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The environmental performance of enhanced metal recovery from dry municipal solid waste incineration bottom ash

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ABSTRACT

This study assesses the environmental performance of the municipal solid waste (MSW) incineration bottom ash (IBA) treatment plant in Hinwil, Switzerland, a large-scale industrial plant, which also serves as a full-scale laboratory for new technologies and aims at an optimal recovery of metals in terms of quantity and quality. Based on new mass-flow data, we perform a life cycle assessment that includes the recovery of iron, stainless steel, aluminium, copper, lead, silver and gold. Fraction-specific modelling allows for investigating the effect of the metal fraction quality on the subsequent secondary metal production as well as examining further metal recycling potentials in the residual IBA. In addition, the implications on the landfill emissions of IBA residues to water were quantified. The impact assessment considered climate change, eco- and human toxicity and abiotic resource depletion as indicators.

Results indicate large environmental savings for every impact category, due to primary metal substitution and reduction of long-term emissions from landfills. Metal product substitution contributes between 75% and >99% to these savings in a base scenario (1′000-year time horizon), depending on the impact category. Reductions in landfill emissions become important only when a much longer time horizon was adopted. The metal-based analysis further illustrates that recovering heavy non-ferrous metals – especially copper and gold – leads to large environmental benefits. Compared to the total net savings of energy recovery (215 kg CO_2-eq per tonne of treated waste, average Swiss plant), enhanced metal recovery may save up to 140 kg CO_2-eq per tonne of treated waste.

1. Introduction

Municipal solid waste (MSW) is a major waste stream of today's urbanized society. In Switzerland, about 4 million tonnes of MSW are incinerated each year, producing about 800′000 tonnes of incinerator bottom ash (IBA) (Christen and Fasko, 2017). In Europe, IBA production has been around 20 million tonnes per year in the recent past (cewep, 2019; Muchová and Rem, 2006). Studies have shown that IBA is a sink for considerable amounts of metals and mineral compounds (Chimenos et al., 1999; Jung et al., 2004; Xia et al., 2017) including precious metals and platinum group metals (Hu et al., 2009; Morf et al., 2013; Muchova et al., 2009). Morf et al. (2013) demonstrated that an economic recovery is possible if precious metals can be accumulated in specific heavy non-ferrous (HNF) output fractions including metals such as copper, silver or gold. However, despite substantially improved recovery technologies, up-to-date detailed data about metal contents and their (potential) recovery, which would enable to carry out a comprehensive Life Cycle Assessment (LCA) of the metal recovery operation is lacking.

The Global Resources Outlook (IRP, 2019) showed that resource extraction and processing heavily contributes to global environmental impacts. Metal recycling is associated with much lower environmental impacts than primary production, but is limited by scrap availability (IRP, 2019). Under the prospect of strongly rising global metal demands (Elshkaki et al., 2018; van der Voet et al., 2019), there is a pronounced need for metal recycling. Metals production from scrap plays an important role in the global steel, aluminium, copper and other metal markets (Graedel et al., 2011; Schipper et al., 2018; Sverdrup et al., 2015; Yellishetty et al., 2010). The global share of recycling compared to total metal

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production, however, has stagnated in the last decade (IRP, 2019). Making better use of metal residues is hence crucial to create additional savings. Metal recycling including its advantages, challenges and limitations has been widely discussed in the scientific literature (e.g. Graedel et al., 2011; Graedel and Reck, 2012; Haupt et al., 2017; Koffler and Florin, 2013; Reuter et al., 2013; van der Voet et al., 2013). From an environmental and economic perspective the recycling of HNF metals is of particular interest, since their primary production is associated with large environmental impacts and costs (Nuss and Eckelman, 2014). Therefore, treatment attempts to efficiently recover HNF metals from IBA have increased in number (Boeni, 2017; Fierz and Bunge, 2007). According to the 2018 EU circular economy package, recycled metals from MSWI residues can be credited to national recycling rates from 2020 onwards (European Commission, 2018), which will further foster metal recycling efforts from IBA. However, there is, to our knowledge, no Life Cycle Assessment (LCA) study including the recovery of precious metals (silver and gold) from IBA. Furthermore, existing studies operate with rough assumptions (e.g. Allegrini et al., 2015) or neglect the quality of produced metal fractions.

The magnitude of metal recycling benefits, however, hinges on which primary materials are substituted, depending on displacement mechanisms and quality reduction through contamination (Daehn et al., 2017). Vadenbo et al. (2017) proposed a reporting framework to support the systematic estimation of substitution potentials from resource recovery. Another approach by Zink et al. (2016) used partial equilibrium modelling for quantifying displacement rates of primary materials. Rigamonti et al. (2018) underlined the importance of performing the substitution calculation at the point of substitution. Despite awareness of the crucial importance of choosing a realistic substitution option, studies still often use best-case assumptions or other simplifications.

Representative sampling as a basis for sound substance flow analysis (SFA) is another important element regarding the reliability of LCA results. Representative sampling includes three major concepts (Esbensen and Wagner 2014): i) sampling correctness; ii) sampling bias; iii) sampling variance as defined in the theory of sampling. Skutan and Haag (2018) applied the theory of sampling concept in their SFA conducted at the IBA treatment plant in Hinwil, Switzerland, which serves as a basis for our LCA.

Another crucial element of LCA in waste management systems is the assessment of long-term emissions from landfills to water (Finnveden et al., 2009; Hauschild et al., 2008; Laurent et al., 2014), including the choice of the considered time horizon (Finnveden, 1999; Hauschild et al., 2008; Obersteiner et al., 2007; Sundqvist, 1999). Doka and Hischier (2005) underlined the importance of considering the specific chemical composition of the waste material, yet geochemical processes are often not taken into account in LCA when it comes to the long-term behaviour of the landfilled material. Di Gianfilippo et al. (2016), who aimed at comparing the environmental impact of different IBA management options, used leaching test data to predict future landfill emissions. Johnson et al. (1995) have analysed the acid neutralising capacity of MSW IBA and concluded that combined knowledge of geochemical properties of IBA together with the consideration of specific landfill conditions are needed for a more accurate assessment of the potential hazards of long-term heavy metal cation contamination. Geochemical modelling of emissions from IBA landfills has been applied in LCA in the past and more recently (Bakas et al., 2015; Dijkstra et al., 2018; Hellweg, 2000; Sabbas et al., 2003; Slack et al., 2007), but the according results have not yet been included in widely used LCI databases.

This study aims to analyse the environmental impacts and benefits of metal recycling from IBA by conducting an LCA. We investigate both primary metal substitution mechanisms and long-term leaching of metals from the landfilled residual IBA. Fraction-specific modelling allows us to estimate the point of substitution separately for each metal fraction and consider cross-contamination between the different metal fractions in the substitution calculations. Furthermore, using PHREEQC modelling (Parkhurst and Appelo, 2013) we consider IBA-specific geochemical processes in the landfill. The analysis includes the recovery of iron, aluminium and copper each in metallic state (Fe (met), Al (met), Cu (met)), stainless steel, as well as lead, silver and gold as chemical elements. Regarding long-term leaching from landfills, arsenic, cadmium, copper, lead and zinc are considered. This study provides essential insights for future environmental optimization of IBA treatment plant configurations.

