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RECOVERY AND UTILISATION OF MUNICIPAL SOLID WASTE INCINERATION BOTTOM ASH: IMPLICATIONS FOR EUROPEAN WASTE MANAGEMENT STRATEGY

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ABSTRACT
Over the last two decades, the stated intent of European waste management strategy has evolved from a specific focus on landfill diversion to enabling the transition to a circular economy. Widespread introduction of source-segregation alongside deployment of material recovery technologies have improved MSW management practices across Europe. However, with diminishing returns it has become more difficult to achieve further landfill diversion through increased recycling alone, and incineration rates (across the EU-27 as a whole) have continued to increase. The advantages of incineration include the ability to harness the energy content of the waste alongside a sizeable reduction in mass and volume. However, the remaining solid residues, the most substantial being incinerator bottom ash, present a management issue. Exploring the role of incineration and the utilisation of incineration bottom ash, this paper highlights the potential risks of lock-in in the context of evolving waste policy. A simple thought experiment suggests that while increased use of incineration may help member states achieve 2035 landfill diversion targets, it would also carry a substantive risk of placing the 2035 recycling target out of reach. To address this, a long-term vision concerning the future of incineration is required, where it is recommended that policy which focuses on landfill diversion and the recycling of residual wastes should be strengthened through mechanisms that gradually phase out incineration and distinguish between open and closed-loop recycling.

1. INTRODUCTION

As global population and affluence have increased, so has the consumption of goods and services. Although this has improved quality of life for current generations, it is unsustainable; contributing to environmental degradation and associated complex challenges such as resource depletion, climate change, and geopolitical tension (Clark, 2007; Moreno et al., 2016). This has been recognised within the United Nations Sustainable Development Goals, where Goal 12 is to ensure sustainable consumption and production (UN General Assembly, 2015). To address unsustainable consumption, replacing the linear 'take-make-dispose' economic model with a Circular Economy (CE) has been encouraged. In standardising environmental policy across Member States (MS), the European Union (EU) acts as a driving force to improve international standards (Wysokinska, 2016) and can be viewed as being at the forefront of the transition to a CE, having published the Circular Economy Package (CEP) in 2015 (EC, 2015a) and the Circular Economy Action Plan (CEAP) in 2020 (EC, 2020).

To achieve a CE, resource efficiency is promoted through optimisation of production systems, maintenance of resource utility, and promotion of reuse, recycling, and recovery, thereby minimising (and ultimately eliminating) landfilling of waste (Kirchherr et al., 2017). Progressive waste management thus has an integral role to play in the CE transition (Johansson et al., 2020), where one of the most complex to manage waste streams is Municipal Solid Waste (MSW). While MSW constitutes only 10% of total EU waste arisings, it has a high political profile due to its link to consumption patterns and resulting complex composition, where its management is considered an excellent indicator of the quality and efficiency of a MS's waste management strategy (EC, 2015b; Eurostat, 2021).

To date, the northern high-income MSs have been most successful in improving MSW management practices, where the last two decades have seen the accomplishment of "easy gains" (Mihai and Apostol, 2012). For

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example, the widespread introduction of source-segregation alongside deployment of technological approaches (e.g., material recovery facilities and mechanical biological treatment plants) has delivered a substantial increase in recycling and composting (Cook et al., 2015; Eurostat, 2023; Vountatsos et al., 2016). However, with diminishing returns it becomes more difficult to achieve further landfill diversion through increased recycling, where incineration has increasingly been employed to achieve landfill diversion targets (Eurostat, 2021, 2023). The advantages of incineration include the ability to harness the energy content of the waste alongside a sizeable reduction in mass and volume. However, while the mass of waste is typically reduced by ca. 80%, there remain a number of solid residues, the most substantial being incinerator bottom ash (MSW-IBA).

MSW-IBA is a granular, agglomerated material, that typically comprises a heterogeneous mix of brick, concrete, silicate-phase glass, unburnt organics, clinker and metal fragments (Bourtsalas et al., 2015; Chiang et al., 2012). The presence and concentration of elements reflects waste inputs and is dependent on combustion unit type, where the most common elements are calcium, silicon, aluminium, iron, sodium and manganese, and heavy metals such as antimony, arsenic, barium and beryllium may be present (Margallo et al., 2015). For a detailed physico-chemical analysis of MSW-IBA, see Dou et al. (2017).

Historically, MSW-IBA was landfilled; a sub-optimal solution in terms of resource conservation and environmental safety, and one subject to increasing economic costs and limited by capacity constraints (Chen and Lo, 2015). Current management strategies therefore look to realise the recovery potential for resources such as metals and aggregates (Allegrini et al., 2015; Costa et al, 2020). It is now common practice for ferrous and non-ferrous metals to be recovered through magnetic and eddy-current separation (Allegrini et al., 2014, 2015; Costa et al, 2020). Typically accounting for up to 20% by weight (ferrous 5-15%, non-ferrous 1-5%; Šyc et al, 2020), metal fragments can differ in size and quality, which in turn affects recycling efficiencies (Allegrini et al., 2014, 2015). While recovery of ferrous metals is generally around 80%, for non-ferrous metals recovery can be as low as 30% (Allegrini et al., 2014; Boesch et al., 2014), although advanced separation techniques can increase this to 70% (Biganzoli et al., 2013; Grosso et al., 2011).

The removal of metal fragments increases the quality of MSW-IBA for utilisation as an aggregate, where sieving is used to produce size separated materials with good geotechnical characteristics (Karagiannidis et al., 2013; Šyc et al, 2020). However, chemical and mineralogical characteristics, particularly alkalinity, can result in instability and leaching, where further processing is then required (Dou et al., 2017; Lancellotti et al., 2013). Stabilisation is often achieved through weathering or natural aging; the exposure of an open stockpile to the atmosphere to promote carbonation, resulting in the precipitation of minerals such as calcite and a reduction in pH (Chimenos et al., 2000; Yao et al., 2010). Although it can take up to three months to complete carbonation (such that the leaching potential is minimal), the use of a carbon dioxide enriched atmosphere has the potential to reduce this to two weeks (Margallo et al., 2015).

The most common use of MSW-IBA is as an aggregate, where similar physico-chemical characteristics to natural aggregates allow treated MSW-IBA to be employed as a partial substitute in construction applications (Ahmed and Khalid, 2011). For example, MSW-IBA can replace without detrimental effect; up to 20% of natural aggregate as a subbase in road construction (Birgisdóttir et al., 2006), up to 25% of clinker used in cement production (Margallo et al., 2014), and up to 15% of cement in low-strength concrete production (Jurič et al., 2006).

In addition to generating income from product sales, using MSW-IBA as a secondary aggregate has two further advantages; reduction of waste landfilled and substitution of natural resources (Margallo et al., 2015; Blasenbauer et al. 2020). Diverting significant volumes of MSW-IBA from landfill reduces the economic and environmental costs of disposal (Birgisdóttir et al., 2006; Olsson et al., 2006). Likewise, substituting raw materials with MSW-IBA avoids the energy use and other environmental costs associated with extraction and processing, and also contributes to mineral stock protection and conservation (Olsson et al., 2006). Indeed, the use of MSW-IBA as an aggregate may be particularly attractive given increasing demand for construction materials and declining availability of natural aggregates (Abbà et al., 2014). However, a recent estimate indicates that full utilisation of MSW-IBA would displace <1% of primary aggregate demand in the EU, suggesting the main benefit is reduction in required landfill capacity (Blasenbauer et al., 2020).

A number of alternative processing and application options explored in the literature are summarised in Table 1. These include the potential for recovering critical raw materials, and potential use as a growth substrate, in construction related products, in hydrogen gas production, and as a purification agent. However, as these do not yet represent substantial utilisation pathways they are not discussed further here.

