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Investigating Power Module Recycling

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Abstract

Power modules (PM) from waste electrical and electronic equipment (WEEE) represent an underexploited source of strategic secondary raw materials. This study analyzes the material composition of different PM types to identify key challenges and opportunities for their end-of-life management in power electronics. A comprehensive inventory model of PM recycling is developed to identify the processes responsible for the main environmental impacts. Based on this model, the influence of material selection, subsystem design, and assembly on recycling performance is assessed, and alternative recycling scenarios are explored to reduce environmental impacts. A life cycle assessment (LCA) of the end-of-life stage supports the analysis.

1 Introduction

WEEE represents a growing source of valuable materials. In 2022, the global quantity of WEEE reached 62 Mt, with an estimated annual growth rate between 3% and 5%. This volume could rise to 82 Mt by 2030[1-2]. Despite their richness in precious metals (Ag, Au), a large share of their material recovery remains limited: in Europe, only 42.8% of WEEE were sent to recycling facilities in 2022 [2], and recycling is mostly concentrated on a few critical and/or precious materials, leading to very low actual recycling rates. Moreover, although the annual recycled quantity of WEEE is gradually increasing every year, the production of electrical and electronic equipment EEE is growing even faster, resulting in a reduction of the WEEE recycling rate [2]. Among this waste stream, power electronics converters are among the most difficult products to recycle because they are highly integrated on printed circuit boards (PCB), they are made of numerous materials in very diverse quantities, including small to very large parts, difficult to sort out thanks to traditional shredding systems. Within power electronics devices, specific recycling of PM, illustrated **Figure 1** with a cross section and the list of their main parts, remain particularly underexplored because of their low concentrations in precious metals [3]. Also, they are in between traditional process flows, too large to be shredded with PCBs, highly integrated and too heterogeneous and with a poor material affinity to be recycled with WEEE large parts such as casing, housing or even cabinet, etc... Our research aims to investigate, based on existing recycling steps and processes, how PM recycling could be improved, considering the opportunity to conduct a specific process flow for PM. First, PM material share and assembly diversities are pointed out, resulting in poor actual recycling rates. Estimates are given to highlight the limited amount of material that can actually be recycled today from PM. Deriving recycling inventories with respect to PM bill of materials and assembly/disassembly technologies, LCA is used to determine whether increased material recovery rates can be achieved without generating disproportionate environmental impacts (and costs). It is also studied how and at which “costs” it would be possible to maximize PM

material circularity, not only for the critical metals but also for the other materials.

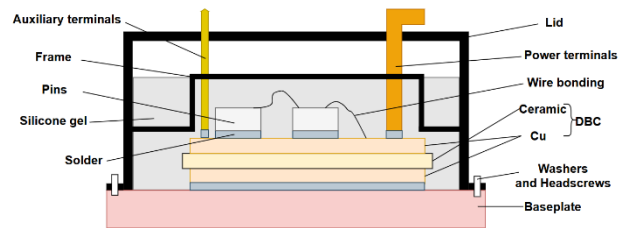


Fig. 1 Cross section schematic of a conventional PM adapted from [4].

2 Challenges in recovering materials from PMs

Given this context, it becomes crucial to look more closely at PMs, which represent a highly complex but largely overlooked fraction of WEEE. The following section examines the present situation of PMs by detailing their material composition, assessing the recycling and reuse rates of the elements they contain, and identifying the technological and structural barriers that currently limit their material recovery. Understanding these limitations is essential to justify the need for improved recycling strategies and more circular design approaches.

2.1 Material composition of PM

Building on this assessment, the next part provides a detailed overview of the different sub-systems that constitute a power module and the wide range of elements associated with each of them. This breakdown underscores an important challenge: a single PM already involves a large diversity of materials, which substantially increases the complexity of recycling processes. This is illustrated on **Figure 2**, with the material breakdown by subpart of a traditional power module from Semikron [3].

RR above 50%, while almost all EoL-RIR values remain below 30%. This indicates that, under currently existing recycling conditions, most materials are not successfully recovered. Even when materials are successfully recovered, their effective reuse potential remains limited due to issues such as insufficient purity or other degradation mechanisms. Consequently, current practices are still far from achieving a meaningful recovery and reuse of materials contained in PMs waste.

