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# Reconciling Twin Transitions with sustainability. A Circular Economy perspective.

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## Introduction

The Twin Transitions concept has garnered significant attention in current policy and economic literature debates (European Commission, 2021; Timmermans et al., 2023). This concept highlights the interconnection between the green transition, geared toward achieving climate goals, and the digital transition, aimed at disseminating digital technologies and infrastructures. In the strategic vision of the European Union (Muench et al., 2022), these transitions form the foundation for delivering a sustainable, fair, and competitive future, not only independently but also through mutual reinforcement.

This notion finds support in recent research providing evidence of the co-adoption of digital technologies, such as artificial intelligence and the Internet of Things, alongside eco-innovations in firms (Barbieri et al., 2020; Cicerone et al., 2023; Montresor and Vezzani, 2023). Additionally, environmental benefits stemming from the adoption of digital technologies have been observed (Del Río Castro et al., 2021; Vinuesa et al., 2020). Over the past decade, a stream of research has recognized the relationship between information technologies and eco-innovations as dual (Faucheux and Nicolaï, 2011) and, even before, the roots of a positive view on the interconnection between technologies and environmental sustainability can likely be traced back to economists of innovation (Freeman, 1992; Rosenberg, 1973) as well as environmental economists (Bretschger, 2005; Dasgupta and Heal, 1974; Grossman and Krueger, 1995).

However, complete compatibility between digitalization and environmental sustainability cannot be assumed. Digital technologies, among other adverse impacts, can increase energy consumption (Lange et al., 2020; Røpke, 2012), emissions (Bianchini et al., 2022), and lead to various forms of rebound effects (Barteková and Börkey, 2022). Thus, the debate on the alignment between the current technological transition and environmental sustainability is highly fragmented and contradictory (Coad et al., 2021).

In this thesis, to systematize the exploration of the relationship between technologies and their environmental impacts, I leverage the concept of circular economy (CE). The CE ultimate goal is to minimize both natural resource use and the flows of waste generated by human activities returning to the environment (Korhonen et al., 2018). CE is grounded in Life Cycle Thinking, an approach that seeks to avoid "problems shifting" in the environment, which involves transferring problems from one life cycle stage, agent, location, or environmental medium to another without solving the issue (European Commission, 2012; Moraga et al., 2019). Therefore, adopting CE as the underlying model for environmental sustainability emphasizes the need for techno-economic systems to develop within the limits imposed by nature — its limited capacity to absorb the waste of human activities and provide resources as production inputs (Brock and Taylor, 2005; Daly, 1990).

In this study, I specifically examine two environmental issues situated at both ends of the technology life cycle: the consumption of mineral resources embedded in technologies and the management of electronic waste.

Concerning the first issue, the definition of the concept of "resources criticality," emerging approximately fifteen years ago through contributions from academics in the fields of Resources Management and Industrial Ecology (Graedel and Reck, 2016), has enabled scholars and policymakers to recognize that high-tech value chains and technologies supporting the green and energy transitions depend on a set of mineral resources, whose supply chain is highly concentrated in a few countries (Carrara et al., 2023; European Commission, 2020; IEA, 2021). Current strategies employed by policymakers to mitigate supply risks for critical raw materials (CRM) primarily focus on diversifying the supply among different countries (European Commission, 2023). However, it is evident that achieving high levels of circularity, both in terms of decoupling techno-economic systems from material resource use and recycling materials, is crucial for finding ultimate solutions. This is particularly true considering that CRM are predominantly supplied by developing countries, where extraction activities contribute to significant environmental and social damages (Martinez-Alier, 2021; Sovacool et al., 2019), while social acceptance of mining activities in developed countries is very limited (Mateus and Martins, 2021). From an academic perspective, the topic of CRM and their use in modern technologies remained confined within the original research streams until very recently[1]. Consequently, there is ample space and a pressing need for policy support and research contributions from additional fields in Economics.

The second environmental issue that I focus on is the management of end-of-life technologies, specifically electronic waste. The widespread integration of electrical and electronic equipment into various aspects of modern life has led to a rapidly growing and massive generation of electronic waste and waste batteries (Forti et al., 2020). The topic has been explored over the past quarter-century, initially to address the high toxicity of these waste types (Pérez-Belis et al., 2015) and the substantial export flows directed towards developing countries (Puckett et al., 2019). Only recently electronic waste and waste batteries have been considered as potential resources (Moïsé and Rubínová, 2023;

[1] Conducting a search on Scopus in December 2023 revealed that the number of papers explicitly mentioning "critical raw materials" in their title, abstract, or keywords within key scientific journals in the fields of Economics of Innovation and Environmental Economics (including Research Policy, Industry and Innovation, Technological Forecasting and Social Change, Industrial Change and Structural Dynamics, Environmental Innovation and Societal Transitions, Ecological Economics, Environmental and Resource Economics, Journal of Environmental Economics and Management) is zero.

Parajuly et al., 2019; Rosendahl and Rubiano, 2019). Indeed, their high concentrations of CRM and their current and future abundance in developed countries with low CRM endowments present a CE opportunity. Consequently, the acknowledgment and exploration of the structural economic dynamics (trade flows, knowledge base, innovations, value chains) that need to change in order to transform this waste into a resource are only hinted at in current literature (Cecere and Martinelli, 2017).
As Kronenberg and Winkler (2009) argued, waste should be viewed in an evolutionary perspective: it is the economic and normative conditions that can transform waste from a form of non-marketable pollution into a marketable, potentially valuable commodity. Acknowledging the significance of this is crucial to unfold CE opportunities that counteract environmental issues associated both with the early and end life cycle stages of (green) technologies. Unfortunately, there is currently a lack of research exploring waste "marketization" processes, their structural impacts on economies and their determinants.

In order to explore the complex connection between Twin Transitions and environmental sustainability and unveil opportunities of reconciliation, the thesis develops three chapters.

In the first Chapter, we investigate the material footprint of the recent technological trajectory. Previous literature in this field has identified three primary environmental challenges: the overall metabolism of the technological system; the growing material complexity of technologies; their reliance on critical or geologically scarce materials. These challenges are often examined in isolation with few studies that strived to develop systematic assessments and policy implications (Santarius et al., 2023). However, the relationship between the evolution of technologies and materials use has been described as complex, i.e. multi-dimensional, and interdependent, i.e. the availability of natural resources shapes the evolution of technologies and vice versa (Berkhout and Hertin, 2004; Boons and Wagner, 2009; Li et al., 2024). Consistently with this vision, to develop a systematic, multi-level framework that simultaneously describes the waves of diffusion of technologies and the mass of materials embedded in each type of technology, we build on the concepts of scale (Gibson et al., 2000) and product ecosystem (Levine, 1999). The former allows to explicitly define the boundaries and levels of our system of analysis, while the latter is useful to represent the fact that technologies are not used in isolation, but rather in combination, and their diffusion co-evolves. By exploiting a database describing the diffusion, the weight and the material composition of a large set of technologies over more than three decades, we are capable to disentangle material footprint patterns at the product level, at the level of products characterized by interrelated functions, and at the aggregate technological trajectory level. Results indicate that strategies improving the material footprint at one level of analysis may have adverse effects at others. We argue that sustainability assessments of the material base of technologies should be incorporated into climate and technology planning. From the theoretical side, the Chapter aims to support a connection between Ecological Economics and Industrial Ecology methodological and conceptual approaches, on the one hand, with Technological and Sustainability Transitions' ones, on the other.

The second Chapter explores the effectiveness of the most systematic policy in the field of electronic devices and waste, namely Extended Producer Responsibility (EPR) (Lauridsen and Jørgensen, 2010). EPR is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product life cycle, that is when the product is discarded by the final user (Lifset, 1993). By shifting waste management (economic, physical and informative) responsibilities from municipalities to producers, the policy is expected to deliver both "downstream" impacts, improving waste management outcomes, and "upstream" impacts, spurring eco-design innovations. In other words, EPR has been conceived for the incorporation of total product life-cycle costs into production decisions. This Chapter provides the first systematic literature review aiming to comprehensively examine the outcomes of EPR implementation in the electronic waste scope, by adopting a product life cycle perspective. It highlights the accomplishment of important downstream goals of EPR on electronic waste, but also a limited upstream effectiveness of the policy. Discrepancies between CE goals and EPR achievements and implementation are classified into seven areas, covering the entire product life cycle and also representing domains of policy recommendations. Moreover, the paper identifies a number of future research directions that would support the alignment of EPR policies and CE objectives in the electronics value chain.

The third Chapter leverages on the results of the previous one and investigates the effect of EPR policies on trade flows of waste, focusing particularly on the case of waste batteries. The literature on international trade in waste typically considers this commodity as a source of pollution, i.e. a negative environmental externality (Kellenberg and Levinson, 2014). In parallel, international circular economy networks offer environmental and economic benefits, such as ensuring sound environmental management of waste flows; recovering materials that can be substituted to virgin ones, alleviating the pressure on resource extraction and lowering the overall environmental impact of value chains; reducing supply risks and price volatility of raw materials. Moreover, the international trade of waste can support the development of economies of scale for waste treatment, of waste management technologies, and an outlet for recycled materials (Egger and Keuschnigg, 2023; Kojima, 2020)

As it emerges from Chapter 2, EPR is considered a key policy for the marketization of waste, i.e. an instrument to set the normative and economic conditions that can effectively turn waste into a resource (Gregson et al., 2013; Kama, 2015). Anyway, the spillover effects of the implementation of EPR regulations on the trade of waste have not yet been empirically investigated.

In particular, we decided to focus on the case of waste batteries because of their nature of hazardous waste, which necessitates advanced and specific treatments, as well as their high contents in CRM. Additionally, batteries have been regulated through EPR mechanisms in a relevant number of countries worldwide. The review of these regulations from a variety of sources also represents a contribution to the literature.

In order to empirically assess the ambiguous effect of EPR on the export of waste batteries, compared to the trend of other waste commodities not affected by the policy, we adopted a combination of gravity and difference-in-differences models on BACI data for international bilateral trade flows. In addition, to explain trade flows, we also made use of data on the number of recycling facilities as well as recycling patents at the country level. Specifically, patent data on waste batteries recycling and recycling of other streams of waste were selected according to the OECD ENV-TECH classification (Haščič and Migotto, 2015).

At a general level, this paper aims to stimulate the discussion on the mechanisms that can support the marketization of waste and how these interplay with international trade. A better understanding of the potential interactions between international trade and waste policies is claimed to ensure an effective global transition to a circular economy (Yamaguchi, 2021).

Ultimately, the Thesis elucidates how both market-driven and policy-led circular economy strategies can engender structural changes capable of driving, at least partially, the Twin Transition dynamics toward achieving effective sustainability outcomes.

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# How technological trajectories affect material consumption: a longitudinal and multi-level analysis of the EU electronics market. [1]

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**Abstract**

Literature dealing with the exploration of technological evolution and its material footprints (MF) suggests three primary challenges: the overall metabolism, i.e. consumption of materials, of the technological system; the growing material complexity of technologies; their reliance on critical or geologically scarce materials. These challenges are often examined in isolation, overlooking their interrelated nature. This study proposes a systematic, multi-level perspective, leveraging concepts from Ecology and Industrial Ecology, such as scale and product ecosystem. Indeed, when investigating empirically the MF of technological trajectory (TT), it is essential to acknowledge the co-evolution of technologies and to explicitly define the limits of the system under scrutiny. The dataset, detailed in technology and material identification, enables a longitudinal, bottom-up analysis disentangling MF patterns at the product, functional group, and aggregate TT levels over three decades. Results indicate that strategies improving MF at one level may have adverse effects at others. The study reveals that reductions in materials consumption stem primarily from changes in the composition of TT rather than functional dematerialization. Notably, technological convergence, the shift from single-function to multi-functional devices, emerges as a significant contributor to reduced materials usage. However, the increasing functionality of technologies is at the same time the main driver of the risk of directing the TT towards a dependence from some specific critical and geologically scarce materials.

**Keywords:** Technological change; material footprint; critical raw materials; sustainable development.

[1] Through its development, this paper has been presented at: Geography of Innovation winter school, Pavia, January 2024; Hiking Ecological Economics Summer School, Pisa - Apuan Alps, July 2023; BETA - Creativity, Science & Innovation seminar, Strasbourg, February 2023; R&D Management Conference, Trento, July 2022; International Research Workshop on Circular Economy in the Era of Digitalization, Brescia, June 2022; Spring Servitization Conference 'Achieving net-zero through Servitization', Florence, May 2022. The paper has greatly benefited from the discussions with the participants to the summer school "Leveraging Ecological Economics to Advance the Sustainability Transition", Pisa, July 2023; in particular, we thank Prof. Tommaso Luzzati for the enlightening discussions. Also, we specially acknowledge Prof. Rasmus Lema, Prof. Stefano Bianchini, Jerome Hambye and Prof. Nicola Saccani for the insightful comments.

*"The volume and rate of pollution as well as the exhaustion of resources depend upon institutional arrangements, upon the choice of technologies […], and the rate as well as the quality of economic growth. In other words, as long as we regard these three factors as autonomous and beyond social control - as independent variables, so to speak - and as long as we treat the assimilative capacity of the environment as infinite and not as a scarce common asset which needs to be protected by public policies and environmental planning, we will not be able to cope with environmental disruption."*

*Karl William Kapp, Environment and Technology: New Frontiers for the Social and Natural Sciences (1977)*

## 1. Introduction

Natural resources have been acknowledged as an essential ingredient of economic development (Graedel et al., 2015; Mayer et al., 2017). Nevertheless, they are scarce, exhaustible or slowly regenerating, unevenly distributed on Earth, and their extraction and use can cause adverse environmental and social impacts (Chazel et al., 2023; Sonter et al., 2023; Sovacool et al., 2019). Hence, scholars discussed on how to achieve a sustainable development, i.e. a path of development that meets the needs of the present without compromising the achievements of the needs of the future, within a "Spaceship Earth" (Dasgupta et al., 2023; IRP UNEP 2017; Wiedenhofer et al., 2020)[2]. In this debate, a prominent and dividing role is played by technology (Franceschini and Pansera, 2015; Rees, 2003; UNEP 2014).

Many scholars expressed their optimism about the disruptive potential of technologies to lead economic systems towards paths of sustainable development (Sachs et al., 2019) and, even, continuous expansion (Perez, 2014; Stern and Valero, 2021). In particular, digital technologies, such as artificial intelligence, cloud computing, and mobile communications undergo a comprehensive overhaul of every industry, reshaping the landscape of work, and introducing significant societal shifts in consumption, habits, and practices (OECD, 2016; Anthes, 2017). This potential can be directed towards environmental sustainability goals, such as reducing resource intensity of production processes, supporting circular economy and enabling zero-carbon energy systems (Barteková and Börkey, 2022). Coherently, the idea of Twin Transitions, in which the digital transition is imagined

[2] Metaphor of Planet Earth as a "spaceship" is a figurative description of the finite nature of planetary resources and is contrasted with the 'cowboy economy', an economy that can expand indefinitely (Ayres, 1999; Meran, 2023).

to support the green one, gained momentum both in the academia (Chatzistamoulou, 2023; Montresor and Vezzani, 2023; Timmermans et al., 2023) and among policy makers (European Commission, 2021; Muench et al., 2022).

It is not the first time, though, that new technologies have strongly modified the world where we live and produce value (Allen, 2017). Innovation scholars already saw in the advent of information and communication technologies (ICT) the potential to spur a wave of sustainable innovations (Antonioli et al., 2018; Cecere et al., 2014), ultimately leading to a new green "techno-economic paradigm" (Kondratieff, 1979). In this regard, Tushman and Anderson (1986) demonstrated that "technology evolves through periods of incremental change punctuated by technological breakthroughs" (p. 439). These technological breakthroughs or discontinuities define the establishment of a dominant design in some industries shaping the present state of production and consumption, but also influencing the future incremental developments of these industries (Utterback and Abernathy, 1978). Chris Freeman (1992) was among the first to propose this forecast, suggesting that the upcoming shift from energy intensive, oil-based technologies towards electronics-based technologies started and marked a new upswing in productivity growth that relates to subsequent reductions in resource use and waste production (Boons and Wagner, 2009; Hayter, 2008; Truffer and Coenen, 2012). According to Carlota Perez (2017), ICT has allowed a reduction of the use of raw materials through data harvesting and sharing, increased precision in design and production, the optimisation of logistics, the redesign of products and the provision of services in replacement of physical products. With regard to this latter aspect, the transformative potential of technology would not be confined to mere efficiency gains, but it would extend into the realm of dematerialization of manufacturing: the so called servitization. Scholars of this topic argue for a shift from tangible to intangible in the satisfaction of societal needs, enabled by technological solutions, that can lead to a strong reduction in the material footprint (MF) of value chains (Beuren et al., 2013; Bressanelli et al., 2018; Mont, 2002)[3].

Thus, it emerges that the concept of Twin Transitions and, in particular, the idea that the evolution of technologies can lead to a decoupling from the use of material resources are deeply rooted in the literature on Economics of Innovation. But not only. Starting already from the 1970s (Bretschger, 2005; Groth, 2007), classical contributions from prominent environmental economists identified in technological progress an opportunity to increase the efficiency in the use of natural resources, i.e. productivity, and an enabler to switch from the use of a scarce resource to another, (Dasgupta and Heal, 1974). According to this perspective, the problem of natural resources

[3] Tukker (2015) resized servitization (or product-service systems) environmental outcomes, concluding that it cannot provide radical boosts in terms of resource-efficiency or economy circularity.

exhaustion (Hotelling, 1931) can be successfully tackled by technological change and the power of the market in adjusting human behaviour (Nordhaus, 1992; Zhang et al., 2018). Indeed, it is interesting to note how the critique to *Limits to Growth* and *Beyond the Limits* (Meadows et al., 1992, 1972), two - at least conceptual – milestones for unfettered economic growth opposers (Haberl et al., 2011), represents a connection between an environmental economist, such as William Nordhaus (1992), and an economist of innovation, such as Chris Freeman (1996). Both scholars claimed that the role of technological innovation was underestimated in Meadows and colleagues' works, but they focus on two different determinants of technological change in their argumentations: the role of environmental externalities and economic incentives, the former, and the role of institutional factors, the latter. Currently, their respective research fields are still focusing on these factors as main determinants of technological change for environmental sustainability (Stern and Valero, 2021; van den Bergh, 2001).

Unfortunately, all that glitters is not gold. In parallel with growing "macro" evidence of the increasing MF of economic systems (Haberl et al., 2020; Hickel and Kallis, 2020; Steffen et al., 2015), an expanding body of literature is discussing threats to social, economic or environmental sustainability coming from technologies evolution or technologies as artefacts (Barteková and Börkey, 2022; Bianchini et al., 2022; Røpke, 2012). In particular, with regard to the issue of the MF of technologies, three main challenges have been identified (Hilty and Aebischer, 2015).

The first challenge refers to the overall demand of materials for the production of technologies. Concerning this challenge, the concepts of decoupling and rebound effect have been discussed for long. Decoupling refers to the de-linking of the trends of a socio-economic factor and its resource consumption, and it can occur in absolute or relative terms. A rebound effect, also known as Jevon's Paradox, occurs when, as a new or incumbent technology improves the efficiency with which it uses a certain resource, total consumption of that resource increases rather than decreases as a consequence of cost savings (Alcott et al., 2012; Missemer, 2012). The Jevon's Paradox anticipated the fact that different ecological outcomes may be observed when considering different levels of analysis, i.e. technology/artifact and macro/aggregate levels. Thus, this first challenge hints to the idea that, since biophysical limits exist on the planet, economic systems should have an optimal scale - or level of metabolism - relative to the total ecosystem (Luzzati et al., 2022).

The second challenge is identified in the growing material complexity of technologies as artifacts (Cecere and Martinelli, 2017; Ljunggren Söderman and André, 2019). The growing physical and digital performances of technologies have been accompanied by an increasing array of resources embedded in them (Graedel et al., 2015). This represents a problem in terms of extension of the lifetime of resources, as the higher the material complexity of end-of-life technologies, i.e. electronic waste, the more difficult the separation of materials through recycling (Althaf et al., 2021; Hagelüken and Goldmann, 2022)[4]. This second challenge, related to the variety of materials contained in technologies, seems to suggest that a trade-off between the evolution of the design of technologies and one of the determinants of their MF, namely recyclability, exists.

Lastly, the alleged scarcity and, especially, the uneven distribution of some material inputs currently used in technological devices worries scholars and institutions (Abraham, 2015; Carrara et al., 2023; IEA, 2021; Kowalski and Legendre, 2023), also in consideration of their current limited substitutability (Ayres and Peiró, 2013; Cenci et al., 2021). Some of these raw materials have been classified as critical because they are exposed to high supply risks (Schrijvers et al., 2020; Wäger et al., 2015). Hence, this last challenge, related to the typology of materials contained in technological devices shows how considerations on the MF of technologies should go beyond a pure mass-based perspective (Zhang et al. 2018). Assessing the dependence of technological trajectories (TT) on scarce and critical materials is of fundamental importance because the long periods required for technologies development and diffusion in combination with the longevity of human-made material stocks create long term path dependencies, which endanger sustainable development if not directed towards low MF goals (Grubler et al., 2016; Krausmann et al., 2017).

These challenges have typically been separately considered, resulting in "single-level" perspectives. In other words, each of the three above-mentioned challenges refer to a different level of analysis that is aggregate materials consumption, the material composition of different electronic devices, and the specific type of the materials within each technology, respectively. This resulted in single-level assessments of the TT-MF relationship and policy implications. Nevertheless, each of these levels of analysis should not be disregarded when we aim to disentangle the variety of phenomena affecting the MF of TTs. Because both economic and ecological systems have a nested structure, when looking at the relationship between socio-economic phenomena and their ecological consequences, a central issue thus is the definition of the system, or scale, for which the relationship is assessed (Peterson, 2000). As already noted by Boons and Wagner (2009), researchers tend to be implicit about the way in which they define their scale of analysis when investigating the nexus between technological innovation and environmental (and economic) performance.

[4] Material complexity can be seen as a measure of entropy, whose relationship with recyclability have been discussed in the literature (Roithner et al., 2022; van Nielen et al., 2022). In this sense, complex technological devices (or electronic waste) are characterized by high entropy.

Moreover, it is not only this interrelated structure of socio-technical and ecological systems that makes the assessment of decoupling trends complicated, but also the uncertainty about the available empirical data (Erdmann and Hilty, 2010; Semieniuk, 2024). This is particularly true when researchers aspire to explore the heterogeneity of technologies and materials (Babbitt et al., 2020).

This study seeks to contribute to the debate on the material footprint of technologies in various ways.

First, by embracing Boons and Wagner (2009) and Berkhout and Hertin (2004)[5] perspectives, this study acknowledges the complexity of the relationship under study and stresses the relevance of adopting systemic perspectives for its assessment. To build this systematic perspective, we leverage on concepts derived from the Ecology and Industrial ecology literatures, namely the one of scale (Gibson et al., 2000) and the one of product ecosystem (Levine, 1999; Ryen et al., 2014). While the introduction of the concept of scale allows us to explicitly define the boundaries and levels of our system of analysis, the one of product ecosystem is useful to represent the fact that technologies are not used in isolation, but rather in combination, and their diffusion co-evolves.

Second, on the basis of the scale and product ecosystem frameworks, we develop a longitudinal multi-level analysis capable of disentangling the different patterns affecting the MF of the TT of our society. To distinguish MF trends occurring at the product, functional groups of products and TT levels, a bottom-up approach for the estimation of MFs is required (Lutter et al., 2016). By exploiting both data on the units and weight of technologies, i.e. electrical and electronic equipment (EEE) (Mazzarano, 2022), placed on the European market and data on their material contents, we are able to simultaneously describe the waves of diffusion of technologies and the mass of materials embedded in each type of technology.[6] To our knowledge, the dataset that we exploit is the most detailed in terms of identification of technologies and materials, providing a twofold advantage. First, we can describe the TT of EEE over about three decades, showing trends of lightning, substitution and convergence of technologies. Second, we can characterize the materials on the basis of their scarcity and criticality, offering insights on the reliance of various technologies on these materials. Indeed, the extent and diversity of this dependence remain uncertain (Diemer et al., 2022).

Finally, our results are discussed in connection to the three challenges described above. We show that, when adopting a multi-level perspective, phenomena improving the MF of the TT at a certain

[5] These authors define the relationship between (information and communication) technologies and the use of resources as complex, interdependent, deeply uncertain and scale-dependent.

