc., can be classified as an S/S process. The process uses water-soluble phosphates and alkali to reduce metals solubility by the formation of insoluble (or less soluble) mineral phases. Treatment by the phosphate may result in insoluble phosphate compounds of lead, copper and zinc. The reduction of Pb leachability was confirmed by tests conducted by the EPA in an evaluation of S/S processes for treating MSW combustion residues [4, 19]. This evaluation did not confirm that Cd was chemically treated. The WES-PHix process is being used at several facilities in the United States.

S/S processes can be effective in treating MWC residues. However, several factors must be carefully considered.

- The high concentration and ultimate fate of soluble salts must be carefully considered in the design of the process. Pretreatment to remove these may be required.
- Evaluation of the S/S process design, performance, and treatment efficiency should be based on a matrix of several testing protocols. A single test such as the TCLP cannot provide the information required to completely evaluate the potential for release of contaminants or physical durability.
- Many of the available S/S processes, particularly those using binders such as portland cement, will increase volume. This could result in added transportation and disposal costs.

Chemical extraction: Processes for chemically extracting metals have been researched and developed at the laboratory scale [20-22]. Chemical extraction has

not been practiced on a commercial scale primarily because of economics. Extraction of soluble salts has also been demonstrated in the laboratory and is less expensive, but also, has not been implemented in the United States.

Vitrification and other thermal processes: Vitrification results in the melting of the ash with glass forming additives to incorporate the contaminants into an alumina-silicate matrix. The process can substantially reduce the volume of materials 60% or more and usually results in a product more resistant to leaching, however, there are several concerns. Costs can be high, ranging from \$100 to \$200 per ton. Release of contaminants during melting (e.g., organics and volatilization of some metals) may require additional air emissions control and subsequent treatment of the collected residues. Vitrification, while researched and demonstrated in the United States has not been implemented for treating MWC residues [22–25].

Heating of the residues can also cause sintering, mineral respeciation or melting to 'fuse' the material into a slag. Bottom ash fusion and perhaps vitrification is practised at several facilities in Japan, but not in the United States.

The necessity for treating residues depends on several factors. These include local regulatory requirements, disposal objectives, specifications required for utilization, liability considerations, economics, and other site-specific factors.

5. Disposal

The prevailing method in the United States for managing combined residues is by monofill disposal. The monofills are lined with clay, synthetic liners or a combination of both and have provisions for leachate collection and treatment. Even before the recent Supreme Court decision which ruled that residues were not exempt from RCRA Subtitle C hazardous waste regulations, some states required testing of ashes to determine if they should be classified as hazardous. Some states already required disposal into landfills very similar in design to subtitle C landfills. In some instances, they were managed as a hazardous waste if they failed these tests. The ruling now means that owners of WTE facilities must determine if ashes are hazardous. If they are, owners will be required to manage the ash as a hazardous waste. This could result in ultimate disposal into a Class C hazardous waste landfill and added costs. On the other hand, if the ashes are determined not to be hazardous they could be classified as a subtitle 'D' non-hazardous waste and disposed into subtitle D, part 258, municipal solid waste landfills. Some states, however, may still require more stringent requirements. Note: At the time this paper was being revised, Congressional legislation was being developed with provisions that would remove the requirement for testing, require disposal in subtitle D, part 258 monofills with an additional synthetic liner, and permit very limited utilization in asphalt paving or similar uses.

In some countries residues are separated into bottom ash and APC residues prior to management. The APC residues are treated, usually with S/S techniques, prior to disposal into landfills. In Germany APC residues may be stored in salt mines. In some instances in Denmark ashes have been placed in fills near the ocean and allowed

to leach into the more saline ocean waters. Some countries are also considering the concept of 'controlled contaminant release'.

The controlled release concept implies that residues would be managed in a manner that controls release rates and the quality of the leachate within preestablished levels that are consistent with surrounding conditions. The leachate is allowed to discharge into the surroundings as it is formed as long as it does not exceed preestablished acceptable levels. Siting is critical. This concept is being considered by some countries as a result of concern that 'dry entombment' in lined and capped landfills will result in greater uncontrolled release of contaminants in the future. Controlled release is not being used or considered, at least on a regulatory basis, in the United States.

