Material flow, economic and environmental assessment of municipal solid waste incineration bottom ash recycling potential in Europe

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Abstract

In 2018 municipal solid waste (MSW) incineration in Europe produced nearly 19 Mt of bottom ash (BA); only 46 %-wt. was treated, often in poorly performing plants, leaving behind 10 Mt of untreated and unrecovered BA, destined to landfill. This work was based on the inventory of BA across Europe, and on the hypothesis to achieve complete BA valorisation through two assumptions: treating 100 % BA and minimizing the loss of valuable fractions due to technical limitations of state-of-the-art processes in comparison to advanced innovative processes. The research involved three phases: characterization of potential secondary raw materials (metals and mineral fraction) currently lost from untreated (the surplus compared to treatment capacity) and unrecovered BA (the fine fraction) through material flow analysis; environmental assessment (energy balance and net GHG emissions) of complete BA valorisation; investigation of the economic feasibility of complete BA valorisation through state-of-the-art technologies. The resulting 2.14 Mt loss of valuable materials included 1 Mt mineral fraction and 0.97 Mt ferrous metals, mostly from untreated BA, and 0.18 Mt non-ferrous metals, mostly from unrecovered BA. The energy balance and GHGs emissions required by the treatment of the currently untreated and unrecovered fractions of BA resulted in energy and GHGs emissions savings. Economic profitability was driven by iron and copper recycling and avoided landfill fees. Profitability was achieved by two thirds of considered countries (average values: NPV 83 M€, ROI 20 %, payback time 11 years) with BA mass flow exceeding 0.02 Mt.

Keywords: bottom ash; circular economy; municipal solid waste; recycling; thermal treatment; waste-to-energy.

List of abbreviations

Abbreviation	Meaning
MSW	Municipal Solid Waste
BA	Bottom Ash
GHGs	Green House Gasses
NPV	Net Present Value
ROI	Return On Investment
EU	European Union
EFTA	European Free Trade Association
LCA	Life Cycle Assessment
W-t-E	Waste to Energy
D10	Incineration on land; according to EU Waste Framework Directive 2008/98
R1	Use principally as a fuel or other means to generate energy; according to EU Waste Framework Directive 2008/98
FA	Fly Ash
GB	Great Britain
BREF	Best Available Technique (BAT) Reference Document
WFD	Waste Framework Directive
EC	European Commission
PTEs	Potentially Toxic Elements
RQ	Research Question
MFA	Material Flow Analysis
MSWI	Municipal Solid Waste Incineration
$\mathbf{I_{E,untreated}}$	GHGs emissions Index for untreated bottom ash

$I_{E,unrecovered}$	GHGs emissions Index for unrecovered bottom ash
$I_{GHGs,untreated}$	Energy consumption Index for untreated bottom ash
$I_{GHGs,unrecovered}$	Energy consumption Index for unrecovered bottom ash
Capex	CAPital EXpenses
A	Amortization
C_0	Initial capital
i	Interest
n	Numbers of years
OPEX	OPErational EXpenses

1. Introduction

The generation of municipal solid waste (MSW) in Europe in 2018 exceeded 300 Mt (in average 489 kg per capita) (Eurostat, 2020), with different contributions: 219.69 Mt from EU-27 member states, 38.42 Mt from EU candidates (Turkey, Montenegro, Macedonia, Serbia, Albania), 12.71 Mt from the European free trade association members (EFTA, Liechtenstein, Iceland, Norway, Switzerland) and 30.79 Mt from the former EU member Great Britain. Data about MSW production in Cyprus, Greece, Iceland, and Ireland in 2018 are not available on Eurostat yet, thus 2017 values were accounted. It is well known that demographic and socio-economic development strongly influence MSW production and management among the member states (Giannakitsidou et al., 2020). The combination of recycling and thermal recovery was proposed as best option for MSW management from a life cycle analysis (LCA) perspective (Cherubini et al., 2009), also together with the reduction of MSW production rate and limitation of greenhouse gas (GHG) emissions (Behzad et al., 2020). The key role for the European context of coupling MSW enhanced recycling practices with thermal treatments according to Circular Economy principles was already analysed (Abis et al., 2020). Considering the classification of MSW management operations defined by the Waste Framework Directive (WFD) 2008/98/EC, incineration (D10) and thermal valorisation (R1) accounted for over 75 Mt of MSW in Europe in 2018 (Eurostat, 2020), leading to the supply of electricity and heat to respectively 18 M and 15.2 M end-users from waste-to-energy (WtE) plants, and to 90 % waste volume reduction (CEWEP, 2017a). The physical outcome of D10 and R1 are bottom ash (BA, accounting for about 25 %-wt of municipal solid waste incinerated, MSWI) (Enzner et al., 2017), and fly ash (FA, accounting for about 3 %-wt of MSWI) (Morf et al., 2002). Residues from 75 Mt of incinerated MSW in Europe during 2018 (58 Mt in EU-27, 12 Mt in GB, less than 5 Mt in EFTA) (Eurostat, 2020) are 18.75 Mt of BA and 2.25 Mt of FA. BA treatment is common in EU, though processes are specifically designed to recover metals (iron, aluminium, copper, zinc) (Astrup et al., 2016; Šyc et al., 2020), which are the most valuable components (Bunge, 2018). However, BA not only encompass recyclable metals; the inert fraction, mostly consisting of the oxides of of silicon (Si), calcium (Ca), aluminium (Al) and iron (Fe) (Astrup et al., 2016), whether not directly sent to landfill has ready-to-market options as sub-base road filling material, replacing mineral aggregates (Minane et al., 2017; Tang et al., 2015) and also perspectives in ceramic manufacturing (Rincon Romero et al., 2018) and as sorbent material (Fontseré Obis et al., 2017). Worth to be mentioned is the potential recovery for glass cullet e.g. as abrasive medium (lowest open loop recycling possibility) (Silva et al., 2017). However, comparing the above-mentioned estimate of BA produced in Europe calculated from Eurostat (18.75 Mt in 2018) with the 8.4 Mt/y BA treatment capacity reported by the new Best Available Techniques Reference Document (BREF) on Waste Incineration (Neuwahl et al., 2019), it becomes clear that less than 50 % of the BA produced in Europe undergo any treatment. A common EC legislation on BA management does not exist at the moment, thus restrictions for material recovery, if existing, are currently set by each country (Blasenbauer et al., 2020). Alongside profits from metals recovery, one of the main drivers towards the optimization of BA treatment is the necessity to comply with WFD targets and to reduce management costs due to landfill tax (Blasenbauer et al., 2020; Bourtsalas, 2012).

Therefore, the actual framework appears highly complex, considering on one side MSW management practices across EU-27 (in 2018: 49 %-wt recycling, 27 %-wt incineration and WtE and 24 %-wt landfilling) (Eurostat, 2020), and on the other side the further efforts urgently required to member states to fulfil the ambitious Circular Economy targets defined by the EC for the next decade. Improving BA management could be, without any doubts, a key issue. Complete and detailed characterisation of BA and of their management was already performed referring to specific countries, as Belgium (Joseph et al., 2018), Denmark (Allegrini et al., 2014), Germany (Enzner et al., 2017), Italy (Funari et al., 2016), The Netherlands (Loginova et al., 2019), Spain (Del Valle-Zermeño et al., 2017) and for EU, Asian and other countries in a review article (Dou et al., 2017). Most applied utilisation pathways are landfill construction, road construction, concrete aggregate, and cement clinker. Long-term experience exists for application of BA in road construction (Di Gianfilippo et al., 2018; Hysk et al. 2019). In the production of cement clinker BA replaces natural, mined material but still requires firing the rotary kiln (Clavier et al., 2020). Compared to previous studies, this work focused on the quantification and characterisation of BA across all Europe (e.g., instead of in specific countries), comparing countries with different attitudes toward MSWI and BA management. Moreover, to our knowledge, two fundamental aspects were not yet analysed from the technical, environmental, and economic viewpoints, considering state-of-the-art technologies and the whole European context: 1. enhancing the amount of treated BA aiming at reaching 100 % production, and 2. minimizing the losses of potential secondary raw materials from treated BA due to technical limitations of state-of-the-art processes in comparison to advanced innovative processes. Considering the first issue, BA treatment allows in average the recovery of 6.3 %-wt ferrous metals and 1.7 %-wt non-ferrous metals (CEWEP, 2017), therefore 0.8 Mt metals lost were estimated in 2017 from untreated BA (Abis et al., 2020). Considering the second issue, BA fine fractions (dimensions below 2-5 mm, accounting for up to 40-50 %-wt) (Enzner et al., 2017) are usually unrecovered and landfilled to avoid any PTEs release, implying the loss of valuable residual materials (metals and mineral fraction). Therefore, this work aims to answer the following research questions (RQ): RQ1. Quantify and qualify through material flow analysis (MFA) the potential secondary raw materials actually lost from BA, considering both the untreated and the unrecovered fractions (respectively the surplus compared to treatment capacity and the fine fraction); RQ2. Assess the environmental consequences of the potential complete valorisation of BA, accounting energy consumption and savings and net GHG emissions; RQ3. Assess the economic profitability of the potential complete valorisation of BA through state-of-the-art technologies (e.g., the technologies implemented in current full-scale plants treating BA). The economic analysis included capital and operational costs, market value of recovered materials, net present value, return of investment and payback time. Research question 1 derives from the hypothesis of treating 100% of produced BA. Research questions 2 and 3 derive from the need to evaluate not only the technical feasibility of the proposed solution, but also its environmental consequences and economic feasibility. The analyses presented in this work refer to 2018 data, the most recent available on Eurostat and in the scientific literature on MSWI.