As mentioned above, it is also important to emphasize that these values originate from broad datasets covering the entire spectrum of WEEE, while PMs represent only a very small fraction of this waste stream. PMs generally contain a wider range of heterogeneous materials and have relatively low mass, which increases the risk of material loss or undesirable mixing during recycling processes. As a result, the actual recovery rates of PMs are likely to be lower than the values listed in **Table 2**. Furthermore, the treatment of waste metals requires careful consideration of incompatibilities between materials [11]. Depending on the configuration of the recycling process, recovery efficiencies can vary significantly. If metals contained in waste PMs are not sorted in advance, recycling facilities typically prioritize the most abundant and valuable metals. Based on the material distribution shown in **Table 1**, waste PMs are usually treated following the copper recovery pathway. Gold and silver, which are compatible with copper, can also be recovered, whereas aluminum, which is incompatible with copper, is more likely to exhibit recovery rates far below those reported in **Table 2**.

Figure 3 above points out that PM waste from various types of PM has two main metals contributors, copper and aluminum. These two metals are very difficult to recover together, and one process is usually preferred for valuable reasons as mentioned before. But the risk is also to mix PM waste with large housing, casing and cabinets coming from WEEE, with larger contributions and concentrations in housing metals such as iron and aluminum. In such a

WEEE mixture, PMs recycling becomes a nightmare, aluminum and copper being difficult to recycle with iron. In the next section, we investigate the opportunity to develop dedicated PM recycling process flows in order to increase the material recovery rates. This is today a fictive way to manage PM waste but it helps identifying the lever we may have in our hands if PM end of life could be managed independently or with other devices, very compatible from recycling point of view, calling for better subsystem sorting at the early stage of the recycling process, thanks to clear identification and effective disassembly pathway [12].

Table 2 EoL-RR and EoL-RIR of selected construction materials of PMs. (Data correspond to Europe; * indicates global data; ** indicates data related to WEEE.)

Raw material	EoL-RR	EoL-RIR	References
Aluminium	60%	21%	[2,9]
Antimony	49%	28%	[9]
Chromium	59%	21%	[9]
Cobalt	33%	22%	[9]
Copper	53%	30%	[9]
Gold	15%**	24%	[8,9]
Iron(steel)	75%	32%	[9]
Lead	52-95%*	83%	[6,9]
Magnesium	14%	7%	[9]
Manganese	40%	9%	[9]
Nickel	60%*	16%	[2,9]
PBT, PPS	0%	0%	[10]
Silica	26%	1%	[9]
Silver	15%**	20%	[7,8]
Tin	4%*	26-32%	[2,6]
Zinc	4%*	40%	[2,9]
Zirconium	12%	0%*	[9]

