

Accelerated carbonation of municipal solid waste incineration fly ashes

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Accepted 8 June 2006

Available online 2 October 2006

Abstract

As a result of the EU Landfill Directive, the disposal of municipal solid waste incineration (MSWI) fly ash is restricted to only a few landfill sites in the UK. Alternative options for the management of fly ash, such as sintering, vitrification or stabilization/solidification, are either costly or not fully developed. In this paper an accelerated carbonation step is investigated for use with fly ash. The carbonation reaction involving fly ash was found to be optimum at a water/solid ratio of 0.3 under ambient temperature conditions. The study of ash mineralogy showed the disappearance of lime/portlandite/calcium chloride hydroxide and the formation of calcite as carbonation proceeded. The leaching properties of carbonated ash were examined. Release of soluble salts, such as SO₄, Cl, was reduced after carbonation, but is still higher than the landfill acceptance limits for hazardous waste. It was also found that carbonation had a significant influence on lead leachability. The lead release from carbonated ash, with the exception of one of the fly ashes studied, was reduced by 2–3 orders of magnitude.

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1. Introduction

The increasing amount of municipal waste produced is a worldwide environmental problem and incineration is a popular management option, particularly where recycling or reuse are not possible (European Union, 1999). Incineration can be used to recover energy and reduce the mass and volume of waste by 70% and 90%, respectively, and the ash produced can be recycled or disposed of to landfill. Currently, there are 13 incinerators in the UK with a total processing capacity of 2.9 million tons per year. During the incineration process, the hot gases produced from the waste burn are used to generate electricity and/or heat. The flue gas is treated with dry, wet or semi-dry lime and activated carbon for the removal of acid gases such as NO_x, SO_x and CO₂ to meet emission targets. Two categories of solid residues are produced in incinerator: bottom ash and air pollution control (APC) residues. Bottom ash is a heterogeneous mixture of slag, metals, ceramics, glass,

other non-combustibles and unburned organics, whilst the APC residues are fine particulates collected by the air treatment systems, and are a mixture of fly ash, lime and carbon. Usually APC residues and fly ash are interchangeable.

Fly ash is generally classified as hazardous waste according to the European Waste Catalogue (European Union, 2000; SEPA, 2003) because of the high levels of soluble salts and heavy metals such as cadmium, lead and zinc. The presence of lime also gives fly ash a high alkalinity, resulting in an increased potential for leaching, which is important when landfill is concerned.

The Incineration Directive (EU 2000/76/EC) has resulted in more stringent controls on gaseous emissions from incineration, so that more hazardous compounds will be retained in the solid residues. It has been reported that 314,000 tonnes of fly ash were produced in the UK during 1996–2000, of which nearly 90% was sent to landfill (Environment Agency, 2002). However, the implementation of the EU landfill Directive (EU, 1999/31/EC) has dramatically reduced the availability of landfill space for these hazardous wastes. Since July 2004, the number of sites registered to accept hazardous wastes has decreased from

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over 200 to less than 10. In addition, the UK government is increasing the standard rate of landfill tax by at least €4.4/1000 kg each year from €22/1000 kg in 2005 in order to encourage the reuse and recycling of waste. The long-term rate of the tax is €51/1000 kg (HM Treasury Budget, 2003). Therefore, the disposal of MSWI fly ash is becoming an increasingly costly management option.

There have been a number of research projects examining the disposal of MSWI ash (Poletini et al., 2001, 2004; Sakai and Hiraokab, 2000; Mangialardi, 2001, 2003; Mizutani et al., 2000; Mulder, 1996). The main techniques currently under investigation, or in use, are thermal treatment, stabilization/solidification (S/S) and washing-immobilization processes. Thermal treatment, such as sintering or vitrification (Mangialardi, 2001; Poletini et al., 2004), is costly and not widely used. The sintered residues are more hazardous than the untreated fly ash and become more difficult to dispose of (Sakai and Hiraokab, 2000). So far this technology is only used in a few countries, such as Japan, Korea and Sweden (Ecke et al., 2000). The S/S process employs cement or other chemical agents (such as furnace slag and soluble phosphate) to immobilize the contaminants (Eighmy et al., 1997). It is one of the most commonly used processes for the treatment of solid waste, sludge and contaminated soil (Poletini et al., 2001; Mizutani et al., 2000; Environment Agency, 2004). New mineral phases are generated during the process and the leaching properties of the waste are changed. S/S, however, may increase the mass and volume of the wastes leading to an extra cost of transportation and disposal. Alternatively, the contaminants in fly ash can be removed by washing (Mangialardi, 2003; Mulder, 1996). Water or acid is used as solvent and most of the hazardous compounds, i.e., soluble salts and heavy metals, are dissolved. Further steps are then followed to treat the leached residues. A major concern with this technique is the large amount of wastewater generated.