2. Materials and methods

2.1. Goal and scope

This study assesses the environmental performance of an IBA treatment plant in Hinwil, Switzerland, which is based on the thermo-recycling concept (Boeni and Morf, 2018). The concept aims at enabling efficient metal recovery from dry-discharged IBA as clean metal fractions. Metal recovery levels according to the measurement campaign of 2017 (*IBA treatment* scenario) are compared to the reference case without IBA treatment (*direct landfilling* scenario). Fig. 1 provides an overview of the treatment layout and the recovered metal fractions including their allocation to the different particle size classes. We aim to analyse the fraction-specific contribution to the net environmental benefit by investigating specific recycling paths.

The dry IBA treatment facility in Hinwil, Switzerland, is the first large-scale industrial plant of its kind worldwide (see Boeni and Morf, 2018 for technical documentation). At no point during the IBA treatment process does the IBA have contact with water to minimize oxidation losses and maximize resource recovery efficiency (dry-discharged). The separation of NF metals is carried out with induction separators (Eddy Current Separators). High separation efficiency is achieved by serial configuration of the separation machines (for fractions < 12 mm particle size) and by circulating the material (fractions > 12 mm). Sequential crushing in three crushing stages enables almost complete exposition of the metals enclosed in IBA agglomerates. Due to the consistent sieving and selective crushing of the IBA, high efficiency is achieved, e.g. 85% recovery of the light non-ferrous (LNF) > 1 mm fraction.

2.1.1. System boundaries

Fig. 2 shows the system boundaries of the LCA study. Both the *IBA treatment* and the *direct landfilling* scenario produce two outputs per 1 tonne of dry-discharged IBA from the incineration of Swiss mixed MSW (plant performance and waste composition as measured during a ten-week time period from May to July 2017): an array of metals (see Table 2 for metal masses) and the IBA disposal on a landfill. Table 1 lists the energy and water requirements of the treatment plant, which is included in the *IBA treatment* scenario (see also Table S7 and, for the infrastructure of the plant, S8). In the *direct landfilling* scenario, primary production is assumed as the metal production source. The net environmental benefit of the *IBA treatment* scenario per tonne of dry IBA is then calculated as the difference between the two scenarios, which is comprised of: (i) credits for substitution of metals, that otherwise need to be produced from metal ores, (ii) credits for reduced long-term landfill emissions to water due to lower heavy metal content that is disposed of within the residual IBA (‘avoided burden’ approach). In a second step, the net environmental benefit

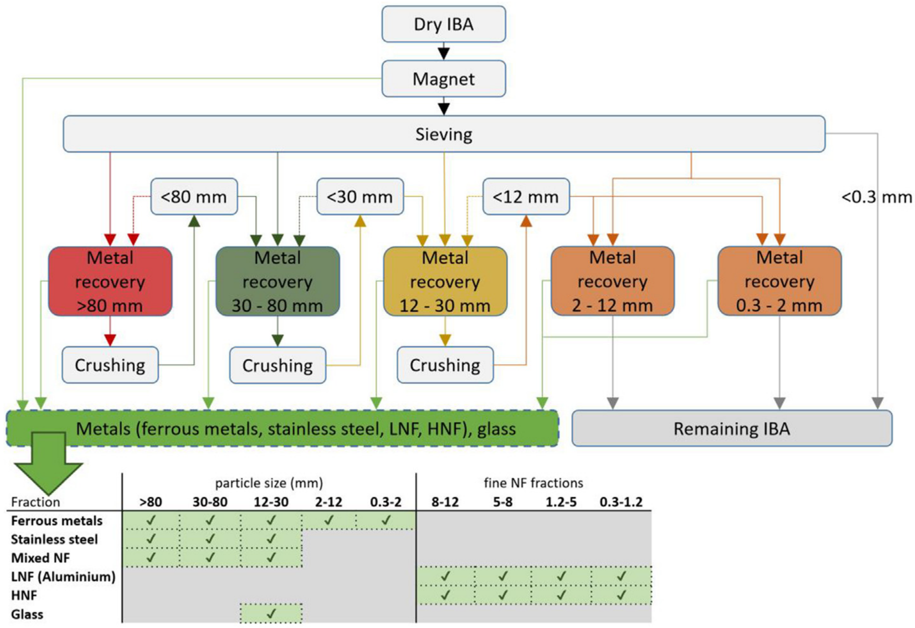

Fig. 1. Layout of the IBA treatment plant in Hinwil, Switzerland. Particle size classes of the recovered metal fractions (including fine NF fractions from NF recovery) are listed below. Glass is not included in the analysis. LNF = Light non-ferrous, HNF = Heavy non-ferrous.

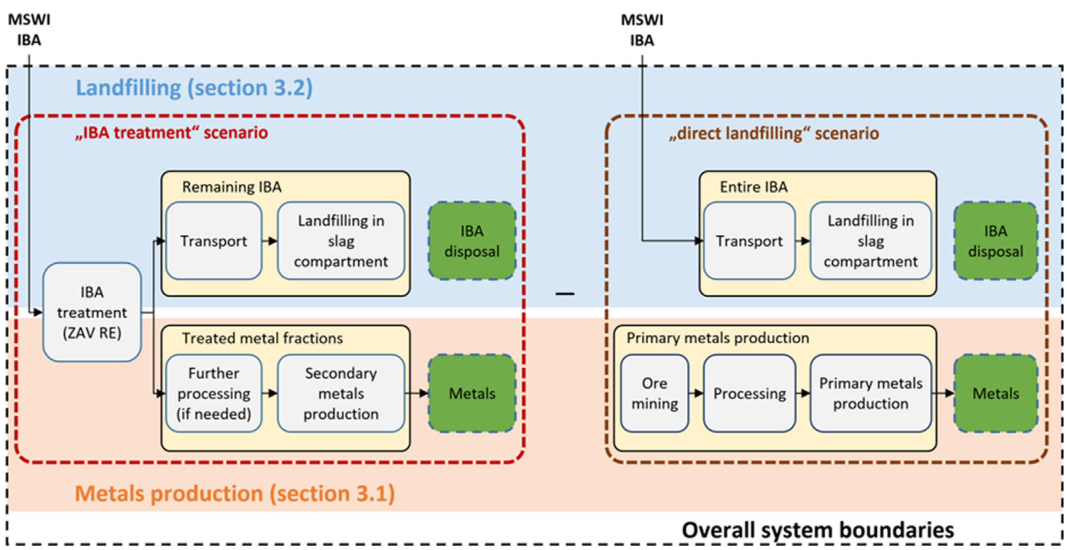

Fig. 2. System boundaries of the LCA ('avoided burden' approach). Products/services of the functional unit are in green. Result chapters of metals production (Section 3.1) and landfilling (Section 3.2) cover subareas of the total system boundaries. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Energy and water use of the IBA treatment plant per tonne of dry IBA for the year of the measurement campaign, 2017.

		2017
Electricity	kWh	15.5
Heat (from MSWI)	MJ	5.1
Cooling energy (from MSWI)	MJ	6.1
Compressed air (from MSWI)	m3	18.5
Water	kg	80.1

of the primary metal substitution is further broken down to the individual metals and metal fractions to detect contribution hotspots (Section 3.1). The analysis additionally covers the residual IBA fractions that are currently landfilled to detect further theoretical environmental improvement potential of the metal recovery operation. The landfilled IBA is analysed for both scenarios with focus on the environmental impact of long-term landfill emissions for specific elements (Section 3.2).