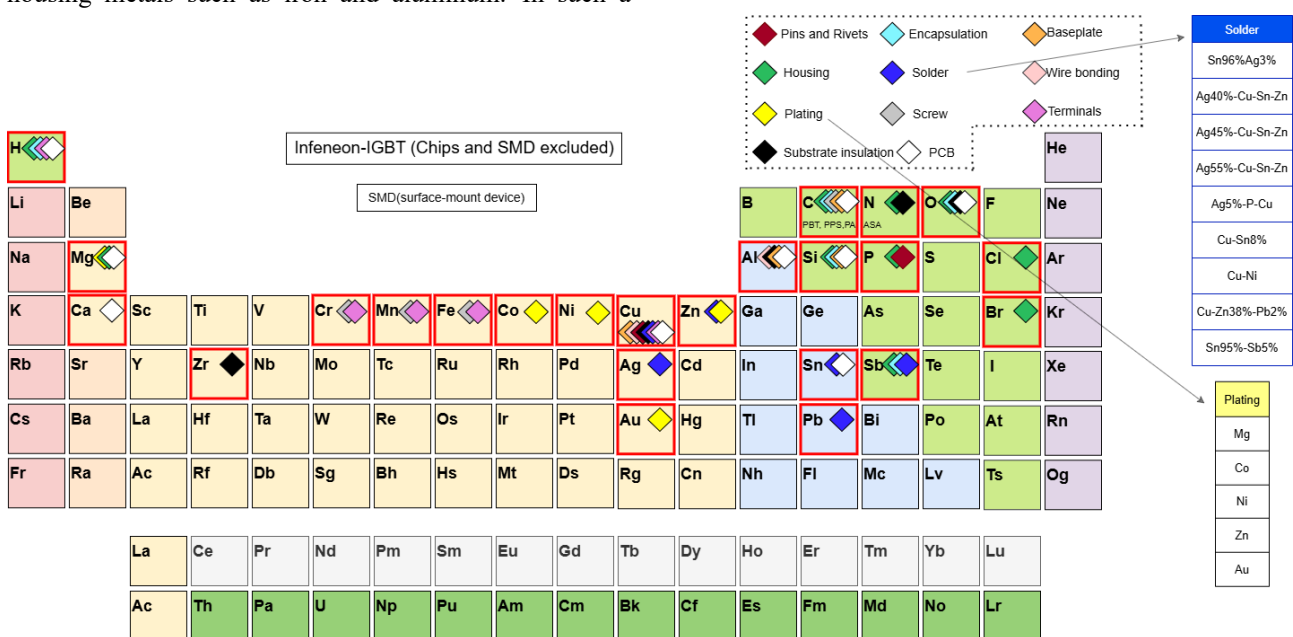


Fig. 3 Color-coded periodic table of elements found in PM materials.

3 PM recycling

To further analyze the recycling performance of PMs within the existing WEEE management system, this study adopts the copper recovery route (currently the most widely used industrial process) as the reference model. This choice is motivated by two considerations: first, copper represents a major fraction of the PM's material composition; second, copper holds relatively high value in the secondary materials market. Therefore, this route provides a realistic approximation of how PMs could be treated under actual end-of-life conditions.

3.1 WEEE recycling process

Since no dedicated recycling pathway currently exists for power modules, PMs are in practice fully integrated into the conventional WEEE recycling stream. This means that PMs enter the same processing chain as large quantities of heterogeneous E-waste, without being identified or selectively dismantled despite their highly integrated metal-ceramic-resin-plastic composite structure. As a result, the recycling efficiency of PMs depends heavily on the mechanical and metallurgical operations commonly used in WEEE facilities. The key steps of this generic recycling chain, which are illustrated in **Figure 4** [6,13], can be summarized as follows:

The first step is the collection step. End-of-life power electronic converters containing PMs are initially collected through standard WEEE channels. While preliminary disassembly of large electrical and electronic equipment (such as vehicles or solar panels) is performed at this stage, to our knowledge, no specific identification or sorting of PMs takes place. Consequently, PMs remain embedded within the larger assemblies they originate from (e.g., converters, inverters, cooling system, casing, busbar...), resulting in a collected waste stream composed of a heterogeneous mixture of plastics, metals (ferrous and non-ferrous), ceramics, and electronic components.

Once collected, the mixed WEEE is fed into mechanical shredding units. This step reduces the size of the elementary waste parts in order to expose the internal materials and facilitate subsequent sorting. Because PMs are particularly robust devices (thick base plate,

encapsulated resin, substrates insulation, solder joints), most of their internal sub-systems are not selectively liberated during shredding: instead, they are fragmented into heterogeneous mixtures of metals, plastics, ceramics and alloys. This mixing strongly limits the recoverability of individual materials.

After shredding, a sequence of sorting operations is applied to separate materials according to their physical properties, typically combining magnetic separation, eddy-current separation, density-based separation, electrostatic sorting or optical sorting. For PMs, this stage mainly allows the extraction of ferromagnetic metals (steel) and plastics. However, due to the small size and high integration of the PM fragments, the efficiency of separation for copper, aluminium, solder alloys, and ceramic particles remains limited. Most of these materials remain mixed within the nonferrous, metallic fraction and are lost a little in the non-metallic fraction.