Alternatively, many researchers have been investigating the natural or accelerated aging of bottom ash, a process which could also be applied to fly ash disposal. Bottom ash can be used as a secondary construction material after several weeks of natural weathering in landfill sites (Environment Agency, 2002). It is known that a series of physical and chemical changes take place during natural weathering including hydrolysis, hydration, precipitation/dissolution, oxidation/reduction and carbonation (Meima and Comans, 1997, 1999; Sabbas et al., 2003). Poletini et al. (2003) claimed that treatment of bottom ash using CO₂, i.e., so called accelerated aging, can result in improved mineralogical, chemical and leaching properties. It has been shown that the formation of carbonate was one of the major changes affecting the leaching behaviour and acid neutralization capacity of the bottom ash (Chimenes et al., 2000).

Therefore, aging and weathering are known to affect the mineralogy, chemical and leaching properties, particularly the immobilization of heavy metals in waste and allow

the mature properties of the ash to develop. Accelerated carbonation may accelerate these natural reactions. In this paper, an accelerated carbonation treatment is proposed for the treatment of MSWI fly ash. The carbonation of fly ash was investigated and optimized, and the changes to mineralogy and leaching behaviour were examined.

The use of carbon dioxide in a treatment step for both bottom ash and fly ash would have the added environmental benefit of the permanent sequestration of carbon dioxide and open up the possibility of trading in carbon credits for the incineration companies.

2. Experiments and analysis

The ash samples were supplied by the Environment Agency and three incinerators in the UK. Their properties and the typical components of ash are listed in Tables 1 and 2. The ash was divided into two parts. One part was subdivided further for analysis. The other was treated by carbonation. The analysis of untreated and carbonated ash included moisture content, carbonate content, mineralogy, pH and leaching test.

2.1. Reaction procedure

Two reactors were used for carbonation. One was a stainless steel chamber incorporating a cooling plate to examine the influence of temperature. Water, used as the coolant, counteracted the heat of reaction and controlled the temperature. Changes in pressure and temperature were monitored using digital gauges. The second carbonation reactor was a closed chamber containing 100% CO₂ at an RH 75% operated at ambient laboratory temperature and was used to ascertain the optimum water/solid (w/s) ratio for ash carbonation.

Table 1
Ash samples

Sample name	Source	Water content	
EA FA1-3	Environment Agency, UK	0–2.5%	Aged
SE FA	SELCHP, Lewisham, London	0.5%	Fresh
CL FA	Cleveland Incinerator, UK	0.7%	Fresh
KI FA	Kirklees Incinerator, UK	0.35%	Fresh

Table 2
Main elements analysis of SELCHP fly ash

Elements	Amount (%)
CaO	36.268
Fe ₂ O ₃	1.054
K ₂ O	2.034
TiO	0.526
MnO	0.036
Pb	0.303
Ba	0.032
Zn	0.752
Cu	0.053
Ni	0.007

Before carbonation, the ashes were dried at 105 °C to constant weight and then mixed with water before being placed into the reactor. For each set of experiments, control samples were prepared in the same manner and then sealed and stored under ambient laboratory conditions for the same time period. The extent of the reaction of the ash with CO₂ was assessed by measuring the gain in weight between the initial dry powder and the dried product, being proportional to the amount of CO₂ that had combined with the sample. The measured value was compared to the weight gain experienced by the control sample.

To examine the influence of reaction temperature on the carbonation, the ashes were mixed with water and then placed in the reactor in a uniform layer of 3 mm. The reactor was closed tightly and filled up to a pressure of 3 bar with a 1:1 mixture of dry CO₂ and N₂. The reaction temperature was controlled by a cooling system. The pressure of CO₂ in the reactor was measured and used to calculate the consumption of CO₂ gas (in moles) by the ash.

To investigate the influence of water to solid ratio (w/s), values from 0.1 to 0.8 on carbonation were studied. Samples of fly ash (5 g) were carbonated for 3 h by exposing the ash to a 100% CO₂ atmosphere at 75% relative humidity and at a pressure of 3 bar. A saturated solution of NaCl was used to maintain a constant relative humidity in the chamber. The weight gain of the dry SELCHP ash was measured in triplicate.

