

Environmental management of bottom ash from municipal solid waste incineration based on a life cycle assessment approach

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Received: 20 December 2013 / Accepted: 8 April 2014 / Published online: 4 May 2014
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Abstract Conventional bottom ash (BA) management consists of a solidification process using inorganic binder reagents, such as cement. However, despite the heavy metal content, the use of BA as a natural aggregate has become increasingly more common. In particular, bottom ash is used as a raw material for clinker, cement mortar or frit production, as a drainage layer in landfills or as a sub-base material in road construction. In this study, the life cycle assessment approach was used to evaluate and compare ash solidification with ash recycling in Portland cement production as a clinker and gypsum substitute. The findings showed that the substitution of ash for clinker resulted in the lowest natural resources (NR) consumption and the lowest environmental burdens (EB). The decrease in the clinker substitution percentage generated a higher NR consumption and an increased EB. In ash recycling, the distance between the incinerator and the cement facility is an important parameter in the decision-making process. Specifically, ash solidification presented less favourable results than ash recycling (with a clinker substitution of 25 %), despite the increasing distance between the incinerator and the cement facility. However, when the clinker substitution decreased to 2.5 % or when ash was substituted for gypsum, the distance played an important role in the water impact.

Keywords Bottom ash · Incineration · Life cycle assessment (LCA) · Municipal solid waste (MSW)

Introduction

Municipal solid waste (MSW) constitutes approximately 10 % of the waste produced, generating approximately 250 million tons of MSW in Europe in 2012, representing an annual per capita generation rate of 492 kg MSW/person (EUROSTAT 2014). The best and most economical method of handling MSW is to minimise its production. However, once waste is produced, it cannot be destroyed, and end-of-pipe treatment, such as landfilling or incineration, is required (Pavlas and Tous 2009). Although landfilling remains the most common practice, incineration has increased in the last 10–15 years, driven by the industry-specific legislation to reduce emissions to air (Margallo et al. 2012). The primary objective of municipal solid waste incineration (MSWI) is to treat waste by reducing the waste mass and allowing energy recovery (Margallo et al. 2013). For this reason, the original designation of 'incinerator' was dropped, and today it is talked about energy from waste (or waste to energy, WTE) (Pavlas et al. 2011). However, incineration results in various types of solid wastes, primarily including bottom, boiler and fly ashes. Fly ashes are fine and are normally characterised by a high content of chlorides and significant amounts of dangerous substances, such as heavy metals or organic compounds. Bottom ashes (BA) have coarser dimensions, and the amount of chlorides and hazardous chemicals is typically significantly lower than in fly ashes. BAs are commonly subjected to a stabilisation process producing a material with physical and mechanical properties that promote a reduction in contaminant release from the residue matrix. These methods use inorganic binder reagents, such as cement, lime and other pozzolanic materials. However, despite the heavy metal content, the use of BA as a natural aggregate has become increasingly common (Margallo

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et al. 2013). In particular, in the cement production, it was studied the use of BA to replace clinker raw materials (Lam et al. 2011) and clay (Pan et al. 2008), as a supplementary cementitious material mixed with coal fly ash (Li et al. 2012), and as a fine aggregate in the cement mortar manufacture (Saika et al. 2008). Other studies analysed concrete production with BA and exhausted sand from a fluidised bed incinerator (Abbà et al. 2014) and with washed BA (Kuo et al. 2013) and treated BA (Pera et al. 1997) both from a grate incinerator. Additionally, Bertolini et al. (2004) assessed the use of bottom and fly ash to replace part of Portland cement to produce concrete. Due to the high mineral content of BA, additional possible management options include frit production (Barbeiro et al. 2010), the utilisation of BA as a landfill cover material (Puma et al. 2013), and as a solid substitute in embankments (Rivard-Lentz et al. 1997). Likewise, BA is now increasingly used for construction. A practice now commonly observed in Denmark, Belgium and the Netherlands is to use BA to repair roads and produce asphalt concrete and permeable pavement (Shih and Ma 2011). The importance of this application is reflected in several studies published over recent years. Specifically, Forteza et al. (2004) characterised BA for its use in a road base in Spain, Toraldo et al. (2013) studied the use of BA in the production of asphalt concrete for road pavement, De Wind et al. (2011) analysed the reuse of BA in a basement of a pilot-scale road, as well as Del Valle-Zermeño et al. (2014) that mixed BA with air pollution control (APC) ashes. Recycling processes have a material and energy consumption associated with the manufacture of the product. However, BA recycling avoids waste disposal and the associated impacts, and replaces non-renewable resources (Margallo et al. 2013). To compare the environmental advantages and disadvantages, a life cycle approach is required. Several life cycle assessment (LCA) studies have been conducted to evaluate the environmental performance of BA recycling in road and pavement construction, highlighting the studies developed in Denmark by Birgisdóttir et al. (2006, 2007), in Sweden by Olsson et al. (2006) and in Taiwan by Geng et al. (2010). Other LCAs have compared the environmental impacts of traditional Portland cement and blended cement production with the addition of BA (Huntzinger and Eatmon 2009) and the utilisation of BA as landfill cover (Toller et al. 2009). However, a comparison based on a process approach between the traditional stabilisation process and the BA recycling process has not been developed. This study applies the LCA methodology with a process perspective to evaluate and compare the environmental impacts of ash solidification with cement and ash recycling in Portland cement production. The LCA employed a novel methodology based on the sustainability metrics developed by the