After sorting, the non-metallic fraction is transported to a separate facility for dedicated processing (waste plastic processing), including washing, drying, and density-based plastic separation. Plastics bonded to metals or containing flame retardants are separated out. Recoverable polymers such as PP, PS, and PET can be extruded into recycled granulates. However, plastics from PMs (including PBT, PPS, and PA) are difficult to recover due to flame retardants and metal bonding, and represent only a small fraction of the WEEE plastic stream. Their low concentration and the high cost of selective sorting make recovery technically and economically unfeasible [13]. Consequently, this fraction is usually incinerated for energy recovery, contributing to additional environmental emissions.

The metal-rich fraction is transported to a metallurgical processing facility for further purification. Depending on the facility design and economic considerations, metals can be recovered through pyrometallurgical processes (smelting, oxidation, refining) or hydrometallurgical methods (leaching, precipitation, electrolysis). Sometimes, a combination of both methods is employed to improve purity and recovery rates. However, both approaches have inherent limitations. Pyrometallurgy separates metals based on their different melting points but is highly energy-intensive and prone to forming undesirable alloys by

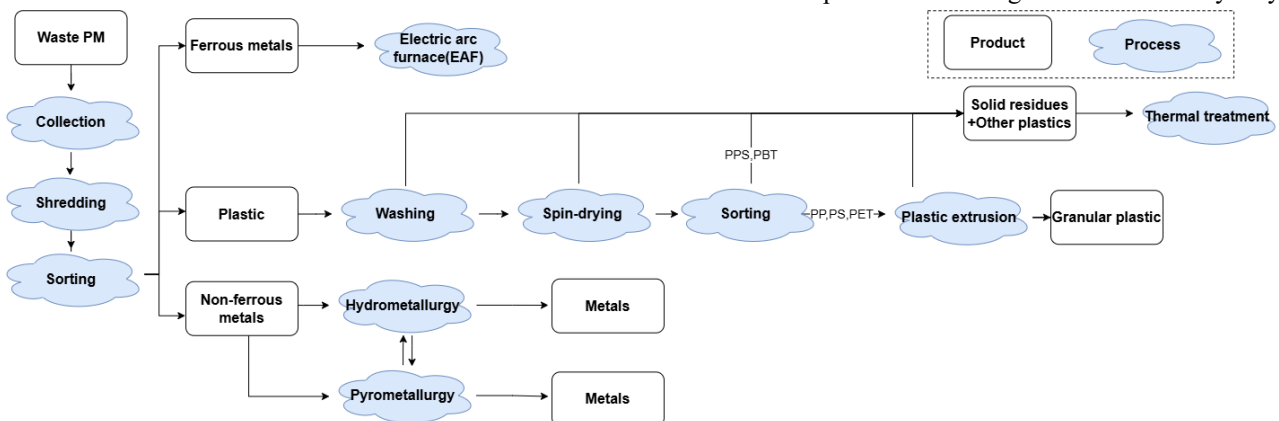


Fig. 4 Current process steps for recycling WEEE.

melting incompatible metals together, which reduces the overall metal recovery rates. Hydrometallurgy, on the other hand, consumes large amounts of solvents and requires multiple distinct solvent treatments, generating substantial liquid waste. Currently, regardless of the method used, copper and nickel can be relatively efficiently separated, but aluminum (which is incompatible with copper) as well as precious metals such as gold and silver, which are present in low concentrations, often suffer significant losses during processing [7,11].

3.2 Life cycle inventory (LCI) of waste PM recycling

The Ecoinvent database (version 3.11, Ecoinvent Association, Zurich, Switzerland), a comprehensive life cycle inventory database widely used for modelling background processes in environmental assessments, was employed as the primary data source [14]. To construct model of the LCI of PM waste recycling, all input and output flows associated with each treatment process must be collected. In order to improve the accuracy and transparency of the model, most process data (including collection, shredding, sorting, thermal treatment, hydrometallurgy, pyrometallurgy and transport between the different processing stages) were obtained from the Ecoinvent database and subsequently adjusted to better reflect the actual conditions of PM recycling in Europe. For processes that are not available in Ecoinvent, or for which the existing datasets differ significantly from the real characteristics of PM waste (e.g., similar processes but applied to devices with very different material compositions), complementary data were gathered from mechanical datasheets and engineering calculations. In this inventory, material losses associated with each process were not explicitly considered. Additional uncertainties were then assigned to these estimated flows to strengthen the reliability of the inventory. For the Ecoinvent-based datasets, uncertainties already included in the database were retained without modification.