Following these initial trials, the ash was carbonated in the closed chamber for 3 days at 3 bar pressure of CO₂ and a relative humidity of 75%. The samples were then dried in order to examine the carbonate content, mineralogy and leaching properties.

2.2. Analytical methods

The moisture content was measured after heating in an oven at 105 °C (BS 1377-2, 1990). The carbonate content was calculated from the weight loss on ignition between 450 °C and 900 °C. Thermogravimetric and differential thermal analysis (TG/DTA) of carbonated ash was performed with a Stanton Redcroft STA-780 thermo analyzer in a temperature range of 20–1100 °C at a heating rate of 10 °C/min. Mineral phases in ashes were determined by X-ray powder diffraction. The pH was examined according to BS 1377 part 3:1990.

The compliance leaching test was carried out in accordance to British Standard BS EN12457: 2002, which is designed to examine the short-term and long-term leaching behaviour for landfills. It is two-step leaching test with liquid/solid = 10 L/kg. The ash was leached at L/S = 2 L/kg for 6 h of end-over-end mixing and then filtered. The residues were leached further at L/S = 8 L/kg for 18 h. The eluate was filtered with a 0.45 µm filter paper and then divided into two solutions. One sample was measured by Ion-Chromatography (DIONEX) for the chloride and sulfate content. The other was acidified with nitric acid to pH < 2 for elemental analysis. The concentration of the

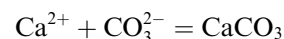
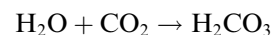
major elements Al, Ca, K, Na and minor elements As, Ba, Cd, Cr, Cu, Fe, Pb, Ni, Zn in leachate were analyzed by ICP-OES (Horizontal). All analyses were carried out in triplicate.

3. Results and discussion

3.1. Carbonation conditions

The variables that have influence on the reaction, such as water/solid ratio, reaction temperature, reaction time, etc., were studied in this project. Previous work about the influence of reaction time and particle size has been published (Fernandez Bertos et al., 2004b).

The carbonation mechanism can be considered a sequential reaction expressed by the following equations (Freyssinet et al., 2002):



It is known that water is necessary to promote the reaction of CO₂, but too much water limits the reaction due to the blockage of the pores in the solid (Fernandez Bertos et al., 2004a). Hydration and dissolution of CO₂ occur in the presence of water, as well as the dissolution of Ca²⁺ ions from the solid phase, which reacts with the CO₂ to form calcium carbonate. At low water–solid ratios, the gas permeability is high and the CO₂ effectively diffuses into the material. However, with the increase in water content, the pores in the ash are effectively sealed off. The diffusion of gas into the pore system is hindered, inhibiting the reaction.

The results of reaction with different w/s ratios are shown in Fig. 1. The optimum w/s ratio is 0.3 by weight of ash. However, some authors found the optimum ratio for the carbonation of cementitious systems to be between w/s 0.06 and 0.20 (Asavapisit et al., 1997; Klemm and

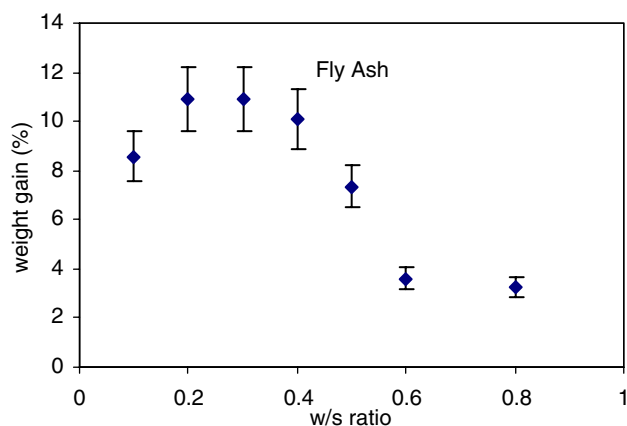


Fig. 1. Carbonation as a function of w/s ratio.

Berger, 1972) and others have successfully carbonated with values up to 0.35 (Yousuf et al., 1993).

The conversion at different temperatures is shown in Fig. 2, which shows the evolution of the reaction with time. As expected, the reaction proceeded more quickly at the beginning and then slowed down. The initial speed of reaction is higher at higher temperatures, although the final conversion achieved is lower. The opposite is the case at lower temperature. The highest weight gain is observed at 21 °C. Consequently, the following reactions were carried out at ambient temperature condition.