While the generation, treatment, and management of MSW-IBA has been extensively discussed (see Margallo et al., 2015 and references therein), the production and utilisation of MSW-IBA as a secondary material in the context of evolving EU policy and practice warrants further exploration.

This policy position paper explores the implications and potential consequences of evolving policy for future waste management within the EU, focusing on the use of incineration and utilisation of MSW-IBA in the context of increasingly stringent targets. To provide context, a review of policy documents and academic literature has been used to understand the evolving situation regarding EU waste management strategy, with a specific focus on landfill diversion and material recycling targets. We then examine the different routes to utilisation of MSW-IBA in the EU, before exploring the possible consequences of a continued reliance on incineration for achieving waste management targets under different MSW-IBA utilisation scenarios. Based on this analysis, we then make policy recommendations for achieving targets and avoiding lock-in in the transition to a CE.

TABLE 1: Alternative uses for MSW-IBA reported within the literature.

References	Use	Study details	Conclusion/Limitations
Material Recovery (Urban Mining)			
	Recovery of REE and CRM		
Allegrini et al. 2014		Detailed MFA (incl. resource recovery potential) of IBA taken from Danish recovery facility.	Conc. of REEs detected in IBA significantly lower than ore. Lack of enrichment options limits recovery of REE's from IBA.
Funari et al. 2016		Used ICP-MS to determine REE concentration in IBA following digestion (novel method).	REE conc. indicate prospective low streams. Several methods identified to facilitate urban mining (from IBA)
Funari et al. 2015		Used XRF/ICP-MS to determine elemental composition and CRM conc. of untreated IBA	Considered a low concentration stream for precious/high-tech metals. Concentration of Mg, Cu, Sb & Zn similar to low-grade ore.
Growth Substrate			
	Green / Brown roofs		
Bates et al. 2015		Six-year experiment, testing effects of recycled aggregate type (including IBA) on the development of vegetation on brown roofs.	IBA is not recommended as a brown (biodiversity) roof growth substrate due to limited capacity to hold moisture but could be used in Sedum green roofs.
Pyroxene ceramics			
	Porcelainized stoneware		
Barbieri et al. 2002		Glassy frits obtained from MSW-IBA compared against glass cullet as sintering promoters in production.	Glassy frits improved water absorption and spot resistance but did not significantly change bending strength.
Schabbach et al. 2012		Replaced feldspar &quartz with IBA (post treatment), characteristics and leaching potential determined.	Mechanical characteristics comparable to commercial products, ISO classification achievable and additional benefits noted.
Verbinnen et al. 2017		Discusses the use of IBA to produce ceramic materials such as tiles and stoneware.	Amorphous matrix reduces leaching. Ceramics using 5-10% IBA, technical properties not influenced, lower firing temp.
Alkali Activated Cements			
	Hybrid cements		
Garcia-Lodeiro et al. 2016		Compared cement mixes (hybrid, Portland, commercial) with respect to leaching potential, mechanical strength, and reactivity.	Alkali activation of hybrid cement lowered leaching potential. Raised concentration of chloride ions in hybrid cement not suitable in manufacture of structural concrete.
Verbinnen et al. 2017		Reports on several studies which replace varying proportions of Portland cement with IBA for use in structured materials.	Advises that replacing between 5-10% of Portland cement has no influence on the structural characteristics. Mixtures made with 40+% IBA, detrimental to concrete strength.
Chen et al. 2020		Assessed the use of MSW IBA as an alkali-activated material as a promising alternative to Portland cement.	Thermal treatment of the IBA (up to 1000°C), eliminated the detrimental effects of metallic AL/Zn and increased crystallinity. Suitable for use as a fine aggregate.
Matsumoto & Takaoka, 2022		Compared five advanced chloride removal methods; addition of Na_2CO_3, addition of Na_2SO_4, accelerated carbonation, aging and acid washing against washing only.	Found presence of Friedel's salt can limit success of washing. Aging and acid washing found to improve utilisation in cement. Concludes that optimal recycling should consider environmental impacts and costs.
	Geopolymers		
Lancellotti et al. 2013		Partial substitution of metakaolin within geopolymers, with chemical, elemental and LOI analysis.	IBA has been demonstrated as suitable source materials for producing metakaolin-blended geopolymers.
Lancellotti et al. 2015		IBA is used as sole source material for geopolymers cured for different lengths of time.	Geo-polymeric networks produced without need for metakaolin. Metallic content may lead to a porous morphology.
Ji & Pei, 2019		Investigates the use of IBA as a raw input for geopolymers, particularly the generation of hydraulic binders with water.	When mixed with DWTR, samples exhibited higher compressive strength than IBA only samples. A ratio of 80% IBA: 20% DWTR was recommended.
Aeration agent			
	Aerated concrete		
Song et al. 2015		Aluminium and silica from IBA used as aerating agent in production of AAC.	Synthesized IBA-AACs had a higher density, compressive strength and shrinkage when compared against standard.
Li et al. 2018		Assessed the feasibility of using IBA as a substitute for quartz sand in the preparation of AAC.	Demonstrated that IBA-ACC had reduced gas-foaming time, compressive strength, density, and thermal conductivity.
Hydrogen gas production			
	Use of Aluminium species to generate Hydrogen		
Saffarzadeh et al. 2016		Identification and characterisation of metallic AL / AL-alloys found in IBA and assessed potential to aid the generation of H gas.	Production of H gas ranged between 8.4 and 38.3 l/kg of dry ash, aided by presence of metallic-AL. Inherent alkalinity noted as key parameter in H gas generating reactions.
Biganzoli et al. 2013		Evaluated the recovery and utilisation of metallic AL, through metal recovery and to generate H gas as a clean fuel.	Successful H gas production, performing better, in terms of overall energy balance, than metal recovery. Economic investment requirements were found to be unjustifiable.
Purification agent			
	Landfill gas purification before energetic valorisation		
Ducom et al. 2009		Pilot plant study assessed qualities of IBA to remove H_2S, CH_4S and C_2H_6S from landfill gas.	IBA successful in sequestering H_2S and CH_4S through acid-base reactions, C_2H_6S retained by physical adsorption.
	Mesoporous silica materials		
Liu et al. 2014		Mesoporous silica materials, synthesised from IBA, evaluated in the removal of heavy metals from aqueous solutions.	Mesoporous silica materials were successfully synthesized and shown to have potential as adsorbents for the removal of heavy metals from aqueous solutions.

Abbreviations: Rare Earth Elements (REE); Critical Raw Materials (CRM); Material Flow Analysis (MFA); Inductively Coupled Plasma - Mass Spectrometry (ICP-MS); X-Ray Fluorescence (XRF); International Standards Organisation (ISO); Loss On Ignition (LOI); Drinking Water Treatment Residue (DWTR), autoclaved aerated concrete (AAC).

2. EVOLUTION OF EUROPEAN WASTE STRATEGY

Two EU directives that have driven significant changes in MSW management (by setting legally binding performance targets) are the Landfill Directive (LD; 1999/31/EC; EC, 1999) and the Waste Framework Directive (WFD; 2008/98/EC; EC, 2008). Both directives were amended by the CEP, with further targeted revision of the WFD (in line with the CEAP) expected in 2023 (EC, 2022a).

During development of the CEP, trilogue discussions between the European Commission (EC), Parliament and Council considered a number of proposed amendments (Figure 1). The final version introduced a ban on the landfilling of separately collected wastes, a maximum MSW landfill target of 10% and a recycling target of 65% by 2035 (EC, 2015a). However, with the compromises reached during trilogue, two key areas of missed opportunity can be identified, neither of which have been addressed in the CEAP.