This approach enables the construction of an LCI that approximates the real environmental profile of PM recycling in Europe to our best knowledge. It has to be clarified that some of the estimated inventories are subject to significant improvement. The dataset shared in this paper are considered as a starting point that is expected to motivate experts in supplying data that are more accurate whenever it applies. A more comprehensive and detailed LCI will be released in future work. The following two examples illustrate how data were collected and adapted during this process.

- **Process available in the Ecoinvent database**

Table 3 illustrates the thermal treatment of plastics. This process exists in Ecoinvent, where it corresponds to the incineration of non-recyclable plastics present in WEEE. Since waste PMs are part of the WEEE stream, this dataset can serve as a first approximation of the treatment conditions relevant to PM plastics.

And then, regional differences in operational practices may lead to variations in environmental impacts. In Ecoinvent, datasets aligned with the European situation are indicated by the geographical code “RER”. Thus, for certain flows (such as water use) the geographical reference was switched from the global (GLO) dataset to the Europe (RER) dataset, without modifying the magnitude of the flow itself, but only adapting its regional context to better reflect European conditions.

Table 3 Input and output flows for incinerating 1 kg of waste plastic [13,15].

Process	Data	Flow type	Decision
Thermal treatment (Treatment of waste plastic, consumer electronics, municipal incineration - ecoinvent)	market for waste plastic, consumer electronics-RoW	Transport	Retain
	1kg plastic waste	Input	Retain
	Anhydrous ammonia	Input	Retain
	Heat	Input	Retain
	Vanadium V	Emission to water	Remove
	Bromine	Emission to water	Retain

Regarding thermal energy requirements, although the proportion of plastics differs between WEEE and waste PMs, **Figure 4** shows that the non-recyclable mixed plastic fraction in WEEE (after removal of recoverable polymers such as PP, PS, and PET) can be considered a reasonable approximation of the plastic composition found in PMs. Moreover, according to the heat transfer equation :

$$Q = c \times m \times \Delta T \quad (3)$$

Where Q is the thermal energy required (J), c is the specific heat capacity of the material ($J * kg^{-1}K^{-1}$), m is the mass of the material (kg), and ΔT is the temperature increase (K). The incineration of halogen-containing plastics must reach temperatures above 1000 °C, a value that can be considered sufficiently well established to be used as ΔT in the calculation, in order to ensure complete oxidation of halogenated compounds [16]. Since the specific heat capacity of mixed plastics is strongly influenced by flame retardants (which are similar in the plastics found in WEEE and waste PMs) the energy demand can be assumed to be comparable in both cases.

During incineration, some pollutants are directly emitted into the environment. Flows corresponding to substances not present in waste PM plastics were removed, while flows corresponding to pollutants relevant to waste PMs were retained with unchanged mass values.

Through these adjustments, all input and output flows associated with the thermal treatment process were collected and adapted, resulting in the final dataset illustrated in **Table 3**.

- **Process not available in Ecoinvent, or significantly different from the real case**

The second example concerns the pyrometallurgy process with an illustration of main input and output flow in **Table 4**. Although Ecoinvent contains datasets for

pyrometallurgical treatment, the available dataset corresponds to end-of-life lithium-ion batteries, whose composition differs substantially from waste PMs. As a result, the associated energy demand cannot be directly transferred to waste PM treatment.

Table 4 Pyrometallurgical process: input and output flows for 1 kg of non-ferrous metal waste [5,14].