3.2. CO₂ uptake and carbonate content

As in the carbonation mechanism mentioned above, carbonate salts were formed during reaction. The DTA curve, shown in Fig. 3, indicates that the loss of pore water in carbonated ash occurred under 200 °C, and the decomposition of calcium carbonate happens between 450 and 850 °C. So the weight loss between 450 and 900 °C was considered to be the carbonate content of the sample. The combustion of organics happens between 300 and 450 °C, as is characterised by a positive ΔT peak (Johnson et al., 2003).

As previously explained, the extent of carbonation was also assessed by measuring the gain in dry weight before and after the reaction. However, there is a difference

between the weight gain recorded and the carbonate as determined by the TG/DTA. The gain in weight is greater than can be explained by the observed increase in carbonate content, suggesting that other reaction products are being generated. Fig. 4 shows the CO₂ uptake and the carbonate content increase of ashes. The individual fly ashes react with CO₂, by different amounts within the range of 2–7% by weight.

3.3. Mineral phases in ash

X-ray diffractograms of the ashes are shown in Fig. 5 and were used to determine the mineralogical changes that took place during carbonation. Table 3 lists the main mineral phases found in ash.

The most obvious differences between the diffractograms of original and carbonated samples are the increase in calcite peak intensity and the reduction of the lime/portlandite/Ca(OH)Cl peaks. Untreated ash contains lime/portlandite and calcium chloride hydroxide (Ca(OH)Cl), as the result of lime addition during gas treatment. These phases disappear after carbonation. There were also many Si–Al–Ca salts identified in the carbonated ashes, including gehlenite, braunite and larnite. For Kirklees fly ash, hydrated products, such as gehlenite, were also found after carbonation. So the difference of weight gain and carbon-

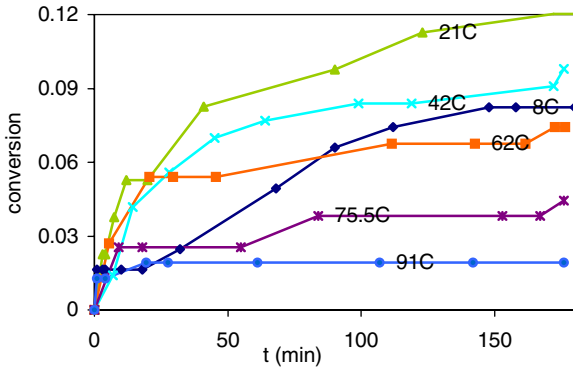


Fig. 2. Evolution of reaction at different temperatures.

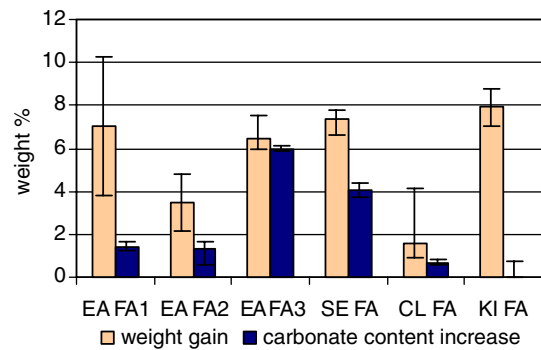


Fig. 4. Weight gain during reaction for the various fly ashes.

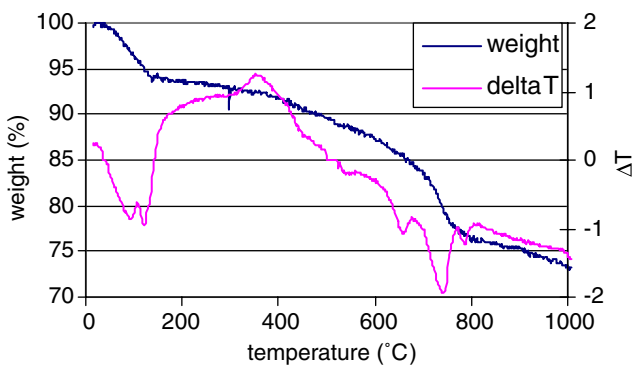


Fig. 3. TG/DTA curve of carbonated SELCHP fly ash.

