

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

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

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15

16 **Abstract**

17 In 2018 municipal solid waste (MSW) incineration in Europe produced nearly 19 Mt of bottom  
18 ash (BA); only 46 %-wt. was treated, often in poorly performing plants, leaving behind 10 Mt  
19 of untreated and unrecovered BA, destined to landfill. This work was based on the inventory of  
20 BA across Europe, and on the hypothesis to achieve complete BA valorisation through two  
21 assumptions: treating 100 % BA and minimizing the loss of valuable fractions due to technical  
22 limitations of state-of-the-art processes in comparison to advanced innovative processes. The  
23 research involved three phases: characterization of potential secondary raw materials (metals  
24 and mineral fraction) currently lost from untreated (the surplus compared to treatment capacity)  
25 and unrecovered BA (the fine fraction) through material flow analysis; environmental

26 assessment (energy balance and net GHG emissions) of complete BA valorisation; investigation  
 27 of the economic feasibility of complete BA valorisation through state-of-the-art technologies.  
 28 The resulting 2.14 Mt loss of valuable materials included 1 Mt mineral fraction and 0.97 Mt  
 29 ferrous metals, mostly from untreated BA, and 0.18 Mt non-ferrous metals, mostly from  
 30 unrecovered BA. The energy balance and GHGs emissions required by the treatment of the  
 31 currently untreated and unrecovered fractions of BA resulted in energy and GHGs emissions  
 32 savings. Economic profitability was driven by iron and copper recycling and avoided landfill  
 33 fees. Profitability was achieved by two thirds of considered countries (average values: NPV 83  
 34 M€, ROI 20 %, payback time 11 years) with BA mass flow exceeding 0.02 Mt.

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

### 37 List of abbreviations

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

<b><math>I_{E,unrecovered}</math></b>	GHGs emissions Index for unrecovered bottom ash
<b><math>I_{GHGs,untreated}</math></b>	Energy consumption Index for untreated bottom ash
<b><math>I_{GHGs,unrecovered}</math></b>	Energy consumption Index for unrecovered bottom ash
<b>Capex</b>	CAPital EXPenses
<b>A</b>	Amortization
<b><math>C_0</math></b>	Initial capital
<b>i</b>	Interest
<b>n</b>	Numbers of years
<b>OPEX</b>	OPERational EXPenses

## 38 1. Introduction

39 The generation of municipal solid waste (MSW) in Europe in 2018 exceeded 300 Mt (in average  
40 489 kg per capita) (Eurostat, 2020), with different contributions: 219.69 Mt from EU-27  
41 member states, 38.42 Mt from EU candidates (Turkey, Montenegro, Macedonia, Serbia,  
42 Albania), 12.71 Mt from the European free trade association members (EFTA, Liechtenstein,  
43 Iceland, Norway, Switzerland) and 30.79 Mt from the former EU member Great Britain. Data  
44 about MSW production in Cyprus, Greece, Iceland, and Ireland in 2018 are not available on  
45 Eurostat yet, thus 2017 values were accounted. It is well known that demographic and socio-  
46 economic development strongly influence MSW production and management among the  
47 member states (Giannakitsidou et al., 2020). The combination of recycling and thermal  
48 recovery was proposed as best option for MSW management from a life cycle analysis (LCA)  
49 perspective (Cherubini et al., 2009), also together with the reduction of MSW production rate  
50 and limitation of greenhouse gas (GHG) emissions (Behzad et al., 2020). The key role for the  
51 European context of coupling MSW enhanced recycling practices with thermal treatments  
52 according to Circular Economy principles was already analysed (Abis et al., 2020). Considering  
53 the classification of MSW management operations defined by the Waste Framework Directive  
54 (WFD) 2008/98/EC, incineration (D10) and thermal valorisation (R1) accounted for over 75  
55 Mt of MSW in Europe in 2018 (Eurostat, 2020), leading to the supply of electricity and heat to  
56 respectively 18 M and 15.2 M end-users from waste-to-energy (WtE) plants, and to 90 % waste  
57 volume reduction (CEWEP, 2017a). The physical outcome of D10 and R1 are bottom ash (BA,

58 accounting for about 25 %-wt of municipal solid waste incinerated, MSWI) (Enzner et al.,  
59 2017), and fly ash (FA, accounting for about 3 %-wt of MSWI) (Morf et al., 2002). Residues  
60 from 75 Mt of incinerated MSW in Europe during 2018 (58 Mt in EU-27, 12 Mt in GB, less  
61 than 5 Mt in EFTA) (Eurostat, 2020) are 18.75 Mt of BA and 2.25 Mt of FA. BA treatment is  
62 common in EU, though processes are specifically designed to recover metals (iron, aluminium,  
63 copper, zinc) (Astrup et al., 2016; Šyc et al., 2020), which are the most valuable components  
64 (Bunge, 2018). However, BA not only encompass recyclable metals; the inert fraction, mostly  
65 consisting of the oxides of silicon (Si), calcium (Ca), aluminium (Al) and iron (Fe) (Astrup  
66 et al., 2016), whether not directly sent to landfill has ready-to-market options as sub-base road  
67 filling material, replacing mineral aggregates (Minane et al., 2017; Tang et al., 2015) and also  
68 perspectives in ceramic manufacturing (Rincon Romero et al., 2018) and as sorbent material  
69 (Fontseré Obis et al., 2017). Worth to be mentioned is the potential recovery for glass cullet  
70 e.g. as abrasive medium (lowest open loop recycling possibility) (Silva et al., 2017). However,  
71 comparing the above-mentioned estimate of BA produced in Europe calculated from Eurostat  
72 (18.75 Mt in 2018) with the 8.4 Mt/y BA treatment capacity reported by the new Best Available  
73 Techniques Reference Document (BREF) on Waste Incineration (Neuwahl et al., 2019), it  
74 becomes clear that less than 50 % of the BA produced in Europe undergo any treatment. A  
75 common EC legislation on BA management does not exist at the moment, thus restrictions for  
76 material recovery, if existing, are currently set by each country (Blasenbauer et al., 2020).  
77 Alongside profits from metals recovery, one of the main drivers towards the optimization of  
78 BA treatment is the necessity to comply with WFD targets and to reduce management costs due  
79 to landfill tax (Blasenbauer et al., 2020; Bourtsalas, 2012).  
80 Therefore, the actual framework appears highly complex, considering on one side MSW  
81 management practices across EU-27 (in 2018: 49 %-wt recycling, 27 %-wt incineration and  
82 WtE and 24 %-wt landfilling) (Eurostat, 2020), and on the other side the further efforts urgently

83 required to member states to fulfil the ambitious Circular Economy targets defined by the EC  
84 for the next decade. Improving BA management could be, without any doubts, a key issue.  
85 Complete and detailed characterisation of BA and of their management was already performed  
86 referring to specific countries, as Belgium (Joseph et al., 2018), Denmark (Allegrini et al.,  
87 2014), Germany (Enzner et al., 2017), Italy (Funari et al., 2016), The Netherlands (Loginova et  
88 al., 2019), Spain (Del Valle-Zermeño et al., 2017) and for EU, Asian and other countries in a  
89 review article (Dou et al., 2017). Most applied utilisation pathways are landfill construction,  
90 road construction, concrete aggregate, and cement clinker. Long-term experience exists for  
91 application of BA in road construction (Di Gianfilippo et al., 2018; Hysk et al. 2019). In the  
92 production of cement clinker BA replaces natural, mined material but still requires firing the  
93 rotary kiln (Clavier et al., 2020). Compared to previous studies, this work focused on the  
94 quantification and characterisation of BA across all Europe (e.g., instead of in specific  
95 countries), comparing countries with different attitudes toward MSWI and BA management.  
96 Moreover, to our knowledge, two fundamental aspects were not yet analysed from the technical,  
97 environmental, and economic viewpoints, considering state-of-the-art technologies and the  
98 whole European context: 1. enhancing the amount of treated BA aiming at reaching 100 %  
99 production, and 2. minimizing the losses of potential secondary raw materials from treated BA  
100 due to technical limitations of state-of-the-art processes in comparison to advanced innovative  
101 processes. Considering the first issue, BA treatment allows in average the recovery of 6.3 %-  
102 wt ferrous metals and 1.7 %-wt non-ferrous metals (CEWEP, 2017), therefore 0.8 Mt metals  
103 lost were estimated in 2017 from untreated BA (Abis et al., 2020). Considering the second  
104 issue, BA fine fractions (dimensions below 2-5 mm, accounting for up to 40-50 %-wt) (Enzner  
105 et al., 2017) are usually unrecovered and landfilled to avoid any PTEs release, implying the loss  
106 of valuable residual materials (metals and mineral fraction). Therefore, this work aims to  
107 answer the following research questions (RQ): RQ1. Quantify and qualify through material