Institution of Chemical Engineers (ICHEME 2002) recommended for the use in the process industries and the threshold values of the European Pollutant Release and Transfer Register (E-PRTR) (E-PRTR Regulation 2006). This methodology reduces the complexity, aiding in the decision-making process. Moreover, a semi-quantitative economic analysis and the study of the distance influence on the environmental impacts were developed to complete the study.

Life cycle assessment methodology

LCA allows the evaluation of processes or products from 'cradle' to 'grave'. LCA should be applied using the ISO 14040 (ISO 2006a) series, which describes the LCA in the following 4-phase process: (a) definition of the goal and scope; (b) life cycle inventory (LCI) analysis; (c) life cycle impact assessment (LCIA) and (d) interpretation (Meyer and Upadhyayula 2014). Figure 1 displays the LCA methodology.

Goal and scope

This step defines the intended application of the study, the system description, the functional unit, the system boundaries, the allocation procedures and the assumptions. The goal of this LCA was to evaluate and compare the environmental impacts of BA solidification and BA recycling.

Functional unit

From the incineration of 1 t of MSW, 57.5 kg of BA was generated (AEVERSU 2011). Because the function of the system is to treat ashes, all of the data are in reference to 57.5 kg of ashes.

System description

A WTE plant located in the north of Spain was selected as a case study. In 2009, 113,338 tons of MSW with a low heating value of 2,800 kcal/kg were incinerated in a roller grate incinerator generating 82,800 MWh (AEVERSU 2011). The system is composed of ash transport and ash treatment. Thermal and flue gas treatment were excluded from the study. These processes are identical in both scenarios (Sc.); thus, the associated impacts are identical and can be neglected in a comparative analysis. Internal transport in the complex, the construction of major capital equipment, and the maintenance and operation of support equipment were excluded from the study. According to Fig. 2a and b in which the system boundaries are shown, three scenarios were analysed.

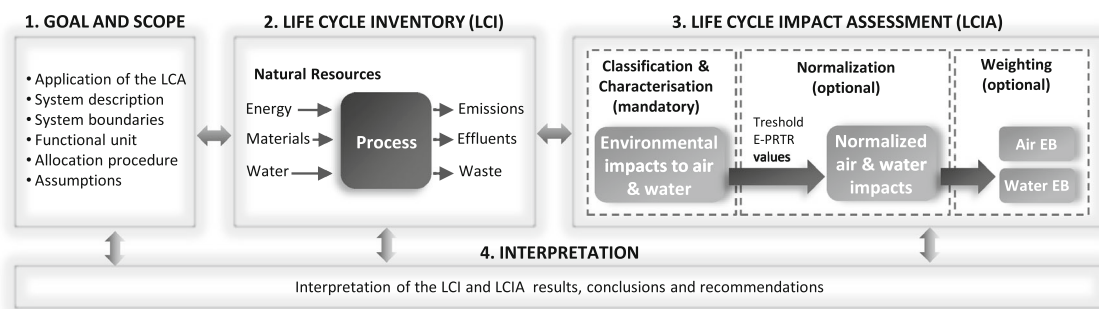


Fig. 1 LCA methodology (based on ISO 2006a)

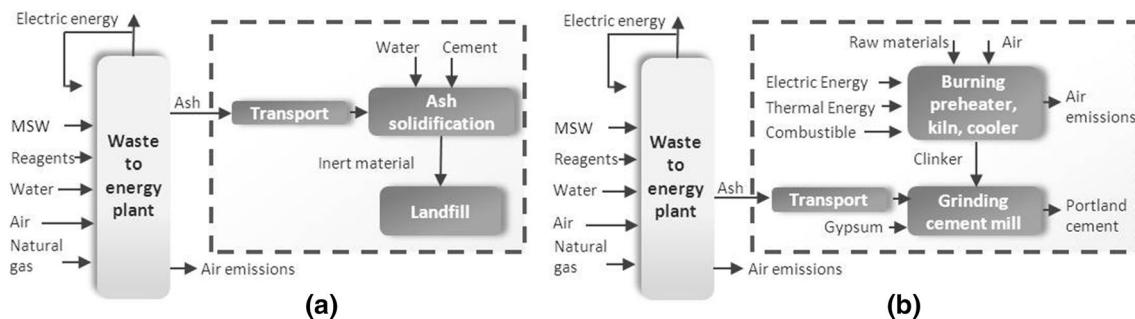


Fig. 2 System description of **a** ash solidification and **b** ash recycling in Portland cement production