2. Methodology

2.1. Quantification of the actual loss of potential secondary raw materials

The quantitative assessment of the actual loss of potential secondary raw materials from untreated (i.e. surplus compared to treatment capacity) and unrecovered (i.e. fine fraction) BA was performed according to material flow analysis (MFA) approach through STAN2WEB open access software (version 2.6.801, http://www.stan2web.net) developed by Technische Universität of Wien according to the Austrian standard ÖNorm S 2096 (Material flow analysis-application in waste management). The MFA was based on the following assumptions: amounts of total available BA in specific countries were calculated as 25 %-wt of MSWI in 2018 (Eurostat, 2020), then compared with current national BA treatment capacity (Blasenbauer et al., 2020) to obtain the amount of untreated BA; unrecovered BA amounts were calculated considering the cut-off particle size of recoverable fraction in each country (Enzner et al., 2017), then multiplied to the corresponding cumulative percentage from a characteristic BA particle-size distribution curve (Šyc et al., 2020) and to the amount of BA treated in the same country. If data about minimum recoverable particle-size were missing for a certain country, the average value 4 mm (50 % cumulative percentage on BA granulometric distribution curve) was considered as technological limit. The result of this evaluation, here-in-after named "unrecovered fraction", assumed that BA treatment plants in Europe (Neuwahl et al., 2019) worked at 100 % capacity (the BREF reports two values for each plant: one referred to the average capacity of each plant and another to 100 % capacity). The material recovery efficiency of BA treatment technologies was assumed 100 %, to estimate the overall theoretical amount of potentially recoverable secondary raw materials. Finally, the hypothesized recovery of potential secondary raw materials involved mineral aggregates or glass recycling (mineral fraction) and secondary smelters (metal fractions), because the technical feasibility of these perspectives was already proven (Buekens, 2013; Bunge, 2018; Clavier et al., 2020; Lam et al., 2010; Neuwahl et al., 2019; Verbinnen et al., 2017).

2.2. Characterization of untreated and unrecovered BA

BA quality was described in terms of macro-components (Neuwahl et al., 2019; CEWEP, 2017a) as follows: 85-90 %-wt mineral fraction, 5-10 %-wt ferrous metals and 2-5 %-wt non-ferrous metals. Several studies (Allegrini et al., 2014; Del Valle-Zermeño et al., 2017; Astrup et al., 2016) highlighted the presence of glass cullet in BA, whose recycling could increase the market value of the mineral fraction. The amount of glass cullet in BA mineral fraction was estimated 11.9 %-wt in 0-2 mm quota (Del Valle-Zermeño et al., 2014) and 8.6 %-wt in above 2 mm fraction (BASH TREAT, 2020)- Ferrous metals were all assumed steel scrap; the amounts of non-ferrous metals were estimated 68 %-wt aluminium and 28 % copper in untreated BA (CEWEP, 2017), and 45 %-wt aluminium and 50 %-wt. copper in unrecovered BA, referring to the amounts detected in fines below 5 mm (Neuwahl et al., 2019).

2.3. Environmental assessment

The environmental assessment of BA valorization was based on two viewpoints (energy balance and GHG emissions), in comparison with extraction and manufacture of construction aggregates, glass and metals from raw materials. In VDI guideline 3925 (VDI, 2016) it was shown that these two viewpoints have highest relevance to the environmental performance of BA treatment whereas other impact categories used for example in life cycle assessment (LCA) such as acidification potential, human toxicity potential or else are negligible **(**Gehrmann et al., 2017).

2.3.1. Energy demand and savings

Specific energy demand of BA treatment (kWh/t) was calculated multiplying the energy required by treatment plants (kWh) published on the new BREF on waste incineration (Neuwahl et al., 2019) to the amounts of BA treated (t) in single countries in 2018 according to the same reference document (see Supplementary Material, Table I). Each treatment plant was fed by different energy sources, categorized as electricity, natural gas, steam (all expressed in MWh) and liquid fossil fuel, reported in liters and converted to MWh (1 L = 9.1 kWh). Energy consumption values of single plants were referred to the corresponding amount of BA treated, obtaining a weighted average value of 8.28 kWh/t, which was comparable with the value (10 kWh/t) obtained by previous studies (Bunge, 2018). The net energy potentially saved, i.e., the difference between the energy necessary for the primary production of materials from natural resources and the energy necessary for materials manufacturing from secondary production, was derived from literature (Appendix, Tables IIa-IId). In details, we considered a saving of energy demand between primary production and recycling equal to 4.11 kWh/t for aggregates (Marinković et al., 2010), 527.8 kWh/t for glass cullet (Larsen et al., 2009), 2166.7 kWh/t for Fe, 51216.7 kwh/t for Al and 4138.9 kWh/t for Cu (Grimes et al., 2008; Norgate and Haque, 2010; Norgate et al., 2007). We assumed that BA treatment results in recycled aggregates ready to use, thereby the potential energy and GHGs emissions savings refer to the energy saved from primary aggregates' production, without considering any further treatments. The literature values employed for the calculation of the net energy potentially saved were published in 2005-2010; we based our analysis, referred to 2018, to the mentioned references in absence of more recent ones. Net energy consumption values were obtained, both for untreated BA and for unrecovered BA, by difference between energy consumption of BA treatment and potential energy savings because of avoided raw materials production (i.e., aggregates, glass, metals). We considered positive an energy balance in which the energy demand necessary to process untreated and unrecovered BA was lower than the energy savings related to the avoided production of corresponding raw materials. To compare different countries, net energy consumption was referred to the specific amounts of untreated BA, through a specific index of energy consumed $I_{E\ untreated}$ (eq. 1), and to the specific amounts of unrecovered BA, through a specific index of energy consumed $I_{E\ unrecovered}$ (eq. 2).

$$I_{E\ \mathrm{untreated}} = \frac{\text{energy consumed} - \text{energy saved}}{\text{untreated material}} \left[\frac{kWh}{t}\right]$$

$$I_{E\ \mathrm{unrecovered}} = \frac{\text{energy consumed} - \text{energy saved}}{\text{unrecovered material}} \left[\frac{kWh}{t}\right]$$

2.3.2. GHG emissions

GHG emissions were calculated as: produced emissions over 100 years related to BA treatment (values were between 0.007 and 1.13 kg $CO_{2\ eq}$/kWh, the analysis considered specific values for each country) (Fruergaard et al., 2009). A country-specific analysis was performed considering the specific GHG emission factors of energy production for non-household consumers (EEA, 2020) (Appendix, Table III). Since emission factors of Albania, Montenegro, Serbia, Turkey, Liechtenstein, Iceland, Norway and Switzerland were missing, an average emission factor 0.569 kg $CO_{2\ eq}$/kWh (Fruergaard et al., 2009) was accounted.

On the other hand, avoided GHG emissions related to raw materials production has been considered as the difference between the emissions due to primary and secondary production, as following: -1.50x10^{-3} kg $CO_{2\ eq}$/kg for aggregates, 0.50 kg $CO_{2\ eq}$/kg for glass, 1.06 kg $CO_{2\ eq}$/kg for Fe, 12.72 kg $CO_{2\ eq}$/kg for Al and 0.97 kg $CO_{2\ eq}$/kg for for Cu (Appendix, Tables IVa-IVd). As the mineral components of the fine fraction were considered inert, no GHG emissions related to their landfill disposal were accounted.

Specific GHG emission indexes were defined for the amount of currently untreated BA, $I_{GHG,untreated}$ (eq. 3) and for the specific amounts of currently unrecovered BA, through a specific index of energy consumed $I_{GHG,unrecovered}$ (eq. 4).

$$\mathrm{I_{GHG.untreated}} = \frac{\text{GHG emissions} - \text{GHG saving}}{\text{untreated material}} \left[\frac{t\ CO_2}{t}\right] \tag{3}$$

$$\mathrm{I_{GHG.unrecovered}} = \frac{\text{GHG emissions} - \text{GHG saving}}{\text{unrecovered material}} \left[\frac{t\ CO_2}{t}\right] \tag{4}$$

2.4. Economic assessment

A cost-benefit analysis compared capital and operational costs with potential benefits (e.g. revenues from potential secondary raw materials sale and savings from avoided landfilling and primary raw material extraction) in order to determine profitability. The total amounts of untreated and unrecovered BA were assumed as operational units. Capital investment costs (CAPEX, eq. 5) (Bunge, 2018) included plant installation and equipment (Appendix, Figure I).