Process	Data	Flow type	Decision
Pyrometallurgical process	transport, freight, light commercial vehicle, fleet average	Transport	Retain
	1kg non-ferrous metal waste	Input	Modify
(Treatment of non-Fe-Co-metals, from used Li-ion battery, pyrometallurgical processing - ecoivalent)	4.3 MJ Heat	Input	Modify
	0.04kg oxygen, liquid	Emission to water	Modify
	Copper ion	Emission to air	Modify
	Sulfur dioxide	Emission to air	Remove
	Fluorine	Emission to air	Remove

To estimate a more realistic value, the mass fractions of the different materials contained in batteries were first collected [17]. And then, the mass fractions of the non-ferrous metal components in waste PMs were recalculated based on **Table 1**, after removal of the plastic and ferrous fractions during sorting. Using these mass fractions and the specific heat capacities of each material, the specific heat capacity c of each material mixture was calculated as shown below :

$$c(\text{mix}) = \sum_{i=1}^n w_i \times c_i \quad (4)$$

Where $c(\text{mix})$ is the specific heat capacity of the mixture ($J * kg^{-1}K^{-1}$), w_i is the mass fraction of component i in the mixture (%), c_i is the specific heat capacity of component i ($J * kg^{-1}K^{-1}$), and n is the number of components in the mixture.

In pyrometallurgical treatment of mixed aluminum and copper fractions, operating temperatures are typically raised to around 1200 °C to ensure complete melting of both metals [18]. This temperature was therefore used to determine the temperature increase ΔT . Based on equation (3), the theoretical thermal energy $Q(\text{theoretical})$ required to heat 1 kg of waste batteries and 1 kg of the non-ferrous metal fraction from waste PMs was calculated separately, yielding values of 0.24 kWh and 0.20 kWh, respectively.

By comparing the theoretical energy demand calculated for batteries with the actual energy consumption $Q(\text{actual-batteries})$ reported in the corresponding Ecoinvent dataset, a correction factor r :

$$r = \frac{Q(\text{actual-batteries})}{Q(\text{theoretical-batteries})} = 6 \quad (5)$$

was derived. This factor was then applied to estimate the actual energy requirement $Q(\text{actual-PMs}) = 4.3$ MJ for pyrometallurgical treatment of the non-ferrous metal fraction from waste PMs. In addition, the difference between was used to define an uncertainty range, reflecting the limitations of the estimation method and increasing the robustness of the subsequent LCA results.

As for direct emissions released to the environment during pyrometallurgical processing, they were adapted using the same approach as for the thermal treatment of plastics: pollutants not relevant to waste PMs were removed, while relevant emissions were retained and scaled by a factor d , defined as the ratio between the theoretical thermal energy required for waste PMs and that for waste batteries, i.e.

$$d = \frac{Q(\text{theoretical-PMs})}{Q(\text{theoretical-batteries})} = 0.85 \quad (6)$$

The resulting dataset provides a reasonable approximation of the actual inflows and outflows associated with pyrometallurgical treatment of waste PMs in Europe.

3.3 LCA of waste PM recycling

Based on PM recycling inventory established in the previous section, the environmental impacts associated with waste PM recycling were evaluated using LCA. **Figure 5** illustrates the relative contribution of each recycling process to the different environmental impact categories. **Table 5** provides a detailed description of each impact category and its corresponding unit, serving as a complementary reference for interpreting **Figure 5**. It is important to emphasize that each impact category has to be analyzed independently, and direct comparisons between the absolute heights of bars across different categories should be avoided, as the units and characterization factors differ between impact indicators.

The results show that waste plastic treatments (washing, drying, sorting, and thermal treatment) contribute significantly to Climate Change and Ecotoxicity: freshwater. Among the recycling steps, thermal treatments are the dominant contributor, mainly due to its high energy demand and the associated CO₂ emissions released during incineration. This result reflects current PM recycling practices, in which plastics that cannot be mechanically recycled (often due to the presence of flame retardants or metal bonding) are directed toward thermal treatment. The high energy demand of incineration, combined with direct gaseous emissions, which make the strong contributions observed for climate change and freshwater ecotoxicity.