First, the waste hierarchy itself has not been revised, where the lack of nuance could have implications in the CE transition (Gharfalkar et al., 2015). Specifically, no distinction is made between open-loop recycling (where often the value of the resource decreases i.e. down-cycling and only one extra lifecycle is achieved) and closed-loop recycling (where value is maintained i.e. re-cycling, or increased i.e. up-cycling, and several lifecycles can be achieved). As such, strategies contributing to targets do not need to consider value maintenance or the number of lifecycles achieved (Bartl, 2014; Gharfalkar et al., 2015).

Second, despite the EC's recognition that increased incineration capacity may jeopardise recycling, no limits (absolute or relative) were introduced. While incineration has a valid role to play in the treatment of other waste streams,

		Disposal	Incineration	Recycle
Existing targets		**When compared to 1995, share of BMW landfilled shall not exceed; 75% by 2006 50% by 2009 35% by 2016**	-	**By 2015, separate collection shall be set up for at least; paper, metal, plastic or glass.** **By 2020, 50% of (at least) paper, metal, plastic and glass from MSW shall be prepared for re-use or recycled.**
Trilogue discussions	Initial proposals by the European Commission.	•Ban on BMW to landfill after separate collection •By 2030, share of MSW to landfill should not exceed 10%.	•Recognise recycling of metals from incineration. •Charges may be established by member states as disincentive.	•Separate collection for paper, metal, plastic and glass. •By 2025, $60\%_{MSW}$ shall be prepared for re-use / recycling •Increasing to $65\%_{MSW}$ by 2030
Trilogue discussions	Response from the European Parliament.	•Ban on BMW to landfill after separate collection •By 2030, share of MSW to landfill should not exceed 5%.	•Quality criteria needed to recycled metal post incineration. •Introduce (or increase) taxes/fees •Ban incineration of separately collected waste. •Place limit on incineration of non-recyclable wastes	•Separate collection for paper, metal, plastic, bio-waste, glass and textiles. •By 2025, $60\%_{MSW}$ shall be prepared for re-use / recycling (with min. 3% total MSW prepared for re-use) •Increasing to $70\%_{MSW}$ by 2030 (with min. 5% total MSW prepared for re-use).
Trilogue discussions	Response from the European council	•Ban on BMW to landfill after separate collection •By 2030, share of MSW to landfill should be 10% or less. AND all wastes suitable for recovery/recycling should not be landfill unless delivers BEO	•Recognise recycling of metals from incineration. •Charges may be established by member states as disincentive.	•Separate collection for paper, metal, plastic and glass. •By 2025, $55\%_{MSW}$ shall be prepared for re-use / recycling •Increasing to $60\%_{MSW}$ by 2030
Adopted CEP targets		**Ban on BMW to landfill that has been separately collected.** **By 2035, share of all MSW landfilled should be reduced to 10% or less.**	**Recognise recycling of metals from incineration.** **Incineration charges may be established by member states as a financial disincentive.** **Separately collected waste should not be incinerated unless pre-teated and delivers BEO.**	**Separate collection for paper, metal, plastic, bio-waste, glass and textiles.** **By 2025, $55\%_{MSW}$ shall be prepared for re-use or recycling.** **This shall increase to; $60\%_{MSW}$ by 2030. $65\%_{MSW}$ by 2035.**

FIGURE 1: Evolution of Circular Economy Package targets during trilogue discussions, with significant differences underlined (EC, 2008, 2015b-c; CEU, 2017a-b).

such as the decontamination of hospital waste (Gielar and Helios-Rybicka, 2013) and the extraction of phosphorus from sewage sludge (Kleemann et al., 2017), the incineration of MSW has negative implications for the CE. Where the CE seeks to maintain and recirculate materials and resources, incineration destroys them (albeit with energy recovery), diverting materials with high calorific value away from recycling pathways. In particular, the management of plastic wastes is largely realised through energy recovery, where reasons for this include complex material composition, inadequate source separation, a lack of automated sorting equipment, and the low cost of waste plastics relative to fossil fuels (Schneider and Ragossnig, 2015). The failure to place limits on incineration thus undermines implementation of the waste hierarchy (Malinauskaite et al., 2017) and incurs the risk of lock-in, potentially stifling the emergence of more sustainable alternatives (Corvellec et al. 2013; Svingstedt & Corvellec, 2018). Indeed, given capital costs up to €180 million and operating contracts exceeding 25 years (Nixon et al., 2013), expansion of incineration infrastructure risks both technological and contractual lock-in, where municipal authorities may be tied-in to supply contracted quantities of waste over decades, irrespective of changes in waste composition, volumes, and policy (Schneider and Ragossnig, 2015; Svingstedt & Corvellec, 2018). Despite these risks being highlighted during CEP trilogue, they are only somewhat obliquely addressed within the final text, with advice to consider the risk of "stranded assets" in investment decisions (highlighting the need to consider feedstock availability over the lifespan of new installations without neglecting separate collection and recycling obligations), while MSs with higher ambition may elect to introduce incineration charges and limits at a national level (EC, 2017). Likewise, the approach taken in the CEAP is to reduce residual waste generation (non-recycled, i.e., landfill + incineration with or without energy recovery) through promoting waste prevention and separation for recycling (rather than to place explicit limits on incineration), encouraging the wider introduction of economic instruments such as landfill and incineration taxes as a mechanism to achieve this (EC, 2020).

While the headline target within the CEAP is to reduce residual waste by half by 2030, this is an EU-wide non-binding commitment. Furthermore, in addition to a lack of incineration targets, as yet there are no specific waste prevention targets. Rather, the EC has placed an obligation on MSs to establish Waste Prevention Plans (WPP). However, where WPPs have been established by progressive MSs, they tend to be reliant on qualitative initiatives, and thus may be less effective (Johansson & Corvellec, 2018). In practice, this means the only well-defined and legally binding target-based drivers are the landfill diversion and recycling targets.

In addition to revising targets, the CEP, and to a greater extent the CEAP, did address broader aspects of consumption and production. Of particular relevance to waste management was the acknowledgement that continued uncertainty regarding secondary materials had restricted their use, thereby limiting resource recovery and landfill diversion (EC, 2015b, 2016). For example, the use of secondary aggregates in road construction has been hindered by perceived performance concerns and additional costs (Huang et al., 2007). In light of this, the CEP and CEAP addressed the further development of secondary materials markets and the strengthening of quality standards such as End of Waste (EoW) criteria (Bartl, 2015, 2020; EC, 2015b, 2020).

