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A toxic legacy

Bottom ash in Europe’s circular economy

Research report
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May 2026
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Contents

Executive summary

Across Europe, approaches to the use of incinerator bottom ash (IBA) in roads and paths, as well as in concrete and cemented blocks, are inconsistent. This waste material, a mixture of dust, ash, glass, sand, stones and tiles, also contains persistent organic pollutants (POPs) and potentially toxic elements (PTEs). These substances can leach into the surrounding environment across a wide range of timescales and conditions.

Regulation is weak and fragmented, while decision-making often favours industrial profit margins over the protection of human and environmental health. Case studies evidence a regulatory "Wild West", with permitting of untreated wastewater discharge, combined with a reliance on industry-devised testing protocols that leniently permit a non-hazardous classification for a material that is demonstrably hazardous.

Microplastics and PFAS are among the many organic pollutants in bottom ash that are inadequately addressed by regulation. Processing methods intended to improve the quality of IBA are non-standardised and insufficient for the removal of toxic substances, often involving trade-offs whereby one improvement measure increases the risk of pollution from other pollutants. In practice, risk pathways vary by jurisdiction.

In unbound applications, the principal concern is leachate release and the uncontrolled dispersal of contaminants over time, particularly where monitoring and end-of-life controls are limited. Long-term structural stability of IBA remains a concern for bound applications due to the potential for swelling and cracking from gas release. Meanwhile, risks to occupational health – through cutting, drilling, grinding, and similar activities involving concrete and cement-based IBA products – are hardly acknowledged.

Legally in Europe, to create bottom ash - incineration of waste - is not considered recycling, nor is the use of its 'mineral fraction' in civil engineering applications. Only the extraction of metals from IBA qualifies as recycling.

The primary driver for the use of bottom ash as a construction material is the high costs of disposal for industry, rather than any environmental benefit. The use of the BA 'mineral fraction' disseminates hazardous substances throughout the built environment, creating a toxic legacy for future generations, despite the European Commission's requirement for any future circular economy to be "*toxic-free*".

Acronyms

(BA) Bottom ash

(BAT) Best Available Techniques

(BREF) Best Available Techniques Reference Document

(IBAA) Incinerator bottom ash aggregate

(CEAP) Circular Economy Action Plan (EC, 2020)

(CEWEP) Confederation of European Waste-to-Energy Plants

(CLP) Classification, Labelling and Packaging of Substances and Mixtures (EU, 2008a)

(EA) Environment Agency (UK)

(EC) European Commission

(EU) European Union

(ECHA) European Chemicals Agency

(EfW) Energy from waste

(EoW) End-of-Waste

(HMVA) Hausmüllverbrennungsasche

(IBA) Incinerator bottom ash

(IED) Industrial Emissions Directive (EU, 2010)

(LOI) Loss on ignition

(MSW) Municipal solid waste

(MSWI) Municipal solid waste incineration

(PFAS) Per-and polyfluoroalkyl substances

(POPs) Persistent organic pollutants

(PTEs) Potentially toxic elements

(REACH) Registration, Evaluation, Authorisation and Restriction of Chemicals

(U-IBA) Unprocessed incinerator bottom ash

(WFD) Waste Framework Directive (EU, 2008b)

1 mg.kg^{-1} = 1 µg.g^{-1} = 0.1 % (mass basis) = ppm (mass basis)

1 ng.g^{-1} = ppb (mass basis) » 1 µg.L^{-1} (of water)

1 ng.kg^{-1} = ppt (mass basis)

Introduction

In Autumn 2026, the European Commission (EC) is expected to adopt a Circular Economy Act. In doing so, it should draw on prior policy, such as the Circular Economy Action Plan (CEAP), which has the stated aim of *"Enhancing circularity in a toxic-free environment"*, alongside alignment with climate neutrality, addressing the presence of hazardous substances, and limiting the environmental release of microplastics (EC, 2020).

The CEAP also set a target to double the (2020) circular material use rate by 2030 (EC, 2020). This target for circularity was subsequently increased to 24% under the Clean Industrial Deal (EC, 2025a).

While the EC definition of 'circular material' is somewhat obscure, the definition of a circular economy is as follows (EU, no date):

> *"A circular economy is a system which maintains the value of products, materials and resources in the economy for as long as possible, and minimises the generation of waste. This means a system where products are reused, repaired, remanufactured or recycled."*

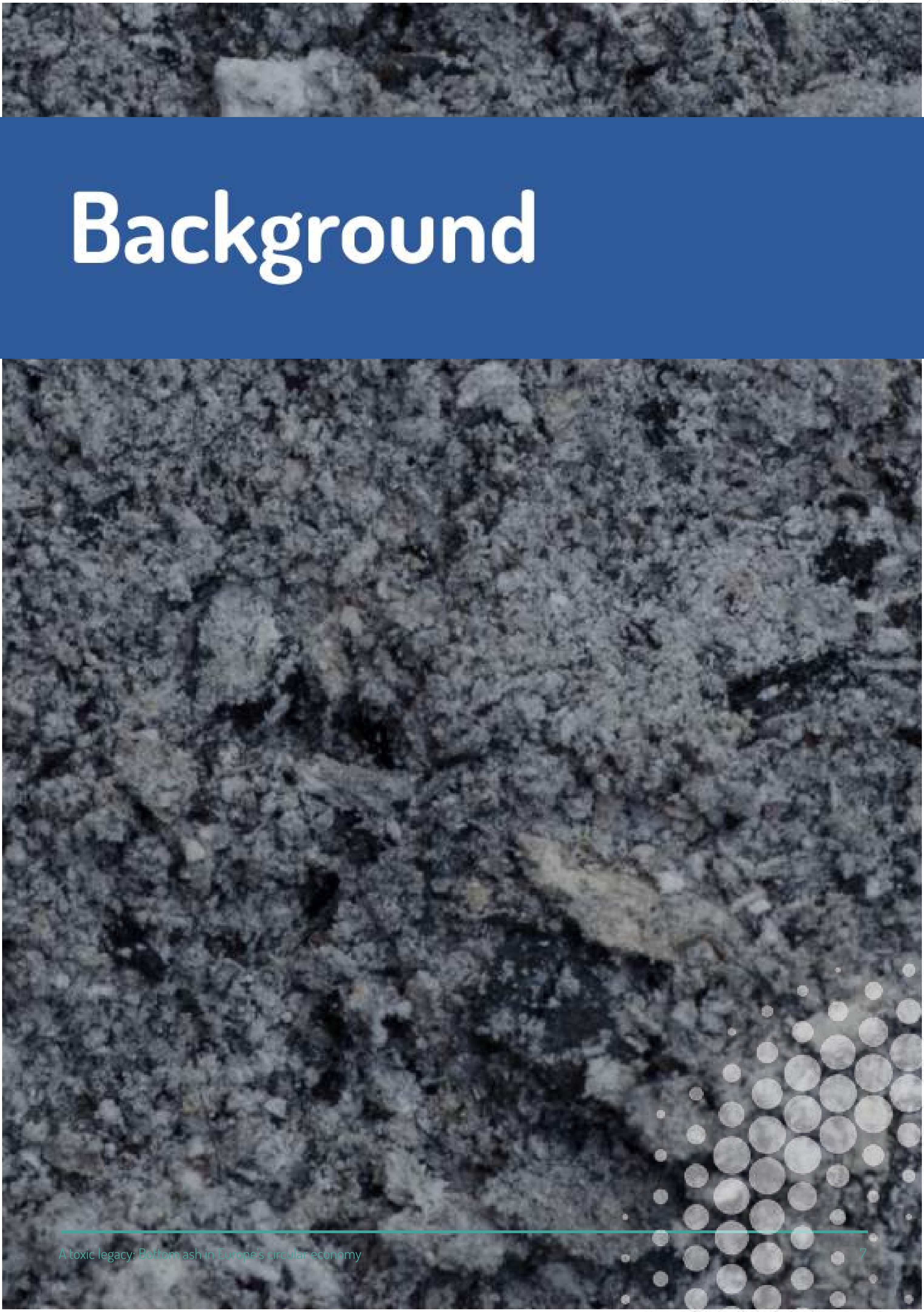
Accepting that a 'circular economy' without the use of resources and energy is an impossibility due to fundamental physical laws and the concept of entropy (Cullen, 2017), this report looks at the EC's attempts to attain it while simultaneously maintaining economic growth, minimising the use of virgin resources, and cutting greenhouse gas emissions, alongside improving human and environmental health. Specifically, it focuses on incinerator bottom ash (IBA) and its place in the EC's ongoing drive towards 'circularity'.