108 flow analysis (MFA) the potential secondary raw materials actually lost from BA, considering  
109 both the untreated and the unrecovered fractions (respectively the surplus compared to  
110 treatment capacity and the fine fraction); RQ2. Assess the environmental consequences of the  
111 potential complete valorisation of BA, accounting energy consumption and savings and net  
112 GHG emissions; RQ3. Assess the economic profitability of the potential complete valorisation  
113 of BA through state-of-the-art technologies (e.g., [the technologies implemented in current full-](#)  
114 [scale plants treating BA](#)). The economic analysis included capital and operational costs, market  
115 value of recovered materials, net present value, return of investment and payback time.  
116 [Research question 1 derives from the hypothesis of treating 100% of produced BA. Research](#)  
117 [questions 2 and 3 derive from the need to evaluate not only the technical feasibility of the](#)  
118 [proposed solution, but also its environmental consequences and economic feasibility.](#) The  
119 analyses presented in this work refer to 2018 data, the most recent available on Eurostat and in  
120 the scientific literature on MSWI.

## 121 **2. Methodology**

### 122 *2.1. Quantification of the actual loss of potential secondary raw materials*

123 The quantitative assessment of the actual loss of potential secondary raw materials from  
124 untreated (i.e. surplus compared to treatment capacity) and unrecovered (i.e. fine fraction) BA  
125 was performed according to material flow analysis (MFA) approach through STAN2WEB open  
126 access software (version 2.6.801, <http://www.stan2web.net>) developed by Technische  
127 Universität of Wien according to the Austrian standard ÖNorm S 2096 (Material flow analysis-  
128 application in waste management). The MFA was based on the following assumptions: amounts  
129 of total available BA in specific countries were calculated as 25 %-wt of MSWI in 2018  
130 (Eurostat, 2020), then compared with current national BA treatment capacity (Blasenbauer et  
131 al., 2020) to obtain the amount of untreated BA; unrecovered BA amounts were calculated  
132 considering the cut-off particle size of recoverable fraction in each country (Enzner et al., 2017),

133 then multiplied to the corresponding cumulative percentage from a characteristic BA particle-  
134 size distribution curve (Šyc et al., 2020) and to the amount of BA treated in the same country.  
135 If data about minimum recoverable particle-size were missing for a certain country, the average  
136 value 4 mm (50 % cumulative percentage on BA granulometric distribution curve) was  
137 considered as technological limit. The result of this evaluation, here-in-after named  
138 “unrecovered fraction”, assumed that BA treatment plants in Europe (Neuwahl et al., 2019)  
139 worked at 100 % capacity (the BREF reports two values for each plant: one referred to the  
140 average capacity of each plant and another to 100 % capacity). The material recovery efficiency  
141 of BA treatment technologies was assumed 100 %, to estimate the overall theoretical amount  
142 of potentially recoverable secondary raw materials. Finally, the hypothesized recovery of  
143 potential secondary raw materials involved mineral aggregates or glass recycling (mineral  
144 fraction) and secondary smelters (metal fractions), because the technical feasibility of these  
145 perspectives was already proven (Buekens, 2013; Bunge, 2018; Clavier et al., 2020; Lam et al.,  
146 2010; Neuwahl et al., 2019; Verbinnen et al., 2017).

147

## 148 *2.2. Characterization of untreated and unrecovered BA*

149 BA quality was described in terms of macro-components (Neuwahl et al., 2019; CEWEP,  
150 2017a) as follows: 85-90 %-wt mineral fraction, 5-10 %-wt ferrous metals and 2-5 %-wt non-  
151 ferrous metals. Several studies (Allegrini et al., 2014; Del Valle-Zermeño et al., 2017; Astrup  
152 et al., 2016) highlighted the presence of glass cullet in BA, whose recycling could increase the  
153 market value of the mineral fraction. The amount of glass cullet in BA mineral fraction was  
154 estimated 11.9 %-wt in 0-2 mm quota (Del Valle-Zermeño et al., 2014) and 8.6 %-wt in above  
155 2 mm fraction (BASH TREAT, 2020)- Ferrous metals were all assumed steel scrap; the  
156 amounts of non-ferrous metals were estimated 68 %-wt aluminium and 28 % copper in

157 untreated BA (CEWEP, 2017), and 45 %-wt aluminium and 50 %-wt. copper in unrecovered  
158 BA, referring to the amounts detected in fines below 5 mm (Neuwahl et al., 2019).

159

### 160 *2.3. Environmental assessment*

161 The environmental assessment of BA valorization was based on two viewpoints (energy  
162 balance and GHG emissions), in comparison with extraction and manufacture of construction  
163 aggregates, glass and metals from raw materials. In VDI guideline 3925 (VDI, 2016) it was  
164 shown that these two viewpoints have highest relevance to the environmental performance of  
165 BA treatment whereas other impact categories used for example in life cycle assessment (LCA)  
166 such as acidification potential, human toxicity potential or else are negligible (Gehrmann et al.,  
167 2017).

168

#### 169 *2.3.1. Energy demand and savings*

170 Specific energy demand of BA treatment (kWh/t) was calculated multiplying the energy  
171 required by treatment plants (kWh) published on the new BREF on waste incineration  
172 (Neuwahl et al., 2019) to the amounts of BA treated (t) in single countries in 2018 according to  
173 the same reference document (see Supplementary Material, Table I). Each treatment plant was  
174 fed by different energy sources, categorized as electricity, natural gas, steam (all expressed in  
175 MWh) and liquid fossil fuel, reported in liters and converted to MWh (1 L = 9.1 kWh). Energy  
176 consumption values of single plants were referred to the corresponding amount of BA treated,  
177 obtaining a weighted average value of 8.28 kWh/t, which was comparable with the value (10  
178 kWh/t) obtained by previous studies (Bunge, 2018). The net energy potentially saved, i.e., the  
179 difference between the energy necessary for the primary production of materials from natural  
180 resources and the energy necessary for materials manufacturing from secondary production,  
181 was derived from literature (Appendix, Tables IIa-IIId). In details, we considered a saving of

182 energy demand between primary production and recycling equal to 4.11 kWh/t for aggregates  
 183 (Marinković et al., 2010), 527.8 kWh/t for glass cullet (Larsen et al., 2009), 2166.7 kWh/t for  
 184 Fe, 51216.7 kwh/t for Al and 4138.9 kWh/t for Cu (Grimes et al., 2008; Norgate and Haque,  
 185 2010; Norgate et al., 2007). We assumed that BA treatment results in recycled aggregates ready  
 186 to use, thereby the potential energy and GHGs emissions savings refer to the energy saved from  
 187 primary aggregates' production, without considering any further treatments. The literature  
 188 values employed for the calculation of the net energy potentially saved were published in 2005-  
 189 2010; we based our analysis, referred to 2018, to the mentioned references in absence of more  
 190 recent ones. Net energy consumption values were obtained, both for untreated BA and for  
 191 unrecovered BA, by difference between energy consumption of BA treatment and potential  
 192 energy savings because of avoided raw materials production (i.e., aggregates, glass, metals).  
 193 We considered positive an energy balance in which the energy demand necessary to process  
 194 untreated and unrecovered BA was lower than the energy savings related to the avoided  
 195 production of corresponding raw materials. To compare different countries, net energy  
 196 consumption was referred to the specific amounts of untreated BA, through a specific index of  
 197 energy consumed  $I_{E \text{ untreated}}$  (eq. 1), and to the specific amounts of unrecovered BA, through a  
 198 specific index of energy consumed  $I_{E \text{ unrecovered}}$  (eq. 2).

$$199 \quad I_{E \text{ untreated}} = \frac{\text{energy consumed} - \text{energy saved}}{\text{untreated material}} \left[ \frac{kWh}{t} \right]$$

$$200 \quad I_{E \text{ unrecovered}} = \frac{\text{energy consumed} - \text{energy saved}}{\text{unrecovered material}} \left[ \frac{kWh}{t} \right]$$

### 201 2.3.2. GHG emissions

202 GHG emissions were calculated as: produced emissions over 100 years related to BA treatment  
 203 (values were between 0.007 and 1.13 kg CO<sub>2</sub> eq/kWh, the analysis considered specific values  
 204 for each country) (Fruergaard et al., 2009). A country-specific analysis was performed

205 considering the specific GHG emission factors of energy production for non-household  
 206 consumers (EEA, 2020) (Appendix, Table III). Since emission factors of Albania, Montenegro,  
 207 Serbia, Turkey, Liechtenstein, Iceland, Norway and Switzerland were missing, an average  
 208 emission factor 0.569 kg CO<sub>2</sub> eq/kWh (Fruergaard et al., 2009) was accounted.