[6] In this paper we look at technologies as products/artifacts, and not as knowledge fields (Bergek et al., 2008). More specifically, our empirical analysis is limited to the direct MF of EEE, i.e. the mass and type of materials embedded in the EEE placed on the market. See Section 3.2 for further details.

level may have adverse effects at different levels of analysis. The same holds for material efficiency strategies and policies focused on single-level approaches.

The next section, first, discusses the limitations of the methodological approaches of previous literature investigating the relationship between the evolution of technologies and their MF and, secondly, introduces the concepts of scale and product ecosystem. Section 3 explains how these concepts are adopted in our empirical investigation; all data sources are also described in this Section. Section 4 presents our empirical results according to three levels of analysis, i.e. aggregation of technologies or materials. Section 5 discusses the evidence relative to the various technological phenomena affecting resource consumption and dependency. Section 6 concludes with recommendations for research and policy making.

# 2. Research gaps and methodological perspective

## 2.1 Limitations of previous literature

The relationship between the evolution of technologies and their MF, or – more generally – environmental impacts, has been acknowledged as complex, i.e. multi-dimensional, and interdependent, i.e. the availability of natural resources shapes the evolution of technologies and vice versa (Li et al., 2024). Nonetheless, few studies strived to develop systematic assessments and policy implications (Santarius et al., 2023b). In this Section, we briefly discuss some research gaps of the two main streams of research that represent the bedrock of both the conceptual and methodological approach of this paper.

The first one is the literature on Sustainability Transitions (STs), which has at its core the study of the nexus between environmental sustainability and the evolution of technologies (Geels, 2010; Jacobsson and Bergek, 2011). These scholars focus on long-term multi-dimensional transformation processes through which established socio-technical systems shift to more sustainable modes of production and consumption (Markard et al., 2012), thereby acknowledging the complexity of the technology-environment relationship.

Nonetheless, in this literature the multi-level perspective is not exploited to disentangle the ecological consequences of technologies development and diffusion. Instead, the scales of analyses are framed on (green) innovation *loci* (such as niches, regimes and socio-technical landscapes) and/or geographical dimensions (Köhler et al., 2019).

Moreover, the field lacks in terms of reflections about why a certain technology should be supported by policy in the first place, i.e. "what an overarching goal might be to which the respective technology might contribute" (Bening et al., 2015, p. 74). Starting from specific transitional goals, papers should reflect on the desirability of the analysed technology. As suggested by Bening et al. (2015), this evaluation should not be done in isolation by innovation scholars, but it should be based on studies from the field of Life Cycle Assessment and other neighbouring fields.

Finally, the STs field is still in its infancy, as demonstrated by the very limited number of publications discussing technological transitions, digitalization and sustainability at the same time (Andersen et al., 2021).

The second stream of research representing the conceptual backbone of this paper is, of course, the one on material footprints, developed within the Ecological Economics and Industrial Ecology literatures. Indeed, an extensive literature on material flows analysis has been developed (Lutter et al. 2016). Unfortunately, the units of analysis traditionally adopted in the MFs literature are not suited to explore the phenomena determining the physical evolution of TTs, namely the diffusion and substitution of technologies as well as the changes in material composition of each of them. Indeed, input-output, which is the typical top-down approach representing material interlinkages between different branches of a national economy or different economies, has the main disadvantage of working on the level of aggregated economic sectors, assuming that each sector produces a homogenous product output (de Koning et al., 2015; Schandl et al., 2018; Zhang et al., 2018). This aggregation of sectors and products implies that, on the one hand, the variety of specific technologies that is possible to track is limited and, on the other hand, within each sector a number of different products with potentially very different material characteristics are mixed together and averaged. Despite the recent attempts to increase the granularity of input-output datasets (Malik et al., 2019), top-down MF indicators remain typically used for contributions at the macro level (Lenzen et al., 2022; Wiedmann et al., 2015).

For the purpose of analysing the MF of TTs, bottom-up approaches centred on products (measured in physical units) and material composition coefficients seem to be more appropriate. In fact, bottom-up approaches have the great advantage of high levels of detail and transparency: the absence of restrictions regarding the definition of sectors or product groups allows performing very specific comparisons of footprints down to the level of single products or materials (Dittrich et al., 2012; Lutter et al., 2016). Nevertheless, the literature exploiting data on technological devices placed on market and their material contents have been mainly devoted to the investigation of end-of-life aspects of technologies, i.e. quantify the mass of electronic waste in a certain geographical area and its economic potential in terms of recyclability (e.g. Forti et al., 2020;Bookhagen et al., 2020; Cucchiella et al., 2015; Panchal et al., 2021); these studies sometimes focus on a single technology (e.g. Bruno et al., 2022; Kasulaitis et al., 2015). Hence, this literature overlooks systematic analyses on the MF of TT. In addition, this methodological approach focused on mineral resources uses of technologies remains nonetheless minoritarian, while assessments focusing on energy consumption and emissions generally tend to prevail (Gossart, 2015; Horner et al., 2016; Lange et al., 2020; Røpke, 2012).

In conclusion, the contradictory indications of previous literature on the MF of TT originates from a lack of systematic approaches pointing to disentangling the complexity of the MF phenomena involving the TT, manifested in unclear or inadequate definitions of system boundaries and in a lack of coherence of levels of analysis.

## 2.2. Ecology approaches to disentangle the complexity of the MF of TTs

Investigating the MF of a TT entails two difficult tasks that must be carried out simultaneously: describing the evolution of technologies and their impact in terms of material consumption. Ecology and connected fields of research allow to elaborate a comprehensive approach able to inform about the three main challenges resulting from the different literatures.

Ecology is a discipline central to the study of human dimensions of global change, focusing on complex, multi-scale systems (Gibson et al., 2000). Ecologists typically try to understand the dynamics at one level of an ecological system as an aggregation of interactions among lower-level units. In their influential paper, Gibson, Ostrom and Ahn (2000) explained that ecological phenomena occurring at any one level are affected by mechanisms occurring at the same level, and by levels below and above. Thus, research on global change processes should examine the world from a multilevel perspective. These choices over the scale and levels of analysis critically affect the type of patterns that will be observed, because patterns that appear at one level of resolution or extent may be lost at lower or higher levels. In other words, making causal statements about particular patterns explicitly or implicitly invoke the definition of a scale of analysis. Moreover, these authors argued that while natural scientists have long understood the importance of scale, and have operated within relatively well-defined hierarchical systems of analysis, social scientists have worked with scales of less precision and of greater variety. Gibson and colleagues claim it is a necessity for social scientists to identify more clearly the effects of diverse levels on multiple scales in their own analyses. Fundamental ecological concepts related to scales are defined in Table 1.

Hence, given the limitations and inconsistency of previous research discussing the MF of TT, a more explicit and accurate integration of the concepts of hierarchies, levels and extent of analysis is useful to disentangle the complexity of involved phenomena and define more consistent patterns.

*Table 1 Key terms related to the concept of scale, adapted from Gibson et al. (2000).*

| Term | Definition |
|---|---|
| Scale | The analytical dimensions (spatial, temporal, quantitative) used to measure and study any phenomenon |
| Extent | The size of the analytical dimensions of a scale |
| Hierarchy | A conceptually or causally linked system of grouping objects or processes along an analytical scale |
| Level | The units of analysis that are located at the same position on a scale |

Another perspective that we integrate in our methodological approach is the one of "product ecosystems." Already adopted by scholars in Industrial Ecology to study the evolution of ecological impacts of consumer electronics (Althaf et al., 2021; Kasulaitis et al., 2019; Ryen et al., 2015), this perspective builds on Levine (1999) seminal contribution. Levine (1999) proposed an ecosystem model centred on products. In the spirit of the Industrial Ecology perspective, the product life-cycle represents a flow of materials and energy. But because products are the output of industrial activity, rather than materials and energy, understanding the history of products, their diversity and diffusion, and their relationships with and dependence on one another can be important to assess and predict ecological impacts of production-consumption systems. A core idea in Levine's view is that products do not exist in isolation from one another, but rather they are consumed in a highly interrelated fashion, where the ownership of one influences the purchase of another. This is especially true for technological devices, i.e. EEE, for which it is the ecosystem of devices used by a consumer that is meeting his own communication, productivity and entertainment needs (Althaf et al., 2021; Ryen et al., 2014). To add further complexity, this view must be put in an evolutionary perspective. Indeed, the components of this products ecosystem come into existence at different times and are therefore in different stages of development and diffusion; new products coexist with mature products, and with others on their way to extinction; many products find themselves in a different economic environment with respect to the one in which they arose (Levine, 1999; Kasulaitis et al., 2019; Ryen et al., 2015). The adoption of a product ecosystem perspective helps visualise the complexity of the phenomena affecting the MF of TTs, and it is compatible with a multi-level approach since it allows to distinguish trends occurring at the EEE community, i.e. aggregate, functional group or species level. The products ecosystem taxonomy adopted in this paper and defined in Table 2 is borrowed from Ryen et al. (2014) who first adapted it from Levine (1999) to study the EEE market.

*Table 2 The product ecosystem taxonomy adopted by Ryen et al. (2014).*

| Concept | Definition in Ecology |
|---|---|
| Community | Collection of organisms living and interacting together in a defined habitat at the same point in time |
| Population | Group of individuals belonging to the same species in a defined habitat at the same point in time |
| Functional group | Grouping of species that share a similar function |
| Species | Group of organisms sharing similar genes |

## 3. Data and methodological approach

### 3.1 Data

This paper exploits data collected and elaborated in the ProSUM project (Huisman et al., 2017). This dataset presents a series of advantages in describing both the evolution of technologies as artefacts, and their material contents. Consistently with the Industrial Ecology literature, we apply a "product-centric" approach by looking at electrical and electronic equipment (EEE) (Mazzarano, 2022).

ProSUM data provide information on the number of units and weight of EEE placed on the market in the EU28 plus Switzerland and Norway in the period 1980-2015[7]. These data on new products, identified by the acronym POM (placed on market), are calculated through the material flow analysis[8] methodology of apparent consumption, typical of bottom-up MF approaches (Bleischwitz et al., 2018; Lutter et al., 2016). More precisely, these data are calculated as EU domestic production of EEE (ProdCom data from Eurostat), plus imports and net of exports (matched to ProdCom data through Combined Nomenclature trade codes). In other words, POM represent the EEE market materials inflow in terms of both number of devices and mass of embedded materials. More details on the POM estimation procedure, already consolidated in Magalini et al. (2015), are available in ProSUM documentation.[9]

ProSUM POM covers all of the 54 United Nations EEE categories (Magalini et al., 2015). Each product category is defined by an UNU-key code, which is reported in Table A1 in Appendix A with the relative description. These 54 product categories define the boundary of the TT that we analyse.

[7] The ProSUM project provided data also for the period 2015-2020 based on estimates. We decided not to make use of this timespan because of the high level of uncertainty.

[8] Material Flow Analysis aims to describe economies in physical units (Fischer-Kowalski et al., 2011).

[9] See Deliverable 3.1 at https://www.prosumproject.eu/project-reports

Clearly, this set of EEE represents a proxy for the set of "all technologies" constituting the current TT. Nonetheless, the comprehensiveness of the UNU-key classification in terms of typologies of EEE devices included, together with its high level of detail of EEE categorization make it suited to develop a general picture of the EEE ecosystem evolution and map out the MF of our TT. Other studies adopting the technologies ecosystem approach focused on a more restricted set of EEE as well as different sources of POM data (e.g. Althaf et al., 2021; Kasulaitis et al., 2019). The same did Horta Arduin et al. (2020) that also used ProSUM data, by restricting their analysis to a subset of UNU-keys.

Besides POM, ProSUM also provides "bill of materials" (BOM) data. BOM present information on the mass of components, materials and chemical elements within each technology. ProSUM BOM dataset, which is definitely one of the most comprehensive and detailed, were estimated through EEE dismantling and chemical characterization, in combination with an extensive literature review (Huisman et al., 2017). In particular, up to 51 chemical elements are detected within the 54 analysed product categories. Moreover, the dataset covers the period 2000-2015 and products composition varies in time, while it is assumed as constant in other sources (e.g. Babbitt et al., 2020). Previous studies investigating the evolution of the material content of EEE took advantage of different sources of data that provide information at the level of components only (Babbitt et al., 2020) or quantify the mass of a limited number of elements (Althaf et al., 2021; Zeng et al., 2016). As a drawback, ProSUM made available BOM information at the aggregation level of the EU 2012 Electronic Waste Directive (WEEE Directive), i.e. six categories of devices with a certain degree of functional homogeneity, and not at the more disaggregated UNU-key level.[10] For this reason, we complement our analysis on the material composition of EEE integrating BOM data at the product level from other sources (Cucchiella et al., 2015; Zhu et al., 2017).

For the purpose of this study, we aggregated both POM and BOM data at the European level, while they were originally separately provided at the country-level. The geographic perspective on Europe also represents a novelty, as studies based on Babbitt et al. (2020) consider the case of the United States, while the wide geographical coverage of our data increases the significance of our results with respect to country-specific analyses (e.g. Parajuly et al., 2017).

Finally, to extend our analysis beyond the pure mass-based perspective, we take in consideration the degree of criticality and geological scarcity of materials. Indeed, MF literature recognized that

[10] The description of the six WEEE Directive categories and their correspondence with UKU-keys is also reported in Tab. A1 in Appendix A.

indicators measuring the weight of materials do not tell the full story, since the extraction and consumption of different resources entails different environmental, social and economic consequences (Zhang et al., 2018). In particular, first, we characterize materials on the basis of their classification as critical or non-critical as defined by the European Commission (2020); this is to obtain an indication about the supply risk of the resources contained in the various EEE. Second, we make use of the Crustal Scarcity Indicator (Arvidsson et al., 2020), a mineral resource impact assessment method that captures the perspective of long-term global scarcity of elements.

### 3.2 Application of the multi-level and product ecosystem approaches

By applying the multi-level and product ecosystem approaches to both POM and BOM data, it is possible to disentangle the different phenomena affecting the MF of a TT.

In Table 4, first, the extent of our scale of analysis is defined. The use of both POM and BOM data allows to develop a double quantitative dimension of the scale: one is based on the variety of technologies and the other on the variety of materials.

Secondly, our levels of analysis are identified, making use of the product ecosystem taxonomy of Table 3. We start from the most aggregate level, i.e. the *community* level, in which all the *populations* of EEE are considered together. This level of analysis depicts the total number of EEE placed on the European market as well as their aggregate weight, which corresponds to aggregate mass on the materials embedded in these new devices. Hence, the community level of analysis describes the aggregate metabolism of the European EEE market.[11]

Moving to a finer level of analysis, we identify groups of technologies with similar or interconnected functionalities. This allows us to explore the TT in an evolutive perspective, investigating how the interaction of old and new technologies affects their diffusion. Upward causation[12] mechanisms affecting the metabolism of the EEE market and downward causation mechanisms affecting EEE material complexity and dependency on critical or scarce materials will be identified. The functional groups are defined according to the six 2012 WEEE Directive categories, which are characterized by a certain functional homogeneity, as well as on the basis of previous literature (see Section 4.2).

[11] Our MF measures are limited to the weight and type of materials embedded in EEE placed on the market. This is consistent with papers in Industrial Ecology (Althaf et al., 2021; Kasulaitis et al., 2019). However, MF, or Raw Material Consumption, indicators usually refer not only to direct material flows, i.e. the mass of consumed, produced or imported goods or services, but also to the indirect, upstream ones required along the production chain. This perspective originates from the concept of ecological rucksack (Dittrich et al., 2012; Hinterberger et al., 1997). Some databases on indirect material flows have been developed, most notably Wuppertal Institute's material input per unit of service (MIPS). Nonetheless, because of the limited data availability or detail at product level of these data, we decided not to include them in our analysis, limiting it to direct material flows (Wäger et al., 2015).

[12] Upward (downward) causation refers to a pattern or variable being used in an explanation that occurs at a higher (lower) level than the pattern or dependent variables being explained (Gibson et al., 2000).

Finally, at the most disaggregated level of analysis, we focus on the materials contained in the functional groups and in specific technologies. Our focus here is both on the variety of materials embedded in the technologies, which determines their material complexity (Ljunggren Söderman and André, 2019; Roithner et al., 2022), and on their characterization in terms of criticality or geological scarcity.

*Table 3 Definition of the product ecosystem taxonomy adopted in this paper*

| Concept | Application in this paper |
|---|---|
| Community | All EEE belonging to the UNU-key classification, in a certain year (1980-2015), in EU28 + Switzerland and Norway |
| Population | Group of EEE (units) belonging to a UNU-key in a certain year (1980-2015) in EU28 + 2 |
| Functional group | Grouping of UNU-keys with similar or interconnected function(s) |
| Species | UNU-key class |

*Table 4 Definition of the scale of analysis adopted in this paper*

| Term | Definition |
|---|---|
| Extent | • Spatial: EU28 + Switzerland and Norway<br>• Temporal: 1980-2015, yearly data<br>• Quantitative:<br>  ○ 54 UNU-key classes of technologies, measured in units and weight → POM data<br>  ○ 51 chemical elements, measured in weight → BOM data |
| Level | 1. Community level:<br>Aggregate evolution of the technological trajectory, i.e. all EEE without distinction by UNU-key classes<br>2. Functional groups level:<br>Evolution of groups of technologies with similar or interconnected function(s)<br>3. Materials level:<br>Evolution of the material content of technologies (variety of materials; characterization in terms of scarcity and criticality) |

## 4. Results

In this Section, our empirical results are presented following the increasing granularity of our unit of analysis: the community level, the functional groups level, and the materials level.

### 4.1. Community level

Figure 1 represents the evolution in terms of numerosity and mass of the TT at community level, that is when all populations of technologies are considered together.

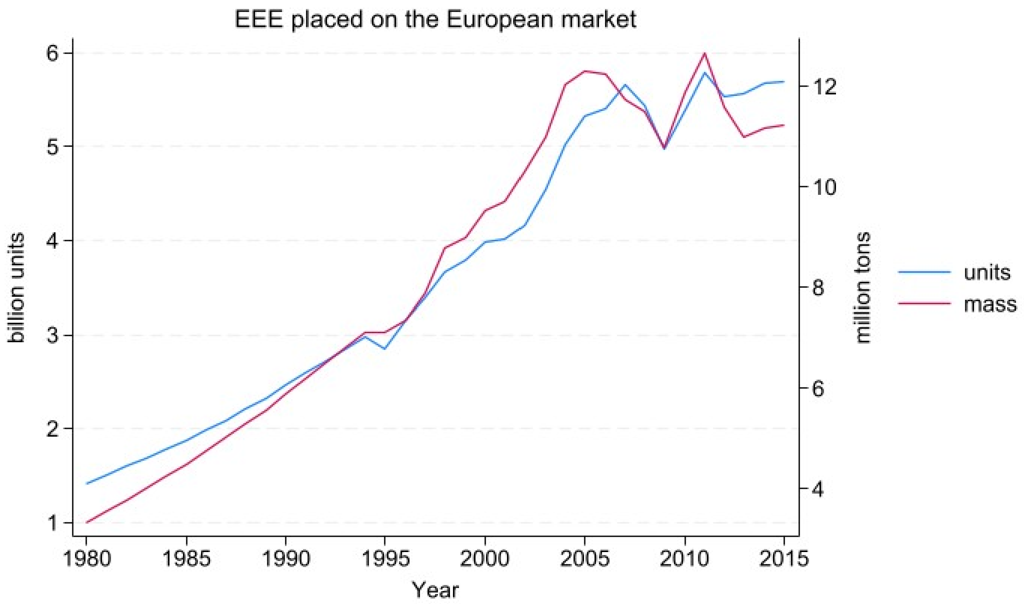


*Figure 1 Number and weight at the community level of new EEE POM in Europe, by year, 1980-2015.*

The total number of electronic devices placed on the European market increased steadily between 1990 and the economic crisis of 2007-2008; subsequently, after a short phase of recovery, the POM units trend almost stabilized. This slowdown of the growth of the EEE community is coherent with what observed by Althaf et al. (2021) for the US and Parajuly et al. (2017) for Denmark, relying on different sources of data. In per capita terms, the total number of EEE POM went from 3 units per person in 1980 to 11 in 2007, remaining stable afterwards except for the negative effect of the economic crisis and the positive one of the subsequent recovery period.

In terms of mass, the EEE community inflow reached the impressive weight of more than 12 million tons of embedded materials at its peak in 2010[13], corresponding to 25 kilograms per European inhabitant. POM weight and units follow a close trend. Nonetheless, while between 1990 and the

[13] The order of magnitude of this figure is compatible with what observed in Althaf et al. (2021), considering that they rely on a restricted set of technologies compared to this paper.

early 2000s the mass of the EEE community grew faster than POM units, after 2005 the average annual growth rate of the former was just half of the one of the latter (see Table 5). This partial de-linking of the two trends is a result of a lightening of the "average", representative EEE POM occurring since early 2000s.

This aggregate level of analysis provides interesting insights in terms of aggregate metabolism of the TT and on its coupling/decoupling trends, but it does not allow to pinpoint the mechanisms affecting these trends. Hence, in the next Section we move to a more disaggregate level of analysis.

### 4.2. Functional groups level

Here we distinguish groups of products – species, in the product ecosystem analogy - characterized by similar or interconnected functions, while keeping material contents (BOM) as aggregate. The aim is to disentangle the various phenomena that are simultaneously, potentially impacting on the aggregate trends described at the community level of analysis. In particular, the community of EEE evolves in time, through a change of the material consumption connected to each technology and a diffusion of new technologies, which act as substitutes or complements of incumbent devices either belonging to a different or the same functional category.

First, we rely on 2012 WEEE Directive to define functional groups (FG)[14]. Table 5 reports that in the period 2000-2015 the highest growth in terms of POM units is observed for the FG of "screens and monitors" (a category also containing televisions, laptops and tablets), followed by "small IT and telecommunication equipment" (a category containing smartphones and featured phones, among other devices). The screens and monitors group maintained its position in the post crisis (2008-2015) period. Interestingly, FG 2 and 6 are also the ones in which the strongest decrease in terms of weight of the representative device placed on market is observed[15] (Col. 4 Table 5). Nonetheless, while the progressive lightening of FG 2 more than compensated the increasing penetration of these technologies, as observed in Col. 3 Table 5, this did not occur for FG 6.

[14] For coherence with our products ecosystem analogy, we define as functional groups what the Directive defines as "categories".

[15] In the case of FG 6, this finding is consistent with Bruno et al. (2022).

*Table 5 Average annual POM units, weight and relative weight variations by functional group, years 2000-2015, in brackets 2008-2015.*

| Functional Group | Avg. annual POM **units** variation | Avg. annual POM **weight** variation | Avg. annual POM **relative weight** variation |
|---|---|---|---|
| FG 1: temperature exchange equipment | 2.6% (-1.8%) | 2.8% | 0.3% |
| FG 2: screens and monitors | 5.1% (3.7%) | -4.6% | -9.3% |
| FG 3: lamps | 1.5% (-4.5%) | 1.4% | 0% |
| FG 4: large equipment | 3.7% (2.8%) | 2.3% | -1% |
| FG 5: small equipment | 2.6% (2.3%) | 1.4% | -1.2% |
| FG 6: small IT and telecommunication equipment | **4.2%** (1%) | 2.3% | -1.7% |
| EEE community: cat.1-6 aggregate | 2.6% (1 %) | 1.3% (**-0.1%**) | -1.2% (**-1%**) |

At this stage, it is important to distinguish two different determinants of POM weight. The first and most commonly considered factor in the literature is *functional dematerialization*, that is the fact that, over time, more function generally becomes possible with less material, as a result of technological improvements (Kasulaitis et al. 2015). Functional dematerialization is seen as a natural consequence of Moore's law (Moore, 1965). The second factor, which is visible at the product, but only when adopting a product ecosystem approach, is the *composition effect*, consisting in the diffusion of new technologies and their interaction with incumbent ones.

To explore how the composition effect may be realized, we focus on the FG of screens and monitors, which is the one that recorded the highest growth in terms of units, but also the highest decrease in weight of POM. As represented in Figure 2, this FG was affected by two technological discontinuities. The early leading technologies within this FG, cathode-ray tube (CRT) televisions and monitors, abruptly exited the EEE market after the advent of flat panel technologies, occurring just before 2005. Because flat panel technologies continued to perform the original function of the respective CRT devices solely, we talk about a process of "*technological substitution*". A second technological discontinuity is connected with the exponential diffusion of laptops starting in the late 1990s and, later, tablets. These devices can perform a variety of functions and serve many purposes. In particular, their characteristic of incorporating a screen component and their ability to display video contents, allowed them to partially replace televisions and monitors, whose POM units have been decreasing since 2010. Hence, because in this case we observe a displacement of single functional devices by multi-functional ones, we define this phenomenon as *"technological convergence"* (Kasulaitis et al., 2021).