- Scenario 1: ash solidification** The BA solidification includes ash transport to a waste manager located 81 km from the WTE plant, the solidification process and ash landfilling. The solidification process employed a mixture of water (30 %), cement (20 %) and ashes (50 %) (Margallo et al. 2013).
- Scenario 2: ash recycling in Portland cement production as a clinker substitute** Portland cement is composed primarily of calcium silicate materials, such as limestone and sand. The raw materials are quarried, crushed and milled into a fine powder that is fed into a rotary kiln. The clinker from the kiln is cooled, and gypsum is added (typically 20 %) to regulate the setting time (Margallo et al. 2013). The amount of clinker can be reduced using certain materials, such as coal fly ash, slag and natural pozzolans. The addition of these materials not only reduces the amount of material landfilled but also reduces the amount of clinker required per ton of cement produced. The strength, durability and life of blended cement using ashes are equivalent to traditional Portland cements with a substitution range of 25–60 %. To ensure an equivalent cement product in this study, a substitution percentage of 25 % by mass was assumed (Huntzinger and Eatmon 2009). This system encompasses ash transport to the cement plant (118 km) and the Portland cement production, which is composed of the burning of raw

materials and the grinding of clinker with gypsum and ashes. Additionally, the clinker substitution percentages of 10 % in Sc. 2.1 and 2.5 % in Sc. 2.2 were studied. These systems are identical to Sc. 2, except for the amount of clinker substituted.

- Scenario 3: ash recycling in Portland cement production as a gypsum substitute** The amount of gypsum required to produce Portland cement has changed in recent years because of the replacement of gypsum with natural or industrial pozzolans (Margallo et al. 2013). In this scenario, ashes substituted 25 % of the gypsum.

Allocations

A system can have multiple functions (multifunctional systems), and to assign the environmental burden associated with each function, allocations are applied. In Sc. 2, 2.1, 2.2 and 3, ashes are recycled to produce blended cement. Therefore, the system has the following two functions: ash treatment and cement production. These multifunctional processes are typically handled through system expansion. To expand the system and subtract the environmental impacts associated with the recovery of recycled materials, the determination of the type of material replacing the recycled material and its equivalence to the virgin material is required. BA replaces clinker in Sc. 2, 2.1 and 2.2, whereas in Sc. 3, BA replaces gypsum in the

Portland cement production. Because the properties of traditional Portland cement are equivalent to those of the blended cement (with bottom ash), a substitution factor of 1 was applied (Margallo et al. 2013).

Life cycle inventory

The relevant mass and energy input and output data are collected in a LCI. Table 1 gives the LCI of ash solidification

(Sc. 1), ash recycling with a clinker substitution of 25 % (Sc. 2), 10 % (Sc. 2.1) and 2.5 % (Sc. 2.2), and with a gypsum substitution of 25 % (Sc. 3). The negative values are associated with an environmental benefit, whereas the positive values indicate a detriment to the environment.

The WTE plant data were provided by the Spanish non-profit company ECOEMBES, which is responsible for the collection and recovery of packaging waste (ECOEMBES 2011), the Spanish association of MSW valorisation

Table 1 LCI of Sc. 1 ash solidification, Sc. 2 ash recycling (25 % clinker substitution), Sc. 2.1 ash recycling (10 % clinker substitution), Sc. 2.2 ash recycling (2.5 % clinker substitution) and Sc. 3 ash recycling (25 % gypsum substitution)