The cost of land for new treatment plants was neglected, due to the high variability within Europe and to perform a non-country-based analysis.

$$\text{CAPEX}\ [€] = 10000 \cdot \text{troughtput}\ [\text{t}]^{0.5} \quad (1)$$

Five years amortization with 10 % interest (Bunge, 2018) was assumed for the investment cost (eq. 6):

$$A\ [€] = C_0 \cdot \frac{i \cdot (1+i)^n}{(1+i)^n - 1} \quad (2)$$

where A is the amortization cost, C_0 is the initial capital, i is the interest and n the number of years considered for amortization.

The operational costs (OPEX) involved the sum of labour (eq. 7), plant maintenance (eq. 8) and energy (eq. 9) costs (Bunge, 2018):

$$\text{labour cost}\ [€] = 6 \cdot \text{throughput}\ [\text{t}] \quad (3)$$

$$\text{plant maintenance cost}\ [€] = 0.08 \cdot \text{CAPEX}\ [€] \quad (4)$$

$$\text{energy cost}\ [€] = \text{energy price}\left[\frac{€}{\text{kWh}}\right] \cdot \text{energy consumption}\ [\text{kWh}] \quad (5)$$

The national prices for non-household electric energy (€/kWh) derived from Eurostat (Appendix, Table V); for the countries not included in the database (Estonia, Hungary, Latvia, Lithuania, Luxembourg, Malta, Slovenia, Albania, Montenegro, Serbia; Liechtenstein, Iceland and Switzerland) the average value 0.117 €/kWh was accounted. Among the operational costs, landfill expenses for the disposal of the mineral part of the fine fraction (considered, according to literature, too contaminated to be recovered) were accounted (Appendix, Table VI) (CEWEP, 2017). Landfill costs for most countries were defined by European Environment Agency (European Environmental Agency, 2014), while for Switzerland landfill tax was 50 €/t (CEWEP, 2017).

Potential incomes from secondary raw materials sale were estimated assigning a specific market value to each fraction. In detail, BA mineral fraction was compared to construction aggregates (average value 9 USD/t per metric ton, USGS, 2020), accounted as 8.2 €/t. The market value assigned to recycled glass was 20 €/t (Rincon Romero et al., 2018). Commercial values of 100 €/t, 500 €/t and 3600 €/t were assigned to iron scrap and non-ferrous metals (aluminium and copper) respectively (Bunge, 2018). Increased BA recovery also implied savings related to reduction of landfilling and primary raw materials extraction. Profitability of untreated and unrecovered BA valorisation was assessed through net present value (NPV), return of investment (ROI) and payback time (Appendix, Table VII).

3. Results and discussion

3.1. Quantification of the actual loss of potential secondary raw materials

MSWI plants are not homogenously spread across across Europe, with only few countries owning three quarters of incineration capacity (Eurostat, 2020). Regulations related to BA recovery are not consistent and uncertainties occur within reported data possibly due to a not univocal definition of recovery, which for mineral materials can imply metal separation either followed by landfilling or recovery as aggregates (Blasenbauer et al., 2020). Specifically considering BA management (Figure 1), the countries where BA production exceeded treatment capacity were: Finland, France, Luxembourg, Portugal, Spain and Sweden. In France and Portugal, the surplus corresponded respectively to 6 % and 15 % of produced BA, whereas in Finland and Luxembourg it was 31 % and the countries with even higher surplus were Sweden (46 %) and Spain (72 %).

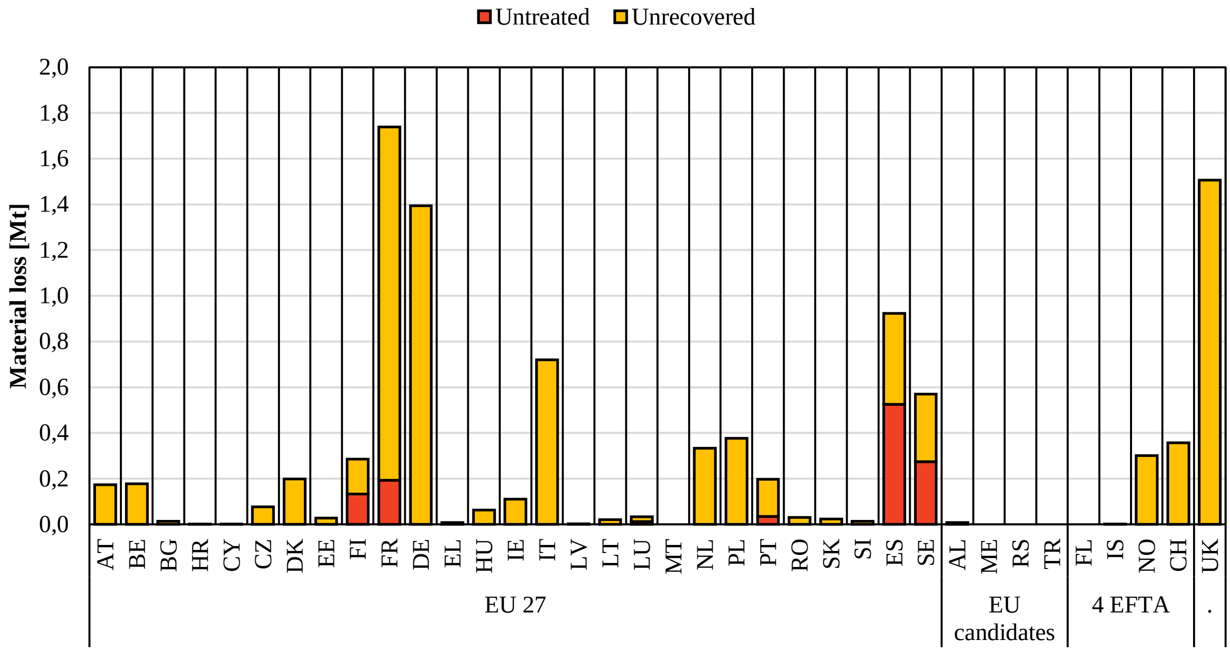


Figure 1. Bottom ash management in Europe in 2018: treatment capacity and untreated and unrecovered fractions (calculated from Eurostat, 2020; Neuwahl et al., 2019) (red: untreated; yellow: unrecovered).

No correlation appeared between the amount of produced BA and the untreated surplus exceeding national treatment plant capacity (R^2 = 0.0307), nor between BA production and installed treatment capacity (R^2 = 0.5216) (Appendix, Figure II). As for unrecovered BA, whose under-exploitment represented the main loss in terms of secondary raw materials, its amount seemed to be related to the amount of produced BA (R^2 = 0.9605) (Appendix, Figure III). This means that the largest contribution to unrecovered BA was associated to the top four producers (Germany, France, Great Britain and Italy), despite them being among the best performing countries in terms of BA treatment, being able to recover BA with particle size down to 2 mm (4 mm in Italy) (Enzner et al., 2017). Nevertheless, although the technological levels reached by each country showed lesser influence, minor BA producers, as Spain and Portugal, were responsible for the production of considerable amounts of unrecovered BA, due to their inability to recover fractions respectively below 5 and 10 mm grain size (Enzner et al., 2017).

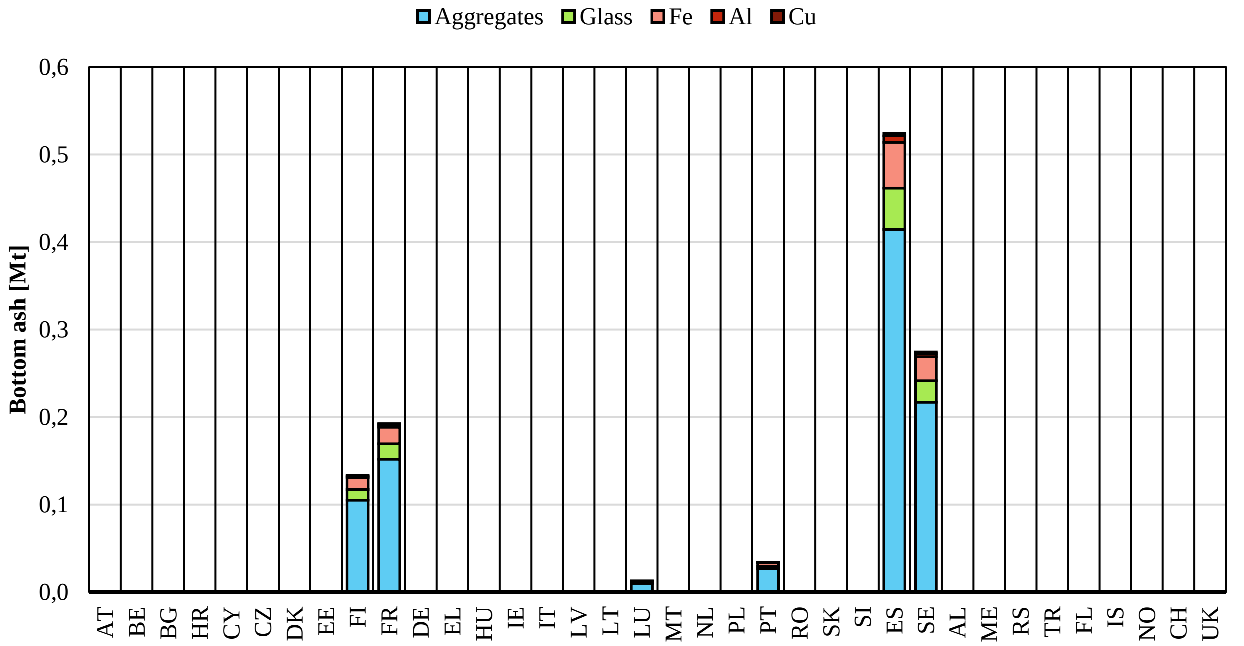
Although the main aim of MSWI is energy recovery, it also plays a key role in reducing the amount of landfilled waste (up to 90 % by volume and 75 %-by weight) (CEWEP, 2019). However, the results of MFA performed on MSWI in Europe in 2018 (Appendix, Figure IV) showed that 54 %-wt (10.24 Mt out of 18.82 Mt) of BA was landfilled, mainly due to underperforming treatment facilities, and 46 %-wt was destined to material recovery.