209 On the other hand, avoided GHG emissions related to raw materials production has been  
 210 considered as the difference between the emissions due to primary and secondary production,  
 211 as following:  $-1.50 \times 10^{-3}$  kg CO<sub>2</sub> eq/kg for aggregates, 0.50 kg CO<sub>2</sub> eq/kg for glass, 1.06 kg CO<sub>2</sub>  
 212 eq/kg for Fe, 12.72 kg CO<sub>2</sub> eq/kg for Al and 0.97 kg CO<sub>2</sub> eq/kg for for Cu (Appendix, Tables IVa-  
 213 IVd). As the mineral components of the fine fraction were considered inert, no GHG emissions  
 214 related to their landfill disposal were accounted.

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

$$218 \quad I_{GHG.untreated} = \frac{\text{GHG emissions}-\text{GHG saving}}{\text{untreated material}} \left[ \frac{t \text{ CO}_2}{t} \right] \quad (3)$$

$$219 \quad I_{GHG.unrecovered} = \frac{\text{GHG emissions}-\text{GHG saving}}{\text{unrecovered material}} \left[ \frac{t \text{ CO}_2}{t} \right] \quad (4)$$

220

#### 221 2.4. Economic assessment

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

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

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

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

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

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

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

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

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

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

235 The national prices for non-household electric energy (€/kWh) derived from Eurostat  
236 (Appendix, Table V); for the countries not included in the database (Estonia, Hungary, Latvia,  
237 Lithuania, Luxembourg, Malta, Slovenia, Albania, Montenegro, Serbia; Liechtenstein, Iceland  
238 and Switzerland) the average value 0.117 €/kWh was accounted. Among the operational costs,  
239 landfill expenses for the disposal of the mineral part of the fine fraction (considered, according  
240 to literature, too contaminated to be recovered) were accounted (Appendix, Table VI) (CEWEP,  
241 2017). Landfill costs for most countries were defined by European Environment Agency  
242 (European Environmental Agency, 2014), while for Switzerland landfill tax was 50 €/t  
243 (CEWEP, 2017).

244 Potential incomes from secondary raw materials sale were estimated assigning a specific market  
245 value to each fraction. In detail, BA mineral fraction was compared to construction aggregates

246 (average value 9 USD/t per metric ton, USGS, 2020), accounted as 8.2 €/t. The market value  
247 assigned to recycled glass was 20 €/t (Rincon Romero et al., 2018). Commercial values of 100  
248 €/t, 500 €/t and 3600 €/t were assigned to iron scrap and non-ferrous metals (aluminium and  
249 copper) respectively (Bunge, 2018). Increased BA recovery also implied savings related to  
250 reduction of landfilling and primary raw materials extraction. Profitability of untreated and  
251 unrecovered BA valorisation was assessed through net present value (NPV), return of  
252 investment (ROI) and payback time (Appendix, Table VII).

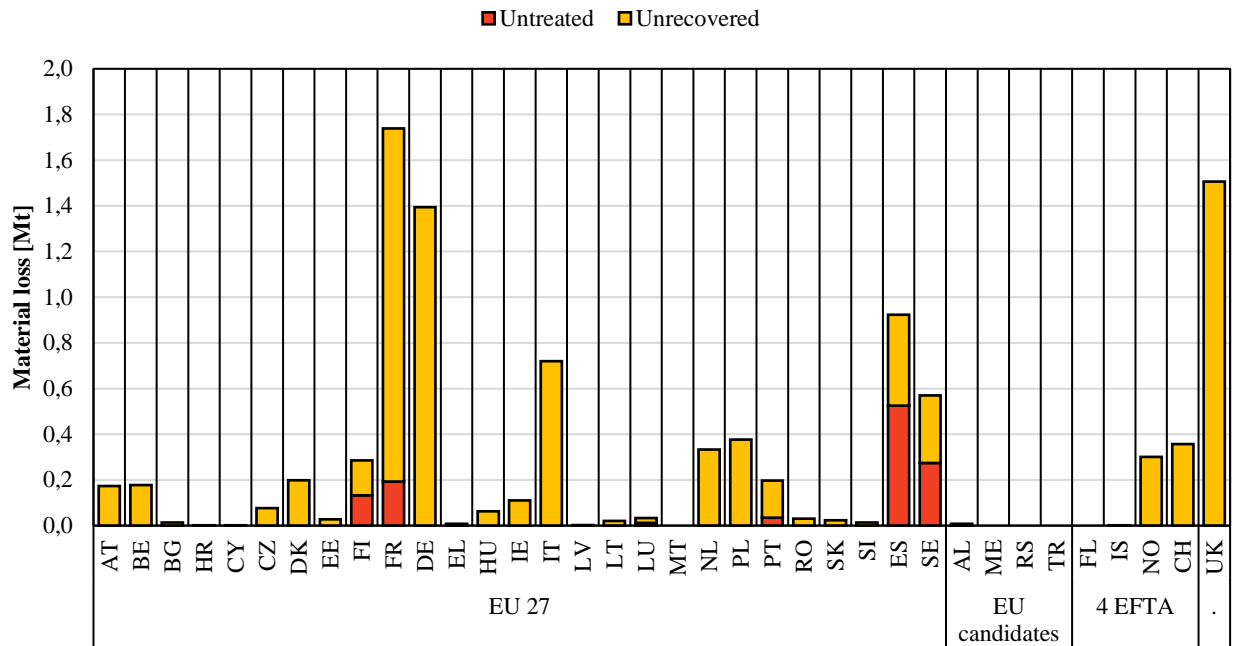
253

### 254 **3. Results and discussion**

#### 255 *3.1. Quantification of the actual loss of potential secondary raw materials*

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

266



267

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

271 No correlation appeared between the amount of produced BA and the untreated surplus  
 272 exceeding national treatment plant capacity ( $R^2 = 0.0307$ ), nor between BA production and  
 273 installed treatment capacity ( $R^2 = 0.5216$ ) (Appendix, Figure II). As for unrecovered BA, whose  
 274 under-exploiment represented the main loss in terms of secondary raw materials, its amount  
 275 seemed to be related to the amount of produced BA ( $R^2 = 0.9605$ ) (Appendix, Figure III). This  
 276 means that the largest contribution to unrecovered BA was associated to the top four producers  
 277 (Germany, France, Great Britain and Italy), despite them being among the best performing  
 278 countries in terms of BA treatment, being able to recover BA with particle size down to 2 mm  
 279 (4 mm in Italy) (Enzner et al., 2017). Nevertheless, although the technological levels reached  
 280 by each country showed lesser influence, minor BA producers, as Spain and Portugal, were  
 281 responsible for the production of considerable amounts of unrecovered BA, due to their inability  
 282 to recover fractions respectively below 5 and 10 mm grain size (Enzner et al., 2017).