Slightly adapting its title from an earlier publication (Rollinson, 2022), this report looks at Europe-wide developments over the last four years through case studies and a review of independent empirical research. A new focus is given to ageing, storage, and processing of IBA in the context of occupational risk, as well as overall human and environmental health impacts, along with dedicated chapters on per-and polyfluoroalkyl substances (PFAS) and microplastics. Political drivers for the re-use of IBA are explored, followed by concluding remarks on future perspectives.

The report's central research question asks whether the use of IBA in civil engineering can assist Europe's transition to 'toxic- free circularity', or whether it is instead creating a *"toxic ticking time bomb"* (Remmers et al., 2025a,b). Quoting Ms Barbara Joziasse, Member of Netherlands Court of Audit:

> *"The signals have been around for decades...but we see that the ministers and state secretaries are slow to respond to this. It's a bit like a teenager's room: you see a huge chaos and ask: 'do you want to do something about that?' Then all the junk is crumpled under the bed. No longer a problem. That is immobilisa [IBA]: disappeared from sight."*

Background



What is bottom ash?

Bottom ash is a heterogeneous material comprising fine particles, bulk items (metal, glass, ceramics) and agglomerates, with high alkalinity (around pH 12) in its raw form. Its odour has been described as "*earthy*" (Environment Agency, 2022). Compositionally, it consists of approximately one-third fines (dust, ash, grit), one-third slag (amorphous/vitreous agglomerate), and one-third silicate-based fragments (glass cullet, stones, ceramics), along with about 10% pure metals. A smaller fraction consists of organics (books, paper, leather, microplastics, and a variety of other substances, some of which are persistent organic pollutants (POPs) [1]) (Figure 1 and Figure 2). However, this composition can vary: for example, the amorphous fraction accounted for over 75% in three out of five IBA samples from Italian municipal solid waste incineration (MSWI) plants (Mantovani et al., 2023).

Figure 1. Raw (unprocessed) incinerator bottom ash.[2] [3]

Elementally, IBA produced in Europe contains approximately 70% calcium, magnesium, and silicon; followed by 2 – 5% each of iron and aluminium; 0.5 – 2% each of chlorine, sulphur and potassium; around 0.1% each of zinc, copper and lead; with a wide range of other elements in smaller concentrations, including antimony, arsenic, barium, beryllium, cadmium, chromium, manganese, nickel, strontium, and vanadium, many of which are potentially toxic (Bunge, 2019; Dacklin et al., 2025; De Matteis et al., 2024; Mantovani et al.,2023). Further information on the elemental composition of IBA is provided later in this study and also in Rollinson (2022).

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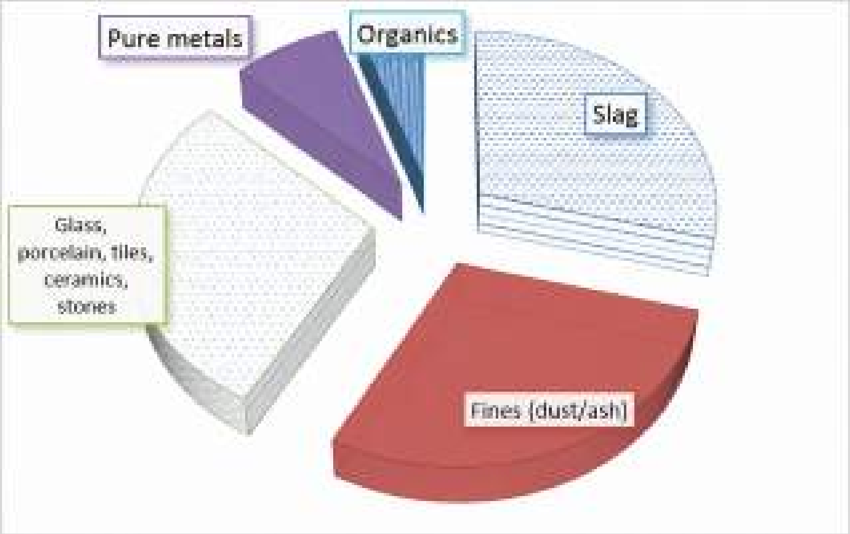
[1] See: Bunge, 2019; Simon and Scholz, 2024; Rollinson (2022); and Section 6 of this report.

[2] Note the high quantity of fine particles (dust/ash) and the larger unburned fragments, particularly those in the image on the left, which appear to be plastic and paper.

[3] Credit: Saxongate Residents Group.

Figure 2. General fractions of raw (unprocessed) IBA.[4]



How is bottom ash made?

The only way to obtain bottom ash is by burning solid waste. This is done via large-scale incineration, with IBA the main residue and the origin of the word's root: 'conversion to cinders'. "Waste to Energy" (WtE) and "Energy from Waste" (EfW) are euphemisms for incineration, and while electricity is recovered (albeit with low efficiency, $\eta \leq 0.3$), greenhouse gases are emitted and bottom ash is produced (Neuwahl et al., 2019).

Contemporary municipal solid waste (MSW) contains large amounts of plastics, which in turn contain many thousands of chemical substances. The latest audit identifies 16,325 chemicals, including 5,776 additives, 3,498 processing aids, 1,975 starting substances, and 1,788 non-intentionally added substances (Monclús et al., 2025). The authors estimate that around 4,200 of these chemicals have intrinsic properties hazardous to human or environmental health. Other studies indicate that the marine ecotoxicity of microplastics is mainly due to these additives (Da Costa et al., 2023).

By the law of conservation of mass, when waste is burned, inorganic constituents and other unburned substances become process residues. Some are light-weight or volatile and become entrained in the flue gas, while denser or less volatile substances fall through the grate and form bottom ash. In the most common type of municipal solid waste incineration (MSWI) in Europe - mass burn, moving-grate systems - approximately 25% of the input waste mass becomes IBA, although the values can range from 14% to 31% (Fletcher and Dunk, 2023; Neuwahl et al., 2019).

Since 2010, MSWI in Europe has been regulated by the Industrial Emissions Directive (IED). This requires compliance with certain operating conditions: the post-combustion gas from non-hazardous waste must be

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[4] Adapted from Vateva et al. (2025), and Paleologos et al. (2023).

heated to at least 850°C for a minimum of 2 seconds under the most unfavourable conditions, and the resulting bottom ash must have a total organic carbon (TOC) content of < 3 wt% or loss on ignition (LOI) of <5 wt% (EU, 2010).[5] [6] The IED sets pollutant limit values only for stack gas emissions and wastewater from air pollution control systems. It has previously been reported that materials such as sewage sludge have been found in IBA, along with exceedances of LOI limits (References in Rollinson, 2022).

How (and why) is bottom ash processed?

A fledgling industry has developed around the processing of IBA in Europe. Methods differ, there is no standard practice, and the activity is usually undertaken off-site by third parties (see Figure 3).

Figure 3. Incinerator bottom ash processing site, Cambridgeshire, UK.[7]

Immediately upon discharge through the incinerator grate, it is common in mainland Europe to quench the hot and reactive ash in a water-filled tank (wet capture). In the UK, dry discharge is more common. Following this, the IBA is invariably left to age, usually in external stockpiles where weathering accelerates chemical changes. At some stage, metals are extracted (using magnets for ferrous metals and eddy current separation for non-ferrous metals), possibly followed by crushing, sieving, and further attempts at metal recovery. Sometimes processing may also involve active washing of the IBA to improve its quality for use as a building aggregate.

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[5] The organic fraction is the source of the loss on ignition (LOI) component: when the IBA is completely burned, the organic substances are 'lost upon ignition' through oxidation and evaporation, thereby reducing the total mass of the IBA.

[6] Hazardous waste incinerators are required to operate at higher temperatures (1100°C). These processes are not part of this study.

[7] Reproduced with permission. All rights reserved.

After primary metals extraction, the remaining material is sometimes referred to as the 'mineral fraction'. However, due to incomplete extraction rates, many metals remain chemically bound within it. This residual material is used in some countries, often contentiously, as a substitute for aggregate in civil engineering, either in bound materials (such as concrete) or in unbound applications (such as road base layer).

A trade-off exists between easing metals recovery and valorising (or reducing disposal costs of) the mineral-based residue. Wet capture increases the stability and safety of the ash but makes metal recovery more difficult as metals become more chemically bound to other substances. Further metals recovery can be achieved by crushing the larger particles; however, this simultaneously diminishes the material's suitability as a replacement for virgin aggregate.