For all processes included in the system boundaries, ecoinvent v3.4 was used as background LCI database. For several processes, ecoinvent v3.4 inventories were adapted to case-study specific conditions as provided in the following sections.

2.1.2. Differentiation of secondary production pathways

Depending on the quality of the recovered metal fractions, subsequent secondary production pathways differ considerably (see Fig. S1 in the *Supplementary Material*). In this study, we consider different pathways of ferrous and LNF fractions and light non-

Table 2
Extracted metal masses (including standard deviation) per tonne of dry IBA as well as current extraction efficiencies of the Hinwil plant (calculated based on Skutan and Haag, 2018). Extraction efficiency is the total percentage of each metal that is extracted from the IBA. The term "in target fraction" depicts the share thereof that ends up in the "correct" fraction (e.g. HNF in the HNF fraction and not in the LNF fraction). Recycling efficiency additionally includes further recycling losses, hence describing the share of each metal that becomes available as secondary material. The recycling efficiency potential is indicated in case of the additional extraction of all free residual metals is listed (calculated based on Skutan and Haag, 2018, same recycling pathways assumed). RMP = Residual metal potential.

		Current metal extraction					RMP free residual metals
	Extracted mass	**Standard deviation**		**Extraction efficiency**	**Extraction efficiency in target fraction**	**Recycling efficiency**	**Recycling efficiency**
	(g/t IBA)			(%)			(%)
Fe(met), >0.3 mm	95′862	±	5′415	91.7%	90.2%	89.6%	94.3%
Stainless steel, >0.3 mm	8′706	±	339	87.7%	83.5%	75.6%	85.3%
Al(met), >0.3 mm	32‘267	±	1′151	84.6%	81.3%	69.9%	72.0%
Cu(met), >0.3 mm	5′636	±	585	64.3%	54.1%	53.6%	69.3%
Pb[1]	300	±	63	22.3%	15.9%	15.7%	21.9%
Ag[1]	22	±	7	57.4%	55.8%	55.2%	67.5%
Au[1]	0.6	±	0.3	28.8%	27.4%	27.2%	39.3%

[1] Total content as chemical element.

ferrous (LNF) fractions (LNF = aluminium). The high-quality coarse ferrous fraction (>80 mm) is delivered directly to the steelworks, whereas the other ferrous fractions need to be pre-processed before utilization in the steel industry. Clean fine fractions, such as the finest LNF fraction, on the other hand, may directly be reused, e.g. as reduction agent, without further processing and remelting. Such differentiation is crucial when determining substitution benefits since a) post-processing processes (e.g. remelting of low-quality fractions) may result in considerable additional environmental impacts and hence decrease recycling benefits and b) the quality of the final metal product may differ from the primarily produced metal product and 1:1 substitution may therefore not be justified.

2.2. Life cycle inventory analysis

The LCA of this study is based on detailed substance flow data from a ten-week measurement campaign commissioned by ZAV Recycling AG (ZAV RE; plant operator) from May to July 2017 at the dry IBA treatment facility in Hinwil, Switzerland (Skutan and Haag, 2018) (see Tables S1 and S2 in the *Supplementary Material*). The campaign included measurements of the mass flows according to Fig. 1 as well as quality measurements of metal fractions and residual IBA, for instance, the grades of pure metal in the metal fractions and the residual metal content of the landfilled IBA. The individual parameters under consideration are Fe (met), stainless steel, Al (met), Cu (met) as well as lead, silver and gold as chemical elements. Regarding landfill emissions, the metals As, Cd, Cu, Pb and Zn contained in the residual IBA are included in the analysis. These metals were chosen due to their toxicity and based on a screening assessment of landfill emissions (see Fig. S3 in the *Supplementary Material*).

The residual IBA consists of a magnetic fraction, two mineral residual fractions from the IBA treatment (0.3–2 mm, 2–12 mm) and a dust fraction (<0.3 mm) which are currently landfilled. Each fraction contains residual metals which are divided into *free* (freely movable metal pieces with little to no contamination) and *enclosed* metals (metal pieces enclosed by or containing much adhesion of mineral IBA particles) (Skutan and Haag 2018). In addition to current metal recovery, the theoretical limits of metal extraction are analysed in two steps. We differentiate between i) potential with the current plant configuration by additional extraction of all *free* metals and ii) total metal potential, only extractable after complete breakdown of IBA aggregates and complete liberation of all metals. Both primary metals substitution (same pathways as for the other fractions) and landfilling were considered. The additional energy consumption of the treatment plant in case of these two scenarios could not be estimated. Therefore, the assessed additional environmental benefit is a theoretical limit rather than a realistic extraction scenario. We call it 'environmental benefit potential'.

Table 2 lists the extracted metal masses per functional unit as well as the extraction efficiencies of the plant (Skutan and Haag 2018).

2.2.1. Primary metals substitution

In the context of the global metal demand increase, it was assumed that all metals extracted from the IBA and subsequently fed into secondary production fully substitute primary metals of equivalent quality (Elshkaki et al., 2018; IRP, 2019).

The secondary production paths are as follows (see also Tables S3–S5 in the *Supplementary Material*): (i) the coarse ferrous fraction (>80 mm) is directly brought to a Swiss steelwork, whereas all other ferrous fractions require further, quality depending, pre-processing. Higher electricity demand in the steelwork for low-quality steel scrap was considered for the ferrous fractions between 0.3 and 30 mm (Haupt et al. 2017). (ii) Stainless steel fractions are pre-processed before they are remelted in European plants. Direct delivery is currently prevented by the low quality of the stainless steel fractions (up to 10% aluminium and 40% other, mostly mineral, impurities). Pre-processing was simulated by iron scrap processing. For the remelting process, functional recycling of alloying elements in the scrap, such as chromium and nickel, was assumed. (iii) The LNF fractions between 1.2 and 12 mm are mixed and then exported for further processing and remelting. The fine fraction (0.3–1.2 mm) is reused directly as, for example, reduction material in the aluminium industry or additive in steelworks (without further recycling steps) (Boeni and Morf, 2018). (iv) The HNF fractions end up in recycling facilities (e.g. Umicore, Belgium, or Aurubis, Germany), where apart from copper and lead, precious metals, such as silver and gold, or palladium, are recycled (Caffarey, 2012). The multi-input-multi-output recycling process is approximated with metal recycling from electric&electronic scrap in a comparable plant in Sweden (Classen et al., 2009).

For all metals, processing losses up to the point of substitution are considered. The treatment of scrap processing residues, however, was neglected. Concerning primary production, ecoinvent v3.4 market processes were adapted so that only primary production pathways are considered (see also sensitivity analysis in the *Supplementary Material*). Table S3 in the *Supplementary Material* lists all ecoinvent v3.4 processes used or adapted for the primary and secondary metals production. Table S4 lists the adapted global market processes.

2.2.2. Landfilling

Landfills release heavy metals and other pollutants to the environment over long time periods. Metal extraction from IBA prior to landfilling may therefore reduce emissions and related toxic effects.