3. GENERATION AND UTILIZATION OF MSW-IBA IN LIGHT OF THE EU POLICY

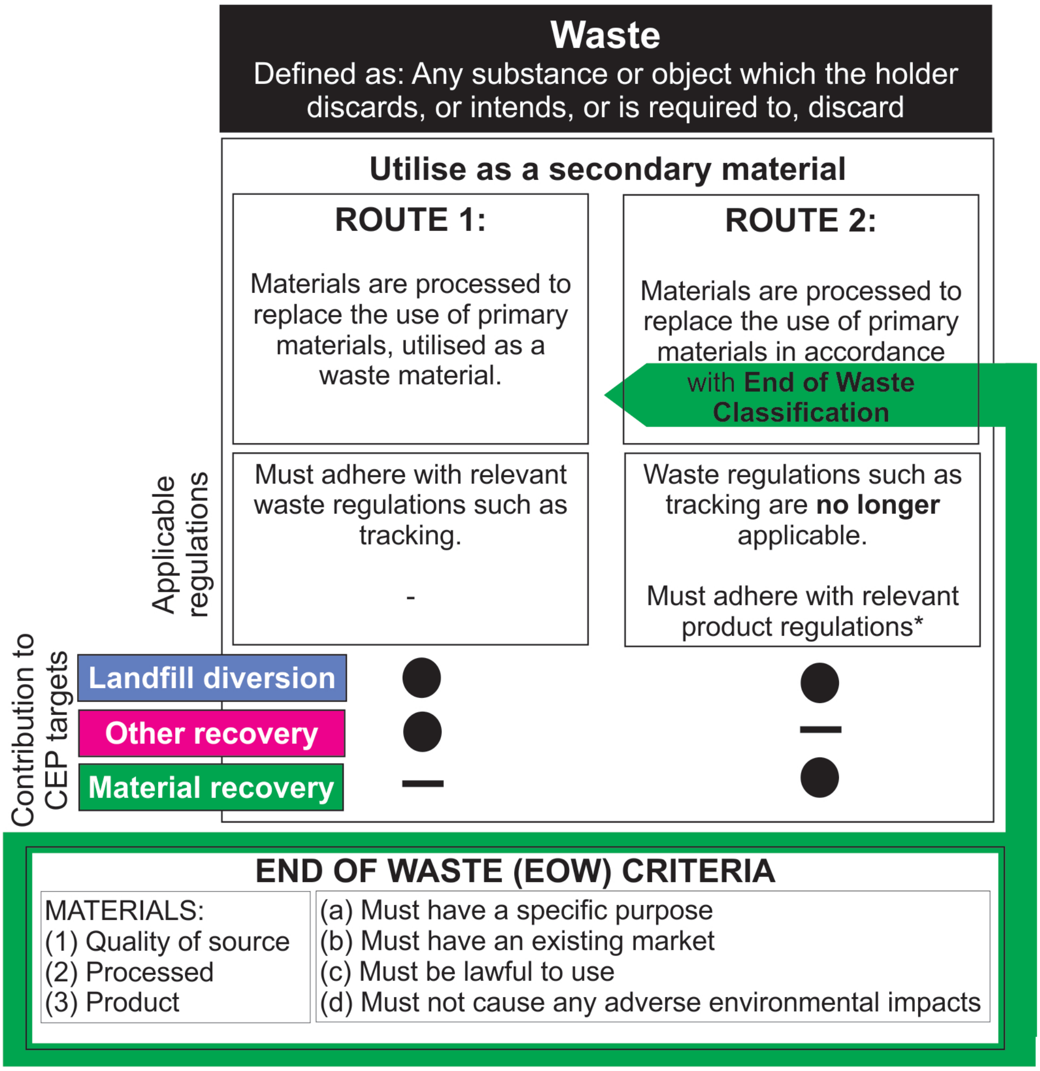
3.1 Recovery and utilisation of MSW-IBA as a secondary material in the EU

Within the EU, MSW-IBA may be utilised via two routes (as a waste or non-waste) with differing implications for landfill diversion and recycling targets (Figure 2).

Under Route 1, secondary materials maintain the status of a waste. As such, the transport, utilisation, and continued monitoring of MSW-IBA must comply with relevant waste legislation, be shown to have no adverse environmental effects, and adhere to restrictions and pre-treatment conditions prompted by national legislation (Kuo et al., 2013; Lancellotti et al., 2013; Van Gerven et al., 2005; van der Sloot et al., 2001). Utilisation through Route 1 contributes to landfill diversion but does not contribute to recycling, instead aligning with the definition of 'other recovery' (EC, 2008).

Alternatively, secondary materials can be utilised via Route 2, where EoW seeks to address known barriers to the development of secondary material markets. Specifically, the common perception that recovered materials are of lower quality than primary materials and the restricted ability to transport materials across national boundaries due to a lack of harmonisation in waste definitions between MSs (Delgado et al., 2009). Successful application of EoW criteria would classify the material as a 'non-waste', removing the need to apply waste regulations. Instead, the secondary material is treated in the same fashion as primary materials, being subject to product regulations, import / export regulations (with free trade within the EU internal market), and where appropriate, regulations concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) (EC, 2006). Achieving non-waste status allows the material to be counted towards both landfill diversion and recycling targets (EC, 2008).

EoW status can be defined at different stages of material recovery depending on the quality of the waste stream and the extent of processing required. Firstly, EoW can be defined for high quality waste materials that require minimal processing, where to date the EC have laid down EU-wide criteria for iron, steel, aluminium, and copper scrap, and glass cullet. However, for lower quality materials such as MSW-IBA, achieving EoW will require either processing to meet quality levels equivalent to that of primary materials, or being processed into a recognisable and marketable product. In all cases, the material / product must also adhere to the four qualifying criteria for EoW (Figure 2). A recently completed scoping assessment (carried out under the CEAP) has identified plastics and textiles as priorities



** Product regulations include those associated with consumer rules, the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH), Import / Export rules and trade (European Single Market).*

FIGURE 2: Routes to utilisation of recovered wastes as secondary materials and implications for achievement of landfill diversion and material recovery targets (where • is positive, – is negative).

for the development of further EU-wide EoW criteria (EC, 2022a). Beyond this, Article 6 (paragraph 2) of the WFD places the onus on MSs to develop national EoW criteria, where these do not automatically apply across the EU.

Within the EU, MSW-IBA utilisation via Route 1 has become commonplace within construction, with use in cement production, as sub-base in road construction, in other civil engineering projects, and as landfill cover (Table 2). The extent of utilisation is primarily influenced by incentives which encourage use in lieu of disposal (e.g., landfill taxes) in combination with market conditions which dictate the quantities and quality of MSW-IBA required (Villanueva et al, 2006; WRAP, 2006).

Despite this widespread use, no EoW criteria have been established to date (van Zomeren and Velzeboer, 2017). While Denmark has considered developing EoW for MSW-IBA, it was concluded that it would be inappropriate in unbound applications (Villanueva et al., 2006). Specific concerns related to traceability, where removal of waste tracking and monitoring requirements has the potential to undermine environmental protection (e.g., risks to groundwater from leaching of MSW-IBA at an unrecorded site with no monitoring) (Villanueva et al., 2006). While it was acknowledged that EoW status could ease administrative and export burdens, it was also highlighted that MSW-IBA has low financial value and tends to be used locally, thus unconstrained export is not necessarily required (Villanueva et al, 2006). Indeed, Denmark uses incineration to treat a large proportion of MSW (between a half and two-thirds; Eurostat 2023a) and achieves high MSW-IBA utilisation rates (Table 2) without the use of EoW criteria.

3.2 EU incineration trends, MSW-IBA production and utilisation rates, and implications for targets

Examination of trends in MSW treatment within the EU clearly shows the impact of the LD (EC, 1999) and WFD (EC, 2008), where a combination of increased material recycling, composting and anaerobic digestion (which col-

TABLE 2: IBA production and utilisation rates in EU countries with MSW incineration plants. For IBA produced, A is calculated from Blasenbauer et al. (2020) and B from CEWEP Country Reports (CEWEP, 2021). The off-landfill utilisation rates are from Blasenbauer et al. (2020). The total (on & off landfill) utilisation rates and method of disposal or utilisation are from CEWEP Country Reports. The total values are calculated from the available data weighted according to the mass of waste incinerated or IBA produced as appropriate.