Input/output data	Units	Sc. 1	Sc. 2	Sc. 2.1	Sc. 2.2	Sc. 3
INPUTS						
Energy resources	MJ/t MSW	1.13E+02	-1.77E+02	-1.05E+02	-3.12E+01	6.01E+00
Material resources	kg/t MSW	1.17E+02	-1.43E+02	-8.69E+01	-2.92E+01	-1.16E+01
Gypsum		1.15E+00	-1.04E-04	-5.49E-05	1.76E-03	-1.15E+01
Inert rock		3.67E+01	-6.54E+01	-3.98E+01	-1.34E+01	-9.41E-02
Limestone		2.58E+01	-4.60E+01	-2.80E+01	-9.43E+00	-9.49E-02
Sand		3.45E+00	-6.16E+00	-3.75E+00	-1.26E+00	-1.29E-02
Water		3.90E+01	7.68E+00	-4.66E+00	-1.54E+00	3.13E-02
Air		1.08E+01	-1.77E+01	-1.07E+01	-3.57E+00	2.36E-02
OUTPUTS						
Emissions to air	kg/t MSW	3.46E+01	-5.96E+01	-3.61E+01	-1.18E+01	3.78E-01
Vanadium (+III)		1.91E-06	-1.73E-06	-8.85E-07	-8.92E-09	4.31E-07
Carbon dioxide (CO ₂)		2.09E+01	-3.57E+01	-2.16E+01	-7.00E+00	3.39E-01
Carbon monoxide (CO)		5.05E-02	-8.68E-02	-5.26E-02	-1.74E-02	3.19E-04
Nitrogen (N-compounds)		5.89E-02	-1.05E-01	-6.40E-02	-2.16E-02	-2.19E-04
Nitrogen oxides (NO _x)		1.42E-02	-1.38E-02	-7.55E-03	-1.09E-03	2.16E-03
Steam		4.95E+00	-8.66E+00	-5.26E+00	-1.76E+00	3.57E-03
Sulphur dioxide (SO ₂)		7.55E-02	-1.34E-01	-8.14E-02	-2.73E-02	-6.78E-05
Dioxins (PCDD)		1.10E-05	-1.97E-05	-1.20E-05	-4.04E-06	-4.12E-08
Non-Methane Volatile Organic Compounds (NMVOC)		1.00E-03	-7.03E-04	-3.57E-04	-5.08E-07	-1.79E-04
Exhaust		8.47E+00	-1.47E+01	-8.96E+00	-2.99E+00	1.26E-02
Dust (PM ₁₀)		4.06E-03	-6.84E-03	-4.17E-03	-1.40E-03	-1.19E-05
Emissions to fresh water	kg/t MSW	2.74E-02	-4.10E-02	-2.42E-02	-6.93E-03	1.76E-03
Chemical oxygen demand		8.98E-04	-8.74E-04	-5.23E-04	-1.60E-04	2.17E-05
Iron		3.84E-04	-6.59E-04	-3.99E-04	-1.30E-04	5.03E-06
Chloride		1.79E-02	-3.00E-02	-1.81E-02	-5.79E-03	3.96E-04
Fluoride		5.54E-04	-9.63E-04	-5.84E-04	-1.92E-04	4.39E-06
Sulphate		2.62E-03	-4.53E-03	-2.74E-03	-8.98E-04	2.87E-05
Solids (suspended)		3.79E-03	-1.95E-03	-6.96E-04	5.99E-04	1.25E-03
Emissions to sea water	kg/t MSW	7.46E-03	-6.48E-03	-3.25E-03	8.74E-05	1.76E-03
Chemical oxygen demand		1.22E-05	-6.98E-06	-2.74E-06	1.63E-06	3.82E-06
Strontium		8.86E-07	2.94E-07	3.71E-07	4.50E-07	4.90E-07
Zinc (+II)		3.45E-07	-2.87E-07	-1.41E-07	9.16E-09	8.47E-08
Chloride		6.84E-03	-6.27E-03	-3.21E-03	-5.63E-05	1.53E-03
Solids (suspended)		4.39E-04	-6.92E-05	3.10E-05	1.34E-04	1.86E-04
Emissions to industrial soil	kg/t MSW	4.80E-05	-6.33E-05	-3.63E-05	-8.39E-06	5.62E-06

AEVERSU (2011), the Spanish Pollutant Release Transfer Register PRTR (2011), the Environmental Integrated Authorisation of the plant, the WTE plant and bibliographic data (Margallo et al. 2013). The ash solidification data were collected from Doka (2003), and the Portland cement production was based on the Reference Document for the Best Available Techniques for the production of cement, lime and magnesium oxide manufacturing industries (BREF 2010) and the Spanish Handbook on the Best Available Techniques for cement manufacturing (BAT 2004).

Life cycle impact assessment

LCIA quantifies the potential for environmental impact over all of the stages involved in the delivery of a product or service (Bare 2010). LCIA is divided into the following four steps: classification, characterisation, normalisation (optional) and grouping or weighting (optional). The LCIA was conducted by following the ISO 14040 (ISO 2006a) and ISO 14044 requirements (ISO 2006b) with the LCA software GaBi 4.4 (PE International 2011), and the environmental sustainability metrics developed by the Institution of Chemical Engineers (IChemE) that provide a balanced view of the environmental impact of inputs (resource usage) and outputs (emissions, effluents and waste). In relation to the outputs, a set of environmental impacts to the atmosphere and aquatic media was chosen. The environmental burden (EB) approach was used to estimate and quantify the potential environmental impacts. In particular, the environmental impacts were classified into atmospheric and aquatic impacts, as shown in Table 2. The indicators used in this study had different units, depending on the environmental impact. To compare the EBs to air and water, the threshold values stated in the European regulation EC No 166/2006 (E-PRTR regulation) were considered as the weighting factors to obtain dimensionless impact indicators (Garcia et al. 2013). The E-PRTR regulation establishes the contaminants for which the European installations must provide notification to the authorities along with the threshold values of those pollutants. The threshold values can be used as an important aid in the normalisation process because they provide an overview of the environmental performance of the installation at a European level.