3.2. Characterization of untreated and unrecovered bottom ash

Several studies analysed BA composition in order to identify potential barriers that could hinder recovery, as content of hazardous substances or leaching behaviour (Kalbe and Simon, 2020; Alam et al., 2019; Schafer et al., 2019; Verbinnen et al., 2017), or to investigate new recovery perspectives (among others: Dou et al., 2017; Šyc et al., 2020; Yang et al., 2018). The knowledge of BA average composition (section 2.2) allowed to estimate specific material losses in European countries related to untreated and unrecovered BA. Considering untreated BA (Figure 2A), it was clear that higher BA production did not necessarily imply larger material losses, since the technological limit that defined the smallest recoverable particle size was essential. As an example, France showed larger material loss than Germany, despite the latter is the European country with largest MSWI capacity and thereby BA production; similarly, Spain, Sweden, and Poland, which produced lesser amounts of BA, contributed to a greater material loss due to their inefficient BA treatment infrastructures. The treatment of unrecovered BA is crucial to reduce pollution potential in case of landfilling and to recover metals to make the process profitable (Allegrini et al., 2014). Management of unrecovered BA could be challenging because of high concern on PTEs. Copper, zinc and other metals showed increasing concentration in BA fine portions (Loginova et al., 2019), however the mineral components of BA fine fraction exhibited high superficial contamination, precluding their recovery. For this reason, this work estimated potential loss of secondary raw materials from unrecovered BA considering only metals (iron, aluminium and copper) (Figure 2B), and presumed landfilling of the remaining mineral fraction.

The overall 2.14 Mt material loss, resulting from untreated and unrecovered BA (without the mineral fine fraction, destined to landfill) (Figure 2A and B) consisted of about 1 Mt mineral fraction (of which 0.9 Mt glass cullet), 0.97 Mt ferrous metals (steel scrap) and 0.18 Mt non-ferrous metals. The main losses were related to unrecovered mineral fraction (42 %-wt of potentially available amount) and steel scrap (45 %-wt of potentially available amount). Worst results were observed for Spain (45 %-wt mineral fraction lost) and for France (18 %-wt ferrous and non-ferrous metals lost). Spain was the only country where, because of the huge gap existing between BA production and treatment capacity and of the different composition of larger and fine BA fractions (see section 2.2), the amount of copper lost within untreated BA (0.0051 Mt) was higher than the amount in unrecovered BA (0.0039 Mt) (Figure 2B). Iron recovery efficiencies reached in standard-level BA treatment facilities were generally medium to high (Bunge, 2018), however, being iron the main metal component in BA (Astrup et al., 2016), the lack of treatment plant capacity caused material loss up to 75 % of available amount because of landfilling of untreated and unrecovered BA (Figure 3A). Besides, despite aluminium is separated from MSW through separate collection, still a considerable amount is found in BA (see section 2.2). Only 0.04 Mt out of total 0.25 Mt present in BA (16 %) was recycled in 2018 (Figure 3B). Similarly, only 40 % (0.02 Mt out of 0.05 Mt) of copper present in BA was recycled (Figure 3C). In this last case the major loss was due to unrecovered BA fine fraction, where copper concentrates, and it could be prevented by upgrading the existing BA treatment infrastructures.

A



B

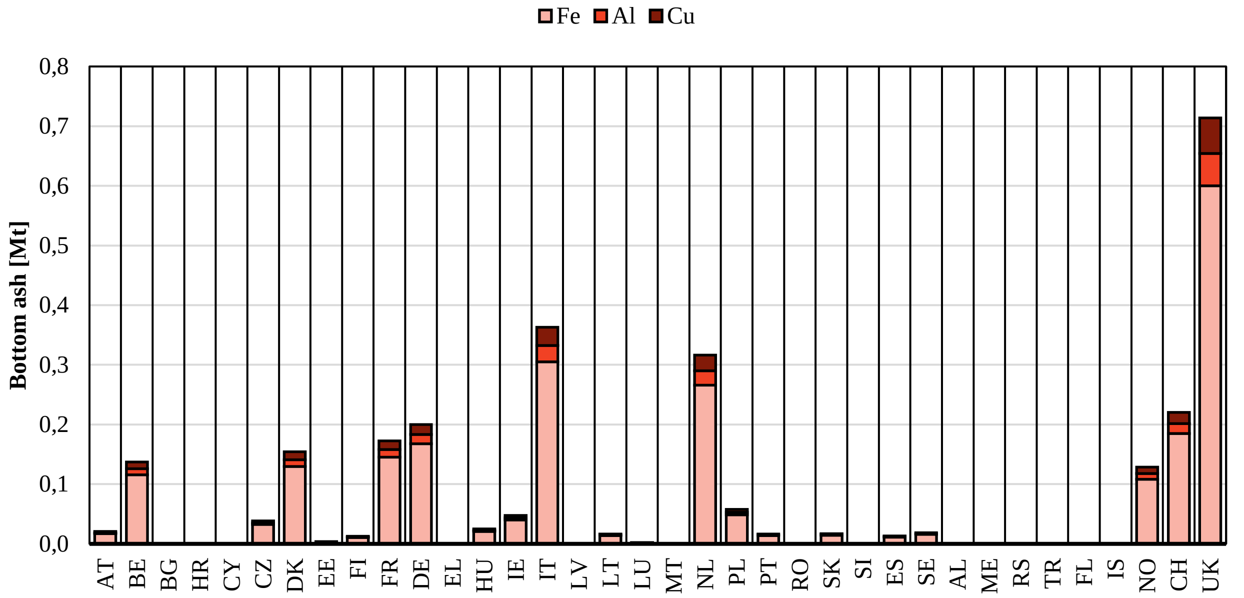


Figure 2. Characterization of: A) untreated bottom ash (in 6 countries, where BA production exceeded treatment capacity) (blue: mineral fraction, green: glass, red: iron, orange: aluminium, brown: copper) and B) unrecovered bottom ash in Europe in 2018 (pink: iron, orange: aluminium, brown: copper)