In contrast, non-ferrous metal treatment processes (pyrometallurgy and hydrometallurgy) dominate the impacts related to the remaining environmental impact categories. Among which, hydrometallurgical processing shows a more pronounced contribution to Material Resources: metals/minerals, as it requires the addition of chemical reagents to treat waste PMs. The production of these chemical reagents increases the demand for mineral and metal resources, thereby leading to higher impacts in this category. whereas pyrometallurgical processing exhibits more significant contributions across the remaining environmental impact categories, with impacts approximately two to three times higher than those of hydrometallurgy. This difference can largely be explained by the high operating temperatures required in pyrometallurgy, typically around 1200 °C [18], to ensure complete melting of mixed aluminum and copper fractions, resulting in substantial energy consumption. Nevertheless, it should be noted that the environmental performance of

hydrometallurgy is strongly dependent on the choice of chemical reagents used for metal separation. In the Ecoinvent dataset, ammonia is used as the separation agent for copper and aluminum, which leads to relatively limited environmental impacts due to its weak basicity. However, alternative hydrometallurgical routes based on strong acids such as sulfuric acid are also used in WEEE recycling, and these would result in significantly higher environmental impacts, potentially exceeding those of pyrometallurgy.

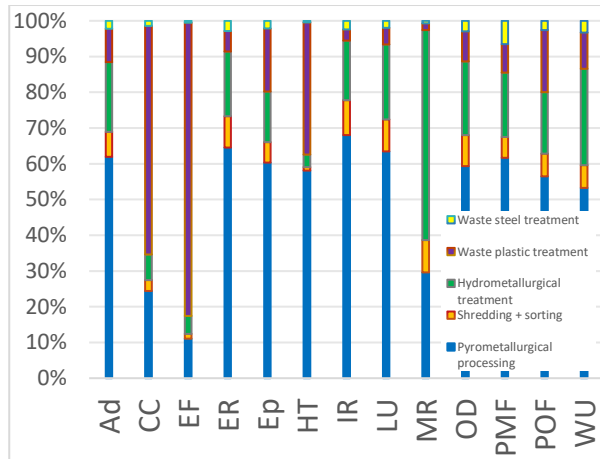


Figure 5 Contribution of individual recycling processes to the environmental impact categories of waste PM recycling.

Table 5 Environmental impact categories and their units used in **Figure 5**.

Abbreviation	Impact category	Unit
Ad	Acidification	mol H ⁺ eq
CC	Climate Change	kg CO ₂ eq
EF	Freshwater Ecotoxicity	CTUe
ER	Energy Resources	MJ
Ep	Eutrophication	kg P-eq
HT	Human toxicity, carcinogenic	CTUh
IR	Ionising Radiation – human health	kBq U235-eq
LU	Land Use	dimensionless
MR	Material resources: metals/minerals	kg Sb-eq
OD	Ozone depletion	kg CFC ⁻¹¹ -eq
PMF	Particulate matter formation	disease incidence
POF	Photochemical Oxidant Formation – human health	kg NMVOC-eq
WU	Water Use	m ³ world eq deprived

Figure 6 presents the influence of uncertainties associated with non-Ecoinvent and estimated data on the overall LCA results. The uncertainty analysis indicates that Energy Resources and Ionizing Radiation – Human Health are the most sensitive impact categories, while uncertainties have a limited influence on the other categories. This behavior can be directly linked to the pyrometallurgical energy modeling: the theoretically calculated thermal energy required to melt the non-ferrous metal fraction of waste PMs was found to be approximately 6 times lower than the actual energy consumption reported in the reference Ecoinvent dataset. However, it should be noted that the current assumption of a 10% uncertainty for the electricity consumption of washing, spin-drying, and sorting processes does not reflect reality. This will be further refined in future improvements, which may lead to an increase in the uncertainty shown in parts of **Figure 6**. Overall, the results indicate that the environmental impacts associated with waste PM recycling are mainly driven by energy-intensive operations, particularly plastic thermal treatment, hydrometallurgical processing and pyrometallurgical processing. The analysis further demonstrates that both process selection and data accuracy (especially assumptions related to energy consumption) play a critical role in shaping the LCA outcomes. The more accurate and representative the collected data, the more reliable the resulting LCA results. These findings support the need to investigate alternative recycling pathways and more targeted PM treatment strategies, which are discussed in the following sections.