Some industry terminology is used to distinguish between processed and unprocessed IBA, although these definitions are fraught with problems due to the non-standard nature of processing. The acronym U-IBA is sometimes used for unprocessed IBA, while in the UK, Incinerator Bottom Ash Aggregate (IBAA) refers to the processed ash. In the Netherlands, a degree of semantic framing is evident in the use of the word *"immobilisate"* to describe processed BA.

How is the use of bottom ash regulated?

Across Europe, approaches to the use of IBA in roadways, concrete and cemented blocks are inconsistent. Some countries use all that they produce, others prohibit its use completely, while some allow only restricted applications. This is accompanied by a wide variety of regulatory protocols, differing limit values for a non-comprehensive range of substances, and fragmented testing methods based on controlled, stable, and short-term conditions (Blasenbauer et al., 2020).

The most permissive countries for using the 'mineral fraction' of IBA are the Netherlands, Denmark, and the UK, followed by Germany, France, Italy, Belgium, Poland and Spain, each operating under their own national guidelines (Fletcher and Dunk, 2023). Austria, Ireland, and many Eastern European states have a more conservative approach, with IBA use being either zero or very limited (*Ibid.*). Switzerland is one of the most restrictive countries for its handling of IBA: metal recovery is compulsory, and the remaining 'mineral fraction' is disposed of in hazardous waste landfills classified as type D, on a scale from A to E, with E representing the most hazardous waste (Dacklin et al., 2025).

Assessing the potential hazards of IBA involves two metrics. One is the 'total concentration' of potentially toxic elements (PTE), along with various organic pollutants. Another is the mobility of these elements and compounds when in contact with liquid (water), referred to as the 'leaching concentration', and defined as the mass of substance per unit volume of leaching fluid. For a discussion on the range and limitations of leaching test methods, see Rollinson (2022) and later sections of this report.

Method and terminology

The content of this report derives mainly from the study and review of independent, peer-reviewed literature published since the first edition of "Toxic Fallout", specifically between December 2021 and April 2026. Its scope is geographically restricted to the European Union, but extended to include countries in the European Economic Area, as well as Switzerland and the United Kingdom.

For purposes of standardisation, only studies involving empirical research using EU-produced IBA were chosen, or where the incineration conditions were stated to comply with the IED. Studies from other regions were used to assist with the interpretation of results, and in the case of microplastics, due to the limited number of European studies available. Only studies involving IBA produced by moving-grate incineration of non-hazardous MSW (*i.e.* household and commercial/industrial waste) were included within the scope.

Bottom ash was the primary subject of this study. Fly ash and air pollution control residues were excluded. A further incinerator residue – boiler ash – was not specifically part of the review, although in some countries it may be treated together with bottom ash (Neuwahl et al., 2019).

The terms 'IBA' and bottom ash (BA) are used interchangeably. Efforts have been made throughout the text to clarify the level of processing and treatment to which IBA samples were subjected prior to analysis, at a minimum distinguishing between unprocessed (raw) and processed materials. However, this was not always possible due to limited disclosure by some authors.

Results of the study

Part 1 – regulation matters

In the Netherlands, a country that ranks among the highest in Europe for the use of the IBA 'mineral fraction', all is not well. The widespread incorporation of IBA into buildings and roadway structures has been likened (cf. **Introduction**) to a *"ticking time bomb"*, and alternatively to the hidden mess of discarded paraphernalia in a teenager's room, where items are scattered and concealed under the bed, difficult to locate and even harder to clean up (Remmers et al. 2025 a, b).

Similar concerns have been expressed in Germany, which, like the Netherlands, uses IBA in civil engineering (Bersi and Schmidt (2023), quoting Mr Peter Gebhardt):

> *"In Germany it is allowed to use bottom ash from incinerators in road construction, under the asphalt. It makes my stomach ache. Rainwater can seep in and toxic pollutants can leach to the environment".*

Is bottom ash inert?

Industry commonly claims that IBA and IBAA are inert.[8] An illustrative example is a processing site in the UK that made this claim despite receiving ash that was still cooling after leaving the incinerator grate (Rollinson, 2025). In some cases, this assertion is qualified by stating that the material is *"largely"* inert, a formulation that should raise concerns, particularly as the non-inert components are not addressed (Blue Phoenix, 2025).

When IBA falls out through the incinerator grate, it is highly chemically reactive and far from inert. Wet capture methods can reduce this chemical reactivity to some extent, but the material is still not stabilised.

A subsequent ageing period of between 6 to 20 weeks is typically recommended, preferably outdoors and with regular turning to accelerate weathering (Neuwahl et al., 2019). Though this stage is often described as a period of stabilisation, IBA remains chemically unstable for many years thereafter, being a highly complex material that continues to undergo chemical changes in response to a variety of external conditions. As such, the leaching of toxic substances can occur across a range of conditions and timescales, and this issue, still not fully understood by science, underpins concerns that the use of IBA in civil engineering is premature.

Is bottom ash hazardous waste?

Whether a waste is classified as hazardous is both a legal and a technical question. In the case of IBA, the matter is also subject to political positioning. Tensions exist for politicians and decision-makers, with industry pulling one way with calls for IBA to be labelled as non-hazardous and for reduced frequency of testing; while

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[8] This means chemically unreactive, or resistant to chemical change.

other stakeholders call for more stringent testing, or even a complete cessation of IBA use until such time, if ever, that it can be shown to be safe over the long-term.

The Basel Convention definition

Ostensibly intended to limit the international movements of hazardous wastes, the Basel Convention also provides technical guidelines on best practice waste management.

For waste to be considered "hazardous" under the Basel Convention, it must fall within a category listed in Annex I ***and*** have one or more listed constituents. IBA meets these criteria under Annex I category Y18 – *"Residues arising from industrial waste disposal operations"* and due to the presence of listed constituents such as: arsenic (Y24), cadmium (Y26), antimony (Y27), lead (Y31), zinc (Y23), and mercury (Y29), as well as congeners of polychlorinated dibenzo-furans (Y43) and polychlorinated dibenzo-p-dioxins (Y45).

The Basel Convention does not prescribe quantitative thresholds at which the presence of a constituent triggers hazardous classification. Instead, its approach allows each country to make its own judgement. Some countries rely on their own national limit values, but this is not required under the Basel Convention.

European waste classifications

The EU waste legislation used to classify IBA as hazardous or non-hazardous is multi-layered. Attempting to follow the process is like opening a series of Russian dolls (or Pandora's boxes), ultimately revealing a very high degree of latitude for decision-making. It is neither straightforward nor robust and, in many ways, not fit for its intended purpose, unless that purpose is to allow great flexibility for industry to arrive at a 'non-hazardous' classification.

IBA is not, by default, 'non-hazardous'; nor is it inherently 'hazardous'. According to the List of Wastes (EU, 2014a), IBA is a 'mirror entry', meaning there are two listings available, one hazardous (19 01 11) and one non-hazardous (19 01 12). The appropriate category must be determined by comparing the waste against fifteen hazard categories described elsewhere (EU, 2014b), as well as considering the presence of a short list of POPs (as listed in EU, 2019). The verifier must then consult additional EU documentation covering substances, their hazard codes, and cut-off limits in the Classification, Labelling and Packaging of Substances and Mixtures Regulation (CLP) (EU, 2008a).

If one hazard property is identified, the waste is classified as hazardous. Similarly, if two wastes are mixed, one hazardous and the other non-hazardous, the whole mixture is deemed hazardous. The determination may be made through expert judgement, animal or plant-based testing, or chemical analysis and calculations (Hennebert, 2019). However, no clear guidance is provided on which analytical method should be used, though a hierarchy exists in which biological tests take precedence over calculations (EU, 2014a).

The hazard property Ecotoxicity (HP14) – relevant to IBA – can be assessed using alternative calculation methods (EU, 2017), but it remains subject to much confusion and debate. For hazard category 8 (HP8, corrosive to skin/eyes), the CLP Regulation states that materials with ≤ 2 pH ≥ 11.5 are hazardous, and IBA commonly exhibits a pH above 11.5. However, an over-ride allows mixtures with higher (or lower) pH to be classified as non-hazardous if buffer testing indicates that a corrosive classification is not warranted.