The long-term emissions from landfills need to rely on models, since the time periods involved exceed existing measurement series (e.g. of leachate concentrations and common batch-leaching tests) (Johnson et al., 1995). In this analysis, we used an updated and new parametrization of the geochemical landfill modelling approach published in Hellweg et al. (2001) to predict future emissions as a function of time. The modelling was based on the coupled transport and reaction model PHREEQC (Parkhurst and Appelo, 2013). Dissolved leachate concentrations of heavy metals are limited by dissolution/precipitation processes of the IBA matrix as a function of solution pH and accompanying component concentrations. The pH value is a function of the minerals contained in the IBA, the partial pressure of CO_2, and the chemical reactions taking place. In all calculations, thermodynamic equilibrium and one-dimensional transport of the pore water through the solid landfill matrix were assumed. We quantified with the model the dissolved concentrations of Cd^{2+}, Zn^{2+}, Cu^{2+}, Pb^{2+} and As in the leachate at equilibrium with a given set of solid phases (see Tables S10–S12). As input data, we used specific chemistry and mineralogy data from the residual dry IBA of the Hinwil plant (see Table 3) (Weibel, 2020). The chemical composition of the residual IBA was determined by X-ray fluorescence analysis (WD-XRF). The crystalline phases of the dried and ground IBA were determined by X-ray powder diffraction (XRD). For the *direct landfilling* scenario, we calculated the chemical composition adding the extracted metal amounts to the analysed IBA-treated landfill composition. For the metals As, Cd and Zn that were not included in the measurement campaign by Skutan and Haag (2018), we used estimates according to Table S9 in the *Supplementary Material*. These estimates are based on results from former chemical testing on similar fractions or comparable bottom ash fractions. Estimated contents of As, Cd and Zn in recovered metal scrap are used in the same way of calculation as the data on Al, Cu, Pb, Ag, Au from Table S2. The model is based on landfill parameters such as infiltration rate or landfill depth (Table S13). Further information about the model, preparation of input data and landfill parameters can be found in the *Supplementary Material, p. 11 ff.*

In the *Supplementary Material* we show the continuous emission concentration curves as a function of time, which can enable the conduction of a risk assessment in future research. For the LCA in this paper, emissions were cumulated and assessed over three emission time horizons: (i) short-term (ST; 100 years); (ii) 1′000a; (iii) long-term (LTmax; indefinite time horizon and total leaching for all metals). The influence of time horizon choice on the overall results is discussed in Section 4.

2.3. Life cycle impact assessment

In the LCIA step, all processes and flows are characterized to determine their environmental impact. The following LCIA indicators were used: climate change (IPCC, 2013), freshwater ecotoxicity and human toxicity (Rosenbaum et al., 2008) and abiotic depletion potential (van Oers et al., 2002), as implemented in ecoinvent v3.4. The selection of indicators aims at covering the most relevant impact categories in the context of waste management and enables a comparison with other studies in the field.

Table 3
Chemistry (WD-XRF, left) and (total digestion, right) of the residual dry IBA. Source: Weibel (2020).

Parameter	wt.%
SiO2	46.6
CaO	22.6
Fe2O3	4.6
Fe (met) > 0.3 mm	0.1
Al2O3	7.1
Al (met) > 0.3 mm	0.7
SO3	1.8
Na2O	4.6
MgO	2.2
TiO2	1.3
K2O	1.3
P2O5	1.0
MnO	0.1
Main Chemistry	94.0
Ba	0.27
Cl	0.51
Cu	0.33
Cu (met) > 0.3 mm	0.21
Zn	0.30
Cr	0.07
Pb	0.17
Sr	0.06
Zr	0.05
Ni	0.03
Sb	0.01
Sn	0.03
W	<0.01
V	<0.01
Br	<0.01
As	<0.01
Ag	<0.01
Co	0.01
Cd	<0.01
TOC	0.60
LOI	3.16
Minor Chemistry	5.8
Sum	99.9

Parameter	Unit	Dry IBA
Dry substance (105 °C)	%	100.0
Residue on ignition (550 °C)	%	97.8
TC	%	1.1
TIC	%	0.5
TOC	%	0.6
N	%	0.1
Al	mg/kg	46,368
As	mg/kg	25.8
Ca	mg/kg	142,500
Cd	mg/kg	4.9
Cr	mg/kg	437
Cu	mg/kg	4,764
Fe	mg/kg	32,502
Ni	mg/kg	183
P	mg/kg	4,140
Pb	mg/kg	1,531
S	mg/kg	4,126
Sb	mg/kg	109
Si	mg/kg	233,750
Zn	mg/kg	3,594

3. Results

3.1. Metals recovery

Secondary metals production is associated with lower environmental impacts compared to the primary production of the same metal product (see Fig. S4 in the *Supplementary Material*). The relative environmental impact of the evaluated secondary production path is, depending on the LCIA method, between 0.00005% and 92% for the associated metals. The relative impact is particularly favourable for copper, lead, silver and gold, as well as for the finest LNF fraction, which is directly reused without energy intensive remelting. In addition, quality differences between the different size fractions of ferrous and LNF scraps are crucial regarding the relative impact of the secondary production path, especially for aluminium. In total, the substitution of primary metals through dry IBA treatment in Hinwil and subsequent recycling leads to net environmental benefits of 780 kg CO_2-eq, 24′700 CTUe (comparative toxic units), $2.1*10^{-3}$ CTUh and 0.2 kg Sb-eq per tonne of dry IBA.

3.1.1. Fraction-based analysis

Our analysis showed that there is no particle size class with a predominant contribution to the total net environmental benefit (see Fig. 3). The largest proportion of benefits can be attributed to the 12–30 mm class (22–36%), more specifically to the aluminium and copper (both in metallic state) therein. Fig. 3 further illustrates that, despite their much lower mass fraction, the fine fractions 0.3–12 mm play an important role. For toxicity and abiotic resource depletion, these fractions contribute to more than half of the total net environmental benefit of the primary metal substitution.

3.1.2. Metal-based analysis

Fig. 4 gives an overview of a) the mass fraction of the recycled metals (left bar) and b) the proportion of the same metals with respect to the total net environmental benefit of the metal recycling operation (right bars). While the ferrous fractions make up close to 70% of the recycled mass, their significance regarding substitution benefits is smaller with between 7% (abiotic resource depletion) and 27% (climate change). LNF makes up 21% of the recycled mass and is especially relevant for climate change (63% of net environmental benefit) due to the avoidance of substantial amounts of CO_2 emissions in primary aluminium production. Copper plays, despite the low proportion in recycled mass (4%), an important role for toxicity impacts, with a contribution to the total net environmental benefit between 45% and 50%, and for abiotic resource depletion (18%). Gold makes up only about 0.0005% of the total recycled mass. However, its recovery is of great importance when looking at toxicity impacts (25% to 26%) and abiotic resource depletion (17%).

3.2. Landfilling

The environmental impact of landfilling consists of the transportation of the IBA to the landfill, the landfill infrastructure and the long-term emissions of the IBA to the environment. Transportation and infrastructure showed to be negligible in case of short transport distances (break-even distance for climate change impacts: 600 km, see Fig. S18 in the *Supplementary Material*).