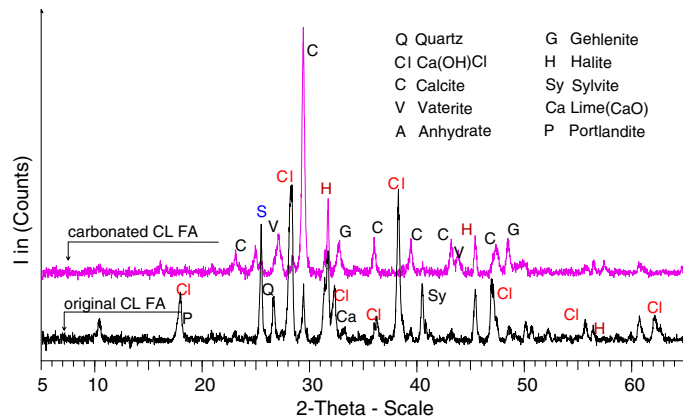


Fig. 5. XRD diffractograms of original and carbonated fly ash.

Table 3
Main mineral phases in ash

Mineral phases	Formula	Untreated	Carbonated
		ash	ash
Lime	CaO	✓	
Portlandite	Ca(OH) ₂	✓	
Calcium chloride hydroxide	Ca(OH)Cl	✓	
Quartz	SiO ₂	✓	✓
Anhydrite	CaSO ₄	✓	✓
Halite	NaCl		
Sylvite	KCl	✓	✓
Friedels salt	Ca ₂ Al(OH) ₆ Cl(H ₂ O) ₂	✓	✓
Nordstrandite	Al(OH) ₃		✓
Calcite/vaterite	CaCO ₃	✓	
Gehlenite	Al ₂ O ₃ · 3CaO · 2SiO ₂		
	Ca ₂ Al ₂ SiO ₇	✓	✓
	Ca ₂ Al(Al · 5B · 5Si · 5Cr · O ₇)		

ate content above could be explained by hydration occurring at the same time as carbonation. It should be noted that there are many chloride bearing phases, such as halite, sylvite and Ca(OH)Cl, in ash. These phases are the main components releasing chloride in leachates (Bodenan and Deniard, 2003) and their high solubility leads to high concentrations of chloride in ash leachates.

3.4. PH of ashes

Due to the existence of lime, MSWI fly ash is highly alkaline, which is detrimental to its reuse. After carbonation, however, the alkalinity of ash is expected to be reduced as the calcium oxide is transformed to calcium carbonate. Fig. 6 shows the pH change after carbonation.

The original pH of ash is around 12–12.5, which is very close to the pH of a solution saturated in portlandite. The alkalinity of aged ashes from the Environment Agency is slightly lower than fresh ash, which may be the result of natural aging. After carbonation, the pH of these aged ashes was lowered to 7–10. The pH drop of fresh ash is, to a large extent, related to their carbonation reactivity, i.e., the amount of CO₂ sequestered. With higher activity, the carbonated SELCHP ash and Kirklees ash show a lower pH at around 7, compared with only a slight drop

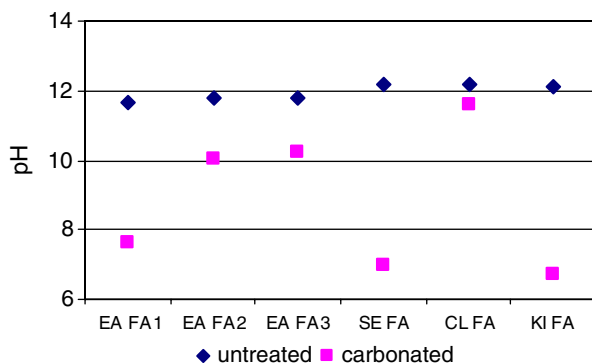


Fig. 6. pH value of original and carbonated ash.

in the pH of the Cleveland ash. Therefore, accelerated carbonation has a major influence on ash pH.

3.5. Leaching properties

One of the most important criteria for disposal to landfill or reuse of wastes is the release of hazardous compounds to the surrounding environment. The Landfill (England and Wales) Regulation, 2004 (Statutory Instrument, 2004) gives criteria for waste landfills, in which the wastes sent to landfill are classified into three categories, 'non-hazardous', 'hazardous' and 'inert' in terms of their leachability and stability. The limit values for landfill are given in Table 4. To determine which elements are of concern, the leachability of metals, such as As, Ba, Cd, Cr, Cu, Mo, Ni, Pb, Zn, and soluble salts Cl, SO₄ from untreated and carbonated SELCHP ash were analyzed. The release of Cl, SO₄, Cd, Cr, Cu, Pb and Zn from the original ash was over the landfill limit values and should be considered.

3.5.1. Soluble salts in leachate

Sulfate and chloride are the main soluble salts released from ash. The existence of large amounts of these salts in fly ash has a major influence on the setting and hydration of cement (Taylor, 1990).