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

288

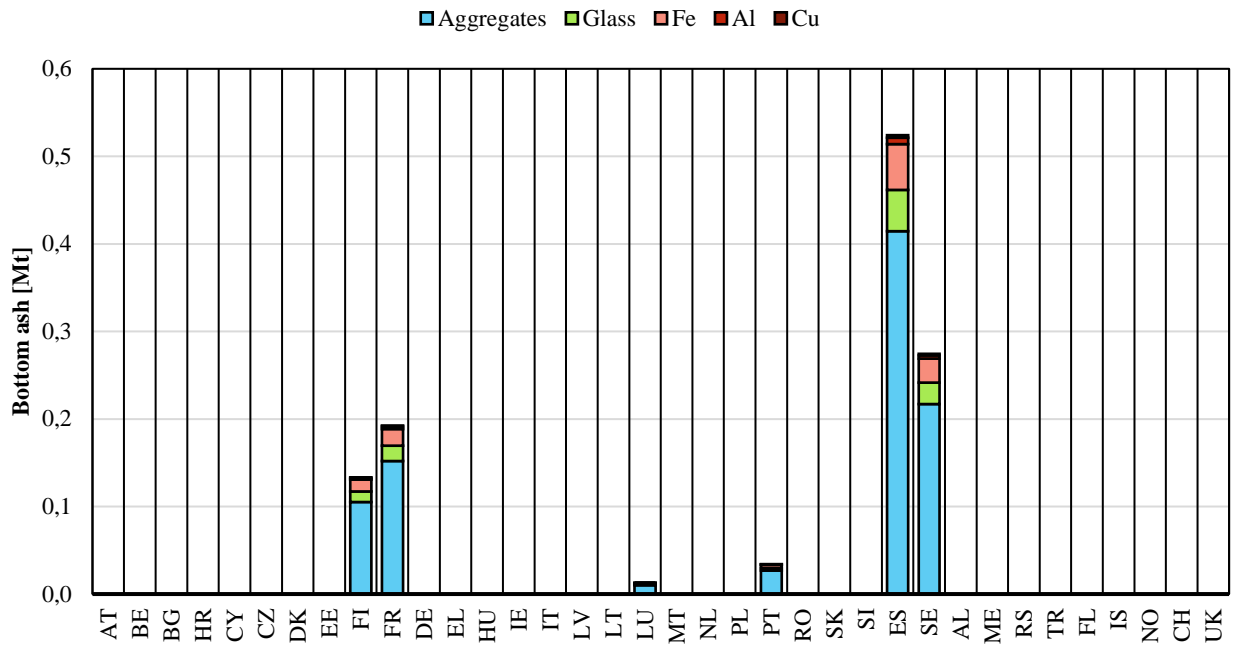
### 289 *3.2. Characterization of untreated and unrecovered bottom ash*

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

308 considering only metals (iron, aluminium and copper) (Figure 2B), and presumed landfilling of  
309 the remaining mineral fraction.

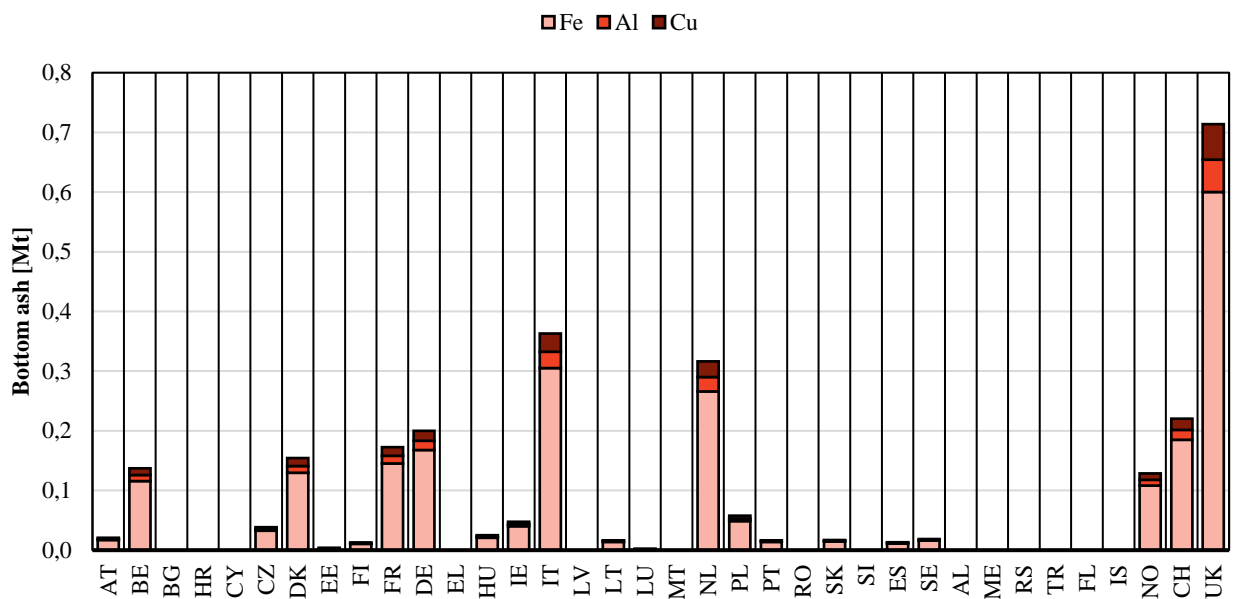
310 The overall 2.14 Mt material loss, resulting from untreated and unrecovered BA (without the  
311 mineral fine fraction, destined to landfill) (Figure 2A and B) consisted of about 1 Mt mineral  
312 fraction (of which 0.9 Mt glass cullet), 0.97 Mt ferrous metals (steel scrap) and 0.18 Mt non-  
313 ferrous metals. The main losses were related to unrecovered mineral fraction (42 %-wt of  
314 potentially available amount) and steel scrap (45 %-wt of potentially available amount). Worst  
315 results were observed for Spain (45 %-wt mineral fraction lost) and for France (18 %-wt ferrous  
316 and non-ferrous metals lost). Spain was the only country where, because of the huge gap  
317 existing between BA production and treatment capacity and of the different composition of  
318 larger and fine BA fractions (see section 2.2), the amount of copper lost within untreated BA  
319 (0.0051 Mt) was higher than the amount in unrecovered BA (0.0039 Mt) (Figure 2B). Iron  
320 recovery efficiencies reached in standard-level BA treatment facilities were generally medium  
321 to high (Bunge, 2018), however, being iron the main metal component in BA (Astrup et al.,  
322 2016), the lack of treatment plant capacity caused material loss up to 75 % of available amount  
323 because of landfilling of untreated and unrecovered BA (Figure 3A). Besides, despite  
324 aluminium is separated from MSW through separate collection, still a considerable amount is  
325 found in BA (see section 2.2). Only 0.04 Mt out of total 0.25 Mt present in BA (16 %) was  
326 recycled in 2018 (Figure 3B). Similarly, only 40 % (0.02 Mt out of 0.05 Mt) of copper present  
327 in BA was recycled (Figure 3C). In this last case the major loss was due to unrecovered BA  
328 fine fraction, where copper concentrates, and it could be prevented by upgrading the existing  
329 BA treatment infrastructures.

330



A

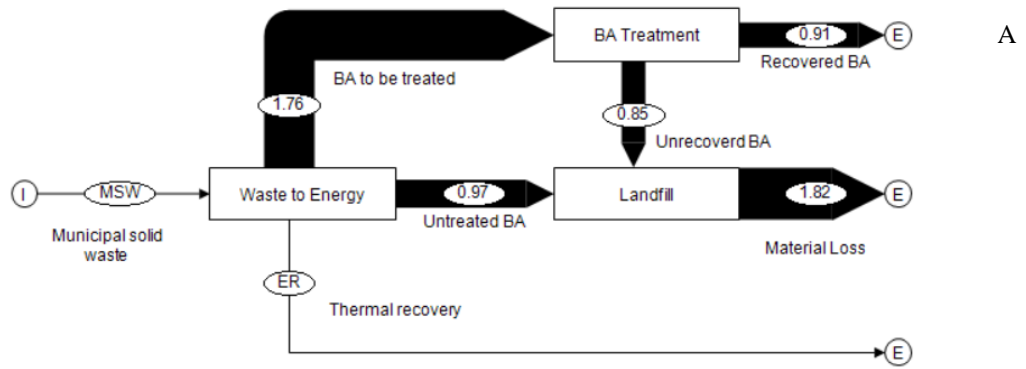
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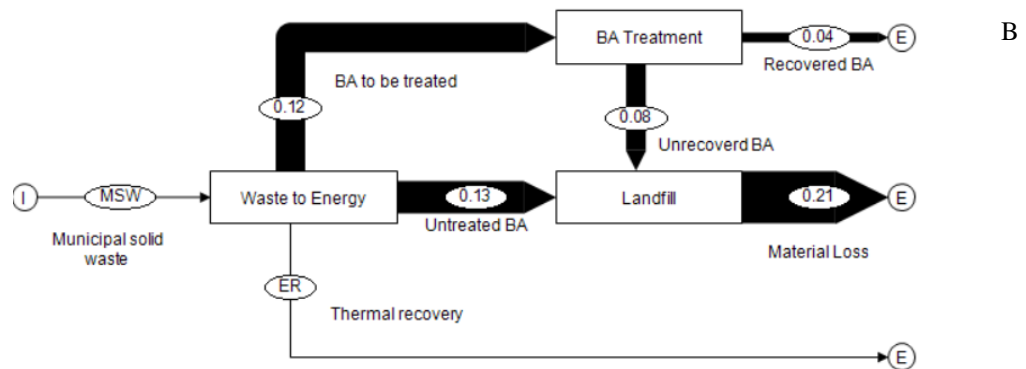
B