Therefore, the evolution of the composition of the screens and monitors FG, with heavy CRT technologies replaced by lighter devices, is a much more determinant driver of the decreasing weight of FG 2 POM, compared to functional dematerialization.

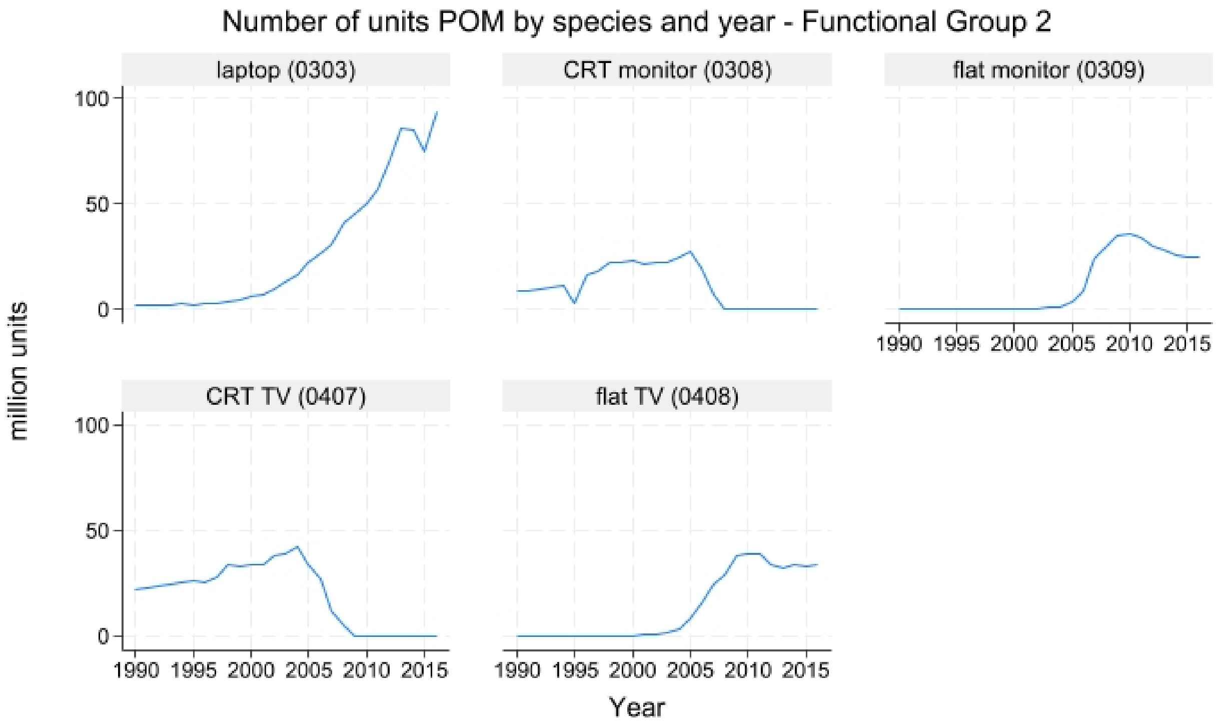


*Figure 2 Units POM within the functional group 2 "screens and monitors", by UNU-key (in parenthesis), 1990-2015.*

The utility of the EEE ecosystem approach is particularly evident by looking at a second case of device convergence, previously hypothesized by Parajuly et al. (2019) and referring to mobile phones and other audio/video players (Figure 3). Mobile phones started significantly penetrating the market in the 1990s. The first generation of "feature phones" offered basic phone and multimedia functionalities, they were relatively cheap and targeted at the low- and mid-end market; a technological discontinuity occurred in the mid-2000s with the appearance of "smartphones", namely handsets equipped with advanced operating systems offering computer-like capabilities, targeted at the high-end market (Giachetti and Marchi, 2017). The ability of smartphones to perform a variety of functions and, in particular, audio and video functionalities led to a convergence of the demand of single functional devices, such as cameras, car navigators and radios, towards that new technology. As smartphones are generally smaller and lighter devices compared to the other technologies included in this FG, this process of convergence resulted in a net saving of embedded materials.

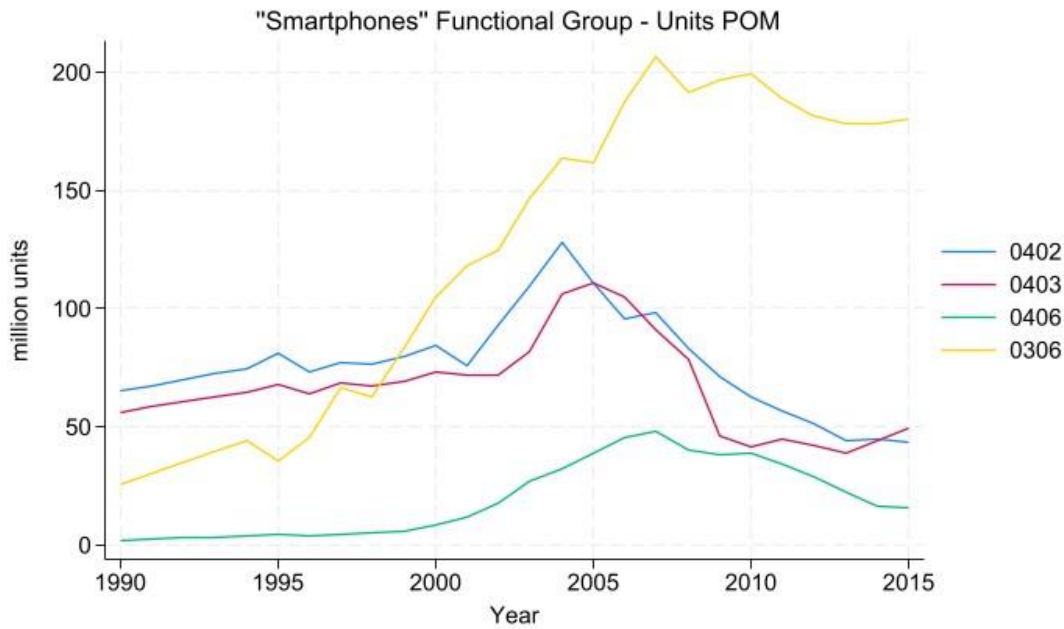


*Figure 3 Units POM within the "smartphones" functional group, by UNU-key, 1990-2015. UNU-keys: 0402 portable audio and video (e.g. car navigation, MP3); 0403 radio, Hi-Fi; 0406 camera; 0306 mobile phone.*

In conclusion, the functional groups level of analysis showed the importance of considering the phenomena of technological substitution and convergence in the assessment of the MF of TT and described the TT of our society as shifting towards lighter, sometimes smaller, multi-functional devices. It remains now to explore further insights when materials are considered in their variety.

### 4.3. Materials level

To measure the material complexity, i.e. the variety of materials (Ljunggren Söderman and André, 2019), within each FG we count the number of chemical elements detected in our BOM data. As reported in Table 6, small IT and telecommunication equipment FG ranks first in terms of material complexity, with 49 elements detected. We consider this figure as a lower bound, as other studies specifically focusing on smartphones BOM estimated the presence of 53 (Bookhagen et al. 2020) to 60 (Tantawi and Hua, 2021) elements in their composition. Screens and monitors, and small equipment FGs follow in the complexity ranking. As it can be noted in the same Table, in terms of POM units, FGs 6, 2 and 5 register a positive growth in the most recent period of observation and, in particular, screens and monitors and small IT have been the fastest growing groups since 2000 (see Section 4.1).

Focusing more in detail on FG 2, laptops are found to be significantly more complex than tablets, which in turn contain a higher number of materials than flat TVs and monitors, while CRT TVs are the least complex (and heaviest) devices, according to the data from Zhu et al. (2017) and Cucchiella et al. (2015). Finally, smartphones and small IT are characterized by a higher complexity than FG 5 technologies - such as portable audio and video, radios, cameras - on average; a much higher variety of chemical elements is also found in smartphones compared to featured phones (Zhu et al., 2017). It seems reasonable to conclude that more complex devices are progressively replacing technologies containing a low variety of materials.

*Table 6 Material complexity by functional group and average annual variation of units POM in (2008-2015).*

| Functional Group | Elements detected | Complexity rank | Avg. annual POM units variation (2008-2015) |
|---|---|---|---|
| FG 1 | 29 | 6 | - 1.8 % |
| FG 2 | 47 | 2 | 3.7 % |
| FG 3 | 40 | 4 | - 4.5 % |
| FG 4 | 39 | 5 | 2.8 % |
| FG 5 | 41 | 3 | 2.3 % |
| FG 6 | 49 | 1 | 1 % |

The second aspect that we investigate at the materials level is the consumption of CRM of the TT. On the basis of the European Commission's (2020) list, we classified the chemical elements present in our BOM data as critical or not. Figure 4 describes the evolution of the aggregate mass of CRM embedded in the TT at the community level. Quite surprisingly, an impressive drop of CRM contents is observed, both in absolute and in concentration terms[16]. This decline occurs quite sharply in the years around 2005. Considering the percentage composition in CRM of each FG, we discover that the trend observed at the community level can be ascribed entirely to FG 2 evolution (Figure 5). In particular, over the total mass of CRM contained in the screens and monitors FG, in 2000 a large share was represented by two heavy metals, i.e. antimony and strontium, while in 2015 they almost entirely disappeared (Figure A1 in Appendix A). This finding is compatible with the much higher use of antimony and strontium - but also cadmium and lead, two non-critical heavy metals - in CRT technologies (Höllriegl, 2019; Zhu et al., 2017).
Therefore, the timing of the drop in CRM contents at the community level, the fact that this result is imputable to FG 2 trend and the specific type of materials exiting the composition of screens and monitors FG indicate that the replacement of CRT technologies led to a significant saving of CRM absorbed by the TT.

[16] Despite a different methodology and the focus on a few minerals only, Shigetomi et al. (2015) predicted declining paths for CRM MFs of Japanese households' consumption that are compatible with our time trends.

Besides heavy metals CRM, other two groups of critical materials attracted a special attention in the literature: platinum group metals (PGM)[17], for their high economic value, and rare-earths elements (REE)[18], for their high supply risk (Figure A2 in Appendix A). At the community level, the PGM share of the EEE ecosystem composition has slowly declined in our period of observation, consistently with Cenci et al. (2021) findings. The two FG with the highest concentration of PGM are small IT, and screens and monitors, once again. Instead, the REE share of the EEE ecosystem composition is slightly higher at the end of the period compared to early 2000s. Mobile phones are the technology with the highest concentration of REE, after lamps (Favot and Massarutto, 2019).
More in general, more complex devices usually also contain a vaster array of CRM.

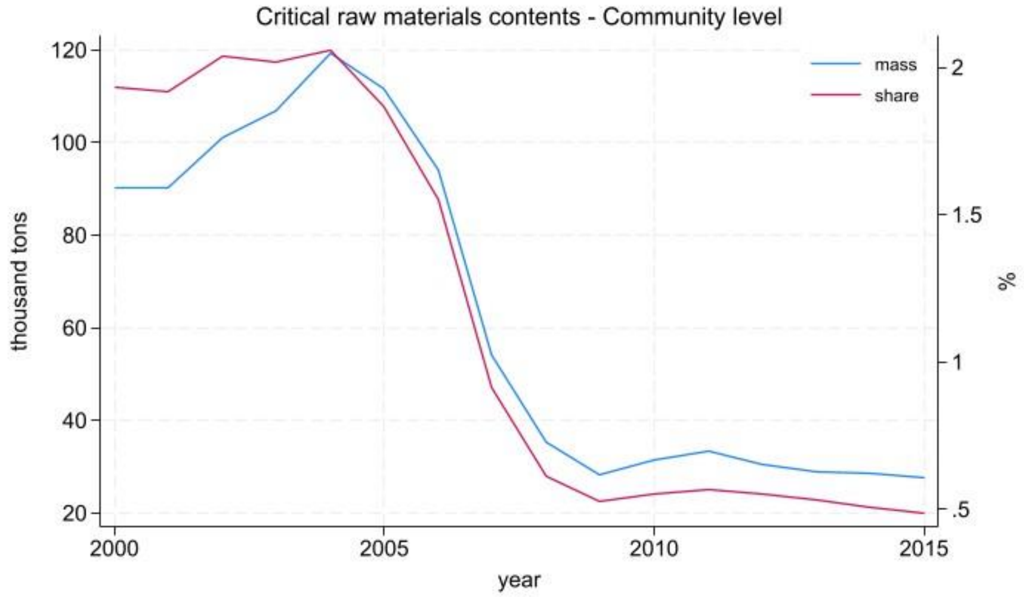


*Figure 4 CRM contents in EEE POM at the community level in mass (left axis) and percentage composition (right), 2000-2015.*

[17] Platinum group metals: Gold, Silver, Iridium, Palladium, Platinum, Rhodium, Ruthenium

[18] Rare earths: Cerium, Lanthanum, Neodymium, Praseodymium, Samarium, Dysprosium, Erbium, Europium, Gadolinium, Holmium, Lutetium, Terbium, Thulium, Ytterbium, Yttrium

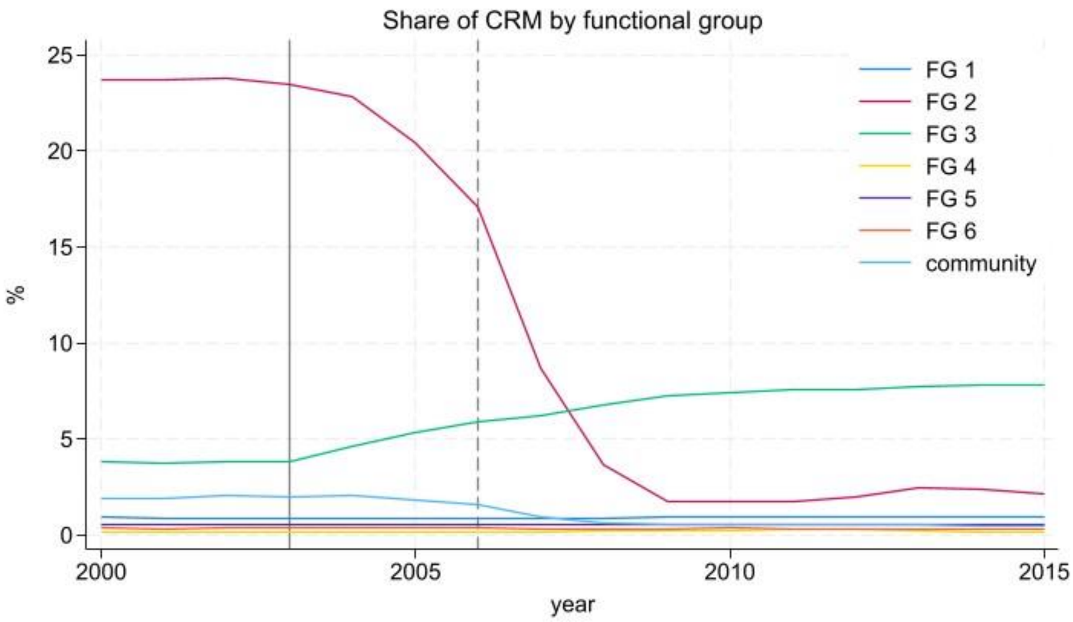


*Figure 5 Percentage composition in CRM by functional group and at the community level, 2000-2015. Solid line indicates the introduction of EU RoHS Directive (2003) and dashed line the introduction of EU REACH Directive (2006).*

In order to deepen our understanding of the material footprint of our technological trajectory, we explore the dependence of the TT from geologically scarce resources.

Arvidsson et al. (2020) proposed *crustal scarcity potential* (CSP) coefficients based on the concentration of chemical elements on the Earth's crust, thereby constituting proxies for long-term global elemental scarcity. On the basis of these CSP[19] and of our BOM data, we adapted Arvidsson et al. (2020) methodology to calculate a *crustal scarcity indicator* (CSI) at EEE community and FGs level:

$$CSI_{l,t} = \sum_{e} CSP_e * mass_{\ e,l,t}$$

where: $l$ stands for the level at which the indicator is calculated, i.e. FG or community level; $e$ is the chemical element; $t$ is the reference period. To reduce the volatility of the indicator, CSIs were calculated on three-years' time windows averages. The calculated CSIs provide two main insights (Table 7). At the community level, the CSI decreased implying a reduced aggregate dependence on geologically scarce resources. Disaggregating the FGs, the three FGs more strongly based on scarce materials are also the most complex ones, and in particular small IT and telecommunications devices have the highest CSI.

[19] CSP are measured in kilograms of silicon (Si) equivalents.

*Table 7 Variations of CSI between 2000-2002 and 2013-2015 periods, and CSI ranking for 2013-2015 period.*

| Functional group (WEEE Directive) | variation $CSI_{l,\ 2000\text{-}2002}$ - $CSI_{l,\ 2013\text{-}2015}$ | $CSI_{l,\ 2013\text{-}2015}$ rank |
|---|---|---|
| FG 1 | + | 5 |
| FG 2 | - | 3 |
| FG 3 | + | 6 |
| FG 4 | - | 4 |
| FG 5 | + | 2 |
| FG 6 | - | 1 |
| EEE community | - | 1.76 e^10 kg Si eq |

Summing up, the analysis at the materials level provided some interesting and, in some respects, surprising evidence. To begin with, the TT is moving towards increasingly complex devices, opening up concern in relation to the sustainability of our technological trajectory. Furthermore, CRM contents have decreased on aggregate, due, in particular, to critical heavy metals substitution. Nonetheless, the consumption of other critical materials, such as rare earths, has increased. Finally, overall, the TT seems to be less dependent on geologically scarce materials, but the most complex devices are the ones exposed to the highest risk of resource depletion.

## 5. Discussion

On the basis of our previous results, we discuss empirical evidences related to the three challenges described in Introduction and their consequences in terms of MF of the TT. In considering the mechanisms for the improvement of the material efficiency of the TT at each level of analysis, we aim to stress the fact that these mechanisms may result in contradicting MF outcomes when adopting a systematic perspective.

### 5.1. Challenge 1: aggregate metabolism

Considering the TT at its community level, we find both worrying and encouraging evidence. On the one hand, the TT has lightened in the last fifteen years of observation, with a decreasing weight of the community of EEE placed on the European market. Moreover, after a phase of recovery following the 2007-2008 economic crisis, the number of new EEE seems to have approached a saturation. On the other hand, the aggregate weight of the EEE community has historically followed and still closely follows the number of units POM. Hence, a robust trend of absolute decoupling is not observed yet and the aggregate mass of materials embedded in the EEE community is still very close to its historical maximum. This is because the lightening of the TT could not keep pace with the diffusion of technological devices, driven by both the expanding demand and supply of EEE.

Ultimately, an absolute reduction of the aggregate metabolism of the TT can occur either through a reduction of the consumption of technologies or through further technological transitions to more resource-friendly artifacts. When exploring this second option and considering the TT at the community level of analysis, two directions appear to benefit its MF. The first is to fasten functional dematerialization, that is to reduce the quantity of materials consumed by a device, ceteris paribus its functional performance. The second is to favour the replacement of incumbent technologies, changing the composition of the EEE community. This can occur through the one-to-one technological substitution of heavier with lighter devices. Alternatively, this can occur through the convergence of technologies: when multi-functional devices enter the TT, they displace incumbent single-functional ones, allowing to reduce the aggregate number of new devices. Hence, technological convergence represents a form of demand control driven by technological innovation and not by consumption control.

These findings claim for more research on consumers' preferences and willingness to pay for multi-functional technological devices (Desmarchelier et al., 2017; Kasulaitis et al., 2021). Servitization processes, in the form of product sharing, can in principle also reduce the proliferation of artifacts placed on market (Pasimeni and Ciarli, 2023). Yet, the effective environmental outcomes of servitization and sharing processes is currently quite contested (Tukker, 2015). Therefore, this represents a further direction for future research.

Summing up, functional dematerialization, technological substitution, technological convergence, and aggregate consumption reduction of artifacts are all phenomena at the functional group or products level that can potentially lead to a reduction of the MF of the TT at the community level (upward causation mechanism, in the scale taxonomy).

### 5.2. Challenge 2: material complexity

The increasing functionality of technologies allows their convergence into a lower number of devices and, in some cases and to some extent, a substitution of tangible with intangible solutions. Nevertheless, this technological convergence phenomenon has a negative downward causation impact at the level of materials. The increasing functionality of artifacts is correlated to their growing material complexity, which hampers the recyclability of end-of-life technologies and, thus, resources life extension (Andersson et al., 2019; Hagelüken and Goldmann, 2022; van Nielen et al., 2022). Indeed, given the current organization of recycling value chains and technologies, it is often impossible to separately recover materials contained in minimal percentages and/or mixed with a high number of different materials because of technical or economic efficiency reasons.

Hence, when focusing on the dimension of material complexity of the TT, improvements to its MF would come from: the design of technologies for life extension and end-of-life (Parajuly et al., 2019; Tecchio et al., 2017); the development of markets for recycled materials (Nicolli et al., 2012; Rosendahl and Rubiano, 2019): to this end, the introduction of recycling contents targets in specific EEE may be helpful (see the case of the EU Batteries Regulation[20]); the development of recycling innovations and value chains (Andersson et al., 2019; Pommeret et al., 2022).

Regarding this last scope, innovation in the European electronic waste recycling sector seems limited, as suggested by the low number of patent applications in the last two decades, both in absolute terms and when compared to the total number of other recycling patents (Figure 6). Germany is by far the most innovative country in the electronic waste recycling sector, with about one hundred patents filed in the period 2000-2018. Investments in this sector are discouraged by technical factors, such as the unfriendly design of EEE and their growing complexity, and by economic profitability factors, such as the difficulty in reaching economies of scale for certain types of waste or materials (Compagnoni, 2022; Magalini and Huisman, 2018). Worldwide, the leaders in electronic waste recycling are located in East Asia (Figure 7). These two Figures are based on PATSTAT data and on the OECD ENV-TECH classification (Haščič and Migotto, 2015) for the definition of the technological fields.