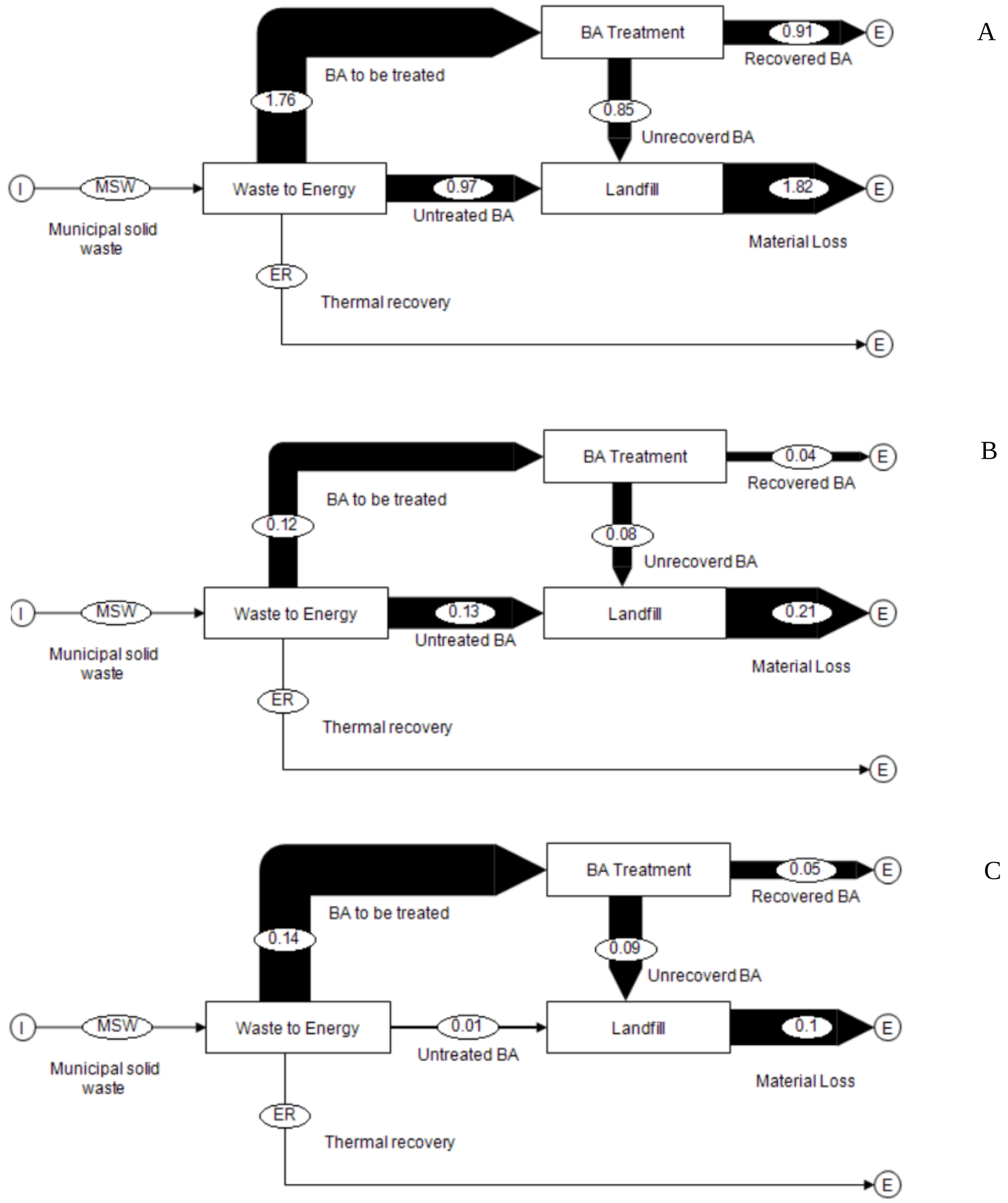


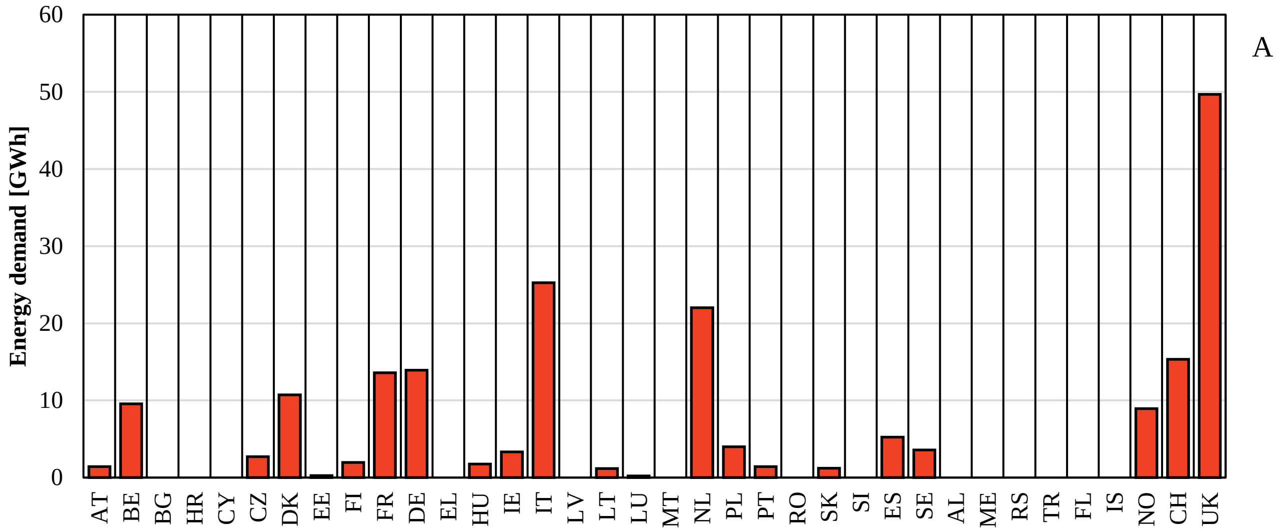
Figure 3. Results of Material Flow Analysis of: A) iron, B) aluminium and C) copper in bottom ash management in Europe in 2018 (MSW: municipal solid waste, BA: bottom ash)

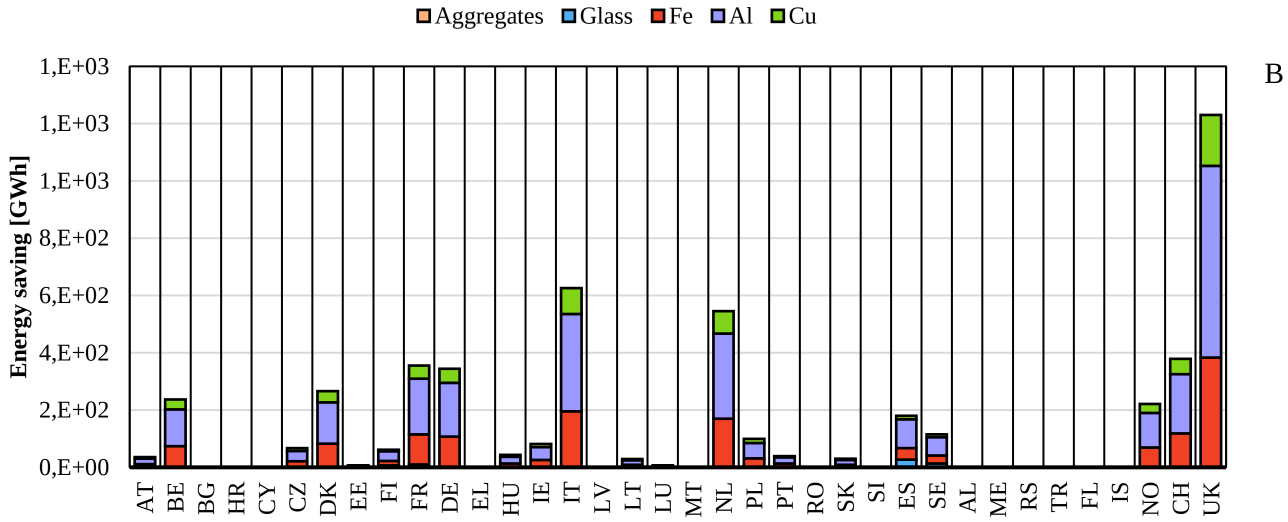
Some general statements about the significance in the European context of the above-mentioned losses of potential secondary raw materials could be formulated. Although being one of the most common metals in earth crust, iron mining in Europe barely accounts 12 %-wt. global production, despite the presence of important steel manufacturing industries in Germany, Italy and France (European Commission, 2017). Copper concentration in BA fine fractions is noteworthy and although it is not currently listed as critical raw material, the only European country in which copper is mined is Poland, accounting only for 2.6 %-wt. global production. Therefore Europe relies almost completely on copper imported from South America (27.6 % Peru, 22.1 % Chile, 9.5 % Brazil and 9.1 % Argentina) and Indonesia (10.9 %), and copper recycling from end-of-life products is highly encouraged (European Commission, 2017).

3.3. Results of environmental analysis

3.3.1. Energy balance

The energy demand values estimated for the complete treatment of BA, considered as sum of the currently untreated and unrecovered fractions, in all European countries are shown in Figure 4A. Whereas, energy savings were estimated calculating the energy required for the extraction and processing of natural resources to produce aggregates, glass, iron, aluminium, and copper. The fine fraction, which was part of the untreated BA and thus contributed to the energy demand, was excluded from energy savings from unrecovered BA because destined to landfill, and due to this issue, it was not possible to obtain a real estimate of the energy balance of the complete BA valorisation scenario.