A good example is provided by Bandarra et al. (2022), who tested six (both processed and unprocessed) IBA samples for ecotoxicity. One sample would have been classified as hazardous (HP10 – toxic for reproduction) due to high concentration of lead, while all samples exceeded hazardous threshold for HP14 (Ecotoxic) based on cumulative calculation formulæ (assessing zinc and copper compounds only). However, bio-tests involving a species of crustacean as well as tests in which IBA was fed to, or applied to the shaven skin of two types of mammal, led the authors to overturn the hazardous classification. Notably, the doses of IBA were diluted beforehand, and the leach test used distilled water, both likely resulting in a more lenient classification.

This study, alongside the regulatory hierarchy, also highlights ethical issues regarding the use of vertebrates and mammals to determine outcomes that could be established through alternative methods. These practices are employed primarily to serve industry interests, whether by avoiding landfill disposal costs or qualifying for tax exemptions, further compounding the matter.

Case Study 1 UK – Europe's regulatory Wild West: a weak 'hazardous' testing protocol

The main testing protocol for IBA classification in the UK is voluntary and was developed by a trade association for the recycling and waste treatment industry (the Environmental Services Association – ESA) with the support of the UK Environment Agency (Environmental Services Association, 2018). The protocol provides multiple routes or 'bites of the cherry' for classifying IBA as non-hazardous, along with many options for reassessing results in order to mitigate or reinterpret findings where a hazard is identified:

The trigger levels for classifying IBA as 'hazardous' based on sampling do not, as one might expect, depend simply on individual samples exceeding limit values. Instead, a waste is considered hazardous only if:

"*6 or more exceedances have occurred in 24 samples*", or

"*any parameter exceeds a hazardous property concentration limit by a factor of 4 (a 4 x exceedance)*"

In other words, this means that up to 21% of samples could exceed the hazardous class limit value, while the IBA is still classed as 'non-hazardous'. Similarly, samples could consistently exceed one *or more* hazardous class limit values, not marginally, but by as much as 3.5 times, and still be deemed 'non-hazardous'.

Furthermore, even where a 4 times exceedance occurs, the protocol advises that a re-test should be undertaken, with results averaged across 22 replicates. In essence this approach may serve to smooth out peaks in concentration.

The protocol also offers a further 'bite of the "non-hazardous" cherry', stating that where a 4 times exceedance is identified:

"*additional chemical speciation tests may be appropriate to refine the original hazard assessment calculation and identify whether the 4-times hazard threshold is still valid and whether the IBA should continue to be classed as non-hazardous*".

The use of term "*refine*", suggests that operators can use judgement to smooth out the hazardous exceedance.

An additional layer of discretion is built into the protocol when waste is deemed 'hazardous'. Historical data may be used to determine whether the result is "*out of character*", alongside consideration of "*any additional relevant evidence that can be used to refine the face value assessment*"

If this were not sufficient, the analyses themselves are conducted in-house and may be limited to only a few hazardous constituents: Full characterisation against all 15 hazard categories and POPs is required only annually, meaning that POPs content in IBA may only be assessed during just one month out of twelve. Also, the minimum parameters to be assessed are limited to pH, pH buffering capacity, and, for potentially toxic elements, only copper, nickel, lead, and zinc. As will be shown in later chapters, this restricted approach fails to capture the full range of toxic substances present in, and leaching from, IBA.

This is further compounded by a relatively hands-off approach to regulatory enforcement. As stated by the Saxongate Residents Group (2025):

> *"The EA does not routinely re-test; it checks paperwork, may take occasional samples, and intervenes only if a classification appears wrong. In practice this means that as long as an operator uses a "recognised" method (ESA protocol or an authorised test house), the Agency will treat the result as valid, even if the frequency of testing is minimal."*

Is using bottom ash 'recycling'?

Only metals extracted from IBA can be counted towards Member States' recycling targets (EU, 2019); the leftover 'mineral fraction' cannot.

The ‘mineral fraction’ does not have EU-wide end-of-waste (EoW) status. While the responsibility lies with Member States to develop their own EoW criteria, none were found to have done so at the time of writing. IBA presents a particular challenge, as achieving EoW status requires demonstrating, *inter alia, that the material does* not adversely impact the environment, taking into account the broad requirement to consider “*any possible*” effects (EU, 2008b):

> “*The criteria [for end-of-waste status] shall include limit values for pollutants where necessary and shall take into account any possible adverse environmental effects of the substance or object.*”

Incineration of waste is the only process by which bottom ash is created, and this itself is not classified as ‘recycling’ under EU policy (EU, 2018). While incineration in some way may facilitate the recovery of some metals by concentrating them in bottom ash, a similar outcome could equally be achieved through improved upstream source separation without the need to destroy waste and generate greenhouse gases in the process.

More recent EU documents introduce some ambiguity. Recital 15 of the Environmental Delegated Act (Sustainable finance framework) 2023/2486, refers to the use of IBA as recycling; however, it does not clearly specify whether this is applied solely to metals. The recital refers to metals and inorganic salts (EU, 2023).

The Critical Raw Materials Act entered into force in May 2024. Its implementing regulation is still in draft form, but currently, IBA is listed as “*List of products, components and waste streams that are considered as having a relevant critical raw materials recovery potential*” (EC, 2025b). This also does not currently differentiate between the metals and the ‘mineral fraction’ of IBA.

In summary, legally in Europe, to create bottom ash - incineration of waste - is not considered recycling, nor is the use of ‘mineral fraction’ in civil engineering applications. Only the extraction of metals from IBA qualifies as recycling.

Can using the ‘mineral fraction’ help meet climate targets?

> “*Energy from Waste and IBA processing facilities are typically located in urban areas, this significantly reduces the CO_2 footprint when compared to primary / virgin aggregates extracted from rurally located quarries, because of reduced haulage distances to the place of use*” (MIBAAA, 2026).

It should come as no surprise that there are questionable claims regarding the potential climate benefits of using the mineral fraction of IBA in civil engineering. Climate targets are now part of EU-wide political strategy and, at the national and local level, must be ‘ticked-off’ for obtaining benefits such as permission to operate. Also, the mineral fraction of IBA carries a negative value for those in possession of it, making it more attractive to paint it ‘green’.

The ‘mineral fraction’ of IBA contains very little organic material. Its major constituents (over 95%) are oxides of sulphur, silicon, calcium, iron, and aluminium, among which are bound numerous other inorganic elements from different chemical groups. These substances do not degrade in landfill to release carbon dioxide or methane in the way organic matter does. As such, there is no apparent ‘climate benefit’ to using it in road construction instead of depositing it in landfill. Indeed, its use as a road sub-base can be seen as a form of landfill by another name, albeit less contained, since engineered landfill include safeguards such as linings to minimise the release of leachate (UNEP, 2022).

The question of whether using IBA as a replacement aggregate has a higher or lower climate impact than virgin aggregate is complex and depends on many variables. These include not just emissions associated with quarrying virgin aggregate, but also factors such as CO_2 capture during stockpiling and the strength and service life of concrete products (Marinković et al., 2023). The aforementioned study found that ‘recycled aggregate’ performed worse overall, with higher non-renewable energy use, greater eutrophication potential[9], and increased photochemical ozone creation. While the global warming potential was slightly lower in comparison to virgin aggregate, the difference was modest – around 10%. Importantly, the ‘recycled aggregate’ in this study was not IBA so did not consider the greenhouse gas emissions associated with the production of IBA itself.

IBA can only be created by burning waste. The process (EfW) has been described as “*the dirtiest way*” to generate power, being up to five times more polluting per unit of electricity than the average, with scientists characterising its continued use as “*a disaster for the climate*” and “*an insane situation*” (Stallard, 2024). Incinerators emit approximately the same amount of greenhouse gases as coal-fired power stations (Bersi and Schmidt, 2023; Stallard, 2024). Nor are these emissions climate neutral: they arise largely from the combustion of plastics and therefore constitute fossil fuel emissions, with the fossil carbon merely having taken a short detour in its life-cycle as a product.

Case Study 2 UK – Europe’s regulatory Wild West: wastewater run-off during stockpiling

Outdoor stockpiling, in many ways, decreases the chemical reactivity of IBA. But, when exposed to wind and rain, and as pH drops through ageing and rainwater ingress, chemicals in ionic form such as calcium, magnesium and sulphur, along with various potentially toxic metals, can leach out (Olea et al., 2024). The leaching of POPs, such as dioxins and PFAS, from IBA is poorly studied and regulated; but as a large and diverse group of chemicals they exhibit a range of solubilities and are known to be present in IBA (Arkenbout and Bouman, 2025):

> “*It is technically and commercially not possible to produce harmless bottom ash, without dioxins*

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[9] Eutrophication is the process by which a body of water becomes overly enriched with minerals and nutrients (like nitrogen and phosphorus).