[20] Regulation (EU) 2023/1542 https://eur-lex.europa.eu/eli/reg/2023/1542/oj

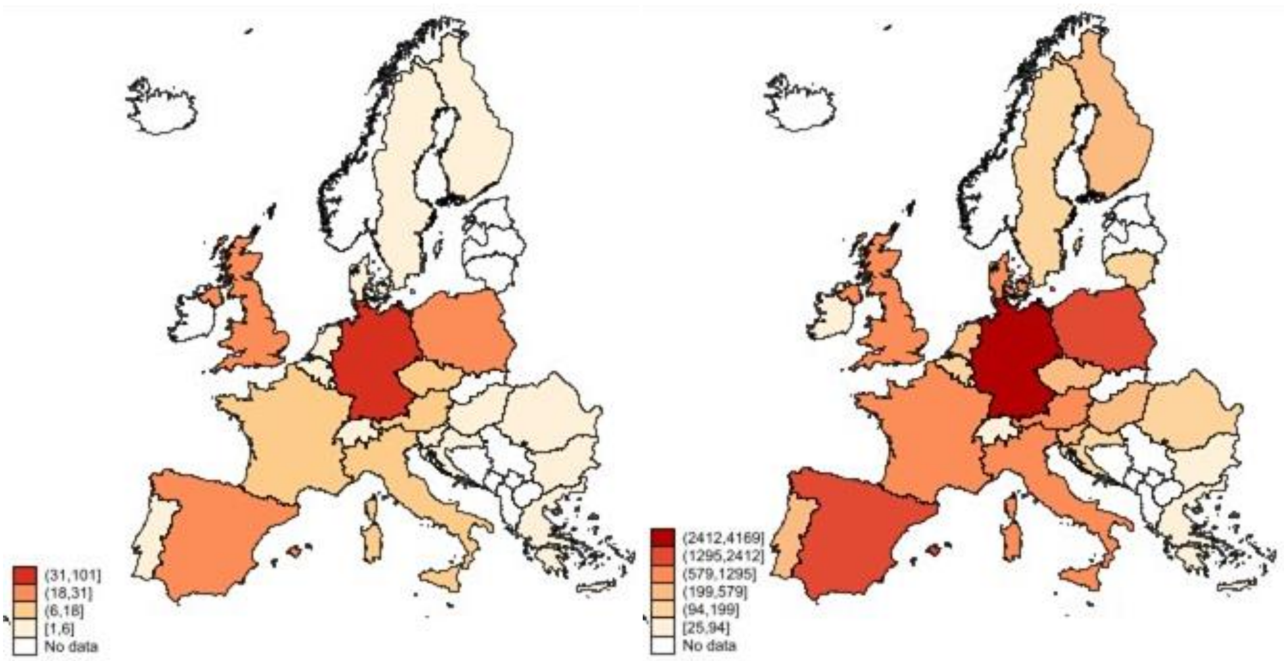


*Figure 6 Map for the total number of electronic waste recycling patents (left) and other recycling patents filed in European countries (right) over the period 2000-2018.*

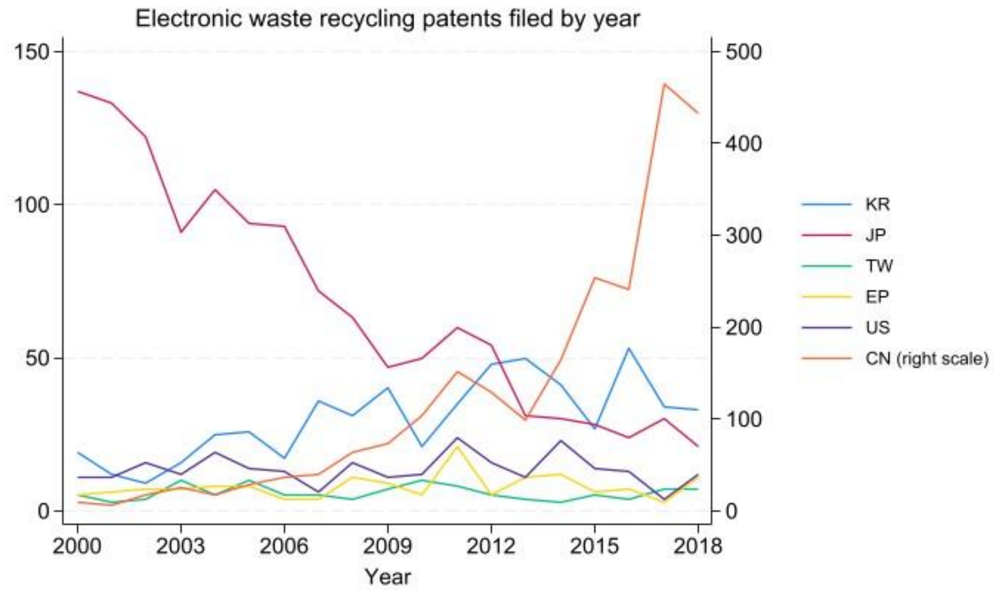


*Figure 7 Number of electronic waste recycling patents filed, top countries worldwide, 2000-2018. Legend: KR South Korea, JP Japan, TW Taiwan, EP European Patent Office, US United States, CN China (right axis).*

## 5.3. Challenge 3: dependence on critical and scarce materials

The analysis of CRM and geologically scarce materials contents revealed some results that partially challenge the general narrative of recent literature. Indeed, according to our data the aggregate mass and percentage composition of CRM in the TT has decreased significantly. In particular, this is due to a replacement of cathode-ray tube technologies, which contained high quantities of critical (and non) heavy metals. The case of this technological discontinuity in the screens-monitors-laptops functional group proves that technological innovation may not necessarily be directed towards an increasing CRM consumption. This is an example of a positive downward causation mechanism due to technological substitution.

It is interesting to speculate on a possible triggering factor of this substitution of critical heavy metals. Some environmental regulations, namely the REACH and RoHS Directives, were introduced in the EU in the early 2000s, restricting the use of certain toxic and pollutant materials in electronics. Given the timing of this shift as well as the type of affected materials, it is possible to advance the hypothesis that these regulations, not only supported the reduction of heavy metals in incumbent technologies (Cenci et al., 2021; Kolias et al., 2014), but they also provided an incentive to fasten the technological discontinuity. The effect of REACH regulation on technological development was investigated by Arfaoui et al. (2014) adopting agent-based model methodologies; empirical analysis should further explore this direction, exploiting, for instance, patent data sources.

In addition to this evidence related to CRM, the TT dependence on geologically scarce resources also diminished, on aggregate.

Nonetheless, adverse trends are also occurring. For specific groups of CRM, such as rare earths, their consumption has increased, showing a limited substitutability of these materials (Cenci et al. 2021), despite their high price volatility (Fernandez, 2017). Moreover, the material complexity of EEE is correlated with the variety of CRM contents, as well as to the consumption of geologically scarce minerals. These findings are particularly worrying, as they suggest that, while technologies become more complex, they could be exposed to higher resource depletion risks and they hamper the recyclability of materials exposed to higher supply risks.

In particular, the transition to Industry 4.0 production processes has been enabled by highly complex technologies, which are only partially tracked in our dataset. This claims for attention by future research in investigating the evolution of Industry 4.0 technologies in terms of complexity, recyclability, and CRM and scarce materials dependence (Carrara et al., 2023). Furthermore, these considerations should be integrated in public policies driving technological developments as well as in EEE industry technology design and innovation investments decisions (Baldassarre et al., 2023).

Thus, this level of analysis highlights two major risks for the MF of the TT of our society: a possibility *lock-in* of some technologies to some specific CRM and the challenge of limited circularity in the use of these resources due to the complexity of technologies.

Consequently, these findings reaffirm the relevance of environmental regulation both supporting material inputs substitution and, possibly, technological transitions in favour of devices characterized by low contents of critical and scarce resources by design.

## 6. Concluding remarks

Literature brings divergent evidence on the environmental impacts of our technological trajectory (Barteková and Börkey, 2022; UNEP, 2014). To shed light on this topic, in this paper we disentangle the complex and interrelated relationship between the evolution of technology and their MF. One of the weak points of the related literature refers to the unit of analysis (Berkhout and Hertin, 2004). To develop a systematic, multi-level framework that simultaneously describes the waves of diffusion of technologies and the mass of materials embedded in each type of technology, we build on the concepts of scale and product ecosystem: the former allows to explicitly define the boundaries and levels of our system of analysis, while the latter is useful to represent the fact that technologies are not used in isolation, but rather in combination, and their diffusion co-evolves.

According to our findings, the major benefits to the reduction of materials consumption came from the evolution of the composition of the TT, rather than from functional dematerialization, that is a reduction of the quantity of materials necessary for an artifact or a component to perform a specific function. Indeed, starting from the 2000s, a replacement of heavier with lighter technologies and of single-function with multi-functional devices is occurring. This latter phenomenon, defined technological convergence, allows the reduction of the aggregate number of devices placed on the market belonging to certain functional groups. Hence, technological convergence represents a form of demand control driven by technological innovation and not by consumption control (Kasulaitis et al., 2021).

Unfortunately, the increasing functionality of technologies is at the same time the main driver of decoupling and of the risk of directing the TT towards a dependence from some specific critical and geologically scarce materials. This is due to their material complexity, that is the variety of the embedded materials, many of which are contained in minor concentrations. As a consequence, the recyclability of these resources is hampered, preventing circular economy strategies for diminishing supply risks to achieve a significant scale.

Sadly, patent data shows weak innovation in the European electronic waste recycling system. The fact that Europe is lagging behind can be related to the effectiveness of policy actions or a lack of knowledge bases (Ashford and Hall, 2018; Hong Nham and Ha, 2022). This point opens rooms for discussion and studies on the role of policy makers at a regional and national level in promoting innovation in the electronic waste recycling sector.

In conclusion, innovation in the form of functional improvements has both positive and adverse effects of the MF of the TT. To counterbalance these adverse effects, "downstream" circular economy strategies focusing on advancements of waste management systems are not enough. Indeed, despite the absolute necessity of enhancing the recirculation of materials through the increase of electronic waste collection rates, an advancement of recycling technologies and the creation of markets for secondary raw materials, recycling inevitably encounters technical limits (Meran, 2023; Reck and Graedel, 2012) leading to a "material loss" (Compagnoni and Stadler, 2021; Merz, 2016) from the economic system, that is the generation of non-marketable materials. Furthermore, the stream of recycled materials cannot keep pace with raw materials demand if decoupling is not occurring. Until the speed of innovation aimed at the development of new products to be placed on the market is seen as the high-tech and electronics industries' key competence and attractiveness for consumers, the goal of decoupling seems difficult to achieve (Lauridsen and Jørgensen, 2010; Santarius et al., 2023a).
Therefore, interventions at the level of technology design seems to be especially needed. For instance, environmental regulations restricting the use of specific materials have been useful, not only to trigger a substitution of these materials, but also to anticipate a switch of technologies designed since their origin with lower contents of the targeted materials. This role of environmental regulations at the materials level as technological transitions push seems to have been overlooked, while instead it may be considered not only for the substitution of pollutant materials, but also of critical or scarce ones. The sustainable use of minerals should be incorporated into climate and technology planning (Pommeret et al., 2022; Sovacool et al., 2020), where greenhouse gasses and carbon pricing received most of the attention (Rosenbloom et al., 2020). These recommendations are in the wake of Sustainability Transitions scholars' perspective: policies are not technology-neutral and they can (or should) direct the purpose of innovation, when market outcomes are not compatible with long-run sustainability goals (Azar and Sandén, 2011). Here, the point of view of Aggeri (2023), arguing in favour of "responsible and frugal innovations", seems to be particularly reasonable: technology innovators should be made more responsible for the long-term, systematic consequences of their projects, and frugal innovation should be promoted, centring on transforming lifestyles and modes of production and consumption in ways that are compatible with planetary boundaries and the needs of future generations.

In terms of research suggestions, we claim that the current momentum that the concept of Twin Transitions has gained should be exploited to critically, systematically and empirically investigate the compatibility of the digital transition with material footprint and, more generally, sustainable development goals, rather than crystalizing on narrow techno-optimist and pessimist perspectives (Kerschner and Ehlers, 2016; Tanguy et al., 2023). To this end, the connection of Ecological Economics and Industrial Ecology streams of research, on the one hand, with Technological and Sustainability Transitions, on the other, can surely support the development of more comprehensive perspectives and empirical approaches.

The limitations of the scope of analysis of this paper, both in terms of MF and of set of technologies, indicate two relevant directions to extend our work, currently hampered by the limited data availability. The first is the integration of indirect MFs of electronic artifacts, such as material inputs per product units. The second is the integration of energy transition technologies, such as wind turbines and batteries, into the analysed product ecosystem, as they strongly rely on CRM.

> *"Information and communication technology can help a great deal in saving energy, materials, and transport costs.* ***Whether it does so, however, is a matter of social and economic policies, as well as of technology and science policy****. Consequently, whether the world economy can move to a new and sustainable pattern of growth remains an open question. It is both a question of new priorities for public and private R&D to nurture a new range of possibilities in such areas as renewable and "cleaner" energy sources and energy and materials conservation devices, and a question of new regulatory mechanisms to ensure their worldwide diffusion. The latter may be the more difficult problem."*
>
> *Chris Freeman, The greening of technology and models of innovation (1996)*

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# Is Extended Producer Responsibility living up to expectations? A review with a focus on electronic waste [1]

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**Abstract**

Extended Producer Responsibility (EPR) is an environmental policy principle conceived for the incorporation of total product life-cycle (PLC) costs into production and consumption decisions. Consequently, EPR is expected to enhance the circularity of the value chains affected by this regulation. The lack of comprehensive evaluations of EPR achievements hampers the possibility to assess the actual alignment of the policy impacts with Circular Economy (CE) objectives. This is true also in the context of electronic waste (WEEE), which has been prioritized by EPR regulations and scientific investigations. This paper provides the first systematic literature review aiming to comprehensively examine the outcomes of EPR implementation in the WEEE scope, by adopting a PLC perspective. The review highlights the accomplishment of important downstream goals of EPR on WEEE, such as the increase of waste collection rates and the development of stable waste management systems. On the other hand, the review highlights limited upstream effectiveness of the policy, which is due to the insufficient allocation of individual responsibility to electronics producers to systematically drive them towards eco-design strategies. Discrepancies between CE goals and EPR achievements and implementation are classified into seven areas, covering the entire PLC and also representing domains of policy recommendations. Finally, the paper identifies a number of future research directions that would support the alignment of EPR policies and CE objectives in the electronics value chain.

**Keywords**: extended producer responsibility; circular economy; electronic waste; systematic literature review; product life-cycle; policy evaluation

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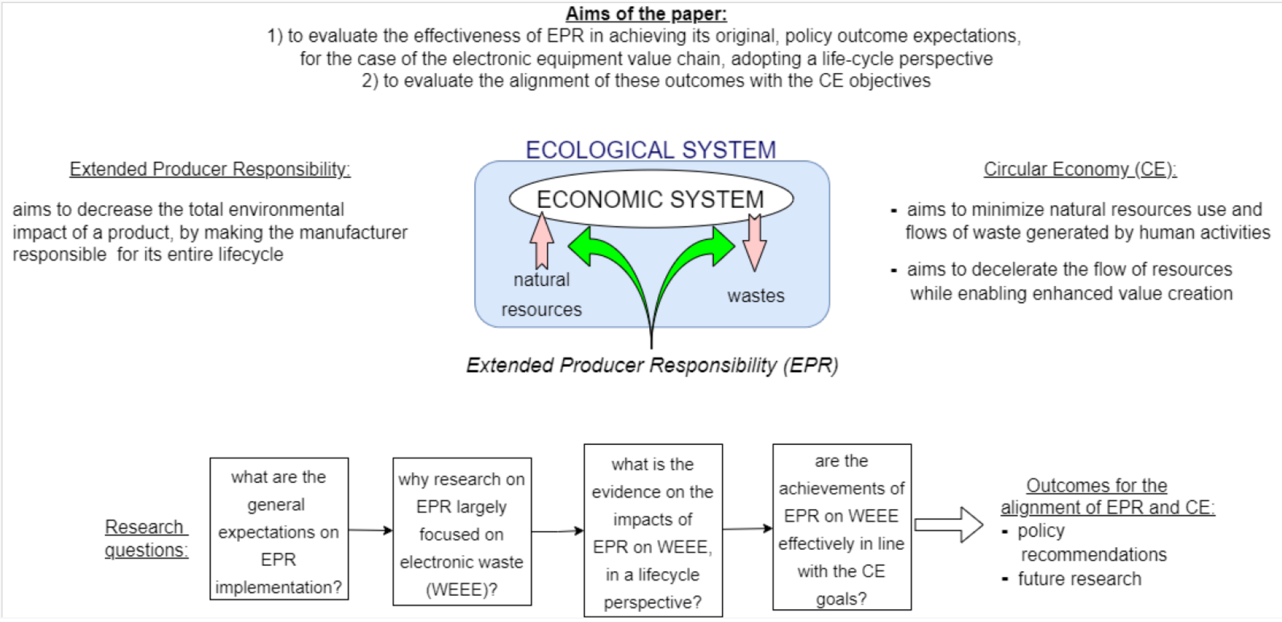
## Graphical abstract




## 1. Introduction

An increase in emissions of pollutants and the exploitation of natural resources has led to increasing pressure on the Earth's System (Steffen et al., 2015). The limited ability of nature to absorb the wastes of human activities and to provide natural resources as production inputs are widely-recognized obstacles to sustainable development (Compagnoni and Stadler, 2021; Meadows and Randers, 2012). It is argued that a systematic transition towards environmental sustainability requires total product life-cycle (PLC) costs to be incorporated into production and consumption decisions (Moraga et al., 2019). The life-cycle approach tries to avoid environmental "problems shifting", which is the transfer of the problem from one life-cycle stage to another, from one agent to another, from one location to another, or from one environmental medium to another, without solving the issue (UNEP, 2005). The principle of Extended Producer Responsibility (EPR) was elaborated based on this approach (Lifset and Lindqvist, 2008).

The concept of EPR, first introduced in the 1990s, is defined by Lindhqvist (2000, p. 37) as a "strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product". In other words, EPR is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a PLC (Lifset, 1993). More specifically, these responsibilities are of the informative, physical, economic, liability and ownership type (Lindhqvist, 2000); EPR asks manufacturers to provide information about the potential environmental impacts of their products, to be responsible for the physical handling, and for covering the cost associated with the end-of-life management of their products. In addition, liability for the damages that a product causes during its life-cycle remains with the producers.

Nowadays, EPR is conceived as a policy approach to promote the circular economy (CE) both in the European Union (EU) (Andersen, 2021; Campbell-Johnston et al., 2021; Pouikli, 2020) and in developing countries (Panchal et al., 2021). CE relies on the systematic application of strategies that can decelerate the flow of resources while enabling enhanced value creation (Bocken et al., 2016). EPR is envisaged as a CE enabler, because of its potential to impact the overall environmental sustainability and resource efficiency of the affected value chains, both upstream, i.e., in production phases, and downstream, i.e., waste management (Gu et al., 2018).
European countries have been the first movers in the implementation of the EPR principle (Corsini et al., 2017) on various waste flows (Tab. 2), among which is electric and electronic waste (WEEE) (Directive 2002/96/EC). The introduction of the policy was justified and advocated (Huisman et al., 2008) in consideration of numerous pressing environmental and economic issues linked to WEEE (Huisman et al., 2008; Pérez-Belis et al., 2015): the rapid growth of generation rate of WEEE worldwide (Parajuly et al., 2019); the low collection and recycling rates before the advent of EPR regulations (Massarutto, 2014); the dangers of improperly treated WEEE, both for the environment (Kiddee et al., 2013) and human health (Parvez et al., 2021); the systematic illicit WEEE trade flows from developed to developing countries (Odeyingbo et al., 2017). In addition to direct environmental impacts, these phenomena entail a limited circularity of resources use, particularly for critical raw materials (Althaf and Babbitt, 2021). Overall, a transition towards CE of the EEE value chain is necessary (Guzzo et al. 2021).

Despite the mature implementation of EPR on WEEE, currently it is difficult to have a comprehensive picture of the actual EPR impacts (Corsini et al., 2017). First of all, this is due to the heterogeneous expectations on EPR implementation, which are related to the entire PLC. However, scientific publications on EPR typically focus either on upstream or downstream effects of the policy, rather than adopting a PLC perspective. Generally speaking, research on the electric and electronic equipment (EEE) value chain rarely adopts a systemic perspective on the different life-cycle phases and related CE strategies (Bressanelli et al., 2020). In addition, the assessment of the impacts of EPR on WEEE is complicated since the heterogeneity of EPR goals is combined with the manyfold WEEE issues for which the policy is expected to provide some benefits and with the complexity of the EEE value chain (Tukker at al. 2016). Due to these constraints in comprehensively evaluating the impacts of EPR, few studies have critically focused on the interplay between EPR and CE (Campbell-Johnston et al., 2021).

The ultimate aim of the paper is twofold. First, adopting a life-cycle perspective, the paper comprehensively evaluates the effectiveness of EPR in achieving its original expectations in terms of policy outcomes, specifically for the EEE value chain. To this end, the paper reconstructs the general rationale and goals of EPR, and clarifies the relevance of WEEE issues. Secondly, the coherence of actual, current results of EPR implementation with the CE objectives is examined. Based on the analysis, the paper provides recommendations for strategic and operational adaptations of EPR systems to ensure an improvement in the circularity of EEE value chains.

This paper presents the first systematic literature review (Tranfield et al., 2003) on EPR based on scientific publications and focusing on the policy impacts along the whole PLC. The only other systematic review on EPR is provided by Cai and Choi (2021), who relied on the Web of Science database to select their publications. The authors adopted an operations management perspective and proposed operational measures to improve the sustainability of EPR systems. This paper differentiates from Cai and Choi (2021) in at least three ways: it adopts a different perspective, i.e., EPR in the context of CE; it stresses the importance of analyzing and enhancing EPR policies in a PLC perspective; it uses Scopus database to select the scientific publications to be analyzed. Other reviews analyze country-level case studies without explicitly referring to the CE concept (Gupt and Sahay, 2015) or the legislative implementation of EPR, either documenting their evolution in time within a specific country (Hickle, 2014; Manomaivibool and Hong, 2014) or comparing national normative differences (Cahill et al., 2011). Finally, other reviews focused on the management of specific waste flows, typically WEEE (Herat, 2021; Jang and Kim, 2010; Li et al., 2021; Premalatha et al., 2014), rather than explicitly on the EPR mechanism.

The work is structured as follows. First, the connection between EPR and CE is discussed. Secondly, the review methodology is described. Third, a bibliometric analysis for the selected literatures is presented. One of the aims here is to demonstrate that WEEE is the waste flow that has most attracted the attention of EPR scholars. Fourth, a content analysis answers four questions: What were the original, general academics and policy makers’ expectations on EPR implementation? Why did research and regulations on EPR prioritize the WEEE scope? What is the current evidence on the impacts of EPR through the life cycle management of EEE? Are EPR achievements aligned with CE objectives? This last question is addressed in Section 5, which is also dedicated to policy recommendations. Finally, the concluding section summarizes the results of the paper, areas of research on the impacts of EPR on WEEE omitted by previous literature and, consequently, it paves the way for various further research.

## 2. Circular Economy and Extended Producer Responsibility

The ultimate aim of a CE is to minimize, on the one hand, natural resources use and, on the other, any flow of waste generated by human activities going back to the environment (Korhonen et al., 2018). Therefore, CE strategies focus on improving the resource efficiency of economic systems, aiming both for their long-run environmental and economic sustainability (EMF 2015; Lieder and Rashid, 2016). The solution advocated by CE is to keep the value of materials and products as high as possible for as long as possible (EEA, 2017), which can be reached via four building blocks of CE transitions (Wasserbaur et al., 2022):

1) Materials and product design, aiming to the reduction of embodied and residual waste impacts, and to lifetime extension (Tecchio et al., 2017) in contrast to planned obsolescence principles (Maitre-Ekern and Dalhammar, 2016).
2) Circular business models (Urbinati et al., 2017), based on the provision of capturing residual value in products and on the substitution of artifacts with services (Tukker, 2015).
3) Reverse supply networks, based on the recovery of used products (Bressanelli et al., 2018; Farooque et al. 2019).
4) Enabling conditions: policies (Mhatre et al. 2021), economically sustainable markets for secondary raw materials (Milios, 2018), consumer awareness (Phulwani et al., 2021), digital technologies (Rosa et al., 2020).

Overall, because of its life-cycle rationale and its orientation to resources conservation, the EPR principle has the potential to improve the circularity of the affected value chains on a global scale (Manomaivibool and Hong, 2014; Mayers et al., 2005). Indeed, EPR regulations typically set waste collection and recycling targets which must be ensured by the respective producers. Moreover, by shifting waste management responsibilities upstream toward producers and consumers and away from municipalities, EPR provides incentives for eco-design to producers (Atasu et al., 2009), that is, the systematic integration of environmental considerations into product and process design (Knight and Jenkins, 2009) (Fig. 1). Hence, EPR is expected to incentivize producers to extend PLCs, prevent waste generation, spur resource efficiency through green innovation and close loops downstream of consumption (Massarutto, 2014). Consequently, EPR can potentially bridge a number of CE objectives.

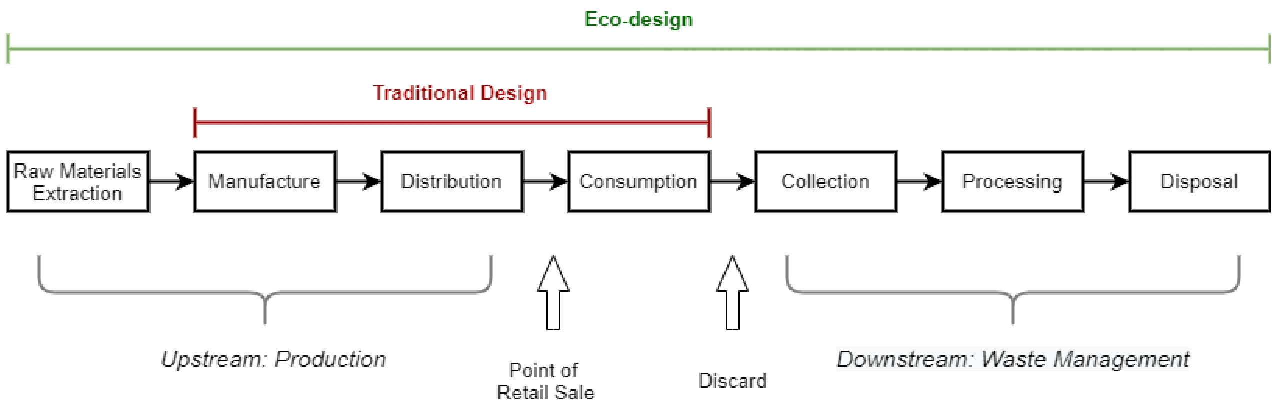



*Fig. 1 Scheme of the product life-cycle and eco-design compared to traditional design. Source: author's elaboration based on Lifset (1993) and Knight and Jenkins (2009).*

## 3. Methodology

This work is based on the three-stages systematic review methodology: planning, searching and reporting (Tranfield et al., 2003). The strength of this review methodology is twofold (Grant and Booth, 2009): it is based on a transparent data collection and analysis protocol, and it seeks to draw together all known knowledge on a topic area.