332

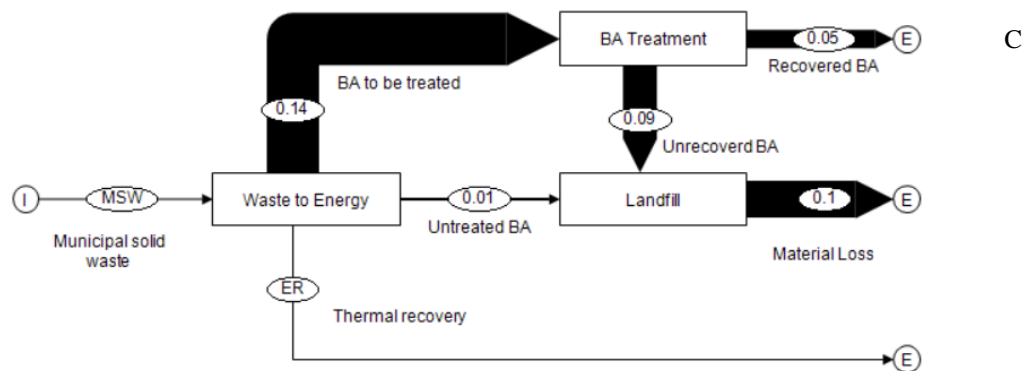
333 **Figure 2.** Characterization of: A) untreated bottom ash (in 6 countries, where BA production  
 334 exceeded treatment capacity) (blue: mineral fraction, green: glass, red: iron, orange: aluminium,  
 335 brown: copper) and B) unrecovered bottom ash in Europe in 2018 (pink: iron, orange:  
 336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



337



338



339

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

342

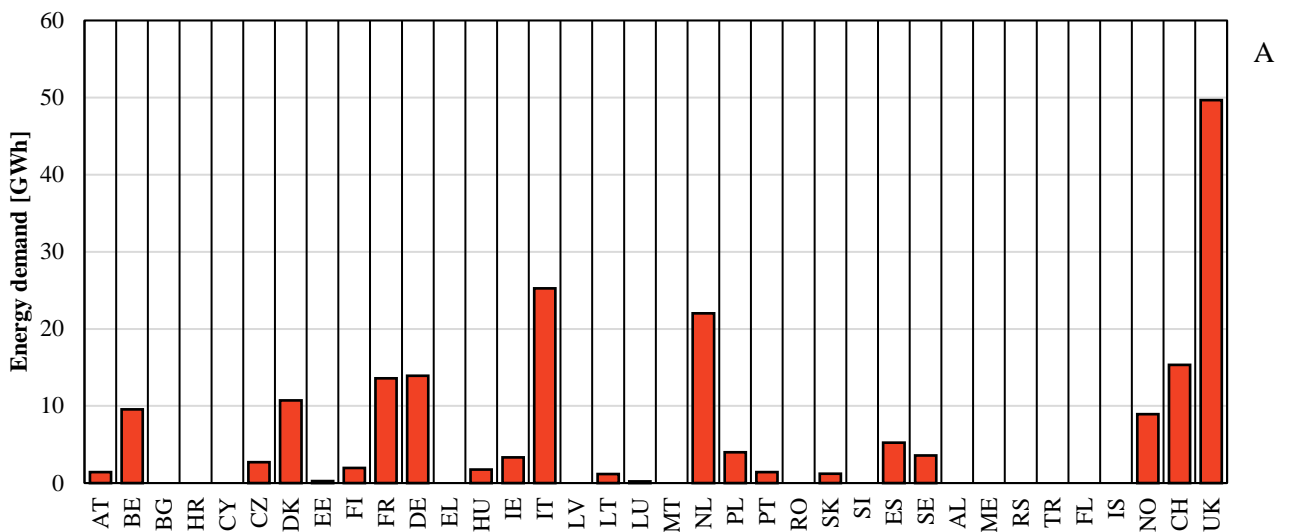
343 Some general statements about the significance in the European context of the above-mentioned  
 344 losses of potential secondary raw materials could be formulated. Although being one of the  
 345 most common metals in earth crust, iron mining in Europe barely accounts 12 %-wt. global  
 346 production, despite the presence of important steel manufacturing industries in Germany, Italy

347 and France (European Commission, 2017). Copper concentration in BA fine fractions is  
 348 noteworthy and although it is not currently listed as critical raw material, the only European  
 349 country in which copper is mined is Poland, accounting only for 2.6 %-wt. global production.  
 350 Therefore Europe relies almost completely on copper imported from South America (27.6 %  
 351 Peru, 22.1 % Chile, 9.5 % Brazil and 9.1 % Argentina) and Indonesia (10.9 %), and copper  
 352 recycling from end-of-life products is highly encouraged (European Commission, 2017).

353 *3.3. Results of environmental analysis*

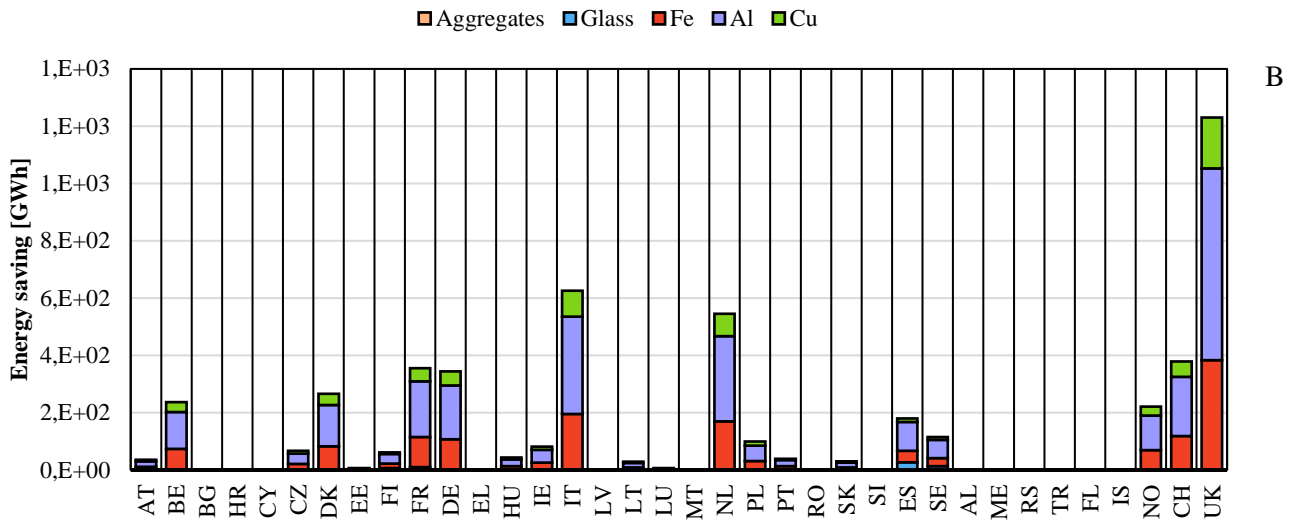
354 *3.3.1. Energy balance*

355 The energy demand values estimated for the complete treatment of BA, considered as sum of  
 356 the currently untreated and unrecovered fractions, in all European countries are shown in Figure  
 357 4A. Whereas, energy savings were estimated calculating the energy required for the extraction  
 358 and processing of natural resources to produce aggregates, glass, iron, aluminium, and copper.  
 359 The fine fraction, which was part of the untreated BA and thus contributed to the energy  
 360 demand, was excluded from energy savings from unrecovered BA because destined to landfill,  
 361 and due to this issue, it was not possible to obtain a real estimate of the energy balance of the  
 362 complete BA valorisation scenario.