> *or other POPs.*"

For these reasons, stockpiling during the ageing process requires careful management. Because IBA is chemically very different to virgin aggregate, its storage, handling, and processing should be approached with a much higher level of caution to minimise fugitive emissions and maintain safety.

Figure 4. Standing water at an IBA processing site in Cambridgeshire, UK.[10]

Yet, even the BREF gives very little guidance as to grey water treatment when stockpiling IBA. It devotes only half a page to the issue, offering a bullet point list of five options for wastewater treatment (Neuwahl et al., 2019):

> *"Waste water comes mainly from the wet process, washing processes, storage areas, and contaminated rainwater where slags and bottom ashes are stored outside. Process waste water contains salts and metals as well as suspended solids and organic substances including PCDD/F."*

In this UK case study, the site operator's permit application proposed the use of "*greywater for dust suppression*", with no measures for wastewater run-off mitigation and no wastewater management plan. It proposed continuous recirculation of water without treatment, meaning that potentially leachable contaminants would accumulate in the greywater, making them highly concentrated (Rollinson, 2025). Figure 3 and Figure 4 show aerial photographs of parts of the site. Figure 5 shows the on-site 'lagoon' which served as the holding tank for greywater to be re-used for dousing the stockpiles of IBA.

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Figure 5. Lagoon at the site of IBA processing in Cambridgeshire, UK, adjacent to stockpiles of IBA.[11]

Due to a history of complaints from residents regarding fugitive emissions from the site, the EA undertook a series of analytical tests on sediments and water at and around the outlet discharging into a small, slow flowing watercourse. The EA did not, initially, disclose the results of these tests, but they were later obtained under a Freedom of Information request. The results of sediment sampling undertaken by the EA at the site are shown in Table 1.

Table 1. Results of sediment sampling around the water discharge outlet of an IBA processing site in the UK.[12]

Note: Bold and underlined font identifies exceedances of UK guidelines, while shaded cells identify higher concentrations at the discharge outlet or downstream in comparison to upstream (for at least one of two samples). * = The values in brackets in this column are said to be from "*ditch*".

Element	UK guideline	Upstream*	Downstream	Discharge point
(mg/kg)				

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[12] Data obtained via the UK Environment Agency, following a Freedom of Information request.

Lithium (Li)	20-40	36	**45**	36
Sodium (Na)	None fixed	700	760	1700
Potassium (K)	None fixed	4000	4900	4500
Copper (Cu)	20-80	**110** (36)	**270**	**260** (64)
Silver (Ag)	<1	**1**	**1.4**	**1.6**
Beryllium (Be)	<2	4.5	1.8	1.3
Magnesium (Mg)	None fixed	3500	4500	4300
Calcium (Ca)	High in chalky soils	110000	72000	58000
Zinc (Zn)	90-200	**440** (110)	**920**	**830** (190)
Strontium (Sr)	200-400	180	190	220
Cadmium (Cd)	0.5-1.0	0.6 (0.33)	1	0.9 (0.64)
Barium (Ba)	<300	120	210	170
Boron (B)	None fixed	2.8	6.6	2.1
Aluminium (Al)	30,000-70,000	23000	29000	24000
Thallium (Tl)	28	**29**	**41**	**70**
Titanium (Ti)	None fixed	100	140	230
Tin (Sn)	<5	**6.4**	**24**	**12**
Lead (Pb)	100-300	63 (32)	12	81 (28)
Phosphorus (P)	500-1000	**2200** (870)	**7200**	**1400** (780)
Vanadium (V)	50-150	43	120	81
Arsenic (As)	15-20	12 (16)	11	16 (9.5)
Antimony (Sb)	<1	**2.4**	**2.4**	**7.2**
Chromium (Cr)	40-100	77 (32)	**170**	51 (52)

Selenium (Se)	<2	<2	<2	**3.4**
Molybdenum (Mo)	None fixed	4.2	2	15
Manganese (Mn)	300–1,000	550	750	**9000**
Iron (Fe)	10,000–50,000	34000	38000	32000
Cobalt (Co)	20–30	15	20	**51**
Nickel (Ni)	<50	44 (24)	**110**	**76** (39)

As can be seen, the analyses indicate higher concentrations of most metals at the effluent discharge point and downstream compared with upstream locations. A secondary, more limited dataset was also collected by the same agency, and these values are shown in brackets. Even in this dataset, concentrations were generally higher at the discharge point than elsewhere in the stream. For both sampling episodes, no detailed methodology was provided other than the information that the analyses were obtained at two different dates in 2025.

Simultaneous data relating to water quality was obtained (also not shown), with results showing UK guideline exceedances at the discharge point for chloride, mercury, zinc, chromium, and possibly copper. Only one heavy metal (nickel) did not exceed the guideline values, although its concentration was still higher at the discharge point than upstream. A brief invertebrate survey was also undertaken (not shown), but the results were inconclusive, showing an equal number of population differences downstream as upstream.

However, the discharge arises from a site-wide water management system rather than the IBA processing operation alone. While the spatial pattern raises concern, the available data do not establish a definitive source, and other site activities or materials may also contribute.

Despite these results and the following toxicity summary provided by the EA, a permit to discharge was granted for the site:

> "*Multiple elements exceed UK sediment baseline levels, especially at the discharge location. The most critical contaminants include cadmium (Cd), copper (Cu), zinc (Zn), nickel (Ni), thallium (Tl), manganese (Mn), and antimony (Sb), all of which pose significant toxicity to aquatic organisms. Toxicological classifications indicate that at least 7 elements are of high or very high environmental concern. Downstream enrichment shows clear transport from the discharge point.*"

Part 2 - Hazard focus

Organic pollutants

There are a variety of POPs present in IBA (Rollinson, 2022), but this section focuses on one group – PFAS, as a number of European studies have reported their presence in IBA in recent years. Microplastics (defined as small plastic particles, typically less than 5 mm) are also included, due to emerging evidence of their presence in European IBA.

Microplastics

"*Don't go looking for trouble*" is perhaps the approach taken by the incineration and IBA processing industries in Europe to the subject of microplastics in bottom ash. Although research is scant, industry does not seem eager to investigate the matter, or at least to report on such investigations. The topic is dismissed as not being applicable in Europe (MVV, 2025).

> "*There is some information on the internet relating to microplastics in IBA in China. Waste management and regulation there is understood to be different to in the UK.*"

This position is not credible, because samples of IBA obtained from processing plants in Sweden and Germany have been found to contain microplastics in finer fractions, at a minimum of 4.6 mg.kg^{-1} (Pienkoß, et al., 2022). Since then, no other European study appears to have investigated the matter.

Further Asian studies have confirmed the presence of microplastics in IBA. One study found that 70% of the microplastics were concentrated in the smallest IBA size fractions (below 4.75mm) with a maximum of 792 ± 181 per gram at the 0.075 – 1mm IBA cut (Lou et al., 2024). Fang and Wang (2026) reported concentrations of 14,280 ± 500 particles per kilogram in bottom ash, with smaller-sized microplastics (in the range 10-100 μm) accounting for 80% of the leached abundance. They also noted a synergetic interaction between heavy metals, chlorides and microplastics during leaching:

> "*The three pollutants are coupled and interact synergistically, which jointly determines the pollutant release patterns and environmental risks of BA during stockpiling, leaching, and resource utilisation*".

In Europe, it is not known whether any country regulates or tests for microplastics in IBA. Similarly, the potential for microplastic release during stockpiling and processing is given a low priority. One study in Asia identified, as might be expected, that the abundance of microplastics in IBA increased with crushing, thus heightening the risk associated with microplastics release during some processing operations (Lou et al., 2024).

Currently, the situation does not align with EU policy. The Commission committed six years ago to address the presence of microplastics in the environment (EC, 2020). While REACH was updated in 2023, its scope is limited to microplastics that "*confer a sought after characteristic*", and therefore does not apply to microplastics in IBA (EU, 2023). But, the EU's Pathway to a Healthy Planet for All sets a target of reducing the release of all microplastics into the environment by 30% by 2030 (EC, 2021).