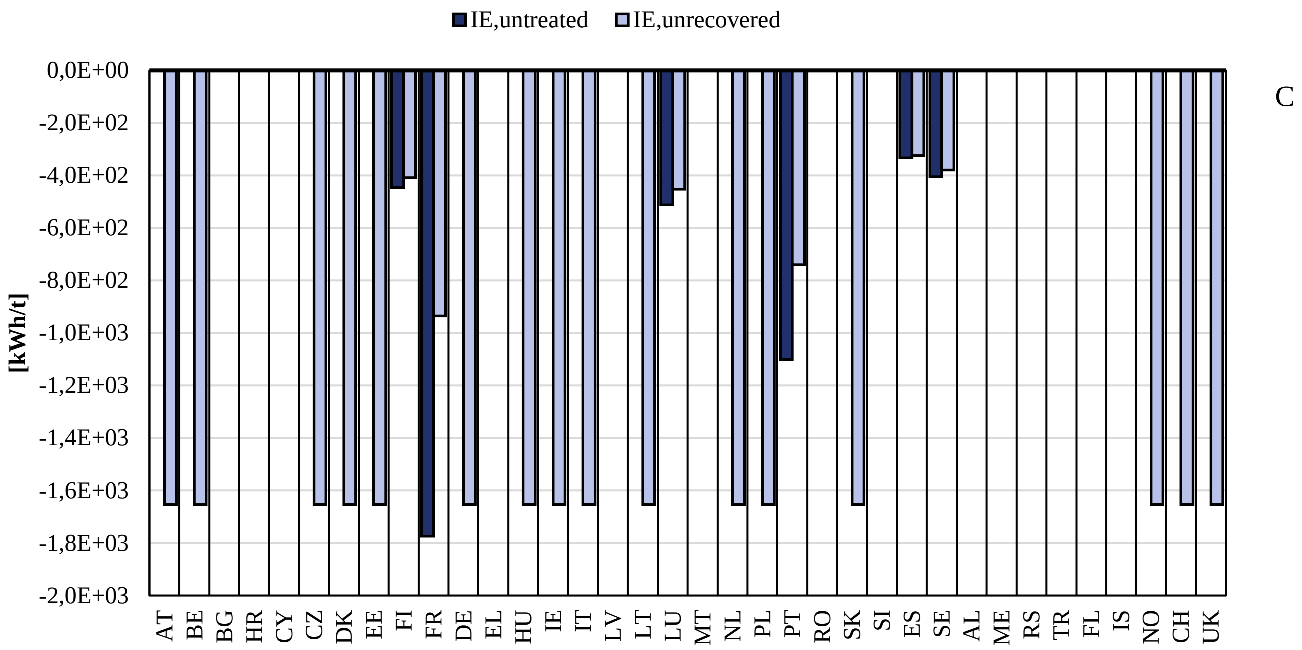
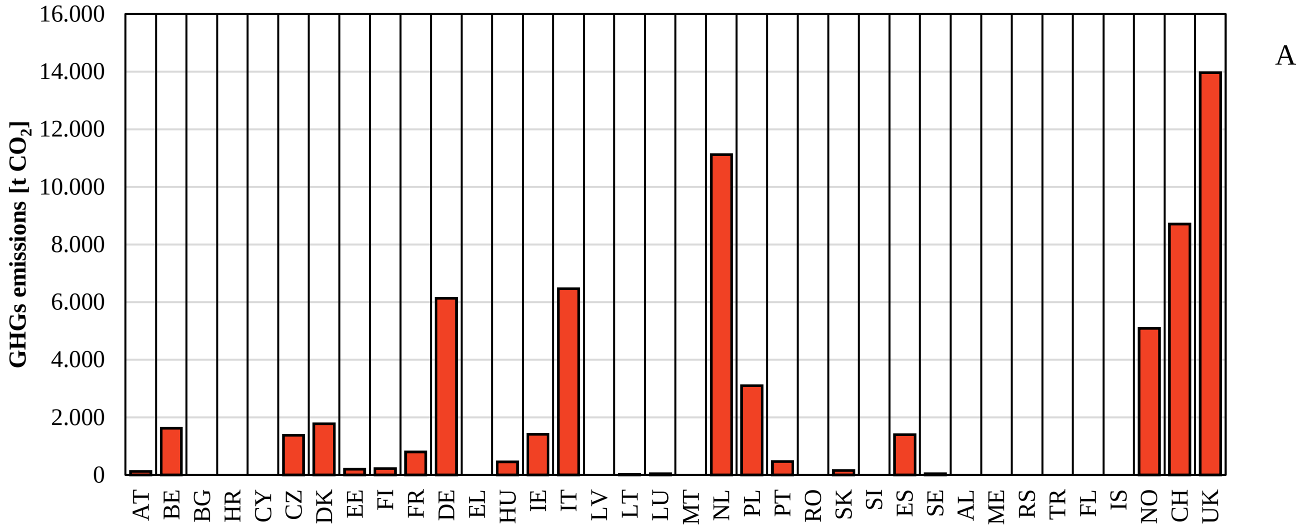


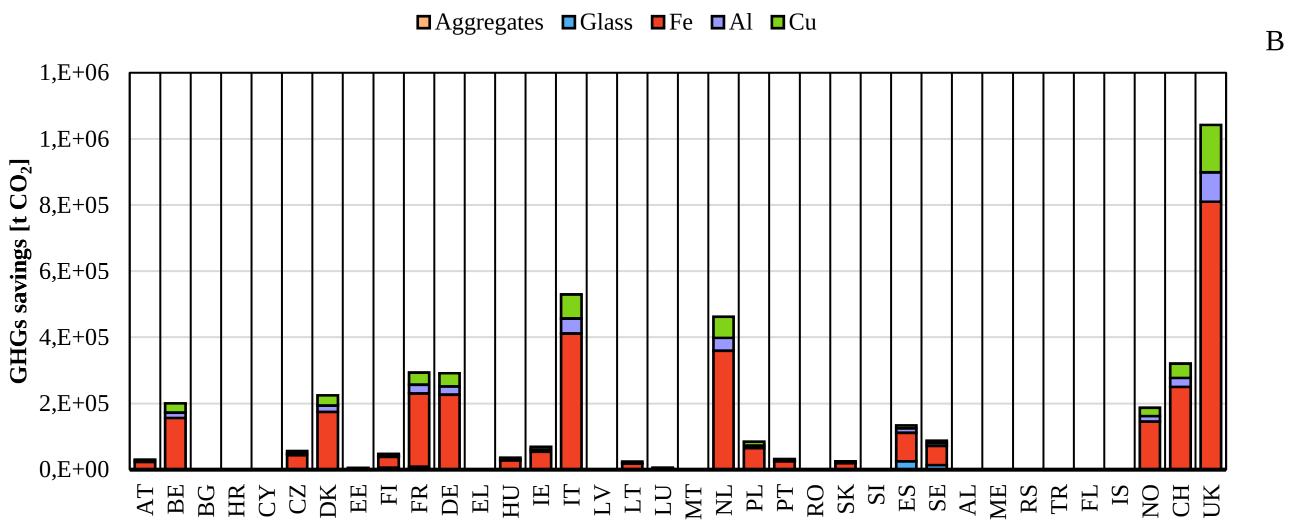
Figure 4. A) Energy consumption for bottom ash management in Europe in 2018 (GWh); B) Energy savings for materials recovery from bottom ash treatment (pink: aggregates, blue: glass, red: iron, violet: aluminium, green: copper); C) Specific net energy consumption per ton of untreated and unrecovered bottom ash in 2018 (kWh/t) (dark blue: IE untreated, light blue: IE unrecovered)

Therefore, the net energy consumption (i.e. the difference between the energy consumption of the treatment minus the energy savings related to materials' recovery) (Figure 4B), calculated as defined in section 2.3.1, appeared a more reliable indicator. The net energy consumption was slightly related to the amount of recovered material (R^2= 0.52), mainly because of the correlation (R^2= 0.54) observed between the amount of recovered material and the energy savings (Appendix, Figure V). Finland, Sweden, and Spain were furthest away from the trend observed for other countries, showing a much smaller net energy consumption compared to what should be expected from their national amount of recoverable material. The rationale of this behaviour could be found in the fact that these countries were among the top producers of untreated BA (Figure 2A), thereby their BA potential recovery was characterised by a considerable amount of mineral and glass components in the coarser fraction, which, being recoverable, entailed energy saving that drastically reduced the net energy consumption.

3.3.2. GHG emissions

GHG emissions were evaluated comparing the avoided emissions related to materials recovery (compared to production from natural resources, in kg $CO_{2\ eq}$/t) (see section 2.3.2), with the emissions produced by the treatment of untreated and unrecovered BA, in t/year (Figure 2).