### 3.1 Planning stage

At the planning stage, the usefulness of a systematic review on the effectiveness of EPR in the WEEE scope was identified in consideration of: the growing number of studies on EPR; the heterogeneity of expectations on EPR implementation; the difficulty of having a clear picture of the effectiveness of EPR in achieving some of those goals in the WEEE context; the paucity of reviews on EPR and, in particular, the absence of reviews reading EPR on WEEE achievements in the light of CE goals.

| SPICE framework feature | Feature in this article |
|---|---|
| Setting – where? | worldwide |
| Perspective – for whom? | (agents involved in) upstream and downstream phases of the EEE life-cycle |
| Intervention – what? | EPR implementation on WEEE |
| Comparison – compared with what? | business-as-usual, ideal situation without EPR |
| Evaluation – with what results/reference point? | coherence with: 1) original, general policy makers and academics' expectation on EPR; 2) CE objectives for the EEE value chain |

*Tab. 1 SPICE framework features of this article.*

As a second step of the planning stage, the SPICE framework (Booth, 2006) is used to clarify the ground of definition of the analysis and of the research questions (Tab. 1).

As a third step and in compliance with the SPICE features defined above, four research questions are defined to gradually achieve the ultimate goals of the paper.

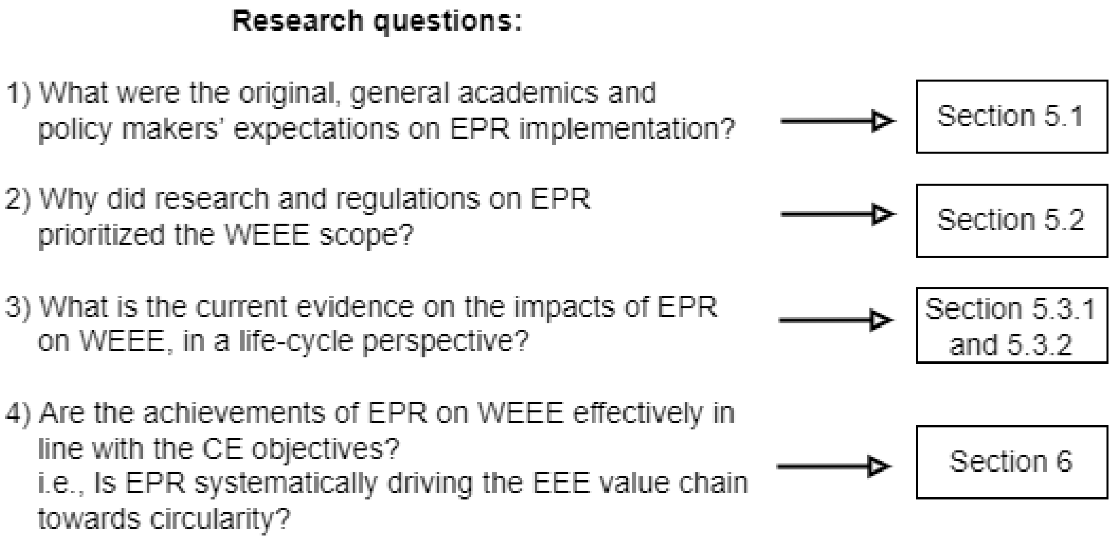



*Fig. 2 Research questions of the paper and Sections in which each of them is addressed.*

### 3.2 Searching stage

This stage discusses the selection procedure for the set of documents to be examined at the following reporting stage. The entire procedure is represented in Fig. 3.

Initially, the keywords to be used for the selection of the documents were defined. It was decided to select two sets of publications. The "EPR dataset" identifies documents reporting the expression "extended producer responsibility" in their title, abstract or authors' keywords. The second dataset, named "EPR and WEEE", collects publications focusing only, simultaneously on EPR and WEEE. Therefore, alongside EPR, a list of secondary keywords related to WEEE (Zhang et al., 2019) is added, via the use of the Boolean connector "AND". The identification of these two datasets was conducted in order to identify relationships between the two respective research fields, as some bibliometrics will show. Moreover, despite research questions 2, 3 and 4 (Fig. 2) focus on the EEE/WEEE scope, papers dealing with the general EPR principle are useful to reconstruct the original expectations addressed to the policy.

Due to their wider coverage, Web of Science (WoS) and Scopus are considered the two dominant scientific publications search engines (Dabic et al., 2020). After a cross-validation, Scopus yield a higher number of contributions, while WoS searches were affected by discrepancies and some relevant studies were not indexed. Hence, it was decided to rely on Scopus as data source, coherently

with many other systematic reviews on sustainability in the field of Economics and Management (e.g., Palumbo et al., 2021; Bressanelli et al., 2020; Acerbi et al., 2021).
After keywords and search engine definition, eligibility criteria were set (Acerbi and Taisch, 2020) and implemented through the "LIMIT-TO" and "EXCLUDE" query operators. The search was carried out in September 2021. Eventually, the identified sets of documents underwent a screening process, operated through abstract and, if necessary, main text reading. The final "EPR" and "EPR and WEEE" datasets are constituted by 549 and 218 publications respectively.
The information reported in the content analysis (see Section 5.3) is mainly extrapolated from the "EPR and WEEE" set with the addition of further 17 documents. These publications were identified through backward snowballing (Wohlin, 2014) during the examination of the documents on EPR and WEEE. Of these publications: 5 were included in the "EPR" set; 4 were not in the "EPR" set, but were indexed in Scopus; 8 were not indexed in Scopus (PhD thesis or scientific reports).

### 3.3 Reporting stage

The reporting stage consists both of descriptive bibliometrics (Sections 4) and content analysis (Sections 5 and 6).

Bibliometrics analyses apply statistical methods to the study of scientific activities in a research field (Donthu et al., 2021). In this paper, bibliometrics aim to provide a number of insights. First, they provide descriptive information on the selected publications datasets. Secondly, they strengthen the rigorousness of the analysis (Donthu et al., 2021). Third, they allow relating the development of the scientific fields of EPR and of EPR on WEEE with the spreading of the relative policies, both in timing and geographical terms. Fourth, they provide insights on the relationship between EPR and CE concepts and policies. Fifth, descriptive bibliometrics show that WEEE has most attracted EPR researchers' attention, among the various waste scopes in which EPR has been implemented.
In this paper, bibliometrics and maps are elaborated with the VOSviewer software (van Eck and Waltman, 2010), in line with previous research using systematic reviews and/or data searches methods (e.g. Caputo et al., 2021; Pizzi et al., 2021; Atanasovska et al. 2022).

Finally, the content analysis aims to synthesize and reframe information from the selected literature in order to sequentially answer the four research questions (Fig. 2). The critical assessment of actual EPR support to the transition of the EEE value chain towards CE is presented in a stand-alone section, which gathers a number of policy recommendations adopting a PLC conceptual perspective (Fig. 9). Directions for future research are illustrated in the concluding section.

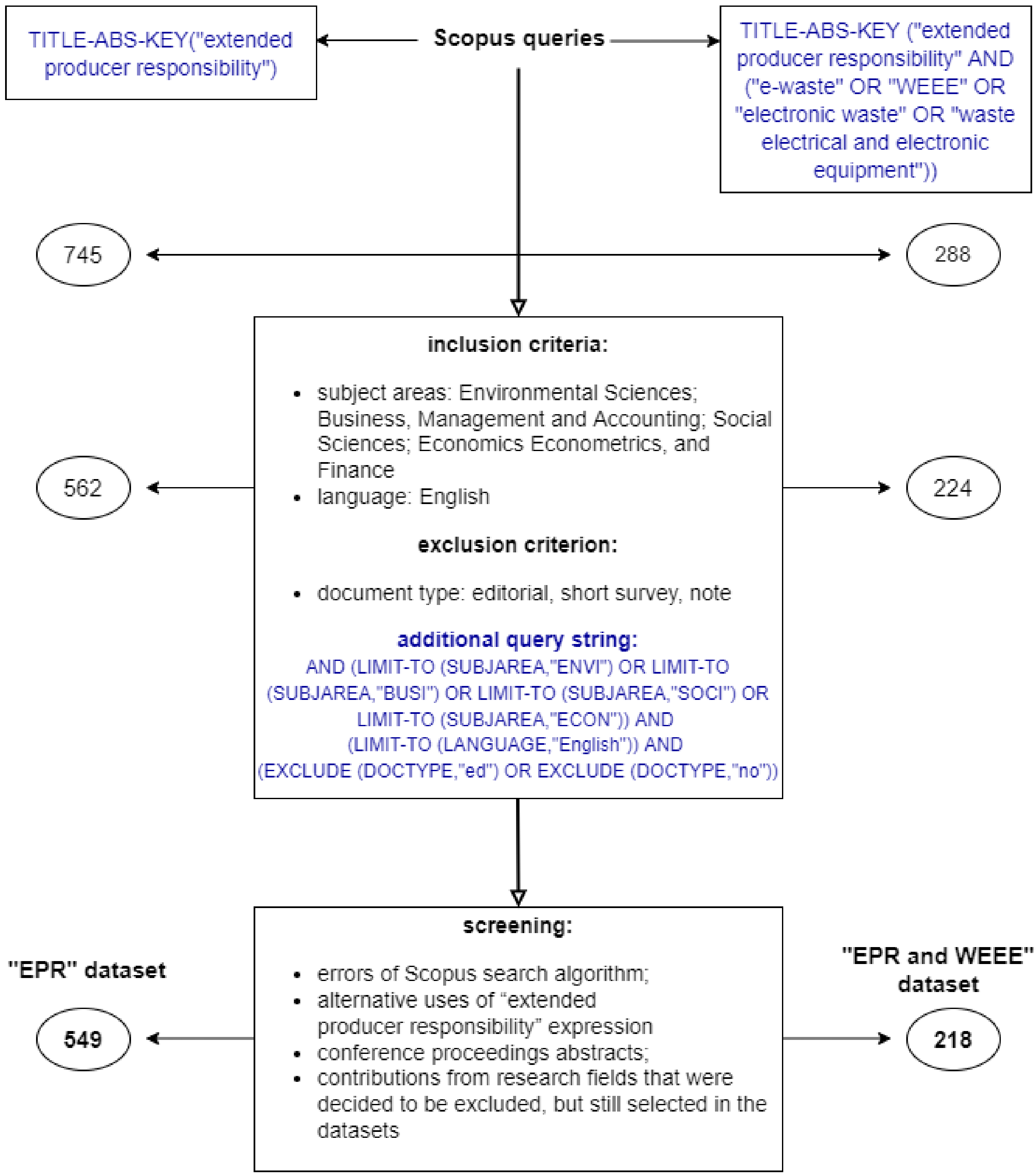



*Fig. 3 Search protocol and numerousness of selected sets of documents.*

## 3. Bibliometric analysis

### 3.1 Publications time trend

After Thomas Lindhqvist's fundamental contributions[2] and Reid Lifset's seminal paper (1993), the EPR concept started appearing in (Scopus indexed) scientific documents, but at an extremely little frequency. Research devoted an increasing attention to EPR starting from the early 2000s (Fig. 4), reacting to the diffusion of EPR regulations. The EU initiatives have been particularly relevant (Tab. 2). Indeed, the EU provided legal frameworks for the implementation of EPR systems for various waste flows – packaging, end-of-life vehicles (ELV), WEEE, batteries (Mayers et al., 2013). Moreover, the EU Waste Framework and Eco-design directives in 2008-2009 and the formal launch of the EU CE strategy in 2015 further stressed the EU effort to adopt a life-cycle perspective to guarantee the environmental sustainability of products distributed or produced in Europe (Compagnoni, 2020; Milios, 2018), coherently with the EPR principle (Kunz et al., 2018; Wiesmeth and Häckl, 2017). Given that the productions affected by EPR are structured on global supply chains, regional norms have had spillover effects worldwide (Scheijgrond, 2011; Wilts et al., 2011; Yu et al., 2006). Eventually, often taking EU initiatives and targets as reference points (Ongondo et al., 2011), many other countries introduced EPR legislations, e.g. China (Hou et al., 2020; Yu et al., 2008), India (Bhaskar and Turaga, 2018), Korea (Jang, 2010), US states (Biedenkopf, 2020), with a special attention to WEEE.

All of these initiatives explain the growth of the number of publications on EPR and on EPR and WEEE observed between 2014-15 and 2019-20. 51% of the selected documents on EPR were published in the last five years, 2016-2021; the same holds for the subsample of publication on WEEE. In the period 2005-2021, the share of documents dealing at the same time with EPR and WEEE, over the total of selected documents on EPR, was 43% on average. This figure proves how relevant is the topic of WEEE within the EPR literature.

[2] The concept of EPR was first formally introduced in a 1990 report to the Swedish Ministry of the Environment: Lindhqvist T., Lidgren K. "Modeller för förlängt producentansvar" ("Models for Extended Producer Responsibility"). Subsequently, a first definition was provided in 1992 in Lindhqvist's report for the Swedish Ministry for the Environment and Natural Resources "Mot ett förlängt producentansvar — analys av erfarenheter samt förslag" ("Towards an Extended Producer Responsibility — analysis of experiences and proposals")

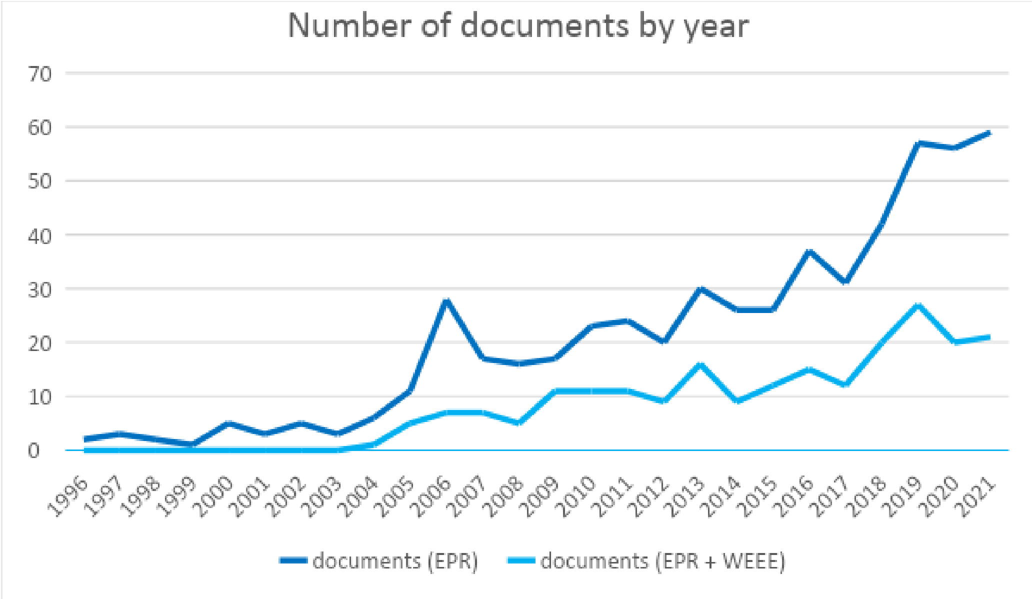



*Fig. 4 Number of scientific publications on EPR (dark blue) and EPR and WEEE (light blue) per year. Source: own elaboration.*

| *Norm* | *Year* | *Amended/abrogated/modified by* | *Scope* |
|---|---|---|---|
| Directive 94/62/EC | 1994 | Directive 2018/852/EU (EU CE Package) | Packaging and packaging waste |
| Directive 2000/53/EC | 2000 | Directive 2018/849/EU (EU CE Package) | End-of-life vehicles |
| Directive 2002/96/EC | 2002 | Directive 2012/19/EU | WEEE |
| Directive 2006/66/EC | 2006 | Directive 2018/849/EU (EU CE Package) | Batteries and waste batteries |
| Directive 2002/95/EC | 2002 | Directive 2011/65/EU | Restrictions on hazardous substances in EEE (RoHS) |
| Regulation 1907/2006 | 2006 | | Chemicals (REACH) |
| Directive 2009/125/EC | 2009 | Directive 2012/27/EU | Eco-design directive |
| Directive 2012/19/EU | 2012 | Directive 2018/849/EU (EU CE Package) | WEEE (“new WEEE directive”) |
| Directive 1999/31/EC | 1999 | Directive 2018/850/EU (EU CE Package) | landfilling |
| Directive 2008/98/EC | 2008 | Directive 2018/851/EU (EU CE Package) | Waste Framework Directive |
| EU strategic documents: | | | |
| Circular Economy Action Plan (COM/2015/0614), 2015 | | | |
| New Circular Economy Action Plan, 2020 (part of European Green Deal) | | | |

*Tab. 2 EU laws and strategic documents formally introducing the EPR principle or providing the legal framework for its application (highlighted in blue) under different scopes, and other EU norms and strategic documents having relevant implications on EPR application under various scopes. Source: own elaboration.*

### 3.2 Sources

The analysis of the sources of the scientific publications provides a picture of the outlets that have most contributed to the development of the research on EPR. Tab. 3, referring to papers on EPR and

WEEE, ranks the scientific journals according to the total number of documents published and their aggregate citations. These metrics show that six journals collected the large majority of the contributions on EPR and provide the greatest number of references to subsequent research: *Resources, Conservation and recycling*; *Journal of Cleaner Production*; *Journal of Industrial Ecology*; *Waste management and Research*; *Waste Management; Sustainability*. By looking, instead, at the average number of citations per document, it is interesting to note that some of the most cited contributions were published by sources collecting few other contributions on the topic. This is the case of relatively old contributions such as Widmer et al. (2005) published on *Environmental Impact Assessment Review*, Atasu et al. (2009) on *Production and Operations Management*, Sthiannopkao and Wong (2013) on *Science of the Total Environment*, for instance. *Resources, Conservation and recycling* and *Waste Management* are the only journals in the top 10 ranking for all the three metrics used.

| source | documents | source | citations | source | avg cit |
|---|---|---|---|---|---|
| resources, conservation and recycling | 22 | resources, conservation and recycling | 1211 | environmental impact assessment review | 492 |
| journal of cleaner production | 20 | environmental impact assessment review | 983 | environmental science and technology | 205 |
| journal of industrial ecology | 19 | waste management | 924 | journal of environmental management | 147 |
| waste management and research | 18 | journal of cleaner production | 679 | production and operations management | 128 |
| waste management | 12 | waste management and research | 675 | science of the total environment | 108 |
| sustainability (switzerland) | 10 | science of the total environment | 538 | management science | 87 |
| environmental science and pollution research | 5 | journal of industrial ecology | 472 | international journal of production research | 83 |
| handbook of electronic waste management: international best practices and case studies | 5 | production and operations management | 383 | waste management | 77 |
| journal of material cycles and waste management | 5 | journal of environmental management | 293 | critical reviews in environmental science and technology | 59 |
| science of the total environment | 5 | environmental science and technology | 205 | resources, conservation and recycling | 55 |

*Tab. 3 Top 10 sources for "EPR and WEEE" sample, according to number of documents, total number of citations, average citations per document. Source: own elaboration.*

### 3.3 Authors

The authors analysis revealed that a total of 1153 authors contributed to the development of the EPR research field, of which 486 also worked on the topic of WEEE. As it can be noted from Tab. 4 and Tab. 5, only a few of the authors extensively providing contributions also rank among the twenty most cited scholars.

| Author (rank 1-10) | documents | Author (rank 11-20) | documents |
|---|---|---|---|
| atasu a. | 13 | wang y. | 6 |
| li y. | 8 | da cruz n.f. | 5 |
| van wassenhove l.n. | 8 | favot m. | 5 |
| mayers k. | 7 | ferrão p. | 5 |
| herat s. | 6 | frey m. | 5 |
| jang y.-c. | 6 | gui l. | 5 |
| lifset r. | 6 | manomaivibool p. | 5 |
| marques r.c. | 6 | subramanian r. | 5 |
| park j. | 6 | wu y. | 5 |
| tong x. | 6 | xu f. | 5 |

*Tab. 4 Top 20 authors ranked by number of documents, sample: EPR. Source: own elaboration.*

| Author (rank 1-10) | citations | documents | Author (rank 11-20) | citations | documents |
|---|---|---|---|---|---|
| widmer r. | 1181 | 2 | kiddee p. | 384 | 1 |
| böni h. | 983 | 2 | naidu r. | 384 | 1 |
| oswald-krapf h. | 979 | 1 | van wassenhove l.n. | 380 | 8 |
| schnellmann m. | 979 | 1 | gupta s. | 364 | 4 |
| sinha-khetriwal d. | 979 | 1 | jang y.-c. | 291 | 6 |
| wong m.h. | 671 | 2 | sthiannopkao s. | 290 | 2 |
| atasu a. | 622 | 13 | lu b. | 283 | 4 |
| nnorom i.c. | 518 | 3 | yang j. | 258 | 4 |
| osibanjo o. | 512 | 2 | burgess s.c. | 257 | 1 |
| subramanian r. | 442 | 5 | ijomah w.; king a.m.; mc mahon c.a. | 257 | 1 |

*Tab. 5 Top 20 authors ranked by total number of citations, sample: EPR. Source: own elaboration.*

Focusing on the publications on EPR in the field of WEEE, it can be noted (Tab. 6) that some of the very top contributors in terms of documents are also in the first positions in Tab. 4. At the same time, authors such as K. Mayers, M. Favot and the Scuola Superiore Sant'Anna Pisa network, led by F. Corsini, gained prominence. By considering the most cited scholars and comparing Tab. 7 and Tab. 5, the top six authors of the two rankings correspond, revealing that these authors provided a limited number of contributions, but highly influential for the whole EPR scientific framework.

| Author (rank 1-4) | documents | Author (rank 4-5) | documents |
|---|---|---|---|
| atasu a. | 7 | wang y. | 4 |
| van wassenhove l.n. | 7 | andersen t. | 3 |
| herat s. | 6 | choi s.o. | 3 |
| favot m. | 5 | kaushal r.k. | 3 |
| jang y.-c. | 5 | li j. | 3 |
| mayers k. | 5 | nnorom i.c. | 3 |
| tong x. | 5 | rizzi f. | 3 |
| corsini f. | 4 | schiller s. | 3 |
| frey m. | 4 | schluep m. | 3 |
| lu b. | 4 | wang h. | 3 |
| manomaivibool p. | 4 | wang j. | 3 |
| park j. | 4 | wiesmeth h. | 3 |

*Tab. 6 Top authors ranked by number of documents, sample: EPR and WEEE. Source: own elaboration.*

| Author (rank 1-10) | citations | documents | Author (rank 11-20) | citations | documents |
|---|---|---|---|---|---|
| widmer r. | 1181 | 2 | naidu r. | 384 | 1 |
| böni h. | 983 | 2 | van wassenhove l.n. | 380 | 7 |
| oswald-krapf h. | 979 | 1 | sthiannopkao s. | 290 | 2 |
| schnellmann m. | 979 | 1 | lu b. | 283 | 4 |
| sinha-khetriwal d. | 979 | 1 | jang y.-c. | 260 | 5 |
| wong m.h. | 671 | 2 | yang j. | 250 | 3 |
| nnorom i.c. | 518 | 3 | song h.t. | 237 | 2 |
| osibanjo o. | 512 | 2 | yu j. | 235 | 3 |
| atasu a. | 468 | 7 | mayers k. | 234 | 5 |
| kiddee p. | 384 | 1 | sarvary m. | 234 | 1 |

*Tab. 7 Top 20 authors ranked by number of citations, sample: EPR and WEEE. Source: own elaboration.*