PFAS

> "*ECHA's two scientific committees support EU-wide action with appropriate derogations and controls to address the growing and long-lasting risks per-and polyfluoroalkyl substances (PFAS) pose to people and the environment.*" (ECHA, 2026)

Termed 'forever chemicals', PFAS are water, oil and dirt repellent substances that exhibit high biological, chemical and thermal stability. They are hazardous, having been linked to multiple health effects: accelerated puberty, bone variations, behavioural changes, interference with the hormone system (thyroid, cholesterol levels), carcinogenic effects, impairment of the immune system, and transfer through the placenta (Vogel et al., 2023).

According to ECHA, 80% of the fluoropolymer waste generated in Europe ends up in municipal waste incinerators (ECHA, 2025), despite evidence that they are not totally effective at destroying PFAS (*Ibid.*):

> "*...100% complete destruction is not to be expected at operational conditions, especially for municipal waste incineration.*"

Björklund et al. (2023) analysed IBA from a Swedish MSWI plant and found that total PFAS concentrations were in the range $0.16 \leq ng.g^{-1} \leq 0.54$. When up to 8% sewage sludge was co-incinerated with MSW, the total concentrations of PFAS increased to $0.81 \leq ng.g^{-1} \leq 1.5$.

In a broader study, analysing the residues from *twenty-seven* Swedish MSWI plants, Johansson et al., 2024 corroborated these findings, recording concentrations of PFAS in IBA of $0.22 \leq ng.g^{-1} \leq 1.6$. These were in a similar range to PFAS concentrations in fly ash. The authors comment:

> "*Incineration is commonly used to dispose of waste contaminated with PFAS despite few experimental data[sets] supporting the efficacy of this technique*".

Another recent study of Swedish MSWI bottom ash measured higher total PFAS concentrations in IBA than in fly ash or boiler ash (Gustafsson, 2025). The values in IBA were relatively high compared to other studies: from $5.4 \leq ng.g^{-1} \leq 7.8$.

Further evidence, from the United States, is relevant as this relates to incineration conditions compliant with the requirements of the IED. Higher concentrations of PFAS were found in bottom ash, with mean total concentrations approximately seven times greater than those found in fly ash (2.2 ng.g^{-1} vs. 0.33 ng.g^{-1}) (Liu et al., 2026).

Studies consistently indicate that short-chain PFAS predominate in bottom ash (Björklund et al., 2023; Liu et al., 2026; Stradberg et al., 2021). Shorter-chain PFAS are often more mobile and/or bioavailable and hence more problematic (EEA, 2024). These compounds are highly water-soluble, increasing the likelihood of their transfer from IBA into the environment through rainwater infiltrations and surface run-off during outdoor ageing. This leachability of PFAS from IBA appears to have not been systematically investigated in Europe. However, one study assessed leaching of PFAS from fly ash and reported that short-chain PFAS were "*consistently leached…likely due to their high water solubility*" (Gustafsson, 2025).

These findings must be viewed in context. The PFAS family comprises over 10,000 distinct compounds, yet only a small subset was analysed in the aforementioned studies. As such, total PFAS concentrations in IBA are likely to be underestimated.

Although the concentrations of PFAS found in IBA are small, their extreme persistence, mobility and toxicity make even low concentrations significant. This is particularly relevant in view of the fact that Persistent, Mobile, and Toxic (PMT) and very Persistent, very Mobile (vPvM) substances were officially added as new hazard classes to the CLP Regulation via Commission Delegated Regulation (EU) 2023/707, published in March 2023. Several PFAS are also endocrine disruptors, and are considered non-threshold chemicals, meaning that any level of exposure may cause harm (Johnson et al., 2023).

The recommended tolerable weekly intake of PFAS, according to the European Food Safety Authority is 0.00063 ng.g^{-1} (Johansson, 2023). This level is two to three orders of magnitude (hundreds of times) lower than the concentrations of PFAS found in IBA. Therefore, this implies that individuals involved in processing of IBA, those exposed to fugitive emissions where during outdoor handling, or construction workers cutting, drilling or grinding IBA derived products, need only ingest one gram of dust to be hundreds of times over the recommended TWI threshold.

Another comparable limit value is provided by the 2020 Drinking Water Directive, which from 12th January 2026 requires all Member States to comply with maximum PFAS concentrations of 0.1 µg.L^{-1} for the sum of twenty specific PFAS, and 0.5 µg.L^{-1} for total PFAS (EU, 2020). These values are in the same range as the concentrations of PFAS measured in IBA in the aforementioned studies, with the units being approximately equivalent.

Despite the above, limit values for the leaching of PFAS from IBA when used in civil engineering were not found to be part of any testing protocol. The only applicable regulation concerns a single group of PFAS (Perfluorooctane sulfonic acid and its derivatives) in the POPs regulation, which sets a high cut-off value of 50,000 ng.g^{-1} (EU, 2019).

Discussion – loss on ignition: a smoking gun

While some modelling or lab-scale studies of pure PFAS mixtures subjected to incineration conditions show high destruction efficacies, these conflict with the analyses of bottom ash from real-world plants operating in accordance with the IED. One proposed explanation is that the IED operating conditions, of 850°C for 2 seconds gas residence time, are too low to volatilise PFAS (Gustafsson, 2025). But if so, this would be expected to be revealed in modelling and lab-scale tests.

A more likely hypothesis is that the incinerator grate experiences uneven temperature and air distribution, resulting in pockets of incomplete combustion. This effect may be accentuated by the presence of heat-stabilising and flame-retardant additives in plastic waste[13]. Evidence to support this 'cold spot' theory comes from Björklund et al. (2023), who observed elevated PFAS concentrations in IBA when sewage sludge was co-incinerated. However, these elevated concentrations may also reflect higher PFAS content in the sewage sludge itself. This matter is an important one for incineration plants that are used to dispose of sewage sludge.

One pilot-scale study showed a near complete destruction of selected pure fluoropolymers at conditions corresponding to those stipulated by the IED (Gehrmann et al., 2024). In this study, the system used was a rotary kiln – commonly used for hazardous waste destruction, which provides enhanced mixing in a rotational drum. This, together with co-combustion of waste with wood chips, oil and natural gas, may explain the better performance in comparison with larger grate-fired, mass burn, solid waste incinerators where unprocessed bulk waste is burned on a moving grate.

In the previous version of this report, an unexpected finding was that LOI values in bottom ash reported by independent studies exceeded the < 5 wt% threshold required for compliance with the IED (Rollinson, 2022). LOI is a key indicator of incomplete burnout, reflecting the extent of unburned material. Elevated LOI can support why materials such as sewage sludge, paper and microplastics end up in IBA, and perhaps also why PFAS do not get completely destroyed.

This pattern is reinforced in the current review: De Matteis et al. (2024), analysing bottom ash from an incinerator in Italy, measured LOI at 19.69 wt%. Similarly, Mantovani et al. (2023), examining five MSWI plants in Italy, found that only 4 out of 46 samples met the IED limit of 5 wt%, with median values ranging from 10 to 20 wt% and maxima reaching 26%. In England, Olea et al. (2024) measured LOI values in BA ranging from 6 to 22 wt% across three incinerators. Only two studies reported LOI values under the IED limits: Simon and Scholz (2023) for plants in Germany (values below 2 wt%), and Liu et al. (2026) for IBA created in the United States.

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[13] While the global incineration reaction is exothermic (heat releasing), other reactions that occur simultaneously or prior to this, such as pyrolysis and the drying of moisture, are endothermic (heat absorbing) and these lower the internal temperatures.

Potentially toxic elements – can processing immobilise them?

To address claims that processing improves IBA sufficiently for its safe use as a building aggregate, this section reviews independent studies based on IBA subjected to current (and advanced) processing techniques. It should be noted that where total concentration and leaching limit values are cited by specific authors, these apply only within the country of origin. There is no consistency across jurisdictions, and what exceeds a limit in one country may not in another.

Washing IBA – pros and cons

In wet capture methods, some percentage of the soluble chemicals in IBA (e.g. sulphur, chlorine, calcium) are removed prior to processing (Blasenbauer et al. 2023). Additional washing of IBA may be an appropriate method to remove more fines and soluble elements, but it generates hazardous wastewater (the high water solubility of some PFAS has been described in the section on PFAS). This introduces additional costs for labour, machinery, and the management and disposal of large volumes of contaminated liquid, raising questions about whether the process is economically justifiable.