Country	IBA produced (wt% treated waste)		Mineral IBA utilisation rate (wt% IBA)		Method of Mineral IBA disposal or utilisation	CEWEP report year
	A	B	off landfill	total		
Austria	20.4%	21.5%	0%	0%	100% landfill	2018
Belgium	14.2%	18.9%	69%	-	Secondary building material	2016
Czechia	30.8%	23.9%	0%	0%	100% landfill	2016
Denmark	16.2%	17.6%	99%	99%	Recycled (road construction, harbours etc.)	2010
Estonia	23.2%		0%			
Finland	18.8%	16.8%	20%	100%	Recycled/recovered – mainly in construction but also in asphalt production and construction block	2018
France	19.7%	20.8%	80%	80%	80% recovery (e.g. road construction); 17% landfill; 3% other	2010
Germany	24.2%	27.0%	30%	-	Road construction, noise barriers & other technical applications, recovery on landfill (ways, shaping)	2018
Hungary	28.6%	21.8%	0%	0%	100% landfill	2018
Ireland	17.5%	15.7%	0%	100%	100% recovery on landfill (cover & engineering material)	2018
Italy	16.9%	17.8%	85%	71%	71% recovery; 29% landfill	2012-13
Lithuania	26.8%		0%			
Luxembourg	16.5%	16.8%	0%	-	Road construction	2018
Netherlands	25.0%	22.7%	100%	100%	40% road construction; 36% noise barriers; 13% landfill construction; 11% other (e.g. bound in products)	2012-13
Poland	21.6%	25.0%	60%	-	Block fabrication; landfill	2010-11
Portugal	16.9%	19.6%	56%	60%	Road construction, landfill cover & backfilling	2018
Slovakia	21.4%		0%			
Spain	18.3%	16.8%	58%	-	Landfill use (ridge, regularization, etc), road construction, cement production	2010-11
Sweden	18.3%	16.3%	0%	100%	100% recovery as landfill construction material	2018
Minimum	**14.2%**	**15.7%**	**0%**	**0%**		
Maximum	**30.8%**	**27.0%**	**100%**	**100%**		
Total	**20.8%**	**22.0%**	**53%**	**79%**		

lectively contribute to recycling targets), and incineration (with or without energy recovery), has reduced the amount of waste landfilled by 55% between 1995 and 2021 (Figure 3; Eurostat, 2023a). Focusing on incineration, while rates have been relatively stable over the last decade in some MSs (e.g. France, Italy), and declined in others (e.g. Germany, the Netherlands), they have increased in the majority of MSs, where incineration capacity in the EU-27 increased by 39% between 2010 and 2020 (from 126Mt/yr to 199 Mt/yr; Eurostat, 2023b), while the amount of incinerated waste increased by 16% over the same period (from 53 Mt/yr to 62 Mt/yr; Eurostat, 2023a). While this expansion in incineration has helped to drive landfill diversion, if it were to continue apace it could place the 2035 landfill diversion (<10% MSW) and recycling (65% MSW) targets at risk.

Here we take a closer look at the potential consequences of continued growth in incineration across the EU. While there is a high degree of variability between MSs, both in terms of landfill diversion and the extent to which incineration is utilised, examination of the data presented in Figure 4 allows four ballpark estimates of the incineration rate (INC) required to meet the 2035 landfill diversion target under current EU waste management practices.

- For all landfill rates (LF) less than $10\%_{MSW}$ (corresponding to data spanning 1999-2021 from ten MSs), the median LF was $1.5\%_{MSW}$, and median INC was $44.9\%_{MSW}$.
- Focusing on the most recent data, the eight MSs individually achieving a LF less than $10\%_{MSW}$ in 2021, have a weighted mean LF of $0.5\%_{MSW}$ and weighted mean INC of $37.6\%_{MSW}$.
- Linear regression of the 2015-2019 data and 2020-21 data indicates a LF of $10\%_{MSW}$ corresponds to an INC of $41.7\text{-}44.0\%_{MSW}$.
- Looking across the 13 best performing (lowest LF) MSs in 2019 (the most recent year for which data from all 27 MSs is available), a collective LF of $9.9\%_{MSW}$ ($<10\%_{MSW}$) corresponds to an INC of $33.3\%_{MSW}$. This compares to an overall 2019 LF of $24.3\%_{MSW}$ and INC of $27.0\%_{MSW}$.

Thus, if the current trend of increasing incineration to achieve landfill diversion were to continue, it seems reasonable to assume (for a first order estimate) that achiev-

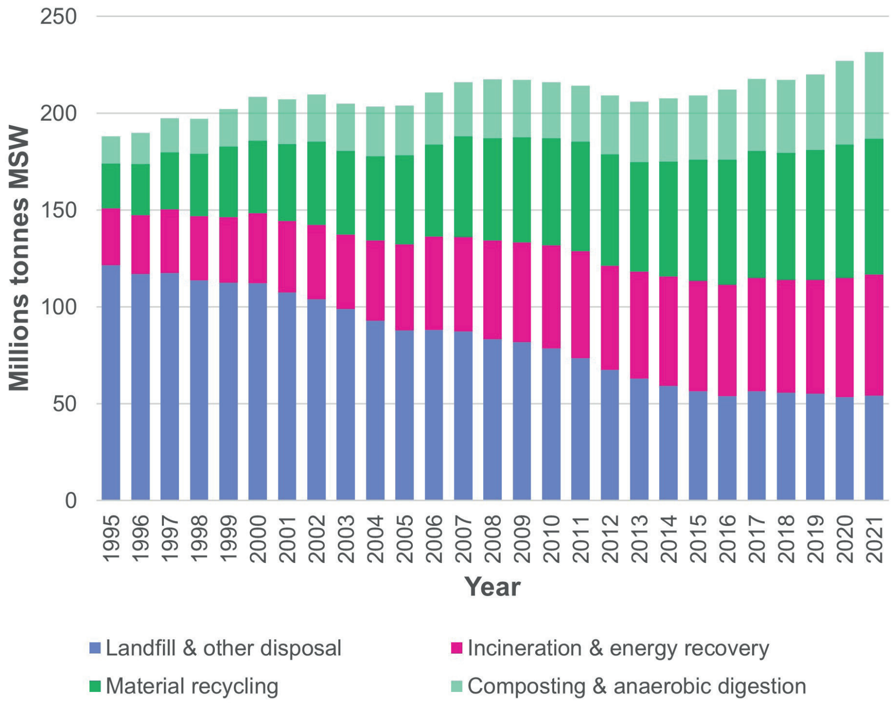


FIGURE 3: Treatment of MSW in the EU-27 from 1995 to 2021 (data from Eurostat, 2023a).