From the mineralogical analysis of ash, the chloride ions occur in halite and sylvite, both of which are highly soluble. As the result, chloride release from fly ash is much higher than the acceptance value for hazardous wastes (Table 4) as shown in Fig. 7. Although carbonation reduced the chloride mobility, it was still higher than the acceptance value.

Sulfate content in the leachate is under the limit for non-hazardous wastes for all of the ash samples. The sulfate concentration was lower after carbonation as shown in Fig. 8. There was no obvious correlation between ash carbonation reactivity and leachable sulfate.

3.5.2. Heavy metals release

The concentrations of Ca, Na and K in leachates are very high and result from the high solubility of minerals bearing these elements, such as halite and sylvite. The release of other elements, with the exception of Cd, Pb and Zn, was under the limit value for non-hazardous waste.

Figs. 9–11 show the change in the release of Cd, Pb and Zn. Table 4 gives the limit value from the landfill accep-

Table 4
Leaching limit values for the acceptance of wastes in landfills (mg/kg)

Components	Hazardous waste	Non-hazardous waste
Cd	1	0.1
Cr total	70	10
Cu	100	50
Ni	40	10
Pb	50	10
Zn	200	50
Cl	25,000	15,000
SO ₄	50,000	20,000

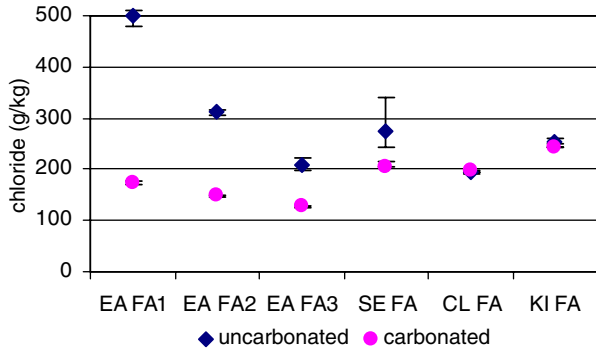


Fig. 7. Chloride leaching from FA.

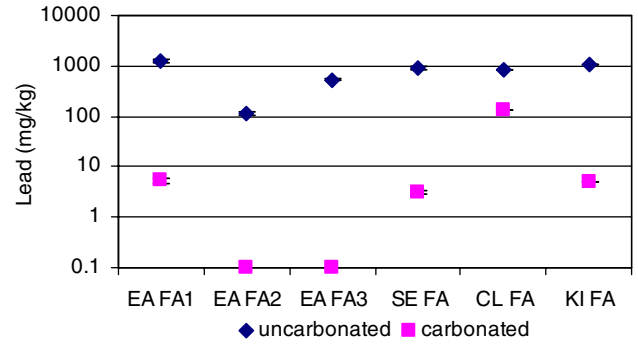


Fig. 10. Lead leaching.

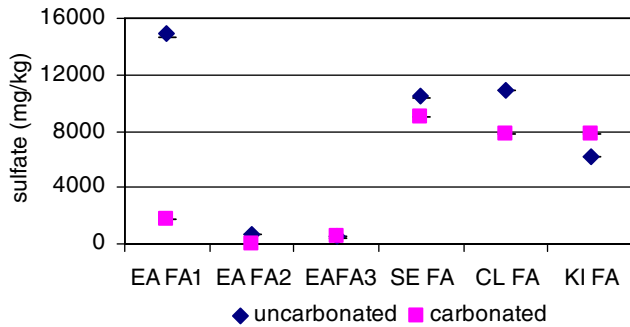


Fig. 8. Sulfate leaching from FA.

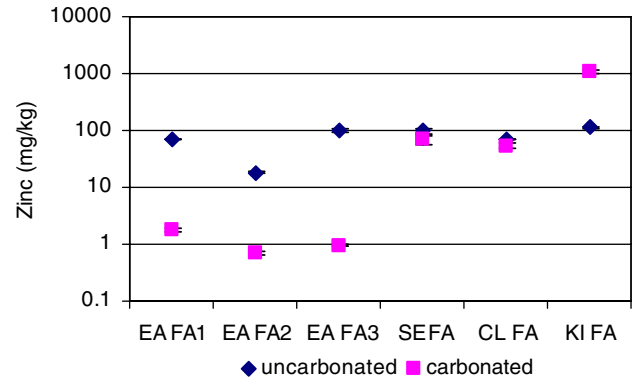


Fig. 11. Zinc release.

tance criteria. Cadmium release from the majority of ashes was lower than the limit value for hazardous wastes (1 mg/kg) but higher than that for non-hazardous wastes (0.1 mg/kg). No obvious relationship between carbonation and cadmium mobility was found. However, it should be noted that for the fly ashes SE FA and KI FA, leachable Cd increased after carbonation. This may result from an increase in cadmium solubility at lower pH (West General Incorporated).