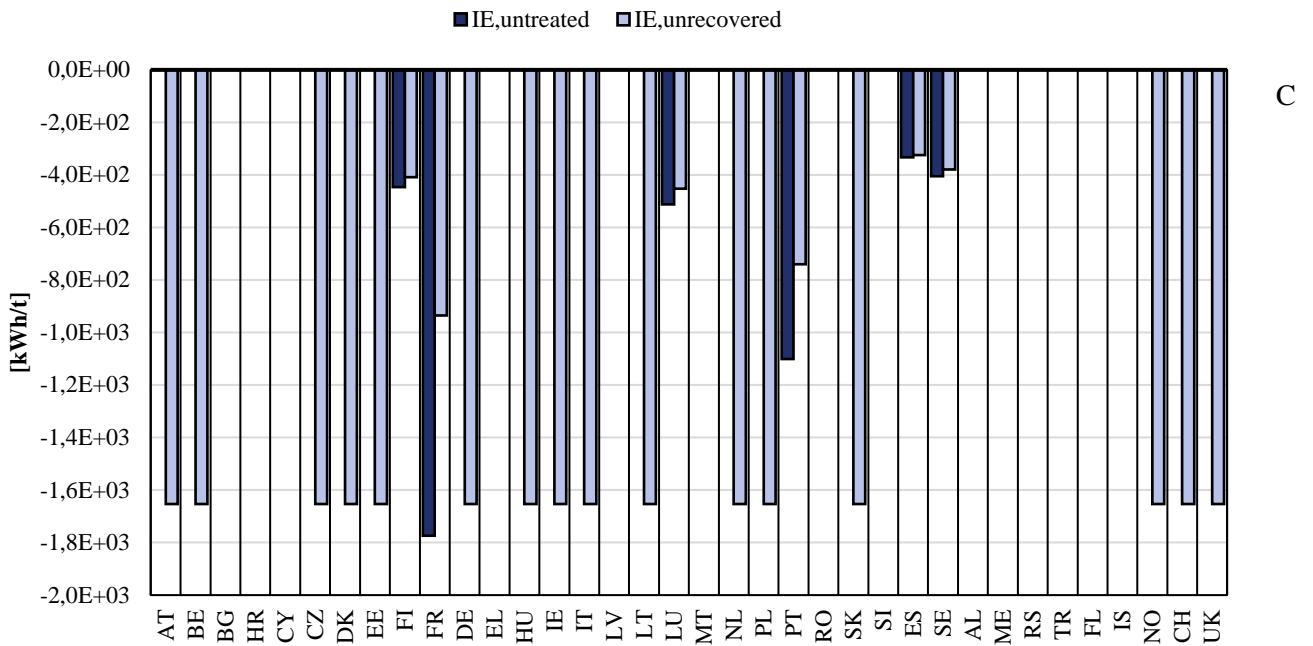


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366

367 **Figure 4.** A) Energy consumption for bottom ash management in Europe in 2018 (GWh); B)

368 Energy savings for materials recovery from bottom ash treatment (pink: aggregates, blue: glass,

369 red: iron, violet: aluminium, green: copper); C) Specific net energy consumption per ton of

370 untreated and unrecovered bottom ash in 2018 (kWh/t) (dark blue: IE untreated, light blue: IE

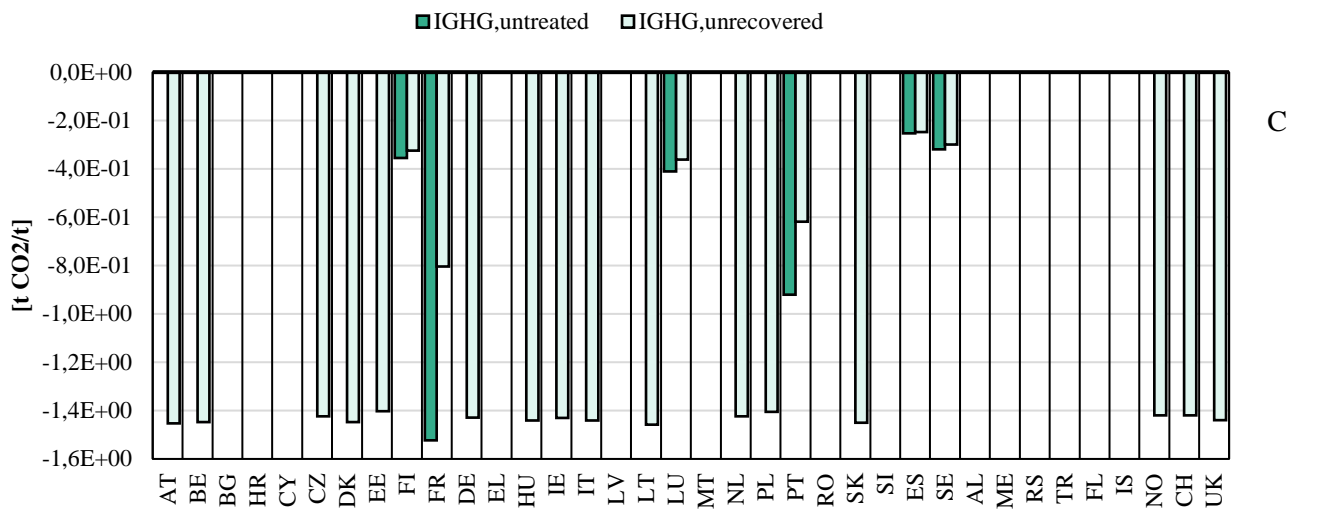
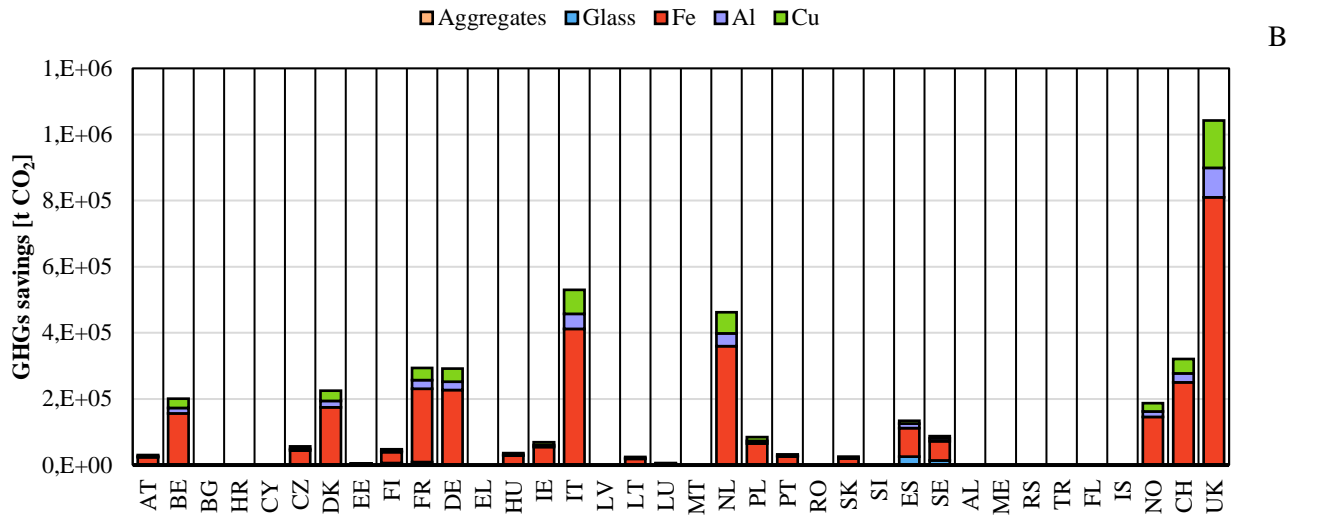
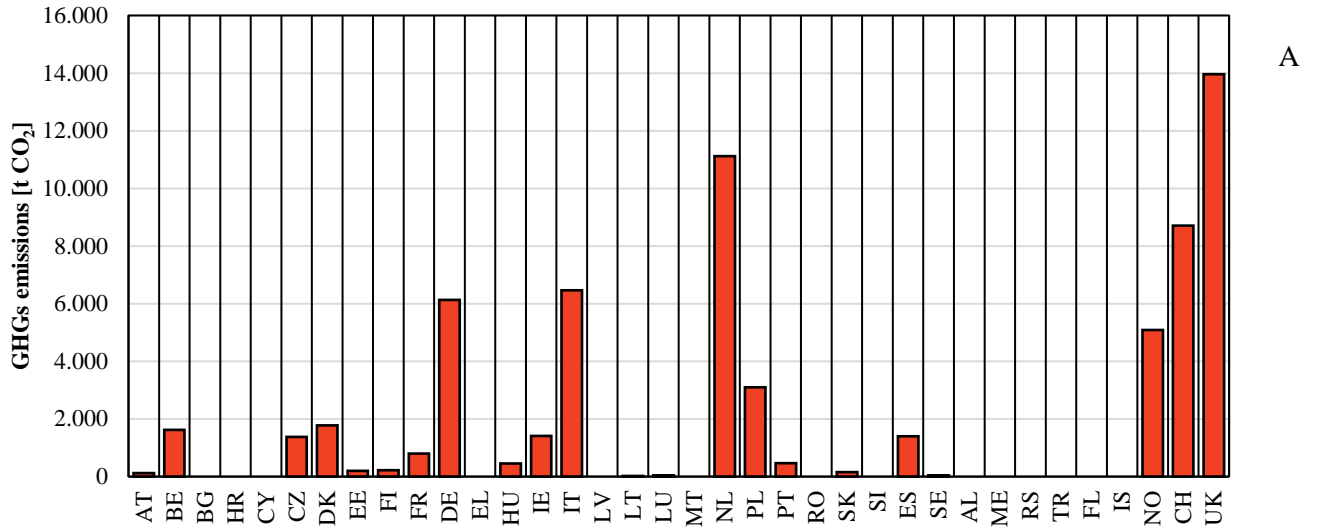
371 unrecovered)