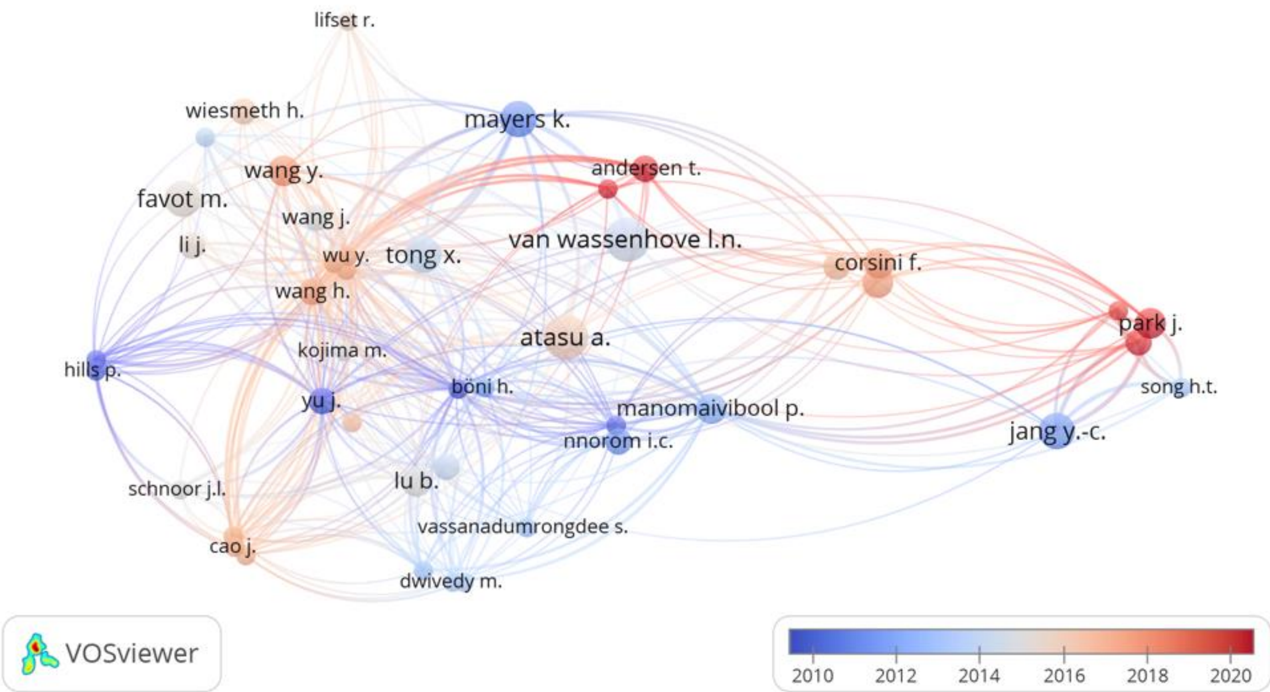



*Fig. 5 Authors map, circles size: number of documents, color: average publication year, sample: EPR and WEEE. Source: own elaboration.*

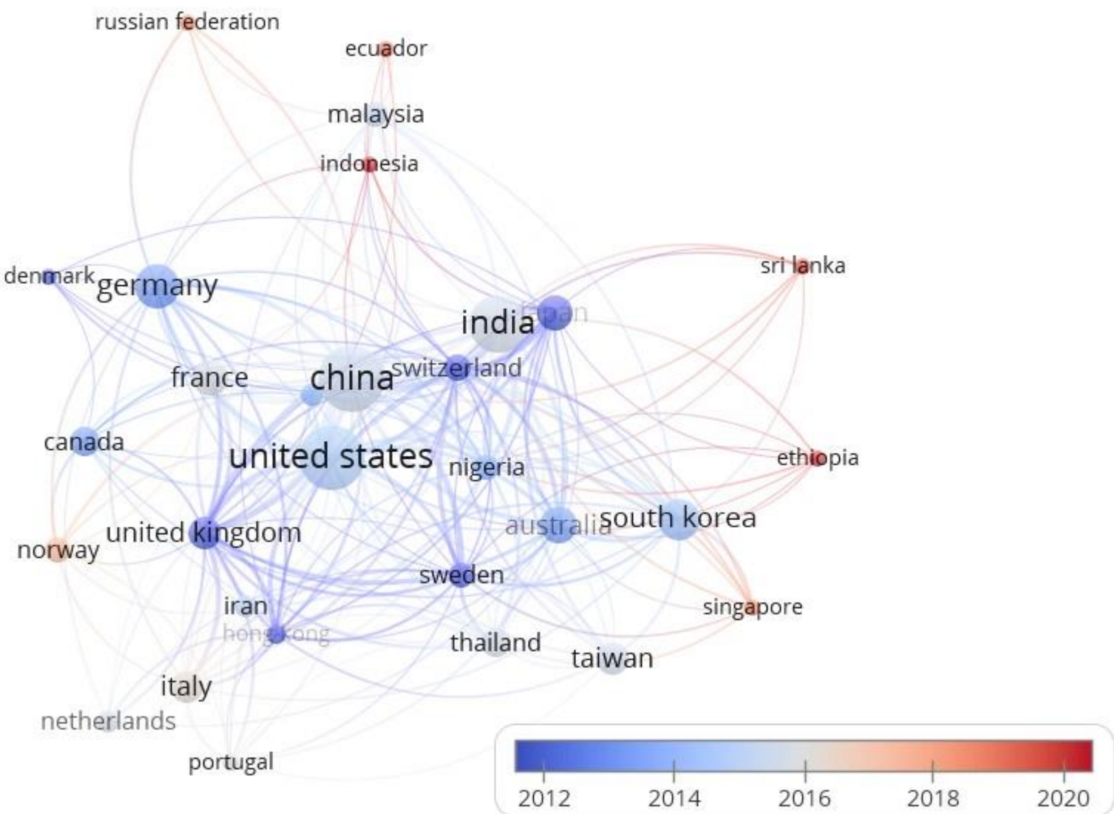



*Fig. 6 Map for authors' countries of affiliation; circles size: number of documents, color: average publication year, sample: EPR and WEEE. Source: own elaboration.*

Lastly, the number of documents on EPR and WEEE as well as their average publication year is analysed by authors' countries of affiliation. The high number of documents published by the US, China and India is not surprising: the map does not weight countries' contribution by population size. The role of European countries is relevant in this literature, which is coherent with the leading role played by the EU in the early introduction of EPR regulations. Instead, more recent publications are mainly provided by developing countries, which are in the process or have recently introduces EPR systems.

### 3.4 Publications

Bibliometric analysis at the level of documents is useful to identify the most relevant publications in a research field. Tab. 8 and Tab. 9 report the most cited documents in the EPR, and EPR and WEEE samples respectively. Moreover, VOSviewer allows analyzing the references cited by the articles in a dataset. In particular, it is possible to count the times a document of the dataset includes or is included in the references of the other documents of the dataset. This measure provides an indication about the embeddedness of each contribution within their respective field (Caputo et al. 2021).

| document | citations | document | links |
|---|---|---|---|
| widmer r. (2005) | 979 | cai y.-j. (2021) | 65 |
| kiddee p. (2013) | 384 | widmer r. (2005) | 61 |
| nnorom i.c. (2008) | 344 | wang h. (2017) | 39 |
| sthiannopkao s. (2013) | 287 | atasu a. (2012) | 38 |
| king a.m. (2006) | 257 | lifset r. (2013) | 38 |
| atasu a. (2009) | 234 | nnorom i.c. (2008) | 37 |
| lee j.-c. (2007) | 228 | manomaivibool p. (2009) | 37 |
| zhang k. (2012) | 205 | tong x. (2013) | 35 |
| yang j. (2008) | 205 | khetriwal d.s. (2009) | 32 |
| khetriwal d.s. (2009) | 202 | atasu a. (2009) | 31 |
| gerrard j. (2007) | 189 | kiddee p. (2013) | 30 |
| lu w. (2011) | 181 | sachs n. (2006) | 30 |
| osibanjo o. (2007) | 168 | kojima m. (2009) | 30 |
| ritchey t. (2006) | 163 | manomaivibool p. (2011) | 29 |
| liu x. (2006) | 158 | massarutto a. (2014) | 29 |
| krikke h.r. (1998b) | 158 | spicer a.j. (2004) | 27 |
| gupta s. (2011) | 152 | mckerlie k. (2006) | 27 |
| atasu a. (2012) | 144 | gu y. (2017) | 27 |
| plambeck e. (2009) | 137 | milanez b. (2009) | 26 |
| spicer a.j. (2004) | 137 | niza s. (2014); gupt y. (2015); campbell-johnston k. (2021) | 25 |

*Tab. 8 Top documents based on total citations and co-citations within selected documents sample; sample: EPR. Source: own elaboration.*

| document | citations | document | links |
|---|---|---|---|
| widmer r. (2005) | 979 | widmer r. (2005) | 54 |
| kiddee p. (2013) | 384 | cai y.-j. (2021) | 35 |
| nnorom i.c. (2008) | 344 | nnorom i.c. (2008) | 30 |
| sthiannopkao s. (2013) | 287 | manomaivibool p. (2009) | 27 |
| atasu a. (2009) | 234 | kiddee p. (2013) | 26 |
| lee j.-c. (2007) | 228 | kojima m. (2009) | 25 |
| zhang k. (2012) | 205 | khetriwal d.s. (2009) | 24 |
| yang j. (2008) | 205 | wang h. (2017) | 24 |
| khetriwal d.s. (2009) | 202 | tong x. (2013) | 23 |
| osibanjo o. (2007) | 168 | manomaivibool p. (2011) | 20 |
| liu x. (2006) | 158 | liu x. (2006) | 19 |
| atasu a. (2012) | 144 | osibanjo o. (2007) | 18 |
| plambeck e. (2009) | 137 | atasu a. (2012) | 18 |
| afroz r. (2013) | 133 | yu j. (2010) | 18 |
| yu j. (2010) | 132 | mayers c.k. (2007) | 16 |
| gottberg a. (2006) | 124 | gu y. (2017) | 16 |
| wath s.b. (2010) | 119 | yang j. (2008) | 15 |
| manomaivibool p. (2009) | 115 | gottberg a. (2006) | 15 |
| li j. (2013) | 102 | lodhia s. (2017) | 15 |
| salhofer s. (2016) | 93 | atasu a. (2009); manomaivibool p. (2014); premalatha m. (2014) | 13 |

*Tab. 9 Top documents based on total citations and co-citations within selected documents sample; sample: EPR and WEEE. Source: own elaboration.*

### 3.5 Keywords

Eventually, keywords analysis is useful for identifying the most relevant thematic areas and topics within a research field (Fakhar Manesh et al., 2021). Tab. 10 reports the author keywords[3] occurring more frequently in the documents of the EPR sample. First of all, it can be noted that WEEE is the second absolute keyword in terms of occurrences: again, this shows that the implementation of EPR on this waste flow has been, and was considered to be, extremely relevant. The keyword "packaging waste" is much less frequent, for example.

| Keyword (rank 1-9) | occurrences | Keyword (rank 10-17) | occurrences |
|---|---|---|---|
| epr | 294 | reverse logistics | 18 |

[3] Keywords were manually harmonized in order to avoid the disaggregation of occurrences due to misspelling, the presence of acronyms and different abbreviations. See thesaurus file.

| | | | |
|---|---|---|---|
| weee | 140 | environmental policy; weee management | 15 |
| recycling | 88 | informal sectors | 14 |
| circular economy | 39 | waste | 14 |
| waste management | 36 | remanufacturing | 12 |
| industrial ecology | 27 | lca | 11 |
| sustainability | 24 | material flow analysis; sustainable development | 10 |
| china | 23 | closed-loop supply chain; developing countries; eee; environmental regulation; legislation; municipal solid waste; packaging waste; pro; product take-back | 9 |
| product stewardship | 18 | | |

*Tab. 10 Top keywords based on total number of occurrences, sample: EPR. Source: own elaboration.*

Considering the average years of publication, it is clear from Fig. 6 that the concept of CE is gaining prominence in recent years, along with related topics, like reverse logistics and remanufacturing, and methodologies, e. g. life-cycle assessment (lca) and material flow analysis. This figure may also provide some clue on the fact that discourses on EPR may be turning from being centered on the end-of-life phase of the PLC towards a more holistic approach to sustainability (“sustainability”, “circular economy”, “industrial ecology”).

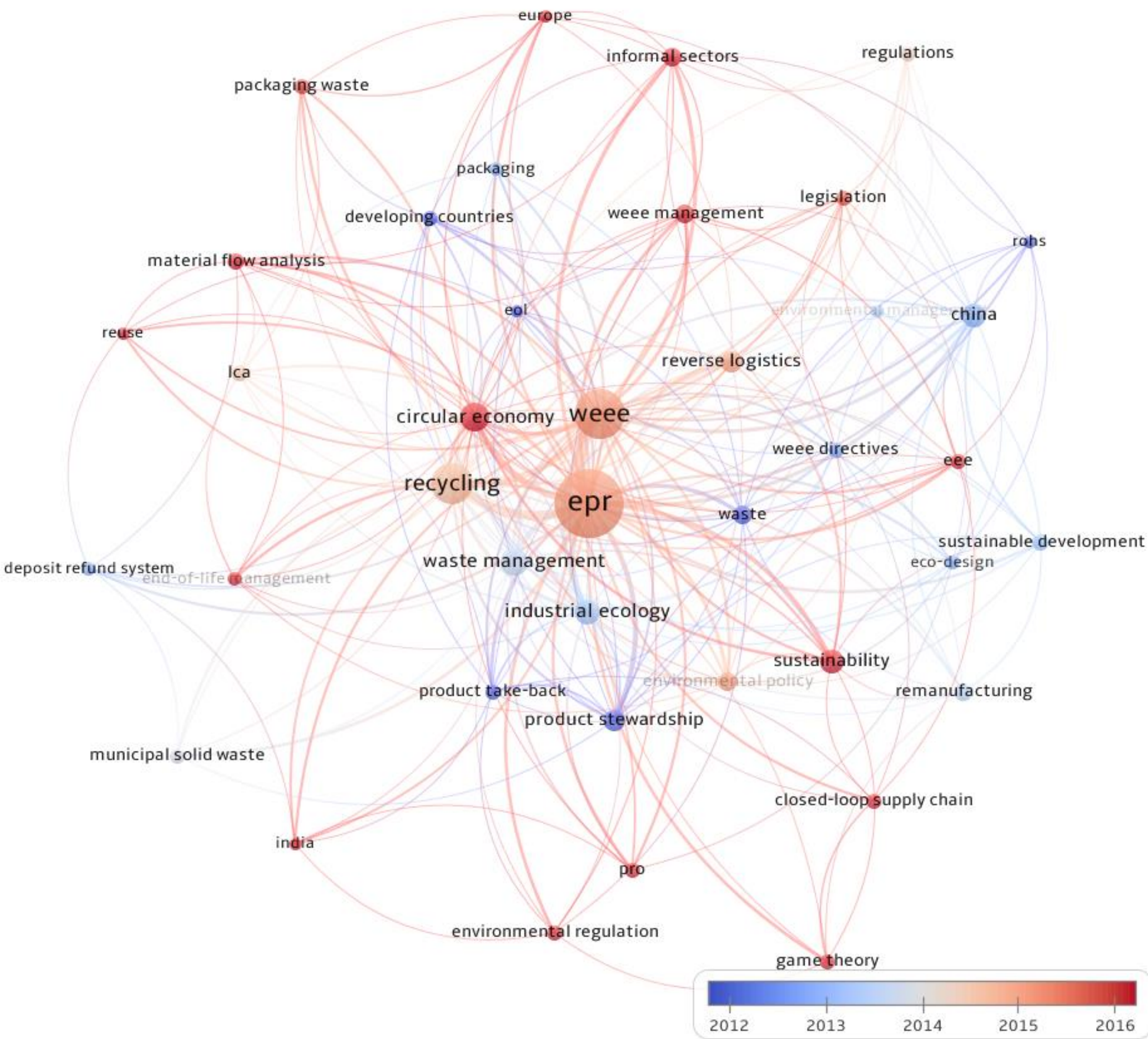



*Fig. 6 Map of top 40 keywords, frame size: occurrences, color: average publication year, sample: EPR.*

This trend can also be observed for the publications on WEEE (Fig. 7). Moreover, the analysis of the most frequent keywords in this subsample highlights some specific scopes of investigation and issues. The keyword “China” is relatively more present than in the general EPR dataset, due to the key role of China in EEE productions and WEEE flows management (Wang et al., 2017; Yu et al., 2006, 2008; Q. Zhang, 2021). A relatively present keyword is “informal sectors”, which alludes to the relevance of management, recycling and trading informal activities around WEEE, especially in developing countries. Developing countries (6 occurrences) are sadly known to be dumping sites for WEEE originating in developed countries (Nnorom and Osibanjo, 2008), but they are also the main suppliers of raw materials essential for electronics productions (EU Comm, 2020c). Eventually, the cluster of keywords connected to legislative aspects – environmental policy, WEEE directives, regulations, eco-design, RoHS – depicts the connections of EPR norms with other environmental regulations and principles, and prove the relevance of EU initiatives (Svensson and Dalhammar, 2018).

| keyword | occurrences |
|---|---|
| weee | 140 |
| epr | 135 |
| recycling | 37 |
| industrial ecology | 18 |
| waste management | 15 |
| weee management | 15 |
| china | 14 |
| circular economy | 14 |
| informal sectors | 12 |
| environmental policy | 9 |
| sustainability | 8 |
| eee; material flow analysis; product stewardship; product take-back; weee directives | 7 |
| developing countries; eol (end-of-life); pro; regulations; reverse logistics; weee recycling | 6 |

*Tab. 11 Top keywords based on number of occurrences, sample: EPR and WEEE. Source: own elaboration.*

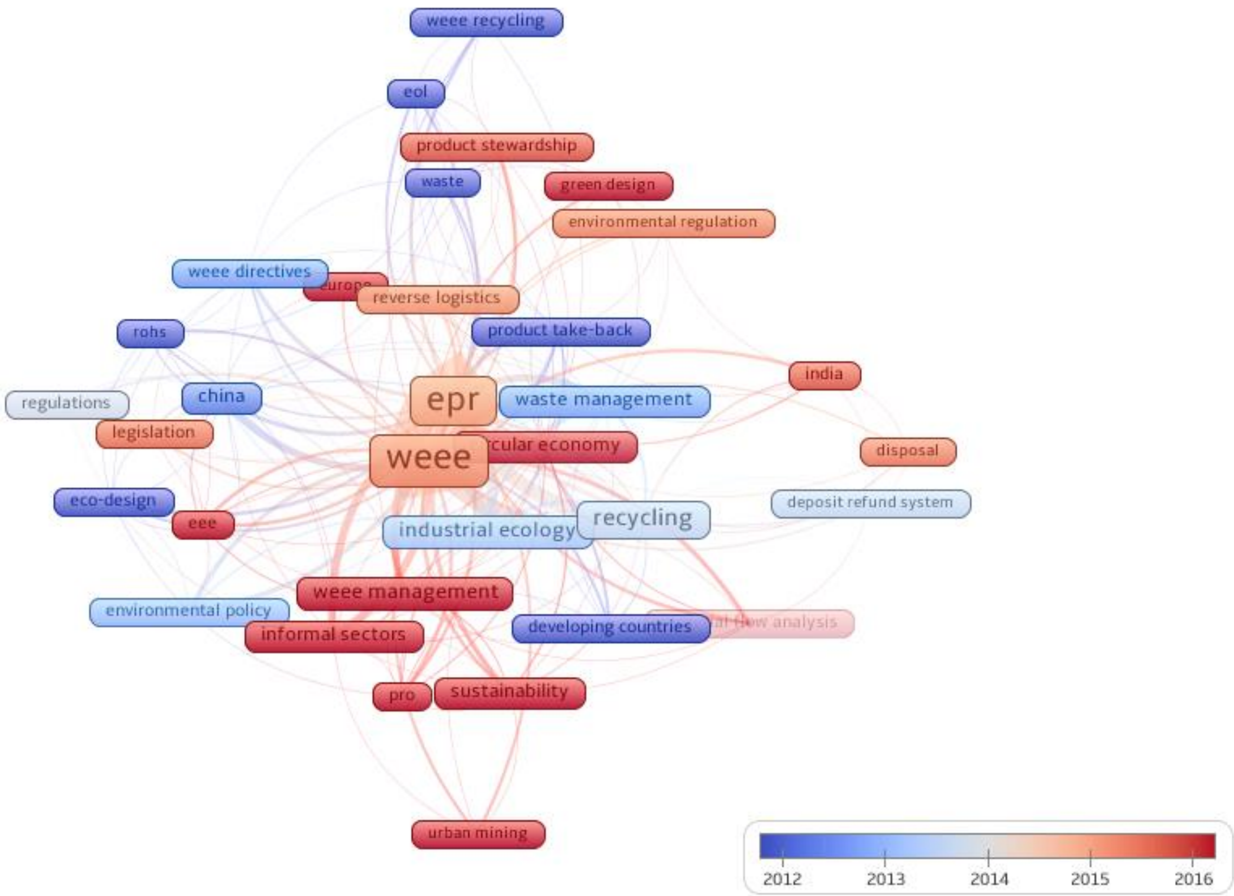



*Fig. 7 Keywords map; frame size: occurrences, color: average publication year, sample: EPR and WEEE. Source: own elaboration*

## 4. Content analysis

The content analysis aims to synthesize and reframe information from the selected literature, in order to sequentially answer the research questions listed in Fig. 2. Research question four and policy recommendations are addressed in Section 6.

### 4.1 General expectations about EPR implementation

A clear understanding of EPR goals is fundamental in order to assess its impacts. Importantly, these expectations can be generalized to all the waste flows/value chains in which EPR has been introduced. The potential benefits of this policy can concern various stages of the PLC (Lifset 1993). Following Lindhqvist and Lifset (1998), EPR goals can be roughly divided in two groups, depending on the stage of the PLC on which they are expected to impact.

"Downstream" EPR goals concern the phases after the discard of the product, that is waste management. Under this perspective, EPR main aims are to divert waste from final disposal (Atasu et al., 2009) and to avoid improper treatments, reducing its direct environmental impact (Kalimo et al., 2014).

The development of formal collective or individual waste management schemes by producers, required by the policy, was expected to lead to the following cascading downstream results:

a) An increase in the collection rate of wastes (Lindhqvist, 2000).
This means improving the separation of waste flows at their origin, with better results from a recovery and recycling point of view (Kiddee et al., 2013), a reduction of externalities generated by improper waste disposal and the drain of dumping flows to developing countries (Nnorom and Osibanjo, 2008; Premalatha et al., 2014).
b) Growing and more stable waste flows and recovery rates would have led to the growth of the secondary raw materials (SRM) market (Milios, 2018), increasing the value of "urban mines" (Panchal et al., 2021).
c) The stable growth of both collection rates and the SRM market would have fostered the recycling sector (Gu et al., 2017) and innovation performance (Atasu, 2019; Massarutto, 2014).

"Upstream" EPR goals concern the stages of the PLC prior to its discard, which are its design and production phases (Lindhqvist and Lifset, 1998). Under this perspective, the EPR focus is on environmental innovation (Lifset, 1993; Mayers, 2007) and waste prevention (McKerlie et al. 2006). By making producers economically and physically responsible for their own waste, the will of the policy makers has been to incentivize producers towards various forms of eco-design and cleaner production:

a) Design for recycling or recovery strategies to decrease the costs and the efficiency of recovery processes (Atasu and Subramanian, 2012).
b) Reduction or elimination of hazardous and pollutant substances (Scheijgrond, 2011) to avoid responsibilities and product bans (Yu et al., 2006).
c) Individual, voluntary product take-back schemes and reverse logistics schemes (Rizzi et al., 2013; Tsai and Hung, 2009) to individually manage their own firm used products or waste.
d) Design for disassembly, remanufacturing (Tojo 2004) or reuse (Dwivedy and Mittal, 2012) strategies to extend PLCs and delay waste generation.
e) Dematerialization (Walls, 2006), that is, the design of products with as few materials and components as possible (Babbitt et al., 2021), to reduce the mass of waste generated.

### 4.2 Issues linked to WEEE

The previous bibliographic analysis showed that WEEE is the waste flow receiving the highest consideration within EPR literature. This is due to the great number of issues linked to WEEE and to the difficulty of evaluating the effects of regulations in this field, caused by the complexity of EEE value chains and WEEE management systems.