Lead release from all of the ashes decreased dramatically with carbonation. The lead concentration in almost all of the untreated ash leachate was much higher than the landfill acceptance value, but after carbonation it was reduced

2–3 orders of magnitudes to lower than 6 mg/kg, with the exception of CL FA.

The change of zinc mobility is more complicated. The zinc release from the carbonated aged ash provided by the Environment Agency was reduced by 1–2 orders of magnitude. For the fresh ashes, some showed only slight change, and in KI FA the zinc is mobilized after carbonation.

4. Conclusions

The fly ash could combine with 7–10% w/w of carbon dioxide during accelerated carbonation. Water/solid ratio and temperature have a major influence on the reaction. As water is a medium of dissolution, ionization and transportation of CO₂, very low or very high water ratios will retard the reaction. High temperatures will improve the reaction velocity but do not improve the sequestration of CO₂ into the ash. The carbonation reaction is optimum at ambient temperatures and a water solids ratio of 0.3.

Mineralogical analysis of ash by XRD showed the disappearance of lime/portlandite and the formation of carbonate salts, such as calcite and vaterite upon carbonation. With ash from the Kirklees incinerator, hydration occurred as well as carbonation. The pH of naturally aged ashes collected from the Environment Agency is near 12, slightly lower than the fresh ashes, which were a pH of 12–12.5. The pH was reduced to between 7 and

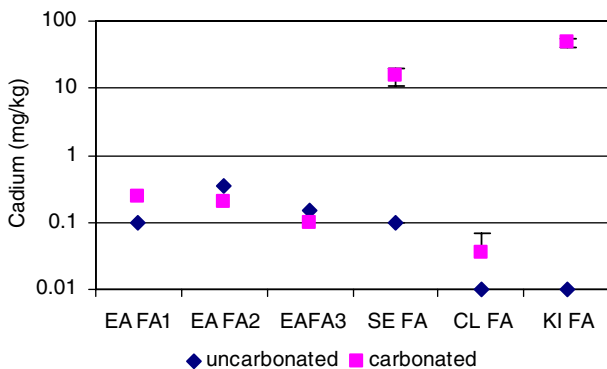


Fig. 9. Cadmium leaching.

9, depending on the reactivity of ash. Therefore, carbonation has a significant neutralizing effect on fly ash.

The release of soluble salts, Cl and SO₄, were reduced by carbonation. The reaction also led to a significant reduction of lead mobility, although cadmium release was increased. It can be concluded that carbonation can reduce the hazardous nature of fly ash, and facilitate disposal and reuse.