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

384

### 385 *3.3.2. GHG emissions*

386 GHG emissions were evaluated comparing the avoided emissions related to materials recovery  
387 (compared to production from natural resources, in kg CO<sub>2</sub> eq/t) (see section 2.3.2), with the  
388 emissions produced by the treatment of untreated and unrecovered BA, in t/year (Figure 2).

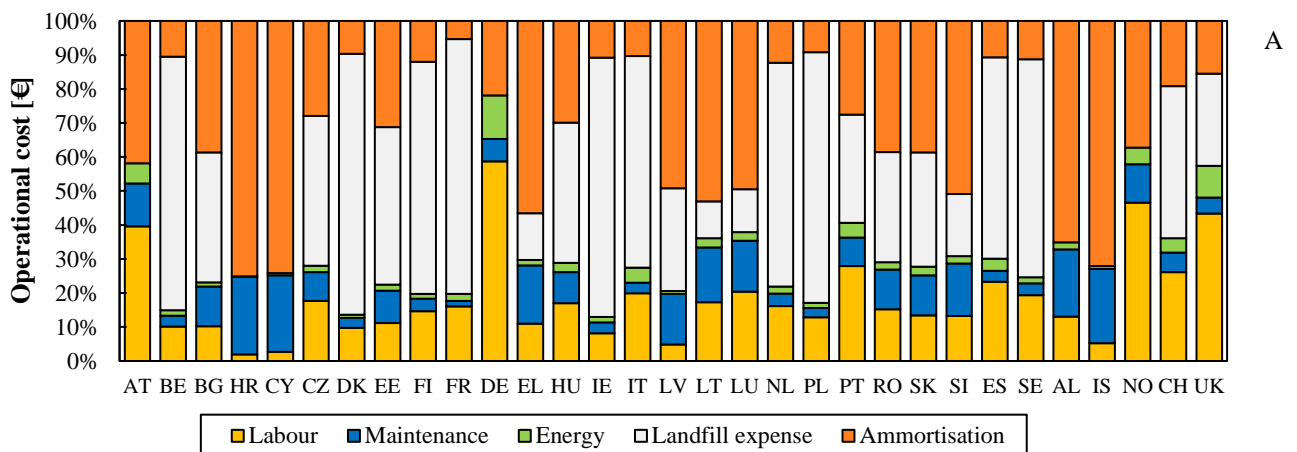


393 **Figure 5.** Bottom ash recycling in Europe in 2018: A) GHG emissions for bottom ash  
 394 management in Europe in 2018 (t CO<sub>2</sub> eq); B) GHG emissions avoided detailed for the different  
 395 materials (t CO<sub>2</sub> eq) (pink: aggregates, blue: glass; red: iron; violet: aluminium; green: copper);  
 396 C) Specific GHG emissions related to untreated and unrecovered bottom ash in Europe in 2018  
 397 (t CO<sub>2</sub> eq/ t) (dark green: IGHG untreated, light green: IGHG unrecovered)

398

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

414

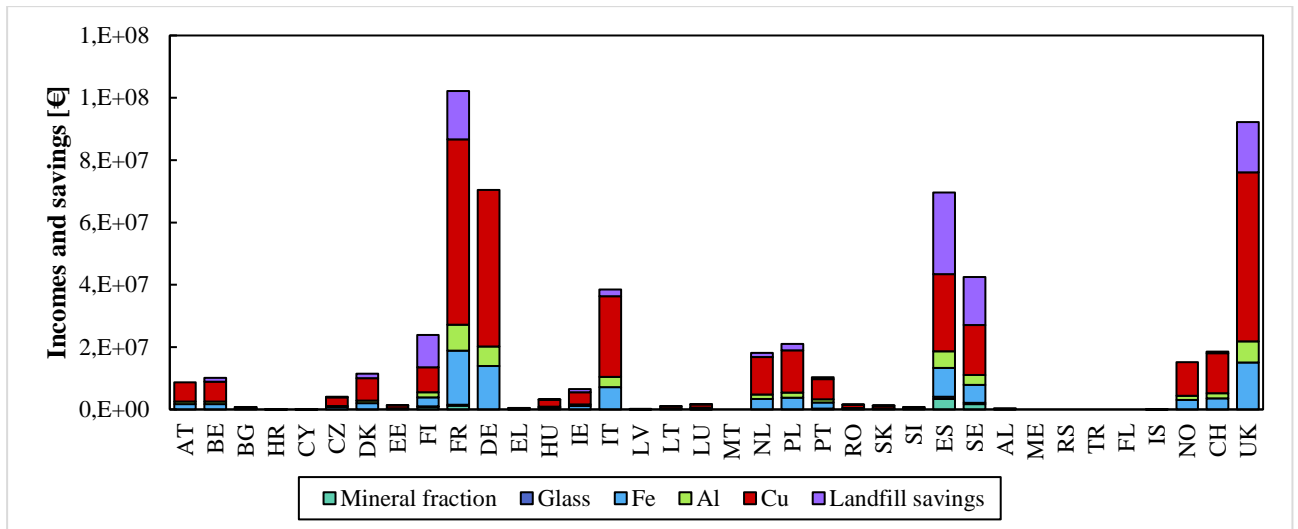


415

416

A

B



417

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

423

### 424 3.4. Results of the economic analysis

425 Total capital costs related to untreated and unrecovered BA (Appendix, Figure VI) were  
 426 obviously dependent from mass throughput, as the highest values were attributed to France,  
 427 Germany, Great Britain, and Spain, which are among the major BA producers. However, the  
 428 amount of the required investment did not depend specifically on whether BA belonged to the  
 429 untreated or unrecovered category. Contrarily, if specific capital costs were considered, the  
 430 countries with the lowest throughput were characterised by the highest values. Assessing the  
 431 operational costs more factors were involved; despite the inverse correlation observed between  
 432 operational cost and treated mass flow, the overall operational cost depended also on country-  
 433 specific parameters as energy cost and landfill tax. Considering country-specific detailed  
 434 operational costs (Figure 6A), the main contribution for most countries was the landfill tax

435 related to the disposal of the fine mineral fraction. However, the framework was not  
436 homogeneous; countries such as Hungary, Albania and Czech Republic imposed low landfill  
437 tax on waste management operators, and in Austria residual waste from WtE plants are  
438 exonerated from landfill fees (CEWEP, 2019). The operational costs due to energy  
439 consumption appeared strongly dependent on the amount of energy required rather than on the  
440 national fee set for non-household energy consumers (Appendix, Table IV). Assessing the  
441 incomes from the sale of recovered materials and the savings from avoided landfilling (Figure  
442 6B), copper recovery was the main economic driver because of its high market value; however,  
443 countries such as Finland, France, Spain, Sweden, and Great Britain, where landfill disposal is  
444 more expensive than the European average, did benefit from the saving of landfill fees. It is  
445 worth noting that iron recovery implied incomes larger than aluminium, notwithstanding its  
446 lower market value. This was due to the fact that, compared to aluminium, iron content in BA  
447 was higher and its recovery requires less effort from the technical and therefore economic  
448 viewpoint. Iron can be recovered easily with magnets whereas for aluminium (with much lower  
449 concentration in BA than Fe) highly efficient eddy current separators (discrete ECS for different  
450 grain size fractions, (Enzner, 2017) usually are necessary. Copper and copper alloys are  
451 separated with the eddy current separators as well. The higher effort is justified because  
452 secondary Al requires much less energy than primary Al (more <90% savings) and, besides  
453 energy savings also for secondary Cu, natural Cu resources are conserved (Simon and Holme,  
454 2016).