Results

Natural resources sustainability (NRS)

NR include energy, material and water consumption. Figure 3 shows that a higher NR consumption was

Table 2 Threshold values from the E-PRTR regulation for normalisation and no of substances included in each impact category (Garcia et al. 2013)

EB		Threshold value (kg/year)	N° substances
Air EB	Atmospheric acidification (AA) (kg SO ₂ eq.)	150,000	6
	Global warming (GW) (kg CO ₂ eq.)	100,000,000	23
	Human health (HHE) (kg benzene eq.)	1,000	52
	Photochemical ozone formation (POF) (kg ethylene eq.)	1,000	100
	Stratospheric ozone depletion (SOF) (kg CFC-11 eq.)	1	60
	Water EB	Aquatic oxygen demand (AOD) (kg O ₂ eq.)	50,000
Aquatic acidification (Aq. A) (kg H ⁺ eq.)		100	4
Ecotoxicity to aquatic life (metals) (MEco) (kg Cu eq.)		50	11
Ecotoxicity to aquatic life (organics) (NMEco) (kg formaldehyde eq.)		50	18
Eutrophication (Eutroph) (kg phosphate eq.)		5,000	8

The units are given in kg equivalents (kg eq.)

obtained in the solidification process (Sc. 1) than in all of the recycling scenarios (Sc. 2, 2.2, 2.2 and 3). This was due to the significant NR consumption during the solidification process in Sc. 1, and the reduction in the amount of required clinker in the cement production in Sc. 2, 2.1 and 2.2. The reduction in the amount of clinker lowers the NR consumption, thus lowering the emissions. Negative NR consumption values were obtained in cement manufacturing (avoided burden), displaying only positive values for ash transport. For the consumption of NR in Sc. 1, 95 % of the energy was consumed in the production of cement, which is used in the solidification process. Cement manufacturing is a high energy-consuming industry, focusing the energy consumption in the decarbonation and clinkering of raw materials (BAT 2004; BREF 2010). The comparison of Sc. 1 and 2 (clinker substitution of 25 %) displayed a relative change of 373, 380 and 120 % in energy, material and water consumption, respectively. The reduction in the percentage of clinker substitution from 25 % (Sc.2) to 10 % (Sc. 2.1) produced a 40 % increase in the NR consumption. This increase was even more noticeable (80 %), when the clinker substitution decreased to 2.5 % (Sc. 2.2). Therefore, a higher clinker substitution indicates a lower NR consumption. Sc. 3

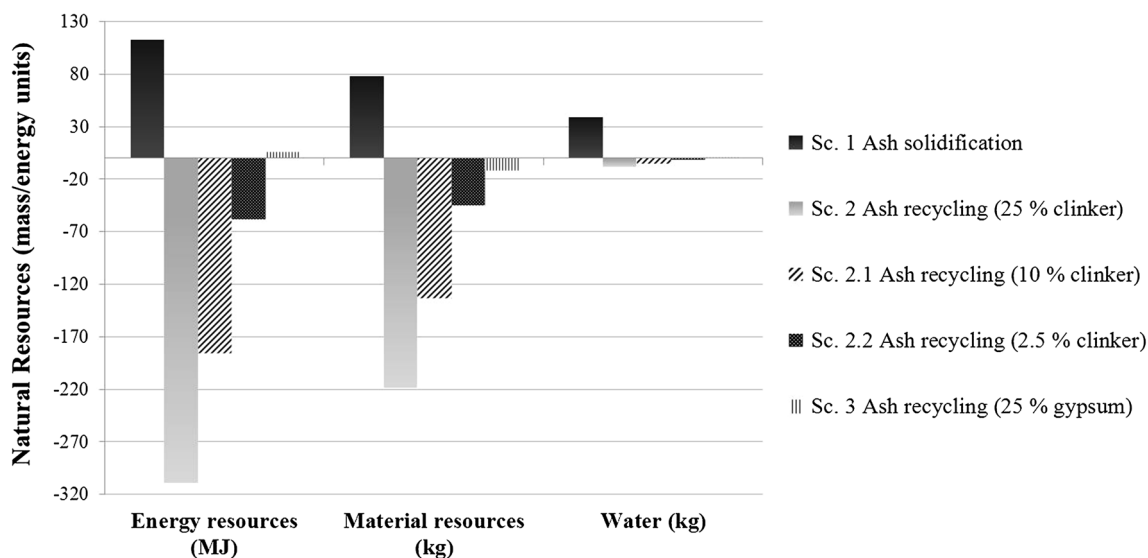


Fig. 3 Natural resources sustainability (NRS)

(25 % gypsum substitution) also showed a lower NR consumption than Sc.1, with a reduction of 95, 85 and 99 % in the consumption of energy, material and water, respectively. However, higher consumptions resulted in Sc. 3 than in Sc. 2, 2.1 and 2.2 because the amount of clinker in this blended cement is identical to that required in the conventional cement; therefore, there is no consumption savings associated with clinker production. The NR savings were only afforded by the amount of gypsum substituted.

Environmental burdens sustainability (EBS)

Table 3 shows the EB values and the normalised results of Sc. 1, 2 and 3. Before the normalisation, global warming (GW) displayed the highest air impact in all of the scenarios due to the emission of greenhouse gases in the clinker production (CO_2 , CO, Volatic Organic Compounds VOC), the consumption of coke and energy in the clinkering of raw materials (CH_4 , CO, CO_2 , NO_x , N_2O), the diesel consumption and the landfill emissions (NO_x , N_2O), and the BA transport (NO_x , N_2O). Ecotoxicity to aquatic life (organics) (NMEco) presented the highest contribution to the water impact in all of the scenarios due to the seawater emissions of ammonia, chloride, benzene, toluene and xylenes in the clinker production, the coke and energy consumption in the clinkering of raw materials, and the diesel consumption in the transport and landfilling of BA. After the normalisation, photochemical ozone formation (POF) had the highest contribution in the air categories, whereas NMEco continued as the highest contributor to the water impact. The primary reason for this result was that although GW had the highest air impact, when the EB was referenced to the threshold value (100,000,000), the

normalised results were reduced by 5 orders of magnitude. Moreover, although POF had a lower EB than GW, a lower threshold value (1,000) was used as the reference value. For the water impacts, there were no significant differences after normalisation because the threshold values are lower than in the air categories.