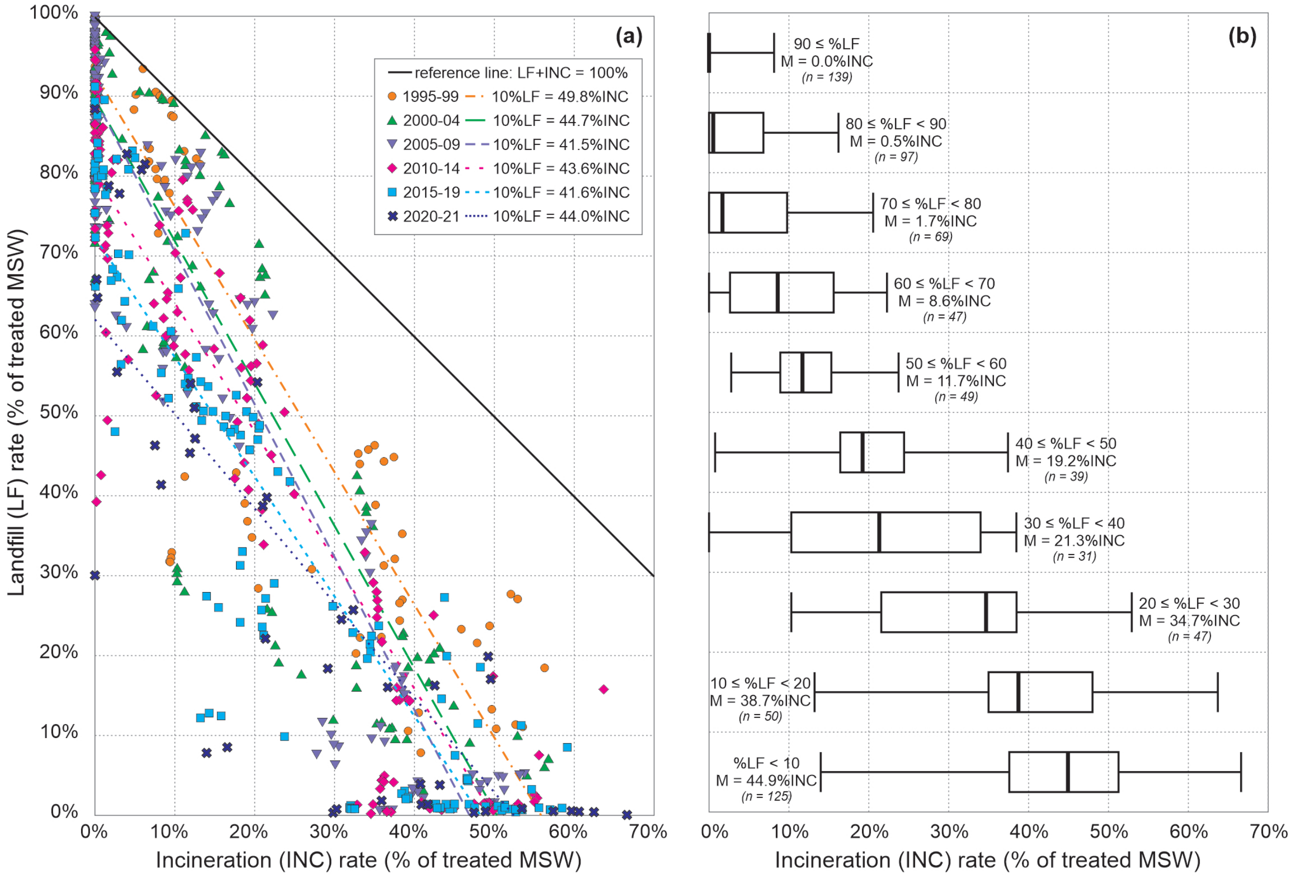


FIGURE 4: Incineration (INC) rate relative to landfill (LF) rate for all EU Member States 1995-2021, expressed as a percentage of treated MSW (data from Eurostat, 2023a). (a) Scatter plot with linear regression of quinquennial (R2 = 0.82 to 0.88) and 2020-21 (R2 = 0.69) data. The distance between the data and the LF+INC = 100% reference line reflects the implementation of other waste management strategies (recycling, composting, anaerobic digestion) that act to decrease LF. (b) Box and whisker plot of INC rate by LF rate decile groups, where M = median INC rate.groups, where M = median INC rate.

ing a LF of ca. 1-10%$_{MSW}$ (5.5±4.5%$_{MSW}$) would correspond to an increase in INC from ca. 27.0%$_{MSW}$ to ca. 33.3-44.9%$_{MSW}$ (39.1±5.8%$_{MSW}$). The consequences of such an increase for landfill diversion and recycling rates are illustrated in Figure 5, where the calculation of the material flows are explained briefly below.

Based on data presented in Table 2, the mass of IBA produced ranges from 14.2% to 30.8% of the mass of incinerated waste (22.5±8.3%$_{INC}$). While metal recovery from MSW-IBA is now relatively common practice, the extent to which it has been implemented across the EU is unclear due to incomplete reporting. Nonetheless, the available data (Table 3) indicates overall recovery rates consistent with the literature (see Section 2.1). Using a gross extraction of 8.2% of the mass of MSW-IBA and a metal fraction of 0.75±0.04 gives a metal recovery of 6.2±0.3%$_{IBA}$. Combining this with the MSW-IBA production rate and expressing it relative to the mass of incinerated waste gives a metal recovery of 1.4±0.5%$_{INC}$ and a residual (mineral) MSW-IBA production rate of 21.1±8.3%$_{INC}$. If an INC of 39.1±5.8%$_{MSW}$ is then applied, the total mass of MSW-IBA produced is 8.8±3.5%$_{MSW}$, with a mineral MSW-IBA production of 8.3±3.5%$_{MSW}$, and a metal recovery of 0.5±0.2%$_{MSW}$. If it assumed that all MSW that is not landfilled or incinerated enters recycling pathways (100-LF-INC = 55.4±7.3%$_{MSW}$), this gives a total recycling rate of 56.8±7.4%$_{MSW}$.

We now consider the implications of the above flows for meeting the landfill and recycling targets under different MSW-IBA utilisation scenarios (Figure 6).

If we consider current practice (Table 2), MSW-IBA is typically landfilled, utilised on landfill, or utilised as a waste in off-landfill construction. If all mineral MSW-IBA were landfilled, MSs would be at significant risk of exceeding their 2035 landfill allowance (Scenario 1). The risk is reduced if utilisation rates (on and off landfill) comparable to today are assumed (Scenario 2a) and is eliminated with full utilisation (Scenario 2b). However, these utilisation pathways qualify as 'other recovery' and do not contribute to recycling targets. As such, incinerating such a large proportion of MSW and maintaining current MSW-IBA management practices would put the 2030 recycling target at risk, and the 2035 target out of reach. Indeed, for both the 2035 landfill diversion and recycling targets to be achievable, a significant fraction (if not all) of the MSW-IBA would need to be utilised off-landfill via material recovery pathways (Scenarios 3a and b). Clearly, maximising metal recovery should be prioritised (given the economic value of metals and that this is a closed-loop recycling pathway), however, the potential gains are relatively small (doubling metal recovery increases the overall recycling rate by 0.5%$_{MSW}$). Conversely, if EoW were achieved for mineral MSW-IBA, then the risk of failing to meet the recycling target would