Fig. 5 shows the eco- and human toxicity impact of IBA disposal per tonne of dry IBA for both scenarios *IBA treatment* and *direct landfilling* for the three time horizons. The total environmental impact is strongly dependent on the chosen time horizon. While impacts are rather low in the *ST* and *1′000-year* time horizons, they become substantially bigger in case of complete leaching (*LTmax*).

In terms of the modelled metals, zinc is the dominating contributor in the *100* and *1′000-year* time horizons for ecotoxicity and in all time horizons for human toxicity. Zinc is washed out the fastest of all considered metals (within the first few thousand years) (see Section 2.2 in the *Supplementary Material*). Copper emissions are the main contributor of ecotoxicity (64%) in the long-term scenario (complete leaching).

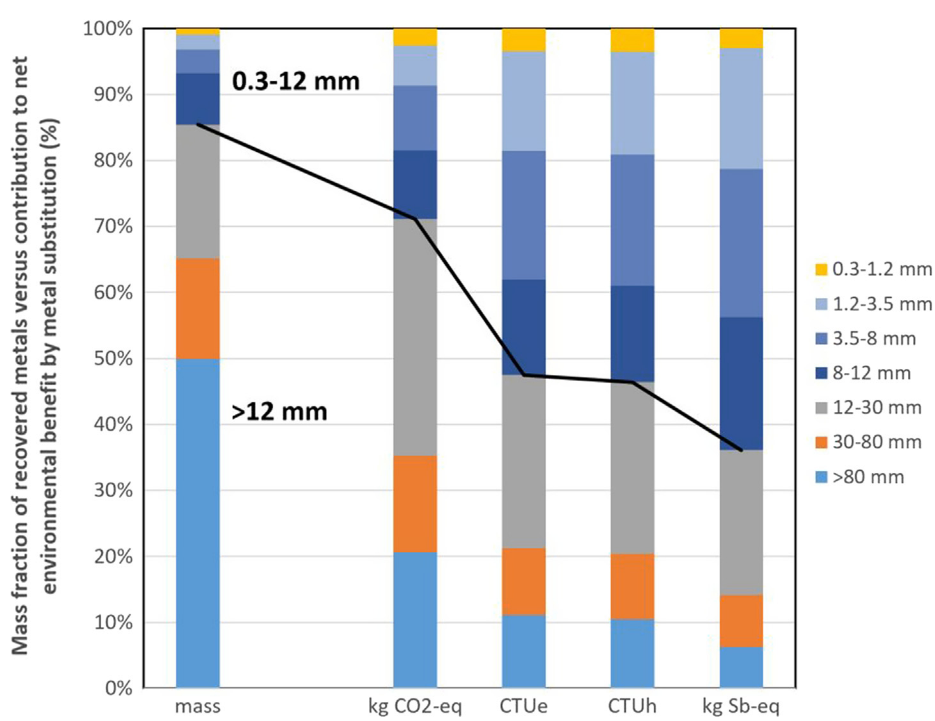

Fig. 3. Particle size class-based mass fraction of recovered metals as well as their contribution to the total net environmental benefit by primary metal substitution. Particle size classes are divided into fine (0.3–12 mm) and coarse (>12 mm) by the black line.

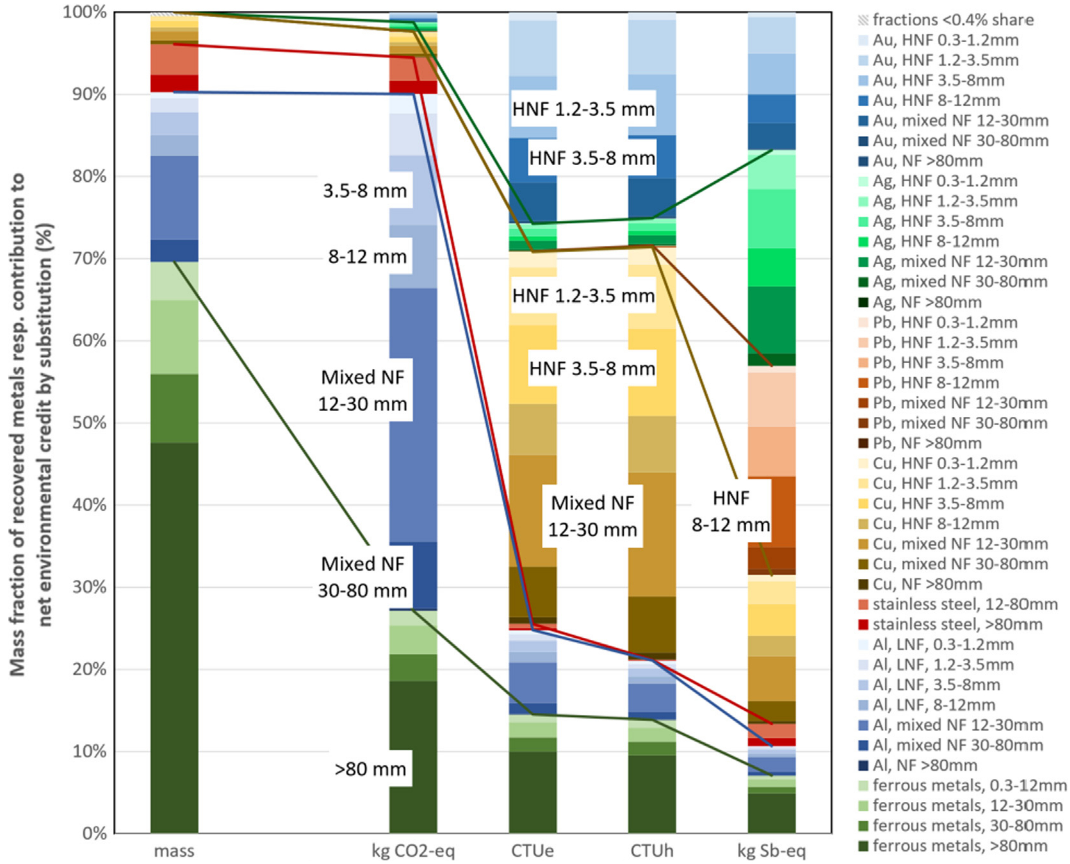

Fig. 4. Mass fraction of recovered metals as well as their contribution to the total net environmental benefit for four impact categories. Metals are divided by colour, additionally subdivided by produced metal fraction. Mass fractions < 0.4% wt. are summarised. Most relevant fraction contributions are additionally labelled.

While the absolute difference in total toxicity impacts between the *direct landfilling* scenario and the *IBA treatment* scenario is small for time horizons $\leq$ 1′000 years, it is much larger for long-term time horizons due to copper and zinc extraction in the IBA treatment (and therefore reduced long-term emissions since the total metal content in the landfilled IBA is smaller).

3.3. Total environmental benefit of IBA treatment

Table 4 shows that the net environmental benefit of IBA treatment is positive for all impact categories. Primary metal substitution benefits are larger than the credit for reduced landfill impacts for all impact indicators when applying a 1′000-year time horizon, and for all impact indicators except for ecotoxicity when considering complete leaching. Therefore, the difference in landfill emissions is negligibly small compared to environmental credits for primary metal substitution in the first 1′000 years after landfilling. In the case of complete leaching (*LTmax*), the share of reduced landfill impacts on the total net environmental benefit is 88.8% for ecotoxicity and 48.1% for human toxicity.