A comparison of the scenarios showed that the ash recycling scenarios (Sc. 2 and 3) had a lower impact than the ash solidification scenario (Sc. 1) in all of the environmental categories. This result was consistent with the NR results because lower NR consumption and emissions result in a lower EB. The primary reason for these results is that although the BA recycling (Sc. 2 and 3) obtained a high impact in the blended cement manufacturing process and a higher transport impact than ash solidification (Sc. 1), the avoided burden associated with BA recycling compensated for these impacts. However, in Sc. 1, cement production for BA solidification had the highest influence in all of the categories. In particular, solidification contributed to the EB to air over a range of 94–99 %. For the water EB, although cement manufacturing is the process with the highest impact, the transport impact reached values of 31 and 36 % in NMEco and MEco, respectively, and water consumption in the solidification process contributed 39 % to the eutrophication. The normalised results are grouped in Fig. 4 into two impacts, EB to air and EB to water. The highest air and water EBs were observed in Sc. 1. Of the recycling scenarios, Sc. 3 presented the highest EBs to air and water, whereas Sc. 2 displayed the lowest air and water impacts. Finally, the reduction in the amount of substituted clinker reduced the air and water EBs.

Table 4 displays the semi-quantitative economic analysis performed with the Gabi 4.4 LCA software (PE International 2011). The symbols “+”, “–” and “0”

Table 3 Environmental burdens and the normalised results of Sc. 1 ash solidification, Sc. 2 ash recycling (25 % clinker substitution) and Sc. 3 ash recycling (25 % gypsum substitution)

	Environmental burdens (EB)				Normalised results		
	Units	Sc. 1	Sc. 2	Sc. 3	Sc. 1	Sc. 2	Sc. 3
EB air							
AA	kg SO ₂ eq.	7.57E-02	-1.34E-01	-6.18E-05	5.04E-07	-8.95E-07	3.63E-16
GW	kg CO ₂ eq.	2.20E+01	-3.71E+01	4.38E-01	2.20E-07	-3.71E-07	4.38E-09
SOD	kg CFC11 eq.	3.44E-07	-6.10E-07	-2.36E-10	3.44E-07	-6.10E-07	-2.36E-10
HHE	kg benzene eq.	2.10E-03	-3.50E-03	9.75E-07	2.10E-06	-3.50E-06	9.75E-10
POF	kg ethylene eq.	5.51E-03	-9.52E-03	5.10E-05	5.51E-06	-9.52E-06	5.10E-08
EB water							
AOD	kg O ₂ eq.	8.95E-06	-1.41E-05	3.41E-07	1.79E-10	-2.82E-10	6.83E-12
Aq. A	kg H ⁺ eq.	1.02E-09	-1.47E-09	5.44E-11	1.02E-11	-1.47E-11	-6.18E-07
NMEco	kg formal. eq.	3.42E-03	-3.14E-03	7.65E-04	6.84E-05	-6.27E-05	1.53E-05
MEco	kg Cu eq.	4.16E-07	-4.35E-07	7.94E-08	8.32E-09	-8.69E-09	1.59E-09
Eutroph.	kg PO ₄ eq.	2.04E-05	-1.97E-05	6.61E-07	4.08E-09	-3.94E-09	1.32E-10