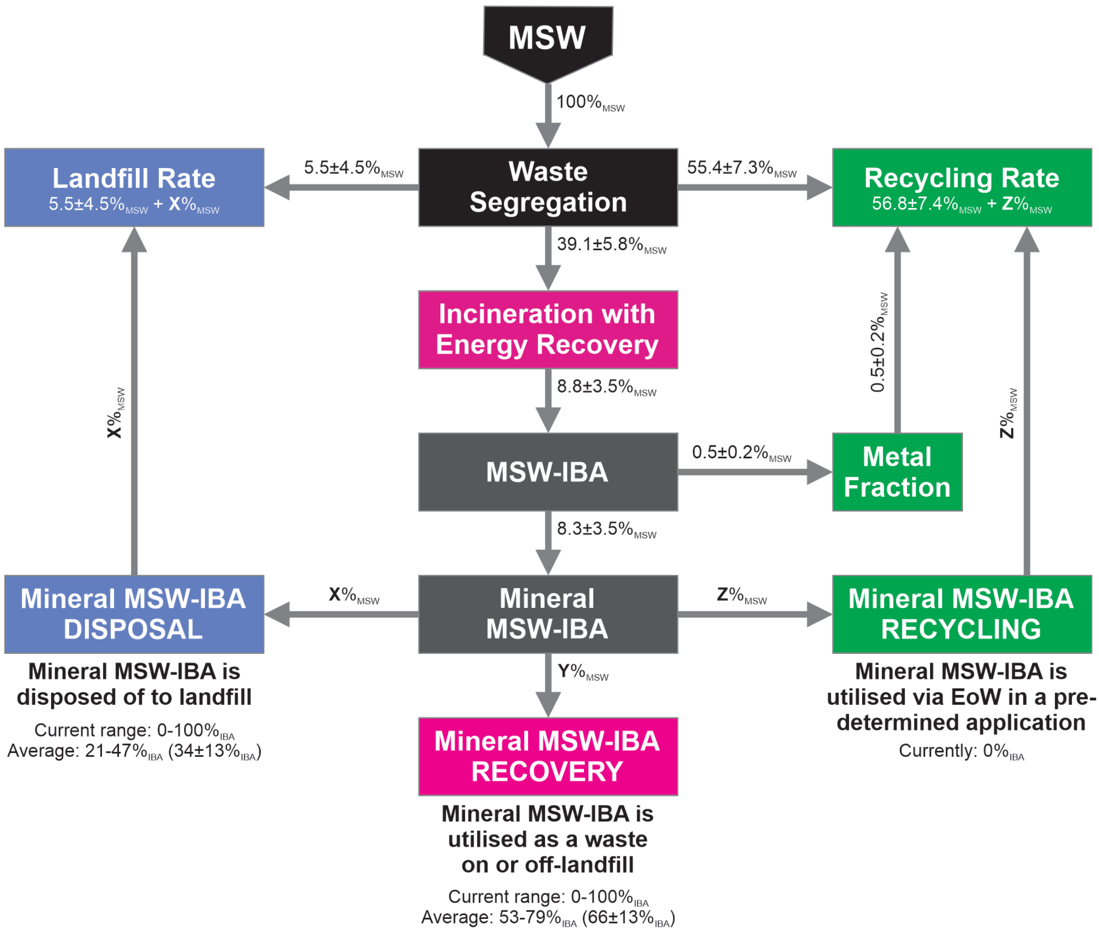


FIGURE 5: Calculation of materials flows based on the current rates of incineration.

TABLE 3: Metal recovery rates for EU countries based on available CEWEP Country Reports (CEWEP, 2021). n.d. Country Report states no data; n.r. not reported. Total gross material extracted ($\%_{IBA}$) calculated across all available data, weighted by mass of IBA produced. Total metal fraction (f_{met}) calculated across all available data, weighted by mass of gross material extracted. M = total metals, F = ferrous, NF = non-ferrous.

Country	Gross material extracted ($\%_{IBA}$)			Metal fraction (f_{met})			CEWEP report year
	M	F	NF	M	F	NF	
Austria	n.r.	3%	n.r.				2018
Belgium [1]	8.0%	7.1%	0.9%				2016
Czechia	5.9%	5.9%	0.0%				2016
Finland [2]	11.3%	6.5%	4.8%			0.29	2016
Germany	9.0%	7.7%	1.3%				2018
Hungary	n.r.	19.5%	n.d.			n.d.	2018
Ireland	10.1%	8.1%	2.0%			0.78	2018
Italy	0.0%	0.0%	0.0%				2012-13
Luxembourg	8.5%	6.9%	1.6%				2018
Netherlands [3]	11.0%	7.9%	3.1%	0.76	0.88	0.46	2012-13
Poland	0.0%	0.0%	0.0%				2010-11
Portugal	6.4%	n.r.	n.r.				2018
Spain [4]	17.1%	(9.9%)	(0.3%)	0.60			2010-11
Sweden	7.2%	5.4%	1.8%				2018
Minimum	**0.0%**	**0.0%**	**0.0%**				
Maximum	**17.1%**	**19.5%**	**4.8%**				
Total (M)	**8.2%**	**-**	**-**	0.71	-	-	
Total (F+NF)	**8.2%**	**6.7%**	**1.5%**	0.80	0.88	0.46	

Notes on Country Report data: [1] States likely underestimated; [2] Company level data; [3] Ferrous and stainless steel given separately; [4] Ferrous and non-ferrous reported as net material extracted.

be relatively low for 2025 and 2030, although a moderate to high risk remains for 2035.

In addition to allowing MSW-IBA utilisation to contribute towards recycling targets, achieving EoW may also help higher off-landfill utilisation rates to be realised, including in bound applications and higher value products, and through further exploration of alternative uses (see Table 1 for example). This is an important consideration for both the recycling and landfill diversion targets due to the expected decrease in the amount of waste landfilled, and thus the capacity to utilise MSW-IBA in landfill construction and backfilling operations. For a conservative estimate of the future reduction in landfill capacity and given the lack of quantitative targets for waste prevention, we might assume that MSW generation stays broadly constant (with waste prevention offsetting the moderate increase in waste generation observed over the past 10-15 years, Figure 3). Under these circumstances, achieving the landfill diversion target would see the amount of MSW sent to landfill decrease by around two fifths (from $23\%_{MSW}$ to $10\%_{MSW}$). Thus, if incineration is employed as a key (although non-optimal) mechanism to achieve landfill diversion, off-landfill utilisation will need to be enhanced. Verbinnen et al. (2017) argue that EoW would improve public acceptance of MSW-IBA derived materials and suggest that introduction of EU-wide criteria would boost recycling by setting unequivocal environmental standards (e.g., leaching limits). However, Blasenbauer et al. (2020) consider the feasibility of developing EU-wide EoW criteria to be low due to country specific situations (where appropriate limit values will vary according to local environmental conditions), and instead suggest a parallel approach, with defined fields of application, a risk-based assessment system for establishing limit values, and standardised test methods.

Irrespective of the approach, whether further utilisation of MSW-IBA is desirable, or would serve to facilitate further progress down a dead-end route towards lock-in of incineration, is an open question. For example, even with full utilisation of MSW-IBA via EoW (Scenario 3b), little to no headroom remains should recycling targets be strengthened in the future, a distinct possibility given that the 70% target proposed by the European Parliament during CEP trilogue (albeit not enacted) was supported by several MSs (EEB, 2017). Indeed, there would be a high risk of failing to meet a future higher ambition target without potentially costly withdrawal from incineration.

4. DISCUSSION AND CONCLUSIONS

While successive EU policies have driven significant improvements in waste management, prioritisation of landfill diversion has resulted in an unbalanced emphasis where mechanisms do not always align with the waste hierarchy. This is illustrated by the increasing prominence of incineration, where several (otherwise) progressive MSs have deployed incineration as a means to achieve landfill diversion targets. Driven by near-term targets, the use of incineration