6. Utilization

In some European countries 50% or more of the bottom ash is utilized but utilization of MWC residues is not routinely practised in the United States. There has, however, been significant interest in evaluation of utilization alternatives including a number of research and demonstration projects. These projects include asphalt pavement, construction blocks, artificial reefs, shoreline erosion control, and similar applications. Research to improve fundamental knowledge of how residues perform in alternative situations include projects to determine the elemental speciation and ultimate life of constituents in untreated and treated residues [26]; mobility of dioxins and furans from stabilized residue in seawater [27]; and environmental evaluation of a boathouse constructed of blocks containing MWC residues [28]. Results from these types of projects and from actual field demonstrations are providing data critical in evaluating, designing and implementing utilization options to protect the environment and human health.

Primary applications under consideration in the United States include the use of bottom or combined ash as an aggregate substitute in portland cement, as bituminous concrete (pavement), as sub-base in roads, as fill material under contained conditions such as covered embankments, and as daily cover in landfills. Marine applications (e.g., shore erosion control and artificial reefs) are also under consideration.

The following summarizes MWC utilization status in other countries:

Canada: Utilization is not currently being considered.

Germany: Approximately 50% of bottom ash is utilized primarily as road bases and sound barriers on autobahns. There are also pilot-scale tests being conducted to evaluate APC residue as grout in coal mines. Another consideration involves use of APC residue to form alinite cement.

The Netherlands: The goal in The Netherlands is to utilize at least 80% of the residues. Currently 40% of the fly ash produced is used as a fine aggregate in asphalt. More than 2 Mt of bottom ash (approximately 60%) have been used as road base, embankments, aggregate in concrete and aggregate in asphaltic concrete. To reach this goal, however, the Netherlands has established a policy and

regulatory framework which establishes standard leaching tests, composition requirements, evaluation protocols for building materials, and certification of residues for utilization.

Denmark: Utilization of bottom ash was initiated in Denmark in 1974. Approximately 72% of the bottom ash, including grate siftings and heat recovery ash, screened, is used as subbase at parking lots, bicycle paths and paved and unpaved roads. Requirements for ash composition for utilization are pH > 9.0 in a 1% slurry, alkalinity greater than 1.5 eq/kg, Pb content below 3000 mg/kg, and Hg content less than 0.5 mg/kg, as determined by HNO₃ digestion. In addition, there are restrictions which include a 20 m minimum distance to water wells, 1-2 m maximum thickness under a pavement, and a maximum of 200 m³ and 0.30 m maximum thickness for unpaved applications.

Sweden: Sweden has adopted a policy that utilization should improve the general environmental conditions and create a smaller environmental impact than disposal. Regulations for MWC ash utilization are currently being developed.

Other countries utilizing bottom ash include Japan, France and Switzerland. In 1992, Japan used approximately 10% of their bottom ash, and is exploring additional use. In 1992, France used approximately 64% of the bottom ash produced, while in 1991 Switzerland used 26% of their bottom ash.

Based on technical data and findings by the IAWG and others, the following factors should be considered in selecting, designing and implementing MWC residue utilization:

- Bottom ash can be effectively utilized as an aggregate substitute in several applications if appropriate engineering (structural) criteria and environmental performance guidelines are met. Large-scale utilization projects processed as structural soil in Europe include its use in land reclamation from the ocean, wind barriers, sound barriers, road sub-bases and parking lot and bike path bases. Demonstrations in the United States have bottom ash as aggregate substitutes in bituminous pavements, sometimes at high substitution rates. In other demonstrations, bottom ash has been used as an aggregate substitute in portland cement applications for marine reefs, shore protection devices and service buildings.
- The high alkalinity caused by the addition of cement in S/S applications influences the release of some metals in a negative manner. This must be considered in the design.
- Use of bottom ash as structural fill in large applications such as embankments may lead to high salt loads and relatively high leachability of salts. Pretreatment to remove soluble salts may be necessary in some applications.
- The use of bottom ash should be investigated in stabilized road-based applications using cold-emulsion bituminous materials, and as final cover at landfills.
- Ferrous and non-ferrous metals should be separated from bottom ash prior to use as an aggregate.