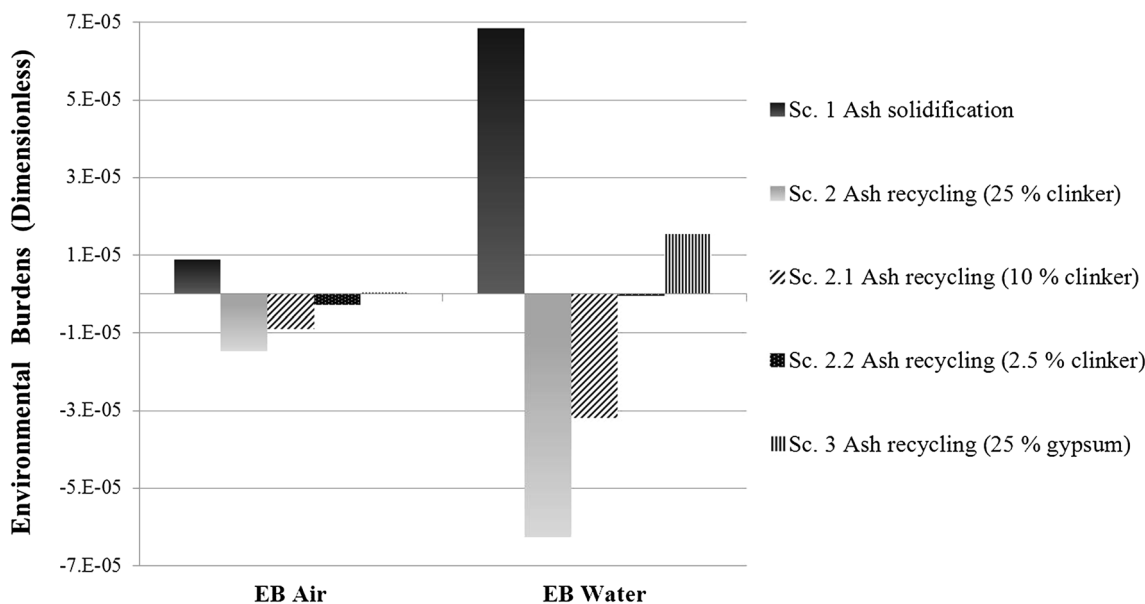


Fig. 4 Air and water EBs of Sc. 1 ash solidification, Sc. 2 ash recycling (25 % clinker substitution), Sc. 2. 1 ash recycling (10 % clinker substitution), Sc. 2.2 ash recycling (2.5 % clinker substitution), Sc. 3 ash recycling (25 % gypsum substitution)

indicate a positive influence (i.e. an economic cost), a negative influence (i.e., an economic benefit) and a low or null influence, respectively. The following three cost variables were evaluated: transport (X_1), raw materials (X_2) and landfill taxes (X_3). The variable X_1 represents the cost of the ash transport to the landfill (Sc. 1) or to the cement plant (Sc. 2, 2.1, 2.2 and 3). Therefore, the distance from the WTE plant to the cement plant or to the landfill has a significant influence on the transport cost, requiring further evaluation. The variable X_2 provides the cost of the raw

Table 4 Semi-quantitative economic analysis of Sc. 1, Sc. 2, Sc. 2.1, Sc. 2.2 and Sc. 3

Cost variables	Sc. 1	Sc. 2	Sc. 2.1	Sc. 2.2	Sc. 3
X_1 (transport)	0	+	+	+	+
X_2 (raw material)	+	-	-	-	-
X_3 (landfill taxes)	+	0	0	0	0
Economic benefit related to Sc. 1		125 %	114 %	102 %	97 %

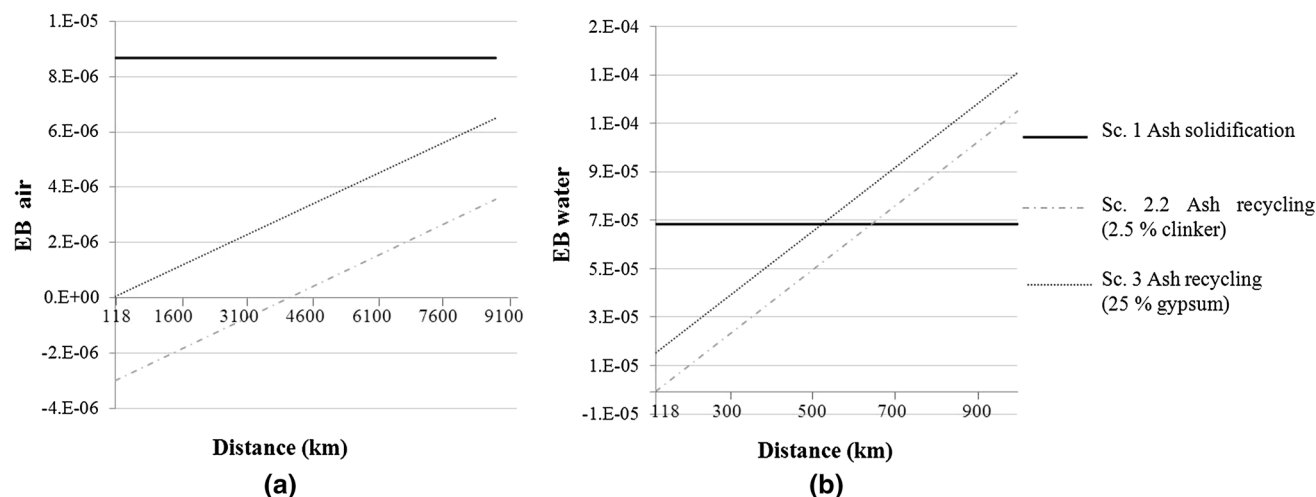


Fig. 5 Evolution of the **a** EB to air and **b** EB to water of Sc. 1, Sc. 2.2 and Sc. 3 as a function of distance