The annual global production of WEEE increased by 21% in the period 2014-19, and is expected to almost double by 2030 (Forti et al., 2020). Asia generated the highest quantity of WEEE in 2019 with 24.9 Mt, equal to 5.6 kg per capita, followed by the Americas (13.1 Mt, 13.3 kg per capita), Europe with 12 Mt (16.2 kg per capita), and Africa (2.9 Mt, 2.5 kg per capita).

According to the United Nations statistics (Baldé et al. 2017), only 42.5% of total WEEE generated in Europe is formally collected and recycled, while this figure decreases to a mere 17.4% worldwide. On a global scale, the officially collected WEEE increased by 1.8 Mt in the period 2014-2019 (+24%): much less than total WEEE generated (+9.2 Mt).

When not separately collected, WEEE ends up in improper waste treatments like incineration and landfilling, if not dispersed in the environment. Therefore, these low collection rates have two kinds of consequences. First, in terms of direct environmental impacts. Secondly, in terms of circularity of the use of natural resources contained in WEEE and necessary for EEE production, that is a form of indirect environmental impact.

WEEE is a complex mixture of materials and components that can cause environmental and health problems (Parvez et al., 2021), indeed it was defined as hazardous by the “Basel Convention on the Transboundary Movements of Hazardous Wastes and Their Disposal” (Widmer et al., 2005). These negative consequences can also occur when WEEE is landfilled or incinerated (Kiddee et al., 2013), Hence separate collection is necessary to prevent them. Coherently, one of the first focuses of research on EPR and WEEE was the chemical characterization of WEEE contents (Pérez-Belis et al., 2015).

It is well documented that substantial flows of WEEE are more or less legally shipped from developed to developing countries (Sthiannopkao and Wong, 2013), in order to avoid the relatively high costs of WEEE disposal and the strict environmental and work regulations of developed countries (Zhang et al., 2012). To limit these flows, the Basel Convention was enacted to ban the transboundary trade of WEEE and other hazardous wastes (Widmer et al. 2005). Despite this international treaty, entering into force in 1992, and the enactment of national bans on WEEE trade by various countries, e.g., China in 2018 (Zhang, 2021), waste flows did not stop (Premalatha et al., 2014). These bans are overcome by declaring WEEE as second-hand goods (Nnorom and Osibanjo, 2008). It is assumed that the volume of transboundary movements of used EEE and WEEE ranges from 7 to 20% of total WEEE generated (Forti et al., 2020).

This phenomenon is particularly worrisome, because in developing countries WEEE is typically managed by the informal sector (Gui, 2020), where WEEE is not treated under sound environmental and safety conditions.

A second problem connected to the limited collection of WEEE and the use of recovery processes not at the best-available-technology level is the limited opportunity to improve the downstream circularity of the EEE value chain. WEEE is considered an "urban mine" (Panchal et al. 2021), representing an important source of SRM, particularly for precious metals and critical raw materials (CRM). CRM are non-energetic raw materials, considered as highly strategic according to two parameters: economic importance and supply risk (Schrijvers et al., 2020). Indeed, these materials are essential for high-tech productions and the ecologic transition (EU Comm, 2020c). Nonetheless, the supply of CRM is exposed to high risks because of the geographical concentration of production and suppliers' unstable geopolitical situations (Althaf et al., 2021). Furthermore, CRM extraction and refining processes are typically carried out in developing countries, where they cause serious negative environmental impacts (Massari and Ruberti, 2013) and are carried out under dangerous job conditions (Ilyassova et al., 2021).

## 4.3 On the effectiveness of EPR in the WEEE scope

Following the structure of Section 5.1 on the general expectations on EPR and focusing on the EEE value chain, the next two subsections summarize the achievements and failures of the policy, while highlighting some research gaps.

### 4.3.1 Evidence on downstream goals

The introduction of EPR is typically associated with a relevant increase in waste collection rates (Walls, 2006; Massarutto, 2014). This also holds for the WEEE case (Cahill et al. 2011), but with some important limitations.

WEEE collection rates have increased significantly in Europe since EPR implementation in the last twenty years (Gupt and Sahay, 2015; Kunz et al. 2018). According to Eurostat data total WEEE collected increased by roughly 30% in the period 2011-2018 in the EU 28. However, despite some improvements, the situation remains dramatic in developing countries (Premalatha et al., 2014), particularly in Africa (Bimir, 2020) and Asia (Herat, 2021). For example, Bhaskar and Turaga (2018) estimate that only 5 to 15% of total WEEE generated in India is channeled through formal processing facilities. The causes of this ineffective implementation of EPR policies in developing countries are

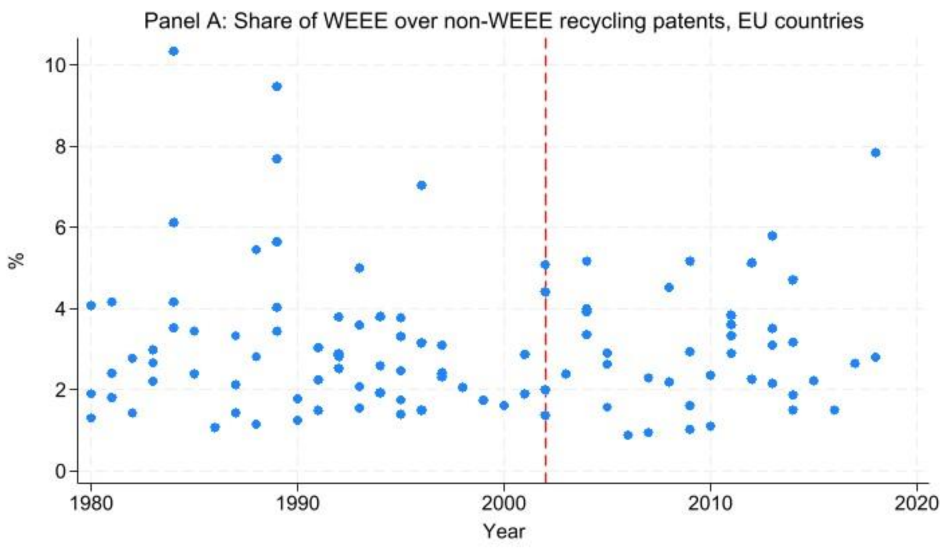
the illicit trade flows (Palmeira et al., 2018), lack of data inventories (Arya and Kumar, 2020) and the limited capability of final consumers to distinguish between formal and informal WEEE collection channels (Cao et al., 2016). From the point of view of the recovery system, the limited availability of treatment technological options and the cannibalization of WEEE flows by the informal sector, which is more economically efficient and widespread than the formal one (Salhofer et al., 2016), remain the causes of relevant adverse environmental and health effects (Ravindra and Mor, 2019).
Moreover, collection rates remain still too low even in the EU. The 2012 WEEE directive (2012/19/EU) set the minimum yearly WEEE collection rate at 45% of the average weight of EEE placed on the market in the three preceding years in each EU country. The same directive increased the target to 65% from 2019. According to Eurostat data, the average EU collection rate was just above the 2016 target (47%) in 2017, i.e., 8.4 kg/inhabitant. Collection rates remain particularly low for some specific WEEE categories, especially small electronic devices (Forti et al., 2020). For instance, this holds for smartphones, even in mature EPR systems such as the EU (Kunz et al., 2018) and Korea (Jang and Kim, 2010).

Considering the treatments which are undergone by the fraction of formally collected WEEE, in Europe 84% of this flow was recycled or reused in 2017 (Eurostat). The 2012 WEEE Directive also sets recovery and recycling targets for each WEEE category. This relatively high efficiency in collected WEEE recovery increased the availability of SRM and the opportunity for materials recirculation. Nonetheless, despite the strategic relevance addressed to WEEE recycling to reduce supply risks (EU Comm 2020c), the share of CRM demand satisfied by SRM in Europe is still extremely low in the majority of the cases (EU Comm, 2020b). Recycled CRM hardly compete with virgin ones for three reasons: the limited availability of cost-efficient recycling technologies (ERECON, 2014); recycling-unfriendly product designs (Richter and Koppejan, 2016; Althaf et al., 2021); low collection rates of specific WEEE categories hampering economies of scale (Jang and Kim, 2010).

With regard to the last sphere of downstream expectations on EPR implementation, limited research tried to assess the connection between this policy and the adoption of innovations in the recycling sector (Massarutto, 2014), particularly in the WEEE scope. Conceptually, this possible effect of EPR could be seen as a form is “Indirect” Porter Hypotheses. In its traditional formulation (Porter & Van Der Linde, 1995), the Porter Hypothesis predicts a possible positive role of environmental regulation in supporting environmental innovation, eventually possibly leading to positive economic outcomes for innovative firms (Ambec et al., 2013). In the case of recyclers, innovation and economic outcomes would be only an indirect consequence of the responsibilities addressed to producers. Franco and

Marin (2017), for instance, showed how environmental regulations can have effects on innovation dynamics occurring at different stages of a value chain with respect to the one which is directly addressed by the regulation. As said, the growth of collection rates and of the SRM market was expected to enhance the economic and innovation performance of the recycling sector. Specific policy actions have, in many cases throughout history, catalyzed the emergence of new innovation paths for sectors, regions, and countries, ultimately giving rise to new industries (Bellandi et al., 2018; Forrer et al., 2022). Among the few studies conducted with a country-sector perspective, Nicolli and Mazzanti (2011) used panel data on patents (1970-2007) to show that the peak of technological innovation in the recycling sector can be placed in the decade preceding the advent of EPR. Apparently, EPR did not prompt a new wave of innovations. Anyway, this study does not focus on the WEEE recycling sector. It is more well documented that innovations in the recycling sector occurred in the end-of-life vehicles scope, both in China (Chen and Zhang, 2009) and Europe (Gerrard and Kandlikar, 2007).

The economic efficiency of recycling sectors in developed countries improved with the advent of EPR, but this seemed to be driven more by the establishment of well-organized cooperation between producers and recyclers, rather than innovation in recycling (Massarutto, 2014). Instead, in developing countries, formal recyclers still suffer from the informal sector's competition (Gui, 2020).



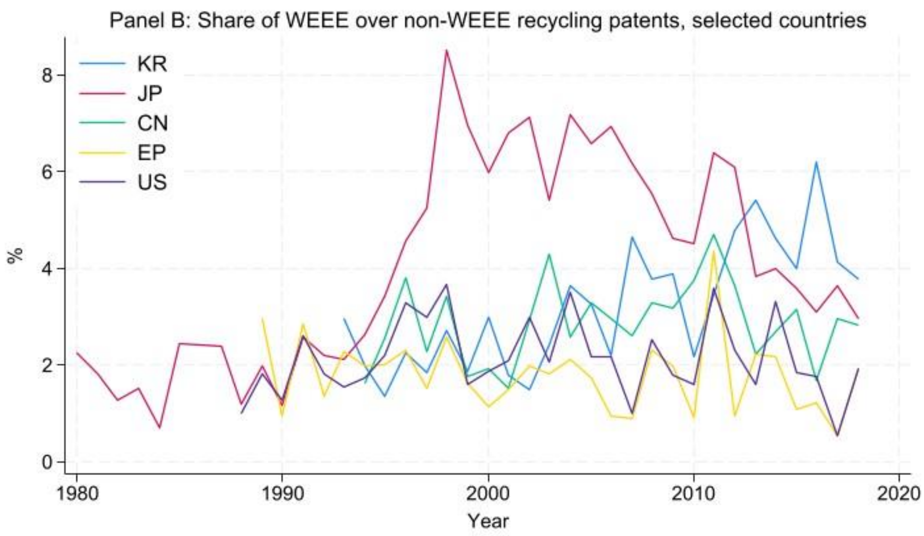



*Fig. 8 Panel A: share of WEEE recycling patents over other recycling patents, EU countries, 1980-2018. Panel B: share of WEEE recycling patents over other recycling patents, top countries worldwide, 1980-2018. Legend: KR South Korea, JP Japan, CN China, EP European Patent Office, US United States. Own elaboration.*
*Methodologic note: the share is calculated as the ratio between WEEE and non-WEEE recycling patents filed each year, after the drop of the observations falling in the lowest ten percent of both WEEE and non-WEEE patent count distributions; this is in order to avoid overweighting small numbers, generating abnormous shares.*

In Fig. 8, patent data taken from PATSTAT are used to investigate descriptively trends in WEEE recycling innovation. The elaboration is based on the OECD ENV-TECH classification (Hascic and Migotto, 2015), which is one of the most commonly used methodologies to identify green patents on the basis of their IPC and CPC codes (Favot et al., 2023). The share in Fig. 8 represents the ratio between the count of patents filed in each country in 1980-2018 with at least one IPC or CPC code related to WEEE recycling over the total number of patents in the recycling field with the exclusion of WEEE recycling. The list of IPC and CPC codes is reported in Table A2 in Appendix A. In the European countries (Panel A), the share of WEEE over non WEEE patents has declined in the 1990s, while received a bust in the 2000s, starting in coincidence with the WEEE Directive. On a global level (Panel B), South Korea and Japan, two countries characterized by mature EPR systems, have the highest specialization in WEEE recycling.

Hence, with regard to a possible role of EPR on innovation in WEEE recycling, it is worth noting some facts. WEEE collection rates have increased in time, starting from small flows when EPR systems were not developed yet, providing an incentive and the knowledge bases necessary to trigger innovation paths only in a longer time frame. Moreover, more recent policy interventions may have had further impacts (e.g. for Europe: Eco-design Directive 2009, Waste Framework Directive 2008, "Circular Economy Package" directives 2018). Last but not least, EPR forced producers-recyclers interaction, possibly stimulating collaborative downstream innovation (Micheaux and Aggeri, 2021).

### 4.3.2 Evidence on upstream goals

On the upstream perspective, research seems to generally agree in finding little effectiveness of EPR policies in incentivizing producers' systematic eco-design strategies (Wang et al., 2017). Companies' responses are mainly driven by their market structure and client requirements, while raw materials cost and supply chain management implications appear to be key challenges for the achievement of EPR eco-design expectations (Yu et al., 2006, 2008). Overall, upstream green innovation seems to be driven more by independent marketing strategies than by EPR (Massarutto, 2014). According to Manomaivibool and Hong (2014), focusing on the Korean case, producers perceive EPR as a mandatory corporate social responsibility rather than an integral part of the competitive core business. The EU and Chinese RoHS directive seem to have been more successful in reducing the presence of hazardous substances in EEE (Tong and Yan 2013).

A few studies contradict this general view on EPR upstream ineffectiveness, providing some scattered evidence in favour of the realization of Porter Hypothesis. Tojo (2004) supports the evidence of EPR as a driver for manufacturers' strategies aimed at the enhancement of PLC environmental impacts. The author investigates multiple countries and sectors case studies. A rare quantitative analysis is provided by Zhao and colleagues (2021), who find evidence of a positive relationship between EPR and green innovations, based on a difference-in-differences model for Chinese manufacturing companies in the period 2010-2018.

Anyway, this literature review highlights the limited availability of systematic, quantitative analysis testing the causality between EPR and eco-design strategies in the EEE sector, as also underlined by Cai and Choi (2021). Evaluations usually rely on case studies, either for single companies (e.g., Mayers, 2007) or specific sectors (e.g., Gottberg et al., 2006), and on qualitative approaches based on interviews (Yu et al. 2006, 2008; Micheaux and Aggeri, 2021; Tojo, 2004). Upstream innovations are difficult to detect and measure (Yu et al., 2008), because of limited data availability (Babbitt et al., 2020) and of supply chains' complexity. As shown by Scheijgrond (2011), the supply chains of big EEE manufacturers are very articulated and comprehend a huge number of suppliers, so that it is difficult for these companies to control and assess sustainability requirements that may be demanded upstream.

Redesign occurred more frequently in other industries/waste flows, e.g., packaging (Walls, 2006) and vehicles (Chen and Zhang, 2009).

One of the main reasons for this failure of EPR policies on WEEE to achieve their expectations in terms of eco-design is the "collective EPR" mechanism – CPR - (Atasu, 2019). Through *producer responsibility organizations* (PRO), manufacturers typically jointly meet their EPR obligations using collective collection, sorting, and processing systems. In this way, PRO are made responsible for the waste management operations (Huisman, 2013). Under CPR, to finance PRO's waste management costs, producers pay fees calculated according to each producer's share of products placed on the market: this mechanism does not take into account product design and eco-friendliness (Mayers et al., 2013). This provides a disincentive for producers to invest in eco-design, since the consequent benefits in terms of reduction of waste management costs would be shared by the other PRO partners, with a free-riding mechanism (Atasu, 2019; Micheaux and Aggeri, 2021). Moreover, these fees turn into a small increase in the price of EEE products (Favot et al., 2018), due to the cost efficiency of PROs reached during the years, with little effect on consumers' demand.

Actually, the EPR mechanism established by the 2002 EU WEEE directive provides for an *individual producer responsibility* (IPR) to finance the end-of-life costs of its own products (Lindhqvist and Lifset, 2003). IPR models offer superior design-for-recycling incentives (Dempsey et al., 2010), but imply the organization of firm-specific take-back schemes and are therefore expensive and present practical issues (Rotter et al., 2011), compared to the cost efficiency of CPR settings (Atasu and Subramanian, 2012). Under an IPR setting, the reduction of waste management costs becomes a ground of competition among producers.

A practical solution advocated to realize the IPR incentives is represented by eco-modulated fees, i.e., fees differentiated depending on the level of eco-design of products, against mass-based collective costs sharing (Atasu, 2019; Mayers et al., 2013). Mechanisms for differentiating fees exist in various countries in the packaging sector (Eunomia, 2020). But modulated fees are infrequent in the EEE sector: in Europe, only France has introduced them, since 2010 (EU Comm, 2017). However, according to Micheaux and Aggeri (2021), eco-modulated fees failed to incentivize producers to eco-design in France. This is due to the small magnitude of the fees and to the fact that France's share of global market demand for EEE is too small to force the main producers to modify the design of their products.

In conclusion, while IPR is advocated as an inescapable mechanism to make producers interiorize their own products life-cycle environmental impacts, its effectiveness may be hampered by the implementation settings.

In alternative to the IPR principle, some authors suggested to reevaluate the role of PRO, as these, not only secure the supply of recyclers (Gui, 2020), but are also the place of interaction between producers, and producers and recyclers. This may facilitate cooperation among these actors, leading to upstream or downstream innovations, either institutional or technological (Micheaux and Aggeri, 2021). Indeed, Corsini and colleagues (2018) show that some European PRO behave in a proactive way seeking for innovative, market-oriented pathways.

## 5. Alignment of EPR implementation with CE

Discrepancies between CE goals and EPR achievements and implementation along the EEE life-cycle management can be classified in seven, interrelated areas, each of them representing a sphere of strategic and operational policy improvement.

### 5.1 Waste prevention and planned obsolescence

The practical implementation of EPR typically faced criticism because of the prevailing of downstream over upstream goals (Campbell-Johnston et al., 2021). The ability to define new products for end users is seen as the EEE industry's key competence effect (Lauridsen and Jorgensen, 2010), leading to accelerating planned obsolescence (Tong and Yan, 2013). Consequently, improvements in the functional capacity of devices, that is product-level material efficiency (Kasulaitis et al., 2015), are outweighed by the globally increasing demand for electronics and by their penetration in other products (Lauridsen and Jorgensen, 2010).

This is not in line with the CE objective of decoupling economies from resources consumption (Mayer et al., 2019). Therefore, EPR regulations should more effectively focus on resources conservation and waste prevention strategies, disincentivizing planned obsolescence.

### 5.2 Waste collection and resources recovery

Under CPR settings, the stakeholders involved in EPR systems prioritized waste management aspects, especially the enhancement of collection rates and of the economic efficiency of waste collection and treatment. Still, recovery rates should further increase to reduce the damages due to improper WEEE management, particularly in developing countries, and to close downstream loops of resources cycles, especially with regard to CRM (Gaustad et al., 2018). Actions to boost collection and recovery rates should mainly focus on four areas: the consumers' awareness on WEEE environmental impacts, disposal systems (Phulwani et al., 2021) and resources contents (Shigetomi et al., 2015); the capillarity of WEEE disposal systems (Favot and Grassetti, 2017); the reduction of illegal WEEE flows and scavenging (Horta Arduin et al., 2020), while not preventing the development of virtuous waste recycling trade networks (Latorre et al., 2021).

Favot and Massarutto (2019) point out that the market competitiveness of secondary CRM largely depends on their recovery costs vis-à-vis virgin resources prices, while WEEE collection and sorting costs are less relevant. Consequently, for mature EPR systems, the promotion of recovery technologies seems to be a more determinant policy to support a market for SRM, rather than a further stringency of EPR collection and recovery targets.

### 5.3 Place-based policy implementation

The EPR principle can be implemented through a variety of policy instruments, that may be of the economic, administrative or informative type (Nnorom and Osibanjo, 2008). The choice of the policy instrument and of the allocation of responsibilities implies different incentives and behaviors of the stakeholders involved in the affected value chain. A detailed comparison of 27 national EPR systems is provided by Gupt and Sahay (2015), who analyzed the waste scope of EPR regulations and the type and allocation of responsibilities to stakeholders. Similarly is done by Cahill et al. (2011) and Andersen (2021) for EU countries and by Wang et al. (2017) for China, Germany, Japan and Switzerland. In brief, cross countries comparisons suggest a higher relevance of operational characteristics for downstream EPR effectiveness, rather than for upstream goals. With regard to the downstream scope, results have been more positive where physical responsibilities lie with local authorities and these have been engaged in the design and implementation of EEE industry-financed EPR systems (Cahill et al., 2011) characterized by competition among PRO (Corsini et al., 2017). For developing countries, the dominant role of informal recycling sectors remains the main obstacle to achieve any downstream goal (Gupt and Sahay, 2015). Regardless of national EPR operating mechanisms, eco-design has not become a focus of attention for EEE producers (Wang et al., 2017). In conclusion, EPR systems should consider local socioeconomic conditions and may set different allocations of responsibilities depending on their level of maturity.

### 5.4 Integration of waste and production policies

According to Tukker et al. (2016), at the EU level, the integration of waste and production policies remains the major challenge to the circularity of the EEE value chain. The recycling targets of EU directives are not focused on specific waste management or environmental requirements for different materials and components within WEEE. This does not give producers clear signals for improved product design (Mayers, 2007). Meanwhile, policy instruments aimed at the design phase are mostly concerned with energy use (Tukker et al., 2016) and not with Design for X (Sassanelli et al., 2020), e.g., design for recycling or reuse. Eco-design standards are needed to provide harmonized indications for the circular design of products to manufacturers (Tecchio et al., 2017). Some scholars underlined that EPR implementation through mass-based collection or recycling targets may even disincentivize design for the durability of products (Althaf et al., 2021; Huang et al., 2019). Therefore, mass-based targets should be flanked by specific environmental and operational standards (Mayers et al., 2005; Milios, 2018).

## 5.5 Information provision and knowledge management

Gerrard and Kandlikar (2007) referred to a third expectation on EPR outcomes, in addition to eco-design and waste management ones, that is an improved information provision. The relevance of this aspect remained relatively overshadowed in EPR literature, while information provision could impact the individual and cooperative behavior of various actors along the PLC.

Again, the definition of eco-design standards and labeling, related to resource efficiency requirements in particular, would simplify the communication of circularity characteristics of products to the consumers (Tecchio et al., 2017). Information provision to final users should drive them towards more environmentally friendly in-use and disposal behaviors of EEE, but a clearer delineation of responsibilities in this role among EPR stakeholders is needed (Corsini et al., 2017).

Information provision is an essential enabler of Design for X strategies, when these actively involve agents downstream in the PLC. This is particularly true for EEE reuse and repair activities, conducted either by consumers (Bovea et al., 2017) or professional third parties (Whalen et al., 2018).

Stakeholder engagement generates knowledge for value co-creation (Wiesmeth, 2020). This objective could be pursued through the orientation of PRO towards the diffusion of collaborative innovation (Corsini et al., 2018; Micheaux and Aggeri, 2021).

In conclusion, knowledge management is an essential enabler of CE (Acerbi et al., 2021, 2022), especially for the optimization of complex systems such as EPR in the EEE value chain (Tukker et al., 2016).