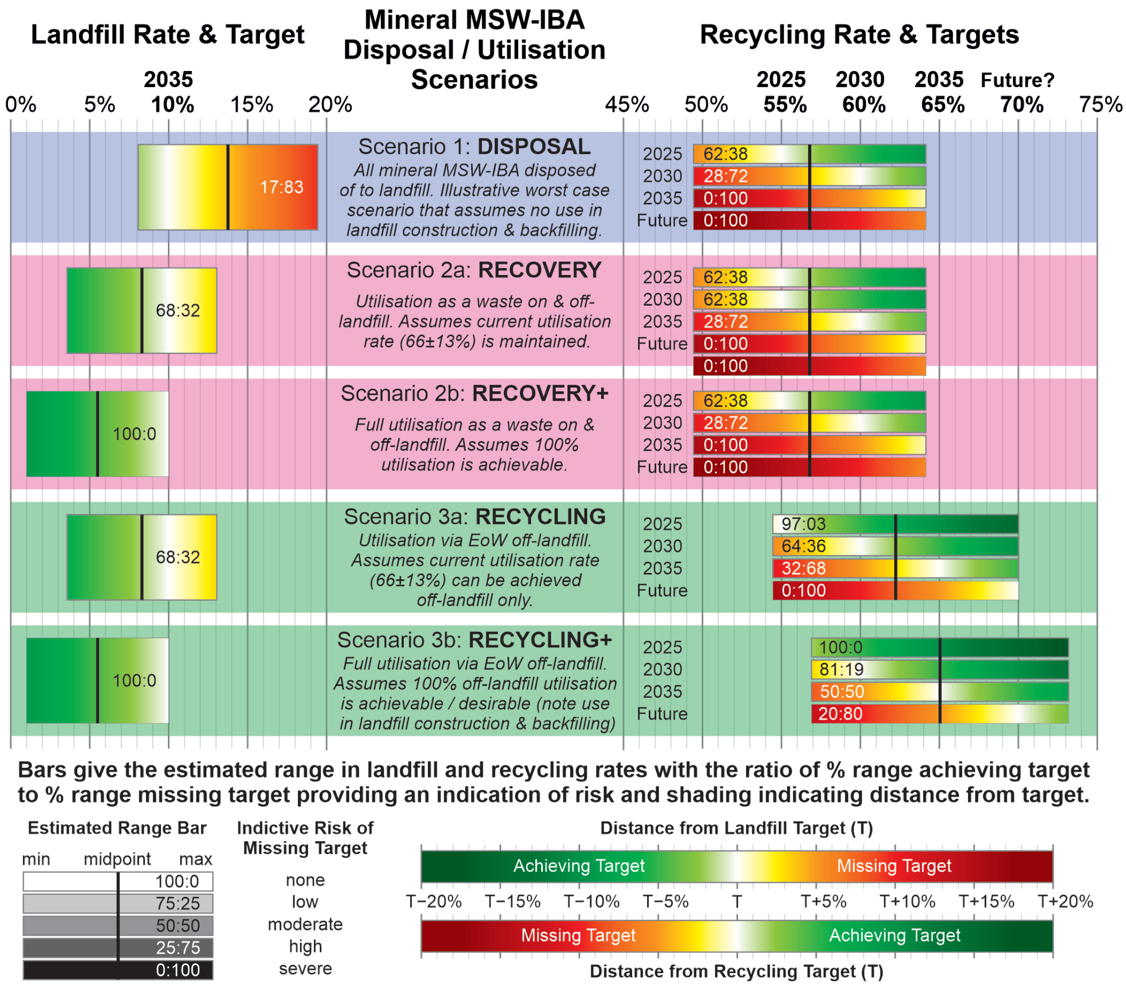


FIGURE 6: Implications for meeting landfill and recycling targets under different MSW-IBA utilisation scenarios.

is now at risk of lock-in, contradicting ambitions set out in the CEAP (through revision of the Industrial Emissions Directive) to "increase investment in new, cleaner technologies […] whilst avoiding lock-in to obsolete technologies" (EC, 2020). In addition, contra to CE principles, high-quality recyclates are diverted from closed-loop recycling routes to ensure that a consistent calorific value of input materials is met for efficient energy production.

That being said, it seems prudent to acknowledge that the use of incineration across the EU is likely to increase in the short to medium term. In light of the current energy crisis, the preeminent policy priority is enhancing security of supply through domestic/regional energy generation while maintaining progress towards climate neutrality (EC, 2022b), with FEAD (the European Waste Management Authority) promoting the potential contribution of energy from waste (EfW) (FEAD, 2022). This highlights the position of incineration at the intersection between waste and energy policy, where competing priorities and the application of different control mechanisms increases the risk of unintended consequences. For example, current discussions concerning the future of the EU Emissions Trading System (EU-ETS) have indicated that EfW incineration plants may no longer be exempt (ZWE, 2022). This introduces uncertainty and a likely reluctance on the part of MSs to establish or retain incineration taxes (which are promoted, but not required, by the CEAP), as inclusion within the EU-ETS would then effectively result in double taxation set against a backdrop of rising energy prices (Recycling Magazine, 2022).

With respect to the achievement of waste targets, it is acknowledged that this study has focused on analysis of aggregated data and does not consider other issues that may influence actual recycling rates, such as the success-rate of source separation initiatives. That being said, the analysis presented here clearly indicates that continuation of the current trend towards increased utilisation of incineration across the EU-27 as a whole carries a substantive risk of placing the 2035 recycling target out of reach. From a purely instrumental perspective, this could be addressed by re-defining the operations that qualify as recycling. At present, utilisation of MSW-IBA only counts towards recycling targets under EoW. However, with no EoW criteria published at either EU or MS level, and with valid questions raised regarding the desirability and operability of EoW for MSW-IBA, development of this route seems unlikely. Nonetheless, high utilisation rates in a variety of construction applications have been achieved (albeit use as a waste), thus reclassifying off-landfill utilisation as recycling would aid MSs in achieving targets. For example, as suggested by Blasenbauer et al. (2020), specific qualifying routes (for different applications) could be defined using a process based on EoW criteria, with additional allowances made for the local context.

While increased incineration would assist MSs to generate energy while achieving ambitious landfill diversion targets, and the proposal above would help achieve recycling targets, it would not help advance the transition to a CE. Additionally, while the CEAP presents a roadmap to 2035, a longer-term vision for achieving a fully CE is lacking. To address this, a clear vision for incineration is required to ensure that today's priority does not becomes tomorrow's status quo when moving beyond the current energy crisis. This vision should consider two missed policy opportunities related to the articulation and implementation of the waste hierarchy which, if unaddressed, may restrict the emergence of more sustainable solutions in the future. First, a lack of EU-wide limits or constraints on incineration (either overall or on specific material streams). Second, a lack of nuance within the waste hierarchy, where no distinction is made between open- and closed-loop recycling. As such, the open-loop utilisation of mineral MSW-IBA after EoW would currently have equal weighting to closed-loop recycling of the feedstock material. Similarly, while it is entirely conceivable that MSW-IBA could be utilised in the same application both with and without EoW, only the former would currently contribute towards recycling targets while the latter would be classed as 'other recovery'. Thus, to aid the CE transition, it is suggested that future policy development should consider the following points:

- To address the risks of technological and contractual lock in, clear policy signals on the future role of incineration within a climate-neutral CE must be formulated, and mechanisms to phase out incineration (by technology and/or of specific waste-streams) on an appropriate timeline should be developed. Given the identification of plastics and textiles as priorities for the development of new EU-wide EoW criteria (EC, 2022a), these represent excellent early candidates for introducing waste-stream specific limits on incineration. This could be similar in formulation to mechanisms within the Renewable Energy Directive, where the use of crop-based biofuels is gradually being phased out from a maximum contribution of 7% in 2020 to 0% in 2030 due to sustainability concerns (EC, 2018).
- To ensure a clear incentive for maintaining value, the definition of recycling should be expanded and the introduction of a weighting system that differentiates between closed and open-loop recycling should be considered. Such a system should reflect the relative value of each utilisation route with respect to the waste hierarchy, consider system maturity, and could also confer credit for utilisation as a waste. For example, a hard-to-treat waste stream might see a weighting <1 for post-incineration utilisation of MSW-IBA as waste material, 1 for post-incineration utilisation of MSW-IBA following application of EoW or direct open-loop recycling (i.e., no incineration), and >1 for closed-loop recycling, thereby providing an incentive for innovation. Again, such weighting mechanisms have been successfully deployed within energy policy for both renewable energy technologies and low emission vehicles (del Rio et al. 2017; EC, 2009).

In conclusion, while acknowledging that incineration will continue to take place, particularly in the near-term, this study argues that to avoid lock-in, policy focusing on promoting diversion of waste from landfill and recycling of residual wastes require bolstering by the introduction of mechanisms that gradually phase out incineration and distinguish between open and closed-loop recycling. Furthermore, to deploy the mechanisms described above, a long-term roadmap is needed, which not only provides an overarching objective for all environmental policy to realise a CE, but also indicates the relevant milestones and feedback loops required for waste management.

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