References

- Asavapisit, S., Fowler, G., Cheeseman, C., 1997. Solution chemistry during cement hydration in the presence of metal hydroxide wastes. *Cement and Concrete Research* 27 (8), 1249–1260.
- Bodenan, F., Deniard, Ph., 2003. Characterization of flue gas cleaning residues from European solid waste incinerators: assessment of various Ca-based sorbent processes. *Chemosphere* 51, 335–347.
- Chimenos, J.M., Fernandez, A.I., Nadal, R., Espiell, F., 2000. Short-term natural weathering of MSWI bottom ash. *Journal of Hazardous Materials* 79, 287–299.
- Ecke, H., Sakanakura, H., Matsuto, T., Tanaka, N., Lagerkvist, A., 2000. State-of-the-art treatment processes for municipal solid waste incineration residues in Japan. *Waste Management and Research* 18, 41–51.
- Environment Agency, 2002. Solid residues from municipal waste incinerators in England and Wales. A report on an investigation by the Environmental Agency.
- Environment Agency, 2004. Guidance on the use of stabilisation/solidification for the treatment of contaminated soil.
- EU Incineration Directive (2000/76/EC).
- EU Landfill Directive (1999/31/EC).
- European Union, 1999. EU focus on waste management.
- European Union, 2000. European Waste Catalogue.
- Eighmy, T.T., Crannell, B.S., Butler, L.G., Cartledge, F.K., Emery, E.F., Oblas, D., Krzanowski, J.E., Eusden, J.D., Shaw, E.L., Francis, C.A., 1997. Heavy metal stabilization in municipal solid waste combustion dry scrubber residue using soluble phosphate. *Environmental Science and Technology* 31, 3330–3338.
- Fernandez Bertos, M., Simons, S., Hills, C.D., Carey, P.J., 2004a. A review of accelerated carbonation technology in the treatment of cement-based materials and sequestration of CO₂. *Journal of Hazardous Materials* 112 (3), 193–205.
- Fernandez Bertos, M., Li, X., Simons, S.J.R., Hills, C.D., Carey, P.J., 2004b. Investigation of accelerated carbonation for the stabilisation of MSWI incinerator ashes and the sequestration of CO₂. *Green Chemistry* 6, 428–436.
- Freyssinet, Ph., Piantone, P., Azaroual, M., Itard, Y., Clozel-Leloup, B., Guyonnet, D., Baubron, J.C., 2002. Chemical changes and leachate mass balance of municipal solid waste bottom ash submitted to weathering. *Waste Management* 22, 159–172.
- HM Treasury, UK, 2003. Budget 2003, Chapter 7 Protecting the Environment. Economic and Fiscal Strategy.
- Johnson, D.C., MacLeod, C.L., Hills, C.D., 2003. Solidification of stainless steel slag by accelerated carbonation. *Environmental Technology* 24, 671–678.
- Klemm, W.A., Berger, R.L., 1972. Accelerated curing of cementitious systems by CO₂. *Cement and Concrete Research* 2, 567–576.
- The Landfill (England and Wales) (Amendment) Regulations, 2004.
- Mangialardi, T., 2001. Sintering of MSW fly ash for reuse as a concrete aggregate. *Journal of Hazardous Materials* B87, 225–239.
- Mangialardi, T., 2003. Disposal of MSWI fly ash through a combined washing-immobilisation process. *Journal of Hazardous Materials* B98, 225–240.
- Meima, J.A., Comans, R.N.J., 1997. Geochemical modelling of weathering reactions in municipal solid waste incinerator bottom ash. *Environmental Science and Technology* 31, 1269–1276.
- Meima, J.A., Comans, R.N.J., 1999. The leaching of trace elements from municipal solid waste incinerator bottom ash at different stages of weathering. *Applied Geochemistry* 14, 159–171.
- Mizutani, S., van der Sloot, H.A., Sakai, S., 2000. Evaluation of treatment of gas cleaning residues from MSWI with chemical agents. *Waste Management* 20, 233–240.
- Mulder, E., 1996. Pre-treatment of MSWI fly ash for useful application. *Waste Management* 16, 181–184.
- Polettini, A., Pomi, R., Sirini, P., Testa, F., 2001. Properties of Portland cement—stabilised MSWI fly ashes. *Journal of Hazardous Materials* 88, 123–138.
- Polettini, A., Pomi, R., Lo Mastro, S., Piacente, E., 2003. Accelerated aging of incinerator bottom ash as a tool for landfill management optimization. In: Ninth International Waste Management and Landfill Symposium, Cagliari, Italy.
- Polettini, A., Pomi, R., Trinci, L., Muntoni, A., Lo Mastro, S., 2004. Engineering and environmental properties of thermally treated mixtures containing MSWI fly ash and low-cost additives. *Chemosphere* 56, 901–910.
- Sabbas, T., Polettini, A., Pomi, R., Astrup, T., Hjelmar, O., Mostbauer, P., Cappai, G., Magel, G., Salhofer, S., Speiser, C., 2003. Management of municipal solid waste incineration residues. *Waste Management* 23, 61–88.
- Sakai, S., Hiraokab, M., 2000. Municipal solid waste incinerator residue recycling by thermal processes. *Waste Management* 20, 249–258.
- Scottish Environment Protection Agency (SEPA), Environment and Heritage Service, Environment Agency, 2003. Hazardous waste, Interpretation of the definition and classification of hazardous waste, Technical Guidance WM2.
- Statutory Instrument, 2004, No. 1375. The Landfill (England and Wales) (Amendment) Regulations 2004. Available from: <http://www.opsi.gov.uk/si/si2004/20041375.htm>.
- Taylor, H.F.W., 1990. *The Chemistry of Cements*. Academic Press, London.
- West General Incorporated. Available from: <http://www.westgeneral.com/outofthebox/compounds/cdsol.html>.
- Yousuf, M., Mollah, A., Hess, T.R., Tsai, Y.N., Cocke, D.L., 1993. An FTIR and XPS investigations of the effects of carbonation on the solidification/stabilization of cement based systems—Portland type V with Zn. *Cement and Concrete Research* 23, 773–784.