A
GHGs emissions [t CO2]
16.000
14.000
12.000
10.000
8.000
6.000
4.000
2.000
0
AT BE BG HR CY CZ DK EE FI FR DE EL HU IE IT LV LT LU MT NL PL PT RO SK SI ES SE AL ME RS TR FL IS NO CH UK


B
Aggregates
Glass
Fe
Al
Cu
GHGs savings [t CO2]
1,E+06
1,E+06
8,E+05
6,E+05
4,E+05
2,E+05
0,E+00
AT BE BG HR CY CZ DK EE FI FR DE EL HU IE IT LV LT LU MT NL PL PT RO SK SI ES SE AL ME RS TR FL IS NO CH UK

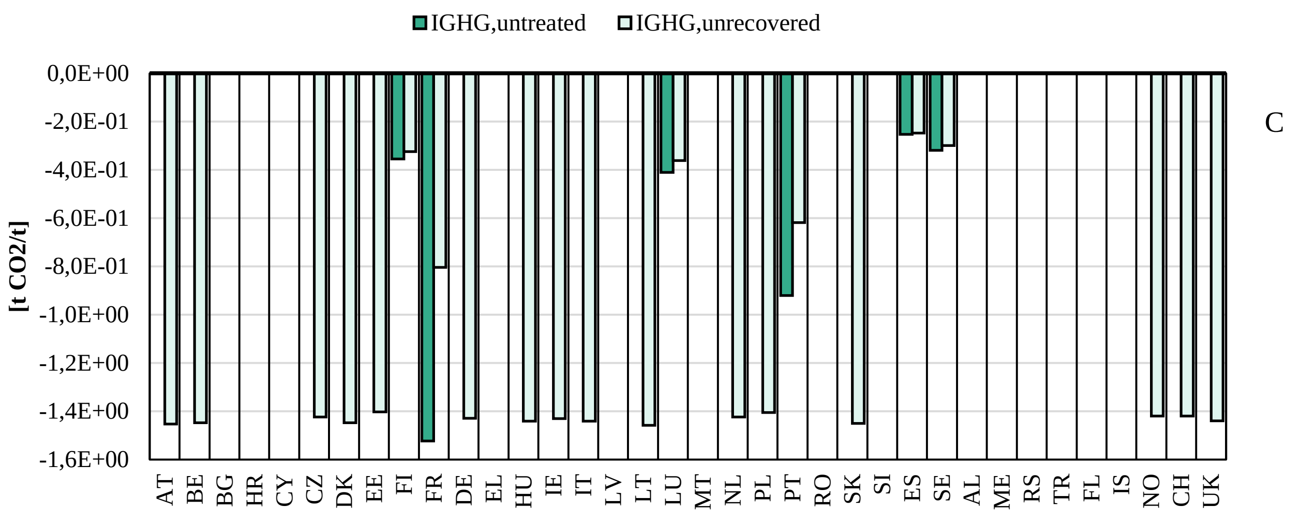
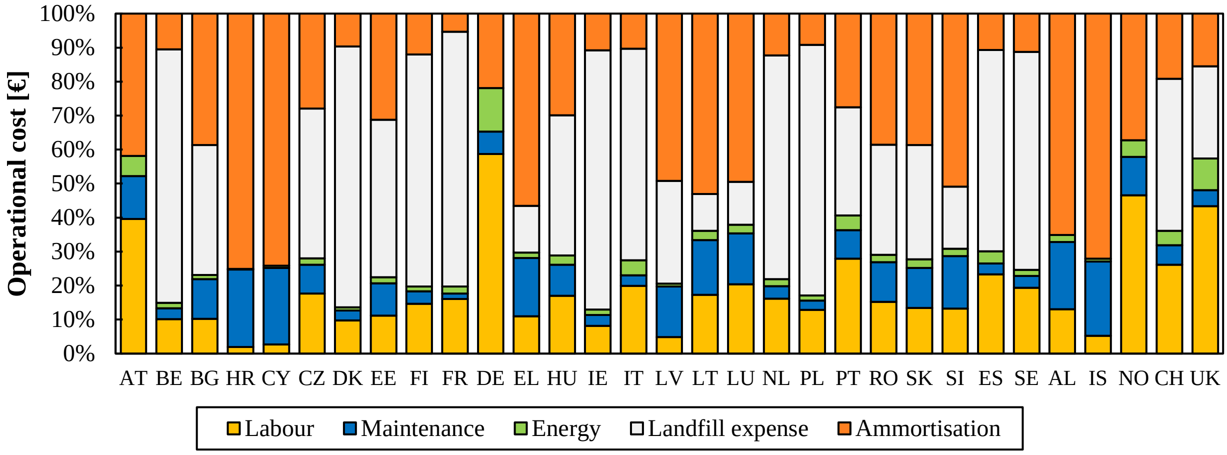

C
IGHG,untreated
IGHG,unrecovered
[t CO2/t]
0,0E+00
-2,0E-01
-4,0E-01
-6,0E-01
-8,0E-01
-1,0E+00
-1,2E+00
-1,4E+00
-1,6E+00
AT BE BG HR CY CZ DK EE FI FR DE EL HU IE IT LV LT LU MT NL PL PT RO SK SI ES SE AL ME RS TR FL IS NO CH UK

Figure 5. Bottom ash recycling in Europe in 2018: A) GHG emissions for bottom ash management in Europe in 2018 (t $CO_{2\ eq}$); B) GHG emissions avoided detailed for the different materials (t $CO_{2\ eq}$) (pink: aggregates, blue: glass; red: iron; violet: aluminium; green: copper); C) Specific GHG emissions related to untreated and unrecovered bottom ash in Europe in 2018 (t $CO_{2\ eq}$/ t) (dark green: IGHG untreated, light green: IGHG unrecovered)

Considering GHG emissions avoided by specific materials' recycling (Figure 5B), consistently for almost all European countries, iron recovery seemed related to the highest absolute GHG emission saving, as it is the dominant metal in untreated BA and its recovery is usually followed by mineral components recovery. Copper, despite being less common than aluminium in BA, showed higher specific GHG emission saving. Since metals recycled from BA should anyway undergo a series of refining treatments, the potential GHG emission savings considered in this study were referred only to the concentration from mineral ore, and in that case copper had a larger impact than aluminium, as demonstrated by several studies (Simon and Holm, 2016; Hanle et al., 2006; Jeswiet and Szekeres, 2016; Norgate and Haque, 2010; Norgate et al., 2007; Nuss and Eckelman, 2014). Net GHG emissions values, calculated considering country-specific GHG emissions (deriving from energy production and the amount of energy required to process untreated and unrecovered BA, see section 2.3.2), showed that material recovery from BA resulted in a far less impacting process than raw materials mining and production, thereby the difference between GHG emissions generated by the two perspectives resulted negative values for all countries.



A

B

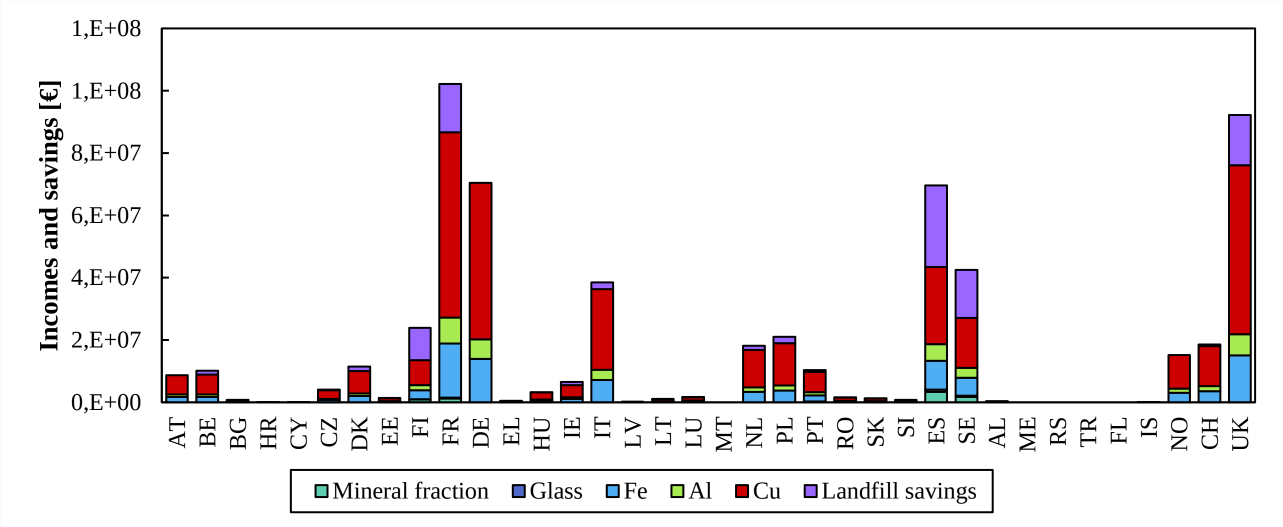


Figure 6. Country-specific details of bottom ash valorisation: A) operational costs (yellow: labour, blue: maintenance, green: energy, white: landfill expenses, orange: depreciation) and B) potential incomes (mineral fraction, glass, Fe, Al, Cu) and savings (saved landfill expenses) (light blue: mineral fraction, dark blue: glass, turquoise: iron, green: aluminium, red: copper, purple: saved landfill expenses)