materials, primarily water, cement and diesel, whereas X_3 displays the landfill taxes. In Sc. 1, the landfill taxes (X_3) and the raw materials (X_2) presented the highest costs. For the variable X_2 , cement consumption in the solidification process had the highest influence. In Sc. 2, 2.1 and 2.2, the cost was associated with the ash transport from the WTE plant to the cement plant (X_1). However, in these scenarios, the variable X_2 assumed an economic benefit because ash recycling reduced the amount of required raw materials, thus reducing the costs. A comparison of Sc. 1 with Sc. 2, 2.1 and 2.2 displayed an economic benefit of 125, 114 and 102 %, respectively. Percentages higher than 100 % indicate that the process generates an economic benefit instead of a cost. In particular, the results showed that the higher amount of clinker substituted, the higher the economic benefit. Similar to Sc. 2, 2.1 and 2.2, ash transport (X_1) was the primary cost in Sc. 3, with the raw material cost (X_2) exerting a very low influence because in the recycling process, ash is only replacing the gypsum. A cost reduction of 97 % was obtained in Sc. 3.

Distance evaluation

The distance from the WTE plant to the cement facility is an important parameter in BA recycling. To evaluate the influence of this distance, the EBs of ash solidification (Sc. 1), BA recycling with a 2.5 % clinker substitution (Sc. 2.2) and with a 25 % gypsum substitution (Sc. 3) were compared, considering several distances (Fig. 5). The negative values in Fig. 5 are associated with an environmental benefit. The EB to air in Sc. 1 was higher than that in Sc. 2.2 and Sc. 3, even though the distance reached 9,000 km. With regard to the individual air impacts, for atmospheric acidification (AA), stratospheric ozone depletion (SOD), human health effects (HHE) and POF, although the

distance increased, the impact of Sc. 1 remained higher than in the ash recycling scenarios. Only for GW, ash recycling reached a higher air impact than Sc. 1 for distances up to 6,800 km in Sc. 2.2 and up to 5,050 km in Sc. 3. In the base case (118 km), the water impacts were higher in all of the categories for Sc. 1. However, the distance variation reversed this situation. When the distance exceeded 651 km in Sc. 2.2 and 523 km in Sc. 3, ash solidification presented a lower EB to water. In particular, for Aq. Ac., when the distance surpassed 2,096 km in Sc. 3 and 3,000 km in Sc. 2.2, Sc. 1 displayed reduced impacts. The same results were obtained for distances up to 2,851 (Sc. 3) and 3,790 km (Sc. 2.2) for AOD; 588 (Sc. 3) and 762 km (Sc. 2.2) for MEco; 520 (Sc. 3) and 644 km (Sc. 2.2) for NMEco and 3,430 (Sc. 3) and 4,123 km (Sc. 2.2) for Eutroph.

Conclusions

This study evaluated the environmental impacts of several treatment alternatives of MSWI bottom ash as follows: BA solidification (Sc. 1), BA recycling in Portland cement production as a substitute for clinker (Sc. 2, 2.1 and 2.2) and gypsum (Sc. 3). For this purpose, the LCA methodology was applied. The primary function of these processes was to evaluate different waste treatments; however, ash recycling presents an additional function (cement production), which was rectified by subtracting the extra processes with a system expansion, generating avoided burdens. Several clinker substitution percentages were studied to evaluate the influence of the avoided burden associated with the cement manufacturing process. The results revealed that a higher clinker substitution indicates a lower EB and NR consumption. According to this premise, Sc. 2

(clinker substitution of 25 %) was presented as the best recycling scenario. The paper also studied the use of ashes as a substitute for gypsum (Sc. 3). In this scenario, an identical amount of clinker was used in the cement production; consequently, no savings were observed in the consumption of material, energy and water. The savings were associated with the gypsum extraction; consequently, a lower impact was avoided. A comparison of the scenarios demonstrated that ash solidification is the least favourable scenario with the highest NR consumption and emissions, resulting in the highest air and water impacts. From all of the recycling alternatives, Sc. 2 with a clinker substitution of 25 % is presented as the best option because the higher the NR savings, the lower the EB. From an economic viewpoint, the following 3 cost variables were evaluated: transport (X_1), raw materials (X_2) and landfill taxes (X_3). Sc. 1 presented the highest costs due to the cement consumption (X_2) and the landfill taxes (X_3). In Sc. 2, 2.1, 2.2 and 3, the raw material cost (X_2) was reduced because ash recycling reduced the required amount of raw materials, thus decreasing the costs. In these scenarios, the ash transport (X_1) was a determining parameter, requiring an in-depth assessment of the influence of the distance from the WTE plant to the cement plant. The distance assessment compared Sc. 1 with the least favourable recycling scenarios, Sc. 2.2 and 3. The results showed that for air EB despite the distance increases up to 9,000 km, Sc. 1 had a higher impact. For water, this situation was reversed when the distance exceeded 651 km in Sc. 2.2 and 523 km in Sc. 3. Therefore, from a LCA approach, the distance between the generation point and the recycling plant was an important parameter. Finally, it is important to highlight the LCA methodology as a powerful tool to be used in the decision-making process within the waste management framework.

Acknowledgments The authors gratefully acknowledge the financial support provided by the project FENIX-Giving Packaging a New Life.

Conflict of interests The authors do not have any potential conflict of interest to declare.

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