- Bottom ash should be stockpiled prior to utilization for an estimated one to three months with adequate moisture so that swelling, hydration, carbonation and oxidation reactions can occur. This benefits structural durability and chemical stability. Appropriate methods should be used to control and manage any leachate and/or runoff during stockpiling.
- The design, construction and implementation of any MWC residue utilization project should be based on a complete understanding and knowledge of the ash characteristics, the ash product behavior in the situation, environmental conditions, physical requirements, regulatory requirements and similar factors. This will necessitate appropriate testing and evaluation protocols, including tests to determine elemental concentration, availability of trace metals and total soluble salts for release, estimates of contaminant release over time and under field conditions, physical durability and similar factors. This cannot be done by relying on a single test such as the TCLP.
- The acceptability of a product for utilization should be based on the amount of anticipated contaminant release (assuming that physical and other specifications are met) over the expected product lifetime. This should take into consideration both contaminant release potential and release rates. The acceptable level of contaminant release should be defined for each utilization – can be based on release rates, the potential of similar products from natural materials, and environmental considerations.
- The time frame of intended use should be considered and an analysis conducted on the ultimate fate of the material following completion of the specified use period. Many of the utilization scenarios for bottom ash have release rates or release potentials similar to natural materials.
- Granular material (e.g., roadbase) will behave differently than monolithic applications and require different testing protocols (e.g., compacted granular leach test). Monolithic ash products should be evaluated using appropriate monolithic leach tests (e.g., tank leaching) and also for durability.
- Appropriate quality control and quality assurance procedures should be developed and implemented to maintain ash product quality.

There are a number of other factors to consider, such as distance to sensitive natural areas, local requirements, monitoring during the product use and similar factors.

Although not routinely practised in the United States, mounting evidence supports the technical argument for separating fly ash from bottom ash and further separation of grate siftings from bottom ash. While this may not be necessary for disposal in lined monofills, as evidenced by the quality of leachates from several disposal facilities, separation definitely appears valid for most potential utilization scenarios. Grate siftings are greatly enriched in the fraction of Pb most available for leaching and Al, which can cause hydrogen generation, subsequent swelling and reduced physical durability.

The APC residues contain relatively high levels of soluble salts and trace metals. Soluble salts are difficult to treat and they readily leach from the matrix.

7. Summary

Combustion to recover energy and reduce the volume of waste that requires landfilling is an important factor in the management of MSW in the United States. This process is applied to approximately 16% of the MSW generated. In other countries, from 30% to 70% is combusted. This generates residues which must be managed in ways that protect human health and the environment.

MSW residues contain varying amounts of trace elements that raise questions covering their classification and management. Cadmium and lead represent chief concerns, although soluble salts could also be problematic if not properly managed.

Some MWC ashes have failed regulatory leach tests such as the TCLP, primarily for Pb and Cd. Technical arguments that these tests do not mimic field conditions are supported by analyses of leachates from ash monofills, which have shown levels of trace metals below the TCLP regulatory limits and, in significant cases, below drinking water standards. The major constituent in the leachates from these sites has been salts.

Residues are primarily managed in the United States by disposal into lined monofills. Their utilization is not routinely practised in the United States. Opportunities and technical basis exists, however, for utilization.

Utilization options being considered and demonstrated include aggregate for fill, aggregate in asphaltic applications, concrete applications (e.g., shoreline erosion control, construction blocks), landfill daily cover, and others.

For utilization, technical data suggest that APC residues and grate siftings should be collected separately from bottom residues. Grate siftings contain enriched levels of Pb, with almost 50% of the Pb available for leaching, and Al, which may result in hydrogen generation. APC residues are enriched in trace metals and contain the largest fraction of leachable Cd. They also have high concentrations of soluble salts.

Residues for utilization should be thoroughly evaluated for constituent concentrations, concentrations of species of concern that are available for leaching, release rates over time, and expected contaminant releases over time for the utilization option under consideration. Protocols are available to determine these factors. One cannot rely on a single test such as the TCLP to make these assessments. Once a class of residues has been properly characterized, one can rely on good quality assurance/quality control programs to assure that required specifications are met.

Based on technical considerations, processed bottom residues should be considered for utilization. Research and demonstrations are providing data which will help verify proper utilization in a manner protective of human health and the environment. The ultimate effect of the Supreme Court's decision on ash management and particularly ash utilization remains unclear. However, certain ashes will now have a legal basis to be classified as not hazardous. This could help alleviate some objections to ash utilization. Utilization, however, must follow sound scientific and engineering principles and be conducted with appropriate measures to assure that it is acceptable to the environment and to human health. Other countries have taken this approach and are utilizing these residues successfully. Why shouldn't the United States?