Total environmental impacts of both scenarios (‘system expansion’ approach) are additionally displayed in Section 2.3 in the *Supplementary Material*.

3.4. Residual metal potential

Fig. 6 gives an overview of a) the relative mass of the additional extraction of residual metals (left bar) and b) the additional environmental benefit potential of their extraction divided by metal type (right bars), both of them in relation to the status quo at the plant in Hinwil in 2017 (for which details were presented in Fig. 4). The recovery of *free* metals and both *free* and *enclosed* (=total) metals is further differentiated.

Increasing the efficiency of the current plant configuration to 100% (=additional recovery of all *free* residual metals; 9% additional mass compared to status quo extraction) would result in an additional environmental benefit potential between 10% (climate change) and 44% (abiotic resource depletion), indicating that the share of environmentally relevant metals such as copper, silver or gold is higher within the *free* residual metals than in the currently extracted IBA fractions. Regarding both *free* and *enclosed* residual metals reveals the theoretical potential beyond current plant configuration under complete liberation of the residual metals in the IBA. The additional environmental benefit potential is then between 24% (climate change) and 89% (human toxicity). Except for climate change, copper and gold have the largest additional potential. Residual metals < 0.3 mm (dust fraction) were considered as non-extractable in any case.

Apart from potentially increasing the substitution benefit, additional metal extraction leads to potentially further reduced long-term emissions from landfills.

3.5. Swiss thermal waste treatment sector

To evaluate the relative environmental importance of metal extraction from IBA in the Swiss thermal waste treatment sector, the results of this study are compared to current data from the Association of operators of Swiss waste treatment plants (VBSA, 2019). We focus solely on climate change, where leachate emissions do not play a role. The enhanced metal recovery according

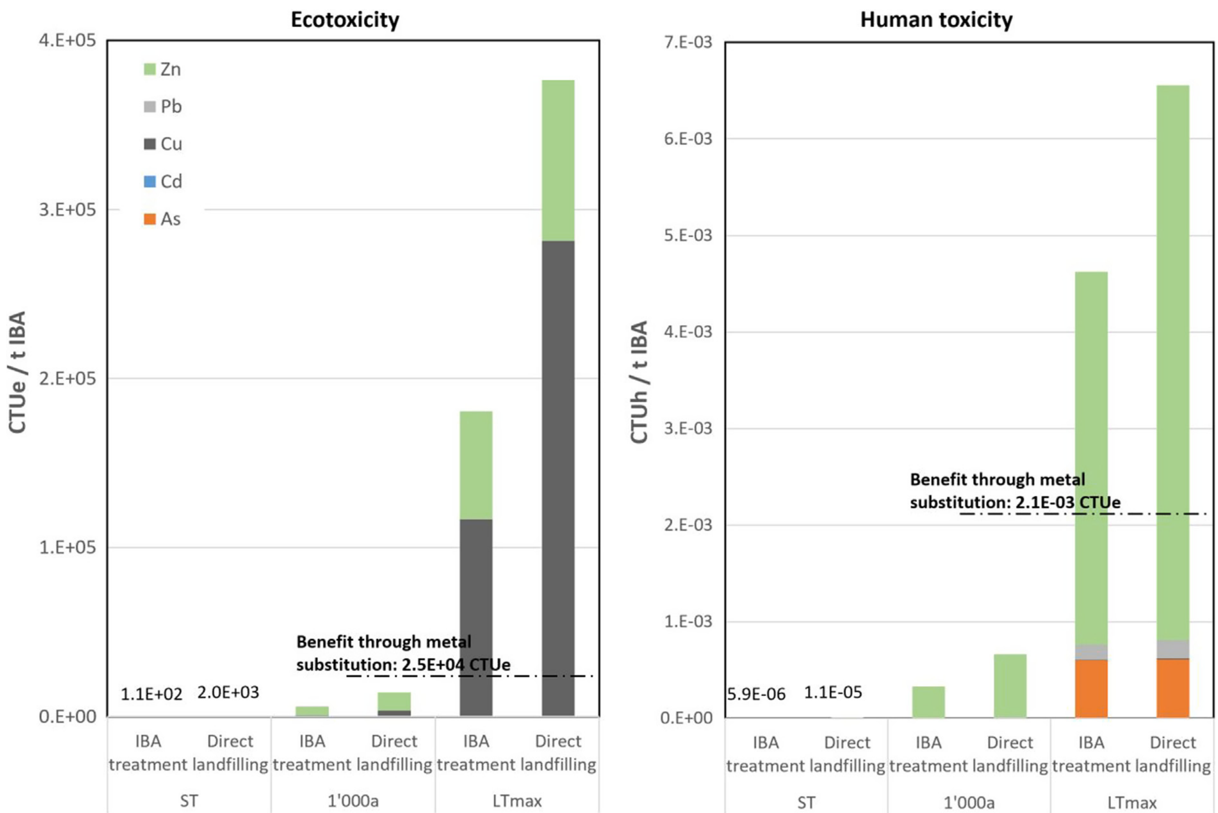

Fig. 5. Ecotoxicity (left graph) and human toxicity (right graph) impact of landfilling: i) residues of IBA treatment (left columns of each pair) and ii) direct landfilling of 1 tonne of IBA (right columns). The environmental benefit through primary metal substitution is displayed for comparison. ST = short-term.

Table 4
Net environmental benefit of IBA treatment operation, compared to the *direct landfilling* scenario, divided into metal substitution ('metals') and reduction of landfill emissions ('landfill'), as well as their share of the total net environmental benefit, for the three landfill time horizons *ST*, *1′000a* and *LTmax*. The higher share is highlighted (orange = metal substitution, blue = landfill).

Impact category	Unit		100a (ST)		1'000a		T=∞ (LTmax)	
			absolute	(%)	absolute	(%)	absolute	(%)
Climate change	kg CO2-eq	Metals	7.8E+02	**99.8**	7.8E+02	**99.8**	7.8E+02	**99.8**
		Landfill	1.2E+00	0.2	1.2E+00	0.2	1.2E+00	0.2
Freshwater ecotoxicity	CTUe	Metals	2.5E+04	**99.6**	2.5E+04	**74.8**	2.5E+04	11.2
		Landfill	1.0E+02	0.4	8.3E+03	25.2	2.0E+05	**88.8**
Human toxicity	CTUh	Metals	2.1E-03	**99.8**	2.1E-03	**86.2**	2.1E-03	**51.9**
		Landfill	5.1E-06	0.2	3.3E-04	13.8	1.9E-03	48.1
Abiotic resource depletion	kg Sb-eq	Metals	4.5E+00	**100.0***	4.5E+00	**100.0***	4.5E+00	**100.0***
		Landfill	1.6E-02	0.0	1.6E-02	0.0	1.6E-02	0.0

*Rounded value.

to this study adapted to the whole IBA amount in Switzerland would lead to a reduction of 559 kt CO_2-eq per year (see Table S22 in the *Supplementary Material*). On a MSWI plant level in Switzerland, environmental benefits from enhanced IBA treatment are 140 kg CO_2-eq per tonne of thermally treated waste in total. This contribution is more than fifteen times the impact reduction by electricity recovery and 64% of total climate change benefits by energy recovery (electricity and heat).