In the Netherlands, washing of IBA is sometimes practised in order to meet compliance levels for its unrestricted use, though it is not mandatory. Because of the cost implications, producers of cleaned IBA reportedly find it difficult to compete with the ready availability of cheaper, unwashed IBA, alongside what is described as insufficient regulatory oversight (Arkenbout and Bouman, 2025). The same source describes the current situation as follows:

> “*Most processors do not clean the bottom ash but mix it with cement or another binding agent and then categorise it as ‘designated building material’. There are hardly any rules for this and there is no obligation to register. This very weak ‘control system’ makes disposal of uncleaned bottom ash open season for the industry*”.

Industry have also opposed proposals for mandatory washing of IBA (Remex, 2025):

> “*Washing not only incurs considerable costs, but also creates a new waste stream: hundreds of thousands of tonnes of contaminated sludge from the process have to be sent to landfill – despite the fact that there is a moratorium on landfilling.*”

But neither is washing totally effective at removing all contaminants. For example, dioxins cannot be removed from bottom ash through washing because they are hydrophobic (Arkenbout and Bouman, 2025). Washing may also increase the long-term leaching behaviour of certain PTEss, namely antimony and vanadium, an effect linked to the loss of ionic calcium in the IBA and a reduction in pH, both of which occur during ageing and exposure to water (Vogel et al., 2024). In other words, the longer that IBA is exposed to outdoor conditions, the greater the potential for antimony and vanadium to leach out of applications in which the ash is used.

When IBA (that had been subjected to ageing and several washing steps) from two Swedish MSWI plants was analysed, the samples failed to meet Swedish guidelines for 'less than minor risk' (*i.e.* unrestricted use). Exceedances were recorded for arsenic, cadmium, chromium, copper, nickel, lead, and zinc across all particle size ranges, sometimes at levels tens to hundreds of times above the limit values. Leachate concentrations also exceeded Swedish limits for chloride and sulphate in seven out of eight samples (Dacklin et al., 2025). One of the two samples was identified as being hazardous waste due to the leaching of antimony. The authors concluded:

> "*None of the sorting fractions achieved the limits set by SEPA (Swedish Environmental Protection Agency), and it is unlikely that these materials can be refined to reach these limits*"

Similarly, even after metals extraction, sieving and washing, IBA samples (obtained from two Austrian MSWI plants) exceeded national concentration limits for use in road construction and as cement clinker, due to elevated levels of cadmium and chromium (Blasenbauer et al., 2023). Chromium and antimony also leached from the IBA samples at concentrations exceeding regulatory limits for use as a road aggregate. The authors comment on the high leachability of antimony being a consequence of their washing technique.

Ageing and advanced treatment of IBA – pros and cons

Outdoor weathering increases the complexity of IBA, one effect being that PTEs form chemical bonds with other elements at grain rims, which in turn influences leaching behaviour. This issue has led to calls for a bespoke investigation based on the provenance of IBA prior to any use, as each MSWI plant creates bottom ash with distinct chemical characteristics (De Matteis et al., 2024).

The evolution of leaching behaviour during ageing - over a period of up to 281 days - was studied by Simon and Scholz (2023). The IBA, sourced from two MSWI plants in Germany, had been subjected to wet capture and both ferrous and non-ferrous metals extraction. Despite this processing, the ash still contained 3.8 to 5.3% iron, 1.5-1.6% aluminium, and well over 500 mg/kg of titanium, copper, zinc, and barium.

The authors observed that sulphate leaching increased with ageing, while the cumulative release of most heavy metals remained almost unchanged over the storage period (ageing). For zinc and lead there was a decrease in leaching tendency as pH dropped from 13 to 9, while the opposite was observed for antimony. They concluded:

> “*[Meeting] Germany’s HMVA-1, with low release of sulfate, chloride, and Sb, are not achievable. The exclusion of the fine fraction 0–2 mm (or 0–4 mm) where chloride and sulfate are enriched might be helpful to reach this goal. However, that would affect approximately 50% of the mass of IBA, which then might be disposed of on landfills. The problem of increasing Sb release as a result of decreasing Ca concentrations would still remain in such a scenario, especially in the long-term view, when most Ca is transformed into* $CaCO_3$”.

Case Study 1 DE - Recent developments in Germany: HMVA-1 and HMVA-2

Germany has recently adopted (August 2023) a detailed technical standard - the Secondary Building Material Ordinance (Ersatzbaustoffverordnung EBV) (BMJV, no date), for regulating the use of IBA as a road sub-base layer. The ordinance has set legal limits on the leaching of certain hazardous substances from IBA (antimony, chloride, chromium, copper, molybdenum, sulphate and vanadium) based on two ‘use classes’: HMVA-1 and HMVA-2[14]. Rather than defining these ‘use classes’ narratively, the EBV categorises them according to installation types (einbauweisen), along with consideration of the proposed location relative to water bodies and quality of groundwater cover layer. Permitted applications include road construction under sealed layers with certain exclusions.

HMVA-1 imposes stricter standards for compliance, with lower limit values for leaching. Under HMVA-2, the leaching requirements are less strict than those applied to the disposal of IBA in inert landfill, despite IBA not being classified as inert waste in Germany (Simon and Scholz, 2023).

With respect to HMVA use classes, the range of PTEs is limited; with certain PTEs usually present in IBA at relatively high concentrations not assigned specific limit values, namely arsenic, lead, cadmium, nickel and zinc, although they are part of the wider ordinance. The ordinance specifies leaching tests (DIN-19528, column and batch methods) using distilled water.

From another study of German IBA (both unprocessed and aged up to one year), all samples exceeded the leaching limit values for HMVA-1 by way of molybdenum, sulphate, chloride and chromium (Vateva et al., 2025). Even after advanced processing, all the IBA samples still exceeded the HMVA-1 leaching limit values for chloride, molybdenum and sulphate. For copper, the advanced mechanical processing actually increased the leaching tendency from the smaller grain sizes. The authors observed:

> “*Despite the further metal separation, there were still residual metals in the mineral fractions such as metallic Al that might pose an obstacle to use in concrete. In the case of the medium fraction, there is a limitation regarding the removal of metallic Al which is related to the efficiency of the ECS [Eddy*

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[14] HMVA is from the German for IBA: hausmüllverbrennungsasche.

> *Current Separation] process. The separation of small particles is more challenging by means of ECS. Furthermore, Al particles that are partly oxidized or covered by mineral particles cannot be removed by means of ECS. A possible solution is a further comminution to release metallic particles from conglomerates. However, by decreasing the grain size, the utilization options will be affected'.*
>
> *"With respect to the use as secondary construction materials, the fractions investigated were associated with high leaching contents of salts and heavy metals such as molybdenum, which posed major obstacles… Salts could be reduced throughout the processing to some degree, in particular by means of wet processing, but remained at relatively high levels, which impairs use in reinforced concrete products".*

The same researchers also produced and tested concrete paving stones made with 2-8mm IBAA replacing virgin aggregate (two samples), along with one sample incorporating 11% IBA fine fraction as a substitute binder. Though leaching was within the regulated (HMVA-1 and 2) limits, concentrations of both heavy metals and soluble salts were significantly higher in the samples containing IBA relative to the control. Moreover, none of the samples met all required technical performance criteria.

In a study of one-month-aged IBA obtained from five different WtE plants in Italy, Mantovani et al. (2023) identified that, across all grain sizes and all 25 samples, leachate limit values were exceeded for chloride and copper (92% of cases). All but one sample also exceeded the limit value for selenium. Noting the "*significant concentrations of chloride and sulphur, especially in the finer grain size*", the authors concluded that further pre-treatment would be necessary before IBA could be considered suitable for use as a cement binder.

Discussion on leaching tests and limitations

An assessment of whether the limit values set by different countries are adequate is not provided here, as the basis on which these limits are established is not clear. While these studies may appear to demonstrate, through numerous limit value exceedances, that the regulatory landscape constrains disposal options for IBA, this must be understood in the context of the wide variations existing across Member States (where such regulations exist) (Blasenbauer et al., 2020).

Overall, prescribed leaching tests for IBA, despite significant methodological differences, typically use distilled water as leachant, at a fixed pH, and assess only a limited range of substances. As such, they provide a stable, controlled snapshot of leaching behaviour, which may underestimate the material's true leaching potential. This issue is discussed in more detail in Rollinson (2022).