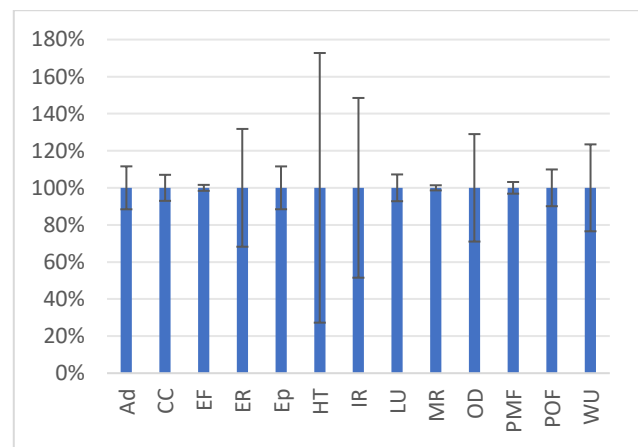


Fig. 6 Influence of inventory data uncertainties on the environmental impact results of waste PM recycling.

4 Discussion

The results presented in Section 3 clearly show that the environmental impacts associated with waste PM recycling are largely driven by energy-intensive end-of-life treatments, in particular plastic thermal treatment and pyrometallurgical processing. These impacts are not inherent to PM recycling itself, but rather arise from the strong material heterogeneity of PMs and from the absence of dedicated recycling routes. Indeed, if only a small share of metals is actually recycled from pyrometallurgy whereas a large amount of material was melted, then the recycling

process is largely inefficient. It would make sense to explore potential levers to improve PM recycling performance by acting at two complementary levels: (i) the design of PM sub-systems, and (ii) the introduction of additional or alternative recycling process steps enabled by redesigned PMs. The objective is to increase subsystem-level recycling rates and more specifically the end of life recycled input rate EOL-RIR, to close material loops, and minimize the pressure on raw material depletion while keeping the environmental burden associated to recycling as minimum as possible.

Material heterogeneity reduction at design phase is probably a perspective, trying rely on a single metal in large share and on a limited number of additional metals, in small shares, and as compatible as possible with the recycling process of the main one. As an example, that has not been yet quantified, using DBA with AISiC base plate and/or DBC with copper baseplate could be one way to avoid two non-compatible metals with large share. Of course, it is clear that material selection answer to many different needs and requirements, for performance but also device lifespan, and all this remains a question of compromise. We want here to highlight that, whenever it is compatible with a minimum impact on other design criteria, material affinity is important.

Also, plastic alloys and metal alloys, although coming with interesting performances, are difficult to recycle properly because each alloy has its own material concentration, leading to waste mixture difficult to reintroduce in the supply chain without multiple recovery steps.

The second PM design improvement toward higher EOL-RIR and recycling rate is related to the ability to disassemble and to separate “easily” sub parts that are difficult to recycle together.

Although lots of work remains to be carried out to improve the PM end-of-life inventories, alternative recycling scenarios could be evaluated using LCA and compared with the current process for recycling WEEE described in Section 3. The results could not only allow the identification of scenarios that reduce the environmental impacts of PM recycling, but also could provide valuable insights for power module designers. In particular, they show that how material selection, reduction of material diversity, avoidance of incompatible material combinations, and design for sub-system-level disassembly influence the overall environmental performance.

From a circular economy perspective, such recycling-oriented design strategies not only promote closed-loop recycling of critical and precious metals, but also enhance the circularity of bulk materials such as copper and aluminum in our quest to limit all raw material depletion. This underscores the importance of integrating environmental considerations at the design stage, rather than relying solely on downstream recycling technologies to compensate for material complexity.

5 Conclusion

This study assessed the recycling of waste PMs within the current WEEE framework using life cycle assessment,

with the aim of identifying the main environmental impact drivers and improvement levers. The results show that recycling impacts are largely driven by energy-intensive processes, particularly plastic thermal treatment and pyrometallurgical processing of mixed non-ferrous metals, which are strongly influenced by material heterogeneity and incompatibility.

From this work, alternative recycling scenarios based on design for recycling principles could be proposed, focusing on reducing material diversity, avoiding incompatible material combinations, and enabling sub-system disassembly. These strategies hold potential to lower environmental impacts and improve material recovery, but further research and detailed assessment are needed to validate their effectiveness. This underlines the importance of considering environmental impacts early in the design phase to foster more sustainable and circular power electronics in the future.

Acknowledgement

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