455 From the simple comparison of overall country-specific costs and incomes and savings related  
456 to BA valorisation, it appeared that in countries with lower BA mass throughput, costs exceeded  
457 potential incomes and saving. Thereby countries as Hungary, Cyprus, Latvia, and Iceland  
458 would record a negative cash flow. The potentially necessary plant size was not the only  
459 element determining the positive outcome of the investment, as among the countries with

460 negative cash flows are also listed Denmark and Ireland, despite their respective BA potential  
461 of 0.2 Mt and 0.11 Mt, which are one order of magnitude higher than Estonia or Lithuania and  
462 two higher than Greece, where the cash flows was instead positive. The justification of this  
463 apparent contradiction was found pointing out that Denmark and Ireland adopted the highest  
464 landfill fees throughout Europe, thereby the expenses due to the management of BA  
465 unrecoverable fraction did not justify other operational costs. Except for Great Britain, where a  
466 discounted landfill fee for the disposal of processed BA is applied, in all other countries landfill  
467 taxes played a dual role on the economic analysis performed in this study, as they represented  
468 a potential saving generated by the recovery of untreated BA and metal components of  
469 unrecovered BA, but still needed to be listed as costs related to the management of mineral fine  
470 unrecovered fraction. Hence, the economic feasibility of BA valorisation was mainly dictated  
471 by how much other factors, such as cost of energy and valuable metals concentration, can shift  
472 the balance to a positive outcome.

473 Considering profitability, net present value (NPV) of the treatment plant exhibited average  
474 value of 83.36 M€(Appendix, Figure VIIA) and was negative in 25 % European countries. The  
475 worst performances were observed among the countries with lower BA production (Bulgaria,  
476 Hungary, Cyprus, Estonia, Ireland, and Iceland), which returned negative NPV after 20 years.  
477 Whereas the highest NPV values were reported among the countries previously identified as  
478 major European BA producers (Great Britain, Germany, Spain, France, Sweden, Italy, Finland,  
479 Norway, and Switzerland). The average return on investment (ROI) was 20 % (Appendix,  
480 Figure VIIB) and the highest values (> 50 %) were observed for Germany, Spain, Sweden, and  
481 Great Britain. Hungary, Cyprus, Ireland, and Latvia were characterised by negative ROI; thus,  
482 the investment was not profitable. Payback time was evaluated for the countries characterised  
483 by positive NPV and ROI values (Appendix, Figure VIIC), and they all met payback time  
484 before 20 years. The average time required by the investment to break even on income-outcome

485 trade-off was 11 years, however, for most countries (67 % of the ones with payback time below  
486 20 years) payback time was shorter. Estonia, Greece, Slovenia, and Albany, despite a consistent  
487 positive outcome with NPV and ROI were characterised by a payback time higher than the 20  
488 years useful plant lifetime, thereby the economic assessment for these countries was defined  
489 not profitable. The countries with profitable scenarios were the ones with the higher amount of  
490 produced BA plus Hungary, Czech Republic, and Luxembourg, which accounted for relatively  
491 lower amounts of produced BA but were characterized by lower-than-average operational costs,  
492 which did justify the investment in improving BA recovery.

493 The economic analysis resulted positive (NPV and ROI >0 and payback time < 20 years) for  
494 66 % of the analysed European countries: Austria, Czech Republic, Finland, France, Germany,  
495 Hungary, Italy, Lithuania, Luxembourg, Netherland, Poland, Portugal, Romania, Slovakia,  
496 Spain, Sweden, Norway, Switzerland, and Great Britain. The minimum BA mass flow among  
497 the countries with positive economic analysis was 0.02 Mt and this was consistent with the  
498 maximum mass flow among the countries where investment was marked unprofitable, except  
499 for Belgium and Denmark where the high landfill tax fees led to a negative economic  
500 profitability.

### 501 *3.5. Policy implications*

502 The positive effects of BA recycling economically and environmentally have also been  
503 recognized by politicians. E.g., in Switzerland, where utilization of the mineral fraction is not  
504 applied, non-ferrous metals must be separated to less than 1% (Schweizerischer Bundesrat,  
505 2015). In the BREF document on waste incineration BAT conclusion 36 lists the best available  
506 technologies to increase the resource efficiency (Neuwahl et al., 2019). These BAT conclusions  
507 are the basis for future legislation on waste management in the EU countries. The present  
508 investigation clearly shows that the extension of BA treatment has positive effects.

509

#### 510 4. Conclusions

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

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

515 - In 2018, 75 Mt of incinerated MSW in Europe generated almost 19 Mt of BA; 54 %-wt,  
516 related both to untreated BA and to technical limitation of treatment facilities (e.g., cut-off  
517 particle size for eliminated fines), was landfilled and 46 %-wt. was processed for material  
518 recovery.

519 - A country-specific inventory at European level of untreated (surplus) and unrecovered (fine  
520 fraction) BA was the first phase of this research. Considering untreated BA, the countries  
521 exhibiting relevant surplus in BA production exceeding local treatment capacity were  
522 Finland and Luxembourg (+31 %), Sweden (+46 %) and Spain (+72 %). Considering  
523 unrecovered BA, its quantity was related to the amount of treated BA (largest contribution  
524 was associated with Germany, France, Great Britain, and Italy), despite the performance  
525 level of BA treatment.

526 - The estimated loss of potential secondary raw materials (2.14 Mt in total) comprised 1 Mt  
527 mineral fraction (0.9 Mt glass cullet), 0.97 Mt ferrous metals and 0.18 Mt non-ferrous metals.  
528 The loss, compared to available amounts of each material in the specific fractions, was  
529 related both to untreated BA (42 % mineral fraction and 45 % ferrous metals) and to  
530 unrecovered BA (84 % aluminum and 60 % copper). Worst results were observed in Spain  
531 (45 %-wt loss of mineral fraction) and France (18 %-wt. loss of ferrous and non-ferrous  
532 metals). The results of MFA showed clearly how higher BA production did not necessarily  
533 imply larger material losses, since the main driver was the technological performance level  
534 that defined the smallest recoverable particle size.

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

537 - Country-specific energy balances and (net) GHG emissions were calculated comparing  
538 complete BA valorization with the extraction and processing of natural resources to produce  
539 aggregates, glass, iron, aluminum, and copper. The energy balance resulted in energy savings  
540 due to the recovery of secondary raw materials from BA.

541 - The evaluation of GHGs emissions showed that the recovery of secondary raw materials  
542 from BA has a much lower environmental impact than mining and processing of natural  
543 resources, with iron implying the highest absolute emission savings and copper the highest  
544 specific emission saving.

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

547 - While CAPEX was subject to the amount of untreated and unrecovered BA (without any  
548 specific dependence to any of the two quotas), country specific OPEX values were mainly  
549 driven by landfill fees regarding the disposal of fine mineral fraction. Incomes were mainly  
550 due to copper and iron recycling and savings to the avoided landfilling of valuable materials.

551 - Economic profitability was achieved by 66 % European countries (Austria, Czech Republic,  
552 Finland, France, Germany, Hungary, Italy, Lithuania, Luxembourg, Netherland, Poland,  
553 Portugal, Romania, Slovakia, Spain, Sweden, Norway, Switzerland, and Great Britain) with  
554 BA mass flow exceeding 0.02 Mt per year, and average values of economic indicators were:  
555 NPV 83 M€ ROI 20 % and payback time 11 years.

556 This work confirmed the strategic significance of optimizing material recovery from MSWI  
557 BA and demonstrated that BA could play a key role in fulfilling European policies based on  
558 Circular Economy. However, country-specific parameters exhibited great influence on the  
559 outcomes of the economic analysis, due to the lack of common legislation across Europe on

560 whether reuse of material recovered from BA is permitted and to the considerable standard  
561 deviation existing among the local landfill fees.

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570 manuscript review: M. Abis, K. Kuchta; F. G. Simon, R. Grönholm, M. Hoppe.

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