Standard leaching tests were not designed for IBA and do not adequately reflect real-life scenarios, particularly the influence of organic compounds present in groundwater. This limitation was illustrated by Olea et al., 2024, who applied a synthetic organic exudate solution to IBA samples that had already been subjected to wet treatment, metal extraction, and four years of storage. The organic leachant caused the release of many potentially toxic elements at pH values around 8 to 10 (a typical range for aged IBA), while leachate concentrations exceeded maximum limit values used in many European countries for cadmium, zinc, copper, lead, and potassium. The authors also commented on the total concentration of elements in IBA:

> “*MSW IBA produced from waste from a major UK city contained PTEs (potentially toxic elements) that exceed most guidelines for unrestricted environmental use*”.

Safety and integrity of IBA aggregate and concrete

IBA is not inert, as chemical reactions within the material can cause gases to evolve, potentially leading to swelling and cracking of structures. One such chemical reaction, producing hydrogen via the interaction with aluminium and calcium hydroxide, has long been recognised (Neuwahl et al., 2019). However, there are other chemical reactions that cause expansion and instability. Sun et al. (2026), showed how the aluminium/calcium/hydrogen reaction was only short-term, and that swelling continued over a period of 7 to 10 months due to sulphate-hydrate crystallisation:

> “*From an engineering perspective, the swelling magnitudes of untreated IBA are far beyond acceptable limits…Previous studies have also shown that even 5–10% swelling is sufficient to cause cracking, heave, and strength loss in foundations and pavements. With confined vertical expansion rates [from this study] exceeding 18%, untreated IBA is expected to undergo even greater free swelling without confinement, posing severe risks of volumetric instability.*“

Case Study 3 UK – Nortrader Incident

The risks associated with hydrogen release from interactions with aluminium and calcium at high pH in IBA are well illustrated by the Nortrader incident in 2017. A cargo of unprocessed IBA, loaded at the MVV Environment Devonport Energy-from-Waste (EfW) plant in Plymouth, was to be shipped to the Netherlands. The material had measured pH values as high as 12.6. In combination with aluminium and water (comprising 14.5% of the IBA weight), this led to a chemical reaction that generated hydrogen gas.

The hydrogen, which has a wide explosive range in air, accumulated in the ship’s holds, resulting in two explosions and causing serious injuries to a crew member. The case was formally investigated by the Marine Accident Investigation Branch (MAIB), whose report stated that the vessel’s charterer had informed the captain that the cargo was “*non dangerous*”. The report also listed a number of previous on-shore incidents, including the cracking of concrete in which IBA had been used (MAIB, 2017).

This, of course, refers to unprocessed IBA, and Sun et al. (2026) state that removing the fine fraction of IBA, along with accelerated carbonation (ageing), can limit swelling to < 0.1%. But they also note that the calcium/sulphate reaction is water induced, advising that: "*at the very least, adequate waterproofing should be implemented*". This recommendation conflicts with the currently widespread practice of outdoor weathering of IBA.

Other authors studying European IBA present a more cautious view. In a study by Vateva et al. (2025), bottom ash obtained from Germany had been subjected to wet treatment upon discharge, followed by metals extraction, crushing and ageing for up to one year. Despite this processing, aluminium concentrations in the fine and medium fractions (up to 8mm) remained above levels considered safe for use in concrete structures in the Netherlands. The authors highlight both the limitations of conventional processing and the current lack of understanding:

> "*The maximum [recommended] concentration of metallic Al is limited to avoid impairment of the concrete to 0.2 wt% in filler material (i.e. fines) and 1.0 wt% in aggregates. In the present case, these limit values are exceeded by 0.1 wt% and 0.2 wt% for the fine and medium fractions, respectively. Therefore, a further reduction of the metallic Al is desirable from an environmental as well as a building material point of view. However, which grain-size specific concentrations of metallic Al actually lead to damage in concrete applications is not yet well investigated and should be the subject of further research.*"

No research study was found that examines the stability of the concrete structures used to stockpile IBA while ageing. There is, however, a case currently under investigation and reported in in the UK press, where a concrete pad used as hard-standing for raw IBA has become repeatedly cracked and damaged for unknown reasons (Cliss, 2026). It is plausible that these outdoor stockpiles of unprocessed IBA, having relatively high concentrations of aluminium and sulphate, could cause expansion in micro cracks in the same way as the freeze and thaw process splits rocks and causes crevices in road surfaces (potholes).

Another potential cause of cracking in concrete made with IBA is the corrosion of reinforced bars due to sulphuric and hydrochloric acids (sulphur and chloride are present in large quantities in raw IBA) (Mantovani, et al. 2023; Vateva et al. 2025). Where iron reinforcement is used in certain applications (such as concrete pads), corrosion can lead to expansion and structural weakening of the iron.

Drivers, and comment on future perspectives

The evidence presented in this report indicates that the continued use of IBA in civil engineering applications is not driven by environmental performance, but by a combination of economic pressures, regulatory gaps, and policy framing.

At a structural level, the generation of IBA is an unavoidable consequence of the expansion and lock-in of waste incineration capacity. Waste generation in Europe remains closely coupled to economic growth, while incineration infrastructure ensures a continuous output of bottom ash requiring management. In this context, the promotion of IBA as a secondary material functions primarily as a downstream solution to a systemic upstream problem.

Economic considerations appear to be a dominant driver. The high cost and regulatory burden associated with landfill disposal, combined with the negative market value of untreated bottom ash, create strong incentives to divert this material into construction applications. This dynamic has been previously characterised as a commercially motivated practice, in which environmental claims act as a secondary justification rather than a primary rationale. The result is a system in which the reuse of IBA is favoured irrespective of its intrinsic hazard profile.

At the same time, the regulatory framework governing IBA use remains fragmented and insufficient. As outlined in earlier sections, the absence of harmonised standards, combined with inconsistent testing methodologies and limited scope of monitored substances, allows materials with significant toxicological concerns to be classified as suitable for use. The reliance on short-term and simplified leaching tests further obscures long-term environmental behaviour, creating a regulatory environment in which risk is systematically underestimated.

Policy narratives around circular economy and resource efficiency also play a role in enabling the continued use of IBA. However, the findings of this report demonstrate that such applications do not fulfil the core principles of circularity. Rather than maintaining material value, the use of IBA disperses hazardous substances into the wider environment, effectively diluting and relocating pollution. This is inconsistent with the objective of achieving a "*toxic-free*" circular economy, and raises fundamental questions about the appropriateness of classifying IBA-derived materials as circular solutions.

From a systems perspective, the issue may be understood as one of redistribution rather than recovery. Incineration concentrates hazardous substances into a single residue stream; subsequent use of IBA in construction redistributes these substances across the built environment, often under conditions where long-term monitoring and control are not feasible. This transition from point-source containment to diffuse exposure represents an increase, rather than a reduction, in environmental risk.

Looking forward, current policy developments at EU level - including the Sustainable Products Initiative and the Circular Economy Act - offer opportunities to address some of these challenges, particularly in relation to improved traceability of hazardous substances and stronger alignment between product and waste legislation. However, the effectiveness of such measures will depend on the extent to which they are grounded in independent scientific evidence and guided by the precautionary principle.

In the absence of such alignment, there is a significant risk that existing practices will be reinforced rather than reformed. In particular, financial incentives or policy support mechanisms aimed at promoting the use of secondary materials could unintentionally accelerate the dissemination of hazardous substances, especially in regions with weaker regulatory controls.

The findings of this report therefore support the conclusion that the use of IBA in civil engineering applications is premature and inadequately justified. As with the conclusions drawn in earlier sections, the available evidence does not support the claim that such use can be considered safe, nor that it contributes meaningfully to circular economy objectives.

In line with the precautionary principle, and pending the development of robust, harmonised, and science-based regulatory frameworks, the continued use of IBA in the built environment should be reconsidered. Priority should instead be given to the containment of hazardous residues, the reduction of waste generation, and the phase-out of practices that perpetuate the circulation of toxic substances.

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Zero Waste Europe (ZWE) is the European network of communities, local leaders, experts, and change agents working towards a better use of resources and the elimination of waste in our society. We advocate for sustainable systems; for the redesign of our relationship with resources; and for a global shift towards environmental justice, accelerating a just transition towards zero waste for the benefit of people and the planet. www.zerowasteeurope.eu

Zero Waste Europe gratefully acknowledges financial assistance from the European Union, Gallifrey Foundation and the United Kingdom Without Incineration Network (UKWIN). The sole responsibility for the content of this material lies with Zero Waste Europe. It does not necessarily reflect the opinion of the funders mentioned above. The funders cannot be held responsible for any use that may be made of the information contained therein.

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Date: May 2026

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