3.4. Results of the economic analysis

Total capital costs related to untreated and unrecovered BA (Appendix, Figure VI) were obviously dependent from mass throughput, as the highest values were attributed to France, Germany, Great Britain, and Spain, which are among the major BA producers. However, the amount of the required investment did not depend specifically on whether BA belonged to the untreated or unrecovered category. Contrarily, if specific capital costs were considered, the countries with the lowest throughput were characterised by the highest values. Assessing the operational costs more factors were involved; despite the inverse correlation observed between operational cost and treated mass flow, the overall operational cost depended also on country-specific parameters as energy cost and landfill tax. Considering country-specific detailed operational costs (Figure 6A), the main contribution for most countries was the landfill tax related to the disposal of the fine mineral fraction. However, the framework was not homogeneous; countries such as Hungary, Albania and Czech Republic imposed low landfill tax on waste management operators, and in Austria residual waste from WtE plants are exonerated from landfill fees (CEWEP, 2019). The operational costs due to energy consumption appeared strongly dependent on the amount of energy required rather than on the national fee set for non-household energy consumers (Appendix, Table IV). Assessing the incomes from the sale of recovered materials and the savings from avoided landfilling (Figure 6B), copper recovery was the main economic driver because of its high market value; however, countries such as Finland, France, Spain, Sweden, and Great Britain, where landfill disposal is more expensive than the European average, did benefit from the saving of landfill fees. It is worth noting that iron recovery implied incomes larger than aluminium, notwithstanding its lower market value. This was due to the fact that, compared to aluminium, iron content in BA was higher and its recovery requires less effort from the technical and therefore economic viewpoint. Iron can be recovered easily with magnets whereas for aluminium (with much lower concentration in BA than Fe) highly efficient eddy current separators (discrete ECS for different grain size fractions, (Enzner, 2017) usually are necessary. Copper and copper alloys are separated with the eddy current separators as well. The higher effort is justified because secondary Al requires much less energy than primary Al (more <90% savings) and, besides energy savings also for secondary Cu, natural Cu resources are conserved (Simon and Holme, 2016).

From the simple comparison of overall country-specific costs and incomes and savings related to BA valorisation, it appeared that in countries with lower BA mass throughput, costs exceeded potential incomes and saving. Thereby countries as Hungary, Cyprus, Latvia, and Iceland would record a negative cash flow. The potentially necessary plant size was not the only element determining the positive outcome of the investment, as among the countries with negative cash flows are also listed Denmark and Ireland, despite their respective BA potential of 0.2 Mt and 0.11 Mt, which are one order of magnitude higher than Estonia or Lithuania and two higher than Greece, where the cash flows was instead positive. The justification of this apparent contradiction was found pointing out that Denmark and Ireland adopted the highest landfill fees throughout Europe, thereby the expenses due to the management of BA unrecoverable fraction did not justify other operational costs. Except for Great Britain, where a discounted landfill fee for the disposal of processed BA is applied, in all other countries landfill taxes played a dual role on the economic analysis performed in this study, as they represented a potential saving generated by the recovery of untreated BA and metal components of unrecovered BA, but still needed to be listed as costs related to the management of mineral fine unrecovered fraction. Hence, the economic feasibility of BA valorisation was mainly dictated by how much other factors, such as cost of energy and valuable metals concentration, can shift the balance to a positive outcome.

Considering profitability, net present value (NPV) of the treatment plant exhibited average value of 83.36 M€ (Appendix, Figure VIIA) and was negative in 25 % European countries. The worst performances were observed among the countries with lower BA production (Bulgaria, Hungary, Cyprus, Estonia, Ireland, and Iceland), which returned negative NPV after 20 years. Whereas the highest NPV values were reported among the countries previously identified as major European BA producers (Great Britain, Germany, Spain, France, Sweden, Italy, Finland, Norway, and Switzerland). The average return on investment (ROI) was 20 % (Appendix, Figure VIIB) and the highest values (> 50 %) were observed for Germany, Spain, Sweden, and Great Britain. Hungary, Cyprus, Ireland, and Latvia were characterised by negative ROI; thus, the investment was not profitable. Payback time was evaluated for the countries characterised by positive NPV and ROI values (Appendix, Figure VIIC), and they all met payback time before 20 years. The average time required by the investment to break even on income-outcome trade-off was 11 years, however, for most countries (67 % of the ones with payback time below 20 years) payback time was shorter. Estonia, Greece, Slovenia, and Albany, despite a consistent positive outcome with NVP and ROI were characterised by a payback time higher than the 20 years useful plant lifetime, thereby the economic assessment for these countries was defined not profitable. The countries with profitable scenarios were the ones with the higher amount of produced BA plus Hungary, Czech Republic, and Luxembourg, which accounted for relatively lower amounts of produced BA but were characterized by lower-than-average operational costs, which did justify the investment in improving BA recovery.

The economic analysis resulted positive (NPV and ROI >0 and payback time < 20 years) for 66 % of the analysed European countries: Austria, Czech Republic, Finland, France, Germany, Hungary, Italy, Lithuania, Luxembourg, Netherland, Poland, Portugal, Romania, Slovakia, Spain, Sweden, Norway, Switzerland, and Great Britain. The minimum BA mass flow among the countries with positive economic analysis was 0.02 Mt and this was consistent with the maximum mass flow among the countries where investment was marked unprofitable, except for Belgium and Denmark where the high landfill tax fees leaded to a negative economic profitability.

3.5. Policy implications

The positive effects of BA recycling economically and environmentally have also been recognized by politicians. E.g., in Switzerland, where utilization of the mineral fraction is not applied, non-ferrous metals must be separated to less than 1% (Schweizerischer Bundesrat, 2015). In the BREF document on waste incineration BAT conclusion 36 lists the best available technologies to increase the resource efficiency (Neuwahl et al., 2019). These BAT conclusions are the basis for future legislation on waste management in the EU countries. The present investigation clearly shows that the extension of BA treatment has positive effects.

4. Conclusions

This work addressed three research questions associated to the assessment of MSWI BA recycling potential in Europe, as follows.

RQ1. Quantify and qualify through material flow analysis (MFA) the potential secondary raw materials lost from BA, considering both the untreated and the unrecovered fractions.

- In 2018, 75 Mt of incinerated MSW in Europe generated almost 19 Mt of BA; 54 %-wt, related both to untreated BA and to technical limitation of treatment facilities (e.g., cut-off particle size for eliminated fines), was landfilled and 46 %-wt. was processed for material recovery.
- A country-specific inventory at European level of untreated (surplus) and unrecovered (fine fraction) BA was the first phase of this research. Considering untreated BA, the countries exhibiting relevant surplus in BA production exceeding local treatment capacity were Finland and Luxembourg (+31 %), Sweden (+46 %) and Spain (+72 %). Considering unrecovered BA, its quantity was related to the amount of treated BA (largest contribution was associated with Germany, France, Great Britain, and Italy), despite the performance level of BA treatment.
- The estimated loss of potential secondary raw materials (2.14 Mt in total) comprised 1 Mt mineral fraction (0.9 Mt glass cullet), 0.97 Mt ferrous metals and 0.18 Mt non-ferrous metals. The loss, compared to available amounts of each material in the specific fractions, was related both to untreated BA (42 % mineral fraction and 45 % ferrous metals) and to unrecovered BA (84 % aluminum and 60 % copper). Worst results were observed in Spain (45 %-wt loss of mineral fraction) and France (18 %-wt. loss of ferrous and non-ferrous metals). The results of MFA showed clearly how higher BA production did not necessarily imply larger material losses, since the main driver was the technological performance level that defined the smallest recoverable particle size.

RQ2. Assess the environmental consequences of the potential complete valorization of BA, accounting energy consumption and savings and GHG emissions.

- Country-specific energy balances and (net) GHG emissions were calculated comparing complete BA valorization with the extraction and processing of natural resources to produce aggregates, glass, iron, aluminum, and copper. The energy balance resulted in energy savings due to the recovery of secondary raw materials from BA.
- The evaluation of GHGs emissions showed that the recovery of secondary raw materials from BA has a much lower environmental impact than mining and processing of natural resources, with iron implying the highest absolute emission savings and copper the highest specific emission saving.

RQ3. Assess the economic profitability of the potential complete valorization of BA through state-of-the-art technologies.

- While CAPEX was subject to the amount of untreated and unrecovered BA (without any specific dependence to any of the two quotas), country specific OPEX values were mainly driven by landfill fees regarding the disposal of fine mineral fraction. Incomes were mainly due to copper and iron recycling and savings to the avoided landfilling of valuable materials.
- Economic profitability was achieved by 66 % European countries (Austria, Czech Republic, Finland, France, Germany, Hungary, Italy, Lithuania, Luxembourg, Netherland, Poland, Portugal, Romania, Slovakia, Spain, Sweden, Norway, Switzerland, and Great Britain) with BA mass flow exceeding 0.02 Mt per year, and average values of economic indicators were: NPV 83 M€, ROI 20 % and payback time 11 years.

This work confirmed the strategic significance of optimizing material recovery from MSWI BA and demonstrated that BA could play a key role in fulfilling European policies based on Circular Economy. However, country-specific parameters exhibited great influence on the outcomes of the economic analysis, due to the lack of common legislation across Europe on whether reuse of material recovered from BA is permitted and to the considerable standard deviation existing among the local landfill fees.

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