4. Discussion

4.1. Case study results

Results indicate substantial environmental savings of metal recycling by processing dry-discharged IBA according to the processes applied in Hinwil, Switzerland, for all investigated impact categories. In the *1′000a* time horizon, metal product substitution

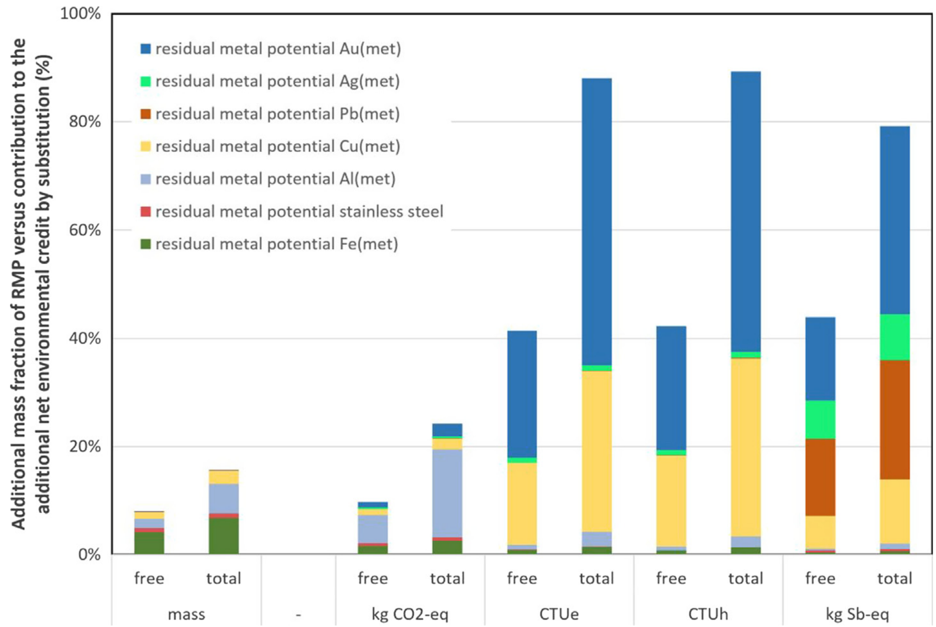

Fig. 6. Additional mass fraction of residual metals as well as their contribution to additional environmental benefit potential through primary metal substitution in relation to status quo extraction of the plant in Hinwil in the year 2017, divided by metals. Theoretical potential with the current plant configuration ('free') and potential under complete liberation of all residual metals ('total') are differentiated.

contributes between 75% (ecotoxicity) and > 99% (climate change, abiotic resource depletion) to this saving. This is in line with Geyer et al. (2016) stating that it is increasingly recognized that the environmental benefits of avoiding primary material production are typically much more significant than the benefits of avoiding landfilling, especially under modern landfill conditions. Allegrini et al. (2015) reported similar findings for climate change but not for toxicity. This discrepancy mainly arises from toxicity impacts of secondary steel production. They reported significantly larger toxicity impacts for secondary steel production compared to primary production based on the ecoinvent inventories at the time (see Table S6 in the *Supplementary Material*). Furthermore, lead, silver and gold recycling was not included in their study and neither in a similar study by Boesch et al. (2014). As we demonstrate, the cumulative contribution of silver and gold to the net environmental benefit of primary metal substitution is almost 30% for both eco- and human toxicity and 43% for abiotic resource depletion. Hence, overall toxicity benefits are much larger due to the inclusion of precious metal recovery.

Secondary metals production is associated with smaller environmental impacts than primary production of the same metal. Due to modified/updated inventories or differing secondary production pathways, direct comparison to the existing literature is often difficult. For aluminium, impacts of primary and secondary production calculated in this study are comparable to the values estimated by Stolz et al. (2016), who compiled updated LCA inventories for aluminium. For steel, on the other hand, Yellishetty et al. (2010) estimated CO_2-emissions of secondary steel production as 28% of primary production, while this study comes to slightly above 10%. This is due to Swiss-specific datasets (low-CO_2 electricity mix) and high plant efficiency (European Commission, 2010). For secondary copper, lead, silver and gold production, there is no other comparable dataset to our knowledge.

When comparing the results of this study to former studies, it must be considered that recycled metal types, metal yields and product qualities are continuously improving. Boesch et al. (2014), for example, analysed environmental benefits through metal recovery from MSWI IBA in Switzerland using metal extraction efficiencies between 40% (Al, Cu, stainless steel) and 80% (Fe). In the meantime, even the mentioned improvement potential of aluminium recovery of up to 70% has been clearly surpassed. The metal yields achieved at the Hinwil plant fall between the two most optimistic scenarios of Allegrini et al. (2015a,b). The electricity demand of the processing facility, on the other hand, is in the same order of magnitude (15.5 kWh in our study compared to 11 kWh per ton of IBA). Since the environmental impact of IBA treatment is negligible compared to savings by primary metal substitution and avoided landfilling, net environmental benefits reported in this study are considerably higher than net benefits in Allegrini et al. (2015a,b).

These results clearly underline the importance of accurate plant configuration and sorting to produce high yields and clean metal fractions. The purity of the recovered fractions directly influences substitution benefits, as is shown for ferrous metals and LNF. Future research could further elaborate on cross-contamination between fractions and its effect on post-processing and remelting processes given that the respective data are collected.

In addition, comparing relative values such as metal yields or relative environmental impacts, e.g. of secondary production compared to primary production, is usually more meaningful than comparing absolute values, since (recoverable) metal contents per ton of IBA differ (Jung et al., 2004; Morf et al., 2013; Muchova et al., 2009; Muchová and Rem, 2006). Silver and gold contents of IBA used in this work slightly exceed findings by Morf et al. (2013), which can be credited to the extended measurement campaign performed by Skutan and Haag (2018). A former study by Muchova et al. (2009) analysed silver and gold contents of IBA in Amsterdam and obtained considerably lower levels.

By comparing the results of this study with current data from VBSA (2019), we aimed to illustrate their importance in the context of the Swiss thermal waste treatment sector. Our results show a substantial potential of CO_2 emission reduction if the dry IBA treatment of Hinwil is extended to all other plants in Switzerland. Total net environmental benefits (compared to direct landfilling) are 1′690% of total electricity recovery from MSWI plants in Switzerland and 64% of total energy recovery. However, this is also due to the low-carbon Swiss electricity mix and therefore rather small credits for avoided electricity (see e.g. Eriksson and Finnveden, 2017). In case of the UCTE electricity mix, the contribution of metal recovery would be considerably larger than that of electricity recovery. Heat and electricity recovery efficiencies in Switzerland vary quite significantly between different plants (Christen and Fasko, 2017), at average values of 29% for heat and 17% for electricity, respectively. In this context, optimised IBA treatment can substantially further increase the climate change mitigation potential of MSWI by complementing further energy recovery efficiency increases, e.g. by better heat utilisation.