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References

- [1] J.V.L. Kiser, Update of US Plants, The IWSA Municipal Waste Combustion Directory: 1993, Integrated Waste Services Association, Washington DC, 1993.
- [2] Energy-from-waste, 1993 Activity Report; Solid Waste & Power, HCI Publications, Kansas City, MO, 1993.
- [3] Toxicity characteristics leaching procedure (TCLP), Federal Register, 1986, Vol. 51, p. 40 643.
- [4] D.S. Kosson, T. Kosson and H. van der Sloot, Evaluation of solidification/stabilization treatment processes for municipal waste combustion residues, EPA/600/SR-93/167, Risk Reduction Engineering Laboratory, US EPA, Cincinnati, OH, 1993.
- [5] C.C. Wiles, in: Proc. National Solid Waste Forum, Portland, OR: ASTSWMO, 1992.
- [6] D.S. Kosson, B.A. Clay, H.A. van der Sloot and T.T. Kosson, in: J.J.J.M. Goumans, H.A. van der Sloot and Th.G. Aalbers (Eds.), Environmental Aspects of Construction with Waste Materials, Elsevier, Amsterdam, 1994.
- [7] D.S. Kosson and F.E. Hoffman, in: Proc. Int. Power Generation Conf., American Society of Mechanical Engineers, Atlanta, GA, 1992.
- [8] W.H. Chesner and F.J. Roethal (Eds.), Proc. Fifth Int. Conf. on Municipal Solid Waste Combustion Ash Utilization, Arlington, VA, 1992.
- [9] C.N. Musselman, T.T. Eighmy, D.L. Gress, M.P. Killeen, J.R. Presher and M.H. Sills, New Hampshire bottom ash paving demonstration US Route 3, Laconia, New Hampshire, Proc. 16th Biennial 1994 National Waste Processing Conf., ASME, New York, NY, 1994, pp. 83–90.
- [10] An international perspective on municipal waste incineration residue characterization and management; Int. Conf. Environmental Implications of Construction Materials and Technology Developments, Maastricht, The Netherlands, 1–3 June 1994, The International Ash Working Group (IAWG).
- [11] H.K. Roffman, in: Proc. Municipal Waste Combustion Conf., Papers and Abstracts from the Second Annual Int. Specialty Conf., Air & Waste Management Association; Tampa, FL, 1991, p. 96.
- [12] H.K. Roffman, in: Proc. Sixth Int. Conf. on Municipal Solid Waste Combustor Ash Utilization, Resource Recovery Report, Arlington, VA, 1993.
- [13] Resource conservation and recovery act of 1976 (RCRA), PL 94-580, 21 October 1976 as amended through PL 102-389, 6 October 1992.
- [14] US Supreme Court, City of Chicago et. al., Petitioners vs. Environmental Defense Fund et al., 114 S. Ct. 1588, 1994.
- [15] R.W. Goodwin, in: Solid Waste & Power, HCI Publications, Kansas City, MO, 1992.
- [16] C.C. Wiles, *J. Hazard. Mater.*, 14 (1987) 5.
- [17] J.R. Conner, Chemical Fixation and Solidification of Hazardous Wastes, Van Nostrand Reinhold, New York, 1990.
- [18] M. Eaton and J. Korn, in: Proc. Fifth Int. Conf. on Municipal Solid Waste Combustor Ash Utilization, Arlington, VA, 1992, p. 141.

- [19] C.C. Wiles, Proc. Municipal Waste Combustion, Tampa, FL, Air & Waste Management Association, 1991.
- [20] I.A. Legiec and D.S. Kosson, in: A.I.Ch.E. 1989 Summer National Meeting, Philadelphia, PA, 1989.
- [21] I.A. Legiec, Design and scale-up of heavy metals recovery from incinerator ashes (B.S. and M.S. Rutgers, Graduate Assistant, 1989 J. Lagrosa Award NJWPCA).
- [22] Solid Waste & Power staff, in: Solid Waste & Power, HCI Publications, Kansas City, MO, 1991.
- [23] T. Kosson et al., in: Municipal Waste Combustion, Proc. Int. Specialty Conf., Williamsburgh, VA, Air & Waste Management Association, Pittsburgh, PA, VIP-32, 1993, p.784.
- [24] L.L. Oden and W.K. O'Connor, ASME/US Bureau of Mines Investigative Program Report on Vitrification of Residue (Ash) from Municipal Waste Combustion Systems, The American Society of Mechanical Engineers, CRTD, Vol. 24, United Engineering Center, New York, NY, 1994.
- [25] D. Wexell, in: 4th Annual ACS Division of Industrial Engineering Technology for Hazardous Waste Management, Atlanta, GA, 1992.
- [26] T.T. Eighmy et al., in: Municipal Waste Combustion, Proc. Int. Specialty Conf., Williamsburgh, VA, Air & Waste Management Association, Pittsburgh, PA, VIP-32, 1993.
- [27] F.J. Roethel and V.T. Breslin, *Chemistry and Ecology*, 10 (1995) 259.
- [28] F.J. Roethel and V.T. Breslin, Proc. Third Int. Conf. on Municipal Solid Waste Combustor Ash Utilization, 1990, p. 237.