## 5.7 Individual and voluntary producer responsibility

Policy makers failed in allocating sufficient individual responsibility to producers to systematically drive them towards the CE goals. The key point for EPR to achieve its CE aspirations is to ensure a financial advantage in improving EEE design (Mayers, 2007). The aim here is to incentivize any of the Design for X abilities identified by Sassanelli et al. (2020), always prioritizing waste prevention strategies. Eco-modulated fees should serve this purpose (Favot et al., 2018), if more widely implemented across countries (Micheaux and Aggeri, 2021). Again, coordination within EPR systems is needed for eco-modulated fees setting.

If product take-back schemes and reverse supply chains would ensure a financial advantage to EEE producers, these would be induced to voluntarily take responsibility for their own end-of-life products (Wang et al. 2017). In turn, this would stimulate the adoption of CE practices and business models.

Recent research showed how digital technologies and servitized business models can support individual and voluntary producer responsibility. Indeed, digital technologies created niches for innovation in business models for post-consumer collection and recycling of WEEE (Tong et al., 2018) and they can help overcome information asymmetry among various agents in EPR systems (Gu et al., 2018). Nonetheless, research in this scope remains very limited, despite the connection between EPR and servitization has been considered since long (McKerlie et al., 2006).

In conclusion, despite the EPR potential to impact on all of the four CE transition building blocks outlined in Section 2, only a minor part of EPR expectations has been achieved. Fig. 9 summarizes the strategic and operational EPR improvements, discussed in this Section, necessary to enhance the circularity of the EEE value chain.

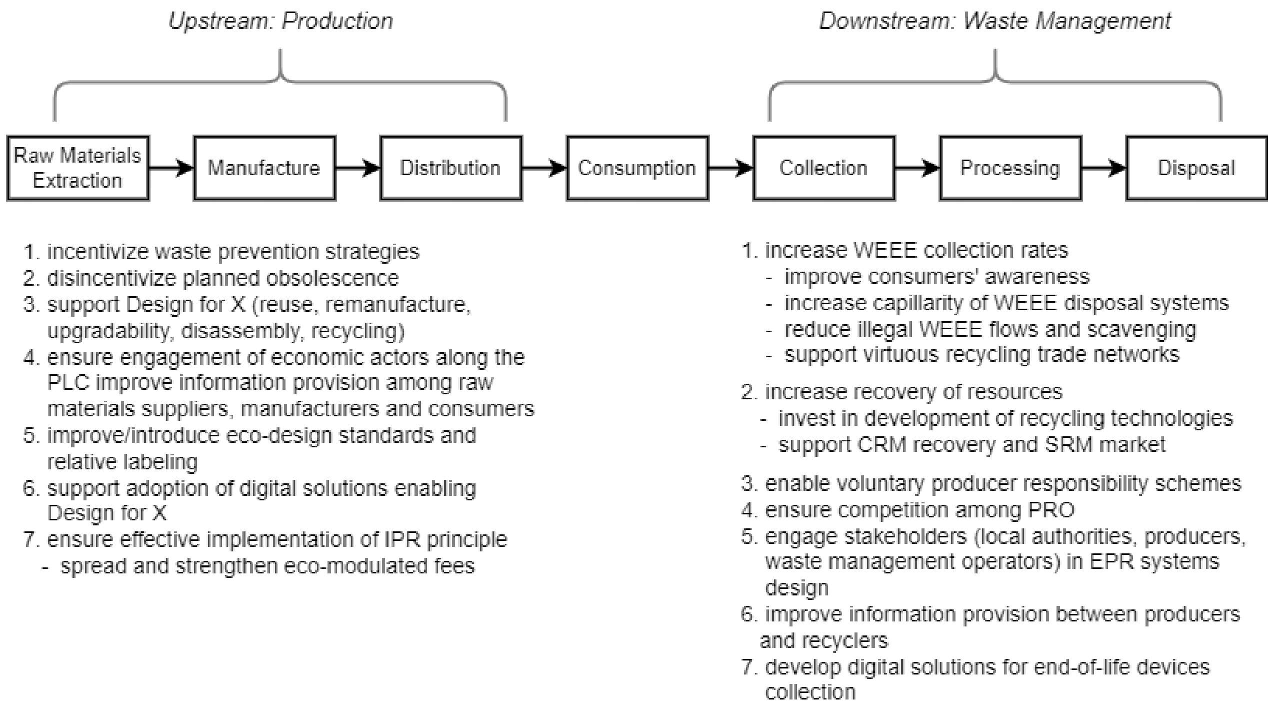



*Fig. 9 Areas of policy improvement to align EPR outcomes and CE objectives for the EEE value chain, classified by PLC stage of impact. Own elaboration.*

## 6. Conclusions and future research

Digitalization is often seen as a mutual strategy for environmental sustainability. Coherently, the concept of Twin, i.e. digital and green, Transitions gained momentum. Nevertheless, it is also acknowledged that the current digitalization trend relies on the exploitation of critical materials and it is responsible for the generation of massive amounts of electronic waste. Therefore, life-cycle strategies, such as the Extended Producer Responsibility, aiming to align digitalization outcomes both in terms of resources exploitations and pollution, seem to be the most appropriate.

This paper represents the first systematic literature review on EPR on WEEE aspiring to comprehensively assess the policy achievements from a life-cycle perspective. The analysis is used to evaluate the actual contribution of EPR in driving the EEE value chain towards circularity. From a practical point of view, the aim is to support policy makers and managers involved in different decision-making processes within the EEE value chain and EPR management systems, especially uncovering discrepancies between current settings of WEEE EPR systems and CE goals, while providing policy recommendations. As for the academic contribution of the article: it contributes to the debate on policies for CE, which typically lacks ad hoc policy assessment studies; it underlines how a life-cycle perspective must be adopted when evaluating policies in terms of CE outcomes; it stresses the double face of WEEE both as a problem and as a possible resource; below, it lists a wide range of future research directions for the field of EPR, at the connection with CE literature.

The EPR principle was conceived to incorporate total life-cycle costs in production and consumption decisions. Indeed, EPR seeks to spur the systematic application of strategies that can reduce the life-cycle environmental impact of products and decelerate the flow of resources. As explained, these objectives are particularly important for the case of electronic devices, in consideration of the number of economic and environmental concerns linked to WEEE. Indeed, the bibliometric analysis showed that WEEE is the waste flow that has attracted the greatest attention within EPR literature.

Moreover, the bibliometric analysis demonstrated that the number of scientific publications on EPR has been increasing over the last twenty years and that discourses are turning from being centered on end-of-life aspects towards a more holistic approach to sustainability and economic aspects. Nonetheless, still few studies focused on the comprehensive assessment of contributions of EPR to CE, i.e. resource efficiency.

The review has highlighted the achievement of important downstream results in mature WEEE EPR systems: substantial increases in WEEE collection rates and the development of stable, formal waste management systems. However, collection rates remain too low in developed EPR systems and extremely low in developing countries, the recovery of CRM is very limited and WEEE dumping flows are still consistent.

With regard to upstream results, EPR on WEEE is generally deemed ineffective, mainly due to the insufficient allocation of individual responsibility to EEE producers.

Adopting a systematic perspective, discrepancies between EC goals and EPR achievements and implementation have been classified into seven interrelated areas, discussed in Section 6, each of them representing a domain of policy recommendations.

Despite the mature implementation of EPR on WEEE, at least in the EU, numerous and relevant impacts of the policy remain obscure. Generally speaking, future research on EPR should focus on the (mis)alignment of the policy outcomes with CE objectives for the EEE value chain, supporting the implementation of the recommendations discussed in Section 6. More specifically, adopting once again the PLC lens, some key areas of investigation are listed below.

On the downstream side, research should better understand the consumers' attitudes towards EEE life extension and WEEE disposal. In addition, evaluations of the SRM markets development and innovation trends in WEEE recycling triggered by EPR introduction have been neglected: these impacts are determinant for the economic sustainability of EPR systems and waste management sector. Therefore, future studies should pay more attention to these economic and innovation outcomes of the policy.

On the upstream side, this review has highlighted the paucity of quantitative investigations on producers' eco-design strategies. Quantitative, country-sector level analyses exploiting innovation data related to EEE, such as patents (Zhao et al., 2021), green designs and trademarks (Ghisetti and Montresor, 2021), could provide more systematic analyses compared to firm-level, qualitative approaches. An alternative approach may consider the use of bill of materials data (Babbitt et al., 2021; Huisman et al. 2017). Moreover, further research should shed light on opportunities for and barriers to the diffusion of voluntary producer responsibility and of circular business models while exploiting the EPR mechanism: the key point here is to show how the implementations of these strategies could ensure a financial advantage for producers, reducing their perceived risk.

Finally, EPR literature should be connected with research focusing on drivers of CE which are transversal to the life-cycle phases. Here, some key fields of investigation are:

- the cooperation among stakeholders involved in EPR systems;
- the integration of waste and production policies;
- the definition of eco-design standards and targets;
- the role of information management within the EPR systems and the EEE value chain;
- the role of digital technologies in improving the efficiency of EPR systems.

Two possible limitations of this study are acknowledged. First, further papers may have been selected for this review by using keywords referring to specific EPR implementation mechanisms, e.g., product stewardship, as listed in Gupt and Shahay (2015). Secondly, this work reviews scientific publications only: useful information may also come from the grey literature offering the perspectives of the industry (e.g., APPLiA Europe), PRO (e.g., WEEE Forum) and recyclers (European Electronics Recyclers Association).

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# Extended producer responsibility and trade flows in waste: The case of batteries*

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**Abstract**

In the debate on international waste trade, the focus on resource efficiency and recycling has gradually begun to accompany the focus on negative environmental externalities. In this context, we examine the impact of Extended Producer Responsibility (EPR) on the export of waste batteries (WB). EPR is considered as a key policy for the "marketization of waste". On the other hand, WB are a hazardous waste that also contain a high concentration of critical raw materials. As such, they are of strategic importance for the recovery of critical resources, while at the same time requiring proper environmental management. Therefore, it is crucial to understand where WB are treated and how this is affected by related policies.

Our results, based on difference-in-difference models in a gravity framework, show a consistent increase in WB exports after EPR implementation compared to the trend for other wastes. This result is likely to be an indirect consequence of the ability of EPR to support growth in waste collection rates, more accurate tracking of transboundary waste flows, and specialization of national waste management systems. In particular, WB exports appear to be directed to countries with more advanced waste management systems, more stringent environmental regulations, and limited endowments of the mineral resources typically contained in batteries.

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**Keywords**: Extended producer responsibility, batteries, trade, critical raw materials, circular economy.

**JEL codes**: K32, Q51, Q53, Q56

# 1 Introduction

The energy transition, the growth of electric mobility and the ubiquitous penetration of digital devices are rapidly increasing the demand for batteries worldwide (IEA, 2022; Salles Martins et al., 2022). In turn, as easy to expect, this will lead to a future growth of waste batteries (WB) (Wang et al., 2018), which are a hazardous type of waste (Mrozik et al., 2021; Winslow et al., 2018). Disposal and processing of batteries, especially if not performed with best available technologies, may result in pollutants emissions in soil, air and water. The high ecotoxicity of these substances is a danger for the wildlife and humans. For instance, the exposure to lead originating from lead-acid batteries has been linked to retarded fetal growth and lower educational achievement (Tanaka et al., 2022). Nickel, another common element in other types of batteries, is the most common allergic metal and it can cause respiratory disorders and cancer at higher concentrations. Cadmium is known for its high carcinogenicity.

Moreover, batteries contain high concentrations of critical raw materials (CRM) (European Commission, 2020; Schrijvers et al., 2020), such as lithium and cobalt (Seck et al., 2022). CRM are classified as such because they are both essential inputs for strategic value chains - for instance they enable the energy and digital transitions (IEA, 2021) - and they are exposed to high supply risks since their extraction and refining is concentrated in few countries (Carrara et al., 2023; Eggert et al., 2016; Liu et al., 2022). This concentration of CRM-related value chains rises the concerns for supply and price volatility in dependent countries (Kowalski and Legendre, 2023). As proof of this, prices of battery metals increased dramatically in early 2022, posing a significant challenge to the electric vehicles industry (IEA, 2022). The European Union (EU), indeed, is trying to regulate the whole life cycle of batteries through its new Batteries Regulation in order to alleviate its dependence on extra-EU suppliers (European Commission, 2019b). In particular the EU claims that it will support the development of a European batteries value chain (Duffner et al., 2020), it will set targets regarding the content of recycled materials, and that WB collection and recycling targets will be updated over time. Alongside the Batteries Regulation, the EU is discussing the CRM Act. The proposed regulation sets benchmarks for domestic extraction, processing and recycling (respectively 10, 40 and 15% of the EU annual consumption), and a diversification of supply by origin within 2030 (European Commission, 2023). The growing pressure on raw materials extraction for batteries production, among other technologies, is responsible of undeniable impacts on local populations and ecosystems around mining sites, typically located in developing countries (Agusdinata and Liu, 2023; Luckeneder et al., 2021).[1] Hence, a pressing problem of environmental justice connected to the green and digital transitions has been recognized (Sovacool et al.,

[1] According to the CRM Act, by 2030 the EU is still projected to rely on virgin CRM, i.e. mining activities, for 85% of its demand.

2019).

Strikingly, research on the current management of waste batteries (WB) is extremely limited. In particular, while a rich literature has investigated the economic potential of battery recycling or reuse (Innocenzi et al., 2017; Liu et al., 2019; Wang et al., 2014), the impact of policies related to WB management has been neglected. This is even more the case for the link between WB management regulations and transboundary movements of WB. A trade perspective has sometimes been adopted, focusing on the upstream part of the battery value chain, i.e. raw material supply and battery manufacturing, to estimate international flows of specific materials (Sun et al., 2017). Therefore, in order to plan *ex-ante* and evaluate *ex-post* circular economy strategies to reduce raw material extraction for battery production, mitigate CRM supply risks and improve the environmental outcomes of the WB management sector, it is crucial to gain a thorough understanding of where WB are treated as a first step.

Over the last thirty years, international environmental agreements and regulations, such as the Basel Convention (1992) and the EU Waste Shipment Regulation (2006), have aimed to reduce transnational flows of hazardous waste, particularly from developed to developing countries (Baggs, 2009; Levinson, 2023; Yamaguchi, 2022; Thapa et al., 2023), where disposal facilities are inadequate and environmental regulations are weaker. Therefore, the priority of regulations on the trade of waste has been to decrease the environmental externalities generated by this phenomenon. Research has usually found these agreements to be rather ineffective (Kellenberg and Levinson, 2014; Rossi and Morone, 2023). In the context of these international environmental agreements, electronic waste, which is often associated to WB because of its materials characterization and hazardousness, has been monitored with a special attention (Khan, 2016). For this specific case, some authors argued that the quantity of electronic waste shipped from developed to developing countries is negligible relying on secondary trade data (Lepawsky, 2015); on the contrary, other studies, based on primary data collection, proved that substantial illegal or questionably legal shipments persist (Bisschop, 2012; Forti et al., 2020; Puckett et al., 2019).

In parallel, it is increasingly considered strategic to strengthen national or international circular economy systems aimed at the recovery of (critical) resources (Kojima, 2020; Pommeret et al., 2022; Rosendahl and Rubiano, 2019). The aim is twofold: to reduce supply risks and to reduce the pressure on resource extraction by reducing the life cycle impact of materials. [2] In particular, within this framework, the EU has tried to implement a strategy of "marketization of waste" (Gregson et al., 2013; Reis, 2016), ready to create the normative and market conditions to turn waste into an economic resource. This is particularly true for CRM-rich waste flows (Theis, 2021).[3] With regard to this marketization of waste, one of the main policy

[2] From a life cycle perspective, recycling batteries reduces energy consumption and greenhouse gas emissions, over and above saving natural resources, when compared to landfilling (Boyden et al., 2016). For this reason, high levels of recycling are also necessary to achieve net zero targets (IEA, 2021). However, informal recycling, which is common in developing countries, is known to be unsafe for workers and the environment (Mrozik et al., 2021).

[3] As explained by Xu et al. (2020) and the International Energy Agency (2022), by 2040 battery recycling could meet a significant 28-50%, 36-71% and 29-57% of lithium, cobalt and nickel demand for new battery production respectively. However, these figures are expected to remain negligible until 2030. Reasons for this include the limited

interventions is considered to be the Extended Producer Responsibility (EPR) (Kama, 2015). EPR is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product life cycle (OECD, 2016). Under EPR regulations, which have been implemented in various countries worldwide starting from the late nineties, producers are typically addressed with three types of responsibilities: a physical, an economic, and an informative one (Compagnoni, 2022). In other words, they are responsible for the physical handling of the end-of-life management of their products, for covering the cost associated with waste management, and to inform the public stakeholders on the quantity of waste collected and its management. The focus on the end-of-life management of batteries shifted from toxic reduction toward resource recovery in the early 2000s, especially thanks to EPR policies discussions (Lindhqvist, 2010; Turner and Nugent, 2016). As we explain in Section 2.2, EPR could have indirect impacts on waste trade networks, both discouraging or boosting exports.

To the best of our knowledge, this paper is the first attempt to empirically investigate the impact of EPR regulations in general, and EPR on WB in particular, on the trade of the affected waste flows.

The relevance and urgency of studying the impact of domestic circular economy policies, such as EPR, on global value chains is explicitly claimed by the OECD (Yamaguchi, 2018). In fact, EPR is rarely considered in open economy settings (Sugeta and Shinkuma, 2014). The few, mostly theoretical, analyses of EPR from a trade perspective focus on illegal waste flows (Bernard, 2015), the role of market power in an international recycling market (Dubois and Eyckmans, 2015) or the relationship between waste trade and demand for natural materials (Joltreau, 2021). In addition to the economic and environmental relevance of (W)B described above, our choice to investigate EPR on WB is due to the fact that this regulation addresses a waste flow that is well defined in the Harmonised System classification for traded goods. Typically, addressing the same research questions to other categories of waste would not necessarily ensure the same level of precision.[4]

# 2 EPR: framework and impact on international trade

## 2.1 EPR: concept and regulations

The concept of EPR, first introduced in the 1990s, is defined by Lindhqvist (2000, p. 37) as a "strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer responsible for the entire life cycle of the product and especially for the take-back, recycling, and final disposal of the product". In other words, EPR is an environmental policy approach in which a producer's responsibility for a product is extended to the post-consumer stage of a product life cycle (OECD,

diffusion of technologies and facilities capable of recovering CRM on an industrial scale and the still relatively low collection rates for WB.

[4] For example, the HS classification has not specifically identified and therefore captured the flows related to the commodity of electronic waste, which is often subject to EPR regulations, until its review in 2022.

2016). To face the challenge of growing volumes and complexity of waste streams, EPR policies sought to shift the burden of managing certain end-of-life products from municipalities and taxpayers to producers. In line with the "polluter-pays" logic, three types of responsibility are usually addressed to producers (Lindhqvist, 2000). The *physical* responsibility imposes an obligation on producers to collect WB and send it for treatment.[5] The *economic* responsibility requires producers to bear the costs associated with waste management and treatment, ideally recycling, as well as those of campaigns to inform consumers about correct disposal procedures.[6] Finally, the *informative* responsibility requires producers to provide information on the environmental characteristics of products and on disposal procedures; in addition, public stakeholders must be informed on the amount of waste collected and how it has been managed.

By implementing EPR regulations[7], policy makers were expected to improve the overall environmental performance of targeted products on a life cycle basis. In general, "upstream" and "downstream" objectives of EPR can be distinguished (Gupt and Sahay, 2015; Lindhqvist and Lifset, 1998). The former refers to the design and production phase of products, where the aim of EPR is for producers to internalise the post-sale environmental costs of the equipment they put on the market, leading to waste prevention strategies (e.g. lifetime extension) or eco-design (e.g. design for recycling). The latter refers to the waste management phase of the product life cycle; from this perspective, EPR aims to increase the (separate) collection of the targeted waste streams and their recycling rates, thus supporting the development of the recycling sector and the market for recycled materials. An overview of the general expectations originally addressed by EPR policies is provided in Compagnoni (2022).

The relevance of EPR regulations and the support the principle has received from policy makers to support the transition to a more circular production and waste management system is highlighted by the global proliferation of EPR regulations, especially since the early 2000s, as shown in Figure 1, Panel A. EPR-type regulations have been applied to a variety of targeted products, most frequently electronic waste, followed by packaging, tires and batteries (Kaffine and O'Reilly, 2015). Globally, the latter are estimated to account for around 11% of EPR regulations (Figure 1, Panel B).

In the next section, we outline the theoretical mechanisms behind the indirect effects that the adoption of EPR could have on the export of waste generated at the end of the life cycle of the products covered by the policy.

[5] Depending on local regulations and the type of waste covered by the EPR, municipalities may remain responsible for organizing waste collection points. In this case, the producers are responsible for the subsequent waste management operations.

[6] These costs could be borne directly by the producer or partially covered by a special charge levied on consumers.

[7] The EPR principle has been implemented though a variety of instruments, ranging from deposit/refund schemes, to upstream combined tax/subsidies, to advanced disposal feel, but product take-back requirements are by far the most common instrument (Kaffine and O'Reilly, 2015). Regardless of the implementation scheme, EPR policies are based on the three pillars of physical, economic, and informative responsibility; therefore, the effects of EPR on trade that we propose in Section 2.2 can be considered as generalizable.

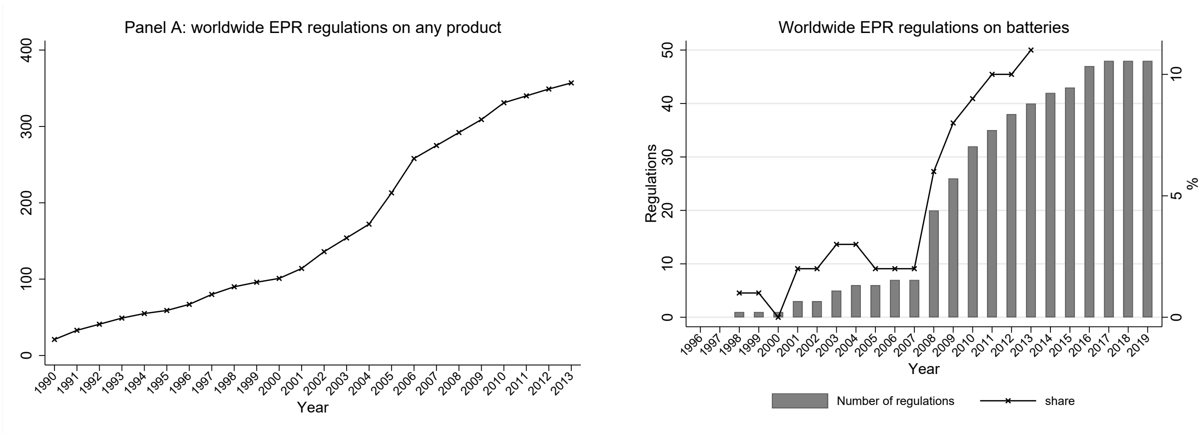


Figure 1: Panel A: overall worldwide number of EPR regulations, 1990-2013; our elaboration on Kaffine and O'Reilly (2015) data. Panel B: worldwide number of EPR regulations on batteries (bars, left scale) and share of EPR regulations on batteries over the overall number of EPR regulations (line), 1996-2019; our elaboration.

## 2.2 The impact of EPR on waste trade

As recalled above, EPR directly addresses the waste collection, management, and reporting responsibilities of producers of several product categories, including batteries, but is not intended to have a direct impact on international trade in the targeted products, nor on trade in waste generated at the end of the life cycle of these products.[8]

The question then arises as to why EPR should be expected to have an impact on waste trade flows. Furthermore, what is the expected sign of the relationship between EPR and waste trade? To answer these two questions, we should outline the rationale for waste trade and how EPR relates to it. The existing literature has extensively shown that bilateral waste trade depends to a large extent on the relative costs of waste treatment in the pair of countries involved in the exchange (Cassing and Kuhn, 2003; Falkowska, 2020; Higashida and Managi, 2014; Kellenberg, 2012). The adoption of an EPR policy increases the monetary and non-monetary costs for domestic producers of targeted products in terms of *physical*, *economic* and *informative* responsibility to treat them more efficiently at the end of their life cycle. Therefore, the adoption of an EPR regulation by a country may change its relative costs of waste treatment with respect to its trading partners and, in turn, affect international trade flows of waste generated by targeted products toward some importing countries (whose relative costs of waste treatment have decreased) and away from others (whose relative costs of waste treatment have increased).

In theory, the adoption of EPR could have both positive and negative indirect effects on trade in waste