While metals extraction from IBA can contribute to the reduced environmental impact of MSW disposal, mineral components still account for the biggest mass fraction in IBA (84% of IBA by weight). Based on the current efficiency of the metal extraction, the remaining mineral fraction needs to be landfilled and is not allowed to be further used in Switzerland, e.g. in road construction. Optimised metal extraction is an important step towards the vision of long-term secure landfills without or with much shorter aftercare time period (Laner et al., 2010). This vision is also part of the waste and resource management action plan of the canton of Zurich (AWEL, 2019). In addition, a much cleaner residual mineral fraction could enable other uses in the long run, e.g. as substitute in road construction, in the cement industry or at least landfilled in an inert landfill.

4.2. Methodology

Allocating environmental impacts to the production of individual metals is not straight-forward due to the highly interconnected system of modern metals production. Many of the metals are derived from multiple ores and as co-products with other metals along the processing stages (Nuss and Eckelman, 2014). Hence, environmental impacts often are a result of allocation algorithms. In case of economic allocation, which is typical in the primary metal sector (ecoinvent, 2018), most environmental impacts are therefore allocated to metals with higher market prices such as gold. Nuss and Eckelman (2014) have performed a sensitivity analysis of different allocation procedures, concluding that there are differences in the resulting environmental impacts for certain metals.

We assumed that the secondary metal products produced from the recovered metal fractions fully substitute primary metal products (of equal quality) on the global metal market. Since the global share of recycled compared to total metal production has stagnated in the last decade (while the total metal demand is ever increasing) (IRP, 2019), this assumption can be assumed to be valid. A sensitivity analysis of displaced primary production shows that climate change results are relatively robust, whereas toxicity and abiotic resource depletion results are more dependent on the choice of primary production process (see Figs. S19 and S20 in the *Supplementary Material*).

Long-term emission modelling in LCA is a particular challenge, since the time periods involved significantly exceed existing measurement series. Often in LCA studies, it is argued that no consensus methodology for assessing long-term emissions has been found and hence long-term assessment is omitted (e.g. Allegrini et al., 2015). One of the advantages of applying geochemical models is that they model concentrations over time and therefore allow for any temporal system boundary. Additionally and in contrast to landfill models operating with transfer coefficients that quantify total leaching amounts as a function of pollutant content in the landfill, they allow for a comparison of the modelled metal leachate concentrations with landfill leachate discharge threshold values (Stengele and Moser, 2012). Assessing leachate concentrations in a risk assessment would complement simply adding up emission amounts over the whole time horizon (denoted as 'total emission potential'), as done in LCA. Another way to include future emissions in LCA studies is the application of leaching test data (E. Allegrini et al., 2015; Di Gianfilippo et al., 2016; Weibel, 2020).

The choice of LCIA methods has been made according to general recommendations (Hauschild et al., 2013) and past studies in the waste management field (e.g. Haupt et al., 2018). Due to the specificity of both data and system model, fully aggregated indicators have been omitted for interpretation reasons (ReCiPe 2016 results are listed in the *Supplementary Material*). Regarding resource indicators, Berger et al. (2019) provide an overview over best-practice methods for different problem formulations. The ILCD-recommended method based on rather short term (economic) reserves used in this work has been disputed repeatedly among the scientific community. However, we chose this method as it focusses on metal resources, compared to other methods assessing abiotic resource use (Rørbech et al., 2014).

4.3. Data quality and uncertainty

MSW as well as IBA are highly heterogeneous (both temporally and regionally), which makes it a great challenge to take representative samples (Morf et al., 2013; Skutan and Gloor, 2014). This especially applies to trace elements such as silver and gold. The ten-week measurement campaign by ZAV RE is a great progress in this regard, applying the theory of sampling and therefore enabling highly representative measurement results (Skutan and Haag, 2018). The greatest uncertainties include the amounts of silver and gold (see Table 2). However, assumptions concerning their contents in the different fractions were confirmed by analysis results of NF metal recyclers, which matched relatively well with the estimated silver and gold contents. The contents even tended to be underestimated in our calculations, hence the probability of overestimating HNF metal extraction amounts (and therefore environmental credits) is small. To get more accurate figures on gold, silver and palladium contents, the sample masses for residual IBA should be considerably increased in an upcoming measurement campaign. In addition, the differentiation of the metal state (in metallic state, element content) in the study by Skutan and Haag (2018) enables a much more realistic estimation of the substitution potential, since chemical transformations in the incineration process such as oxidation can be accounted for.

Especially for toxicity, emissions of subsequent recycling steps (and their disposal) may cause measurable environmental impacts, depending on the contamination of the metal fraction. Hence, the estimated net environmental benefit can be considered as *maximum* credit. The simplification mainly concerns the stainless steel fractions, which are at this stage of operation still of rather poor quality. Since the stainless steel fractions do not play an important role in the overall picture, however, it can be assumed that the results are not heavily influenced.

Regarding geochemical modelling, we used specific data of the dry-discharged IBA after metal recovery to approximate its behaviour on a landfill. For the direct landfilling scenario, we assumed the same mineralogy as in the residual IBA. The model assumes complete accessibility of all landfill constituents. This is a reasonable assumption due to the small particle size and large surface area of landfilled material. Directly landfilled IBA can possibly have larger particle sizes, which was not considered in the model. For the chosen time horizons ($\geq$100 a), however, it can be assumed that this simplification does not affect total leachate amounts. Further analyses (leaching tests, solids analyses, batch tests etc.) will result in even better understanding of dry-discharged IBA behaviour in the future, potentially enabling to model dry-discharged IBA behaviour even more accurately. First results show that neglecting adsorption and complexation processes may overestimate short-term zinc and underestimate short-term copper emissions, respectively (Weibel, 2020).

Case-specific uncertainties of landfilling dry-discharged IBA including long-term emissions are difficult to quantify due to changing climatic conditions and the long time horizons involved (Finnveden, 1999). By adding the potential worst-case of complete leaching ($T = \infty$) to our analysis, we therefore include the maximum overall environmental impact as a sensitivity.

5. Conclusions

Based on the study results, we conclude the following: (i) Enhanced metal recovery from (dry) IBA can, in case of high extraction rates and high quality of the separated metal fractions, considerably decrease the environmental impact of MSWI and disposal. (ii) The emission reduction from the landfilled IBA is negligible compared to primary metal substitution when a time horizon of 1′000 years (or lower) is applied. In case of complete leaching, reduced landfill emissions are the main contributor for ecotoxicity, but not for the other impact categories. (iii) HNF metals are associated with especially high substitution benefits. Hence, their recovery is especially beneficial from an environmental point of view. Because HNF metal pieces are often enclosed by other IBA particles, crushing of the IBA down to small particle sizes (<12 mm) is needed to reveal this potential. (iv) To quantify and compare potentials and recovery efficiencies especially of HNF metals such as copper, silver and gold correctly, better measurement data is necessary (e.g. by applying non-biased sampling methods). (v) Material quality of the recovered metal fractions is crucial for quantifying the correct substitution benefits. We could investigate quality aspects for ferrous metals and LNF, however, future research should examine secondary production pathways of the treated metal fractions as well as the role of cross-contamination in the remelting step in greater detail. (vi) Assessing long-term emissions using geochemical modelling (see script in the *Supplementary Material*) can clearly improve transparency, as it is a mechanistic model. However, questions considering the choice of temporal system boundary and weighting of future impacts in relation to present impacts remain. This issue similarly applies to mine tailings for primary metal production.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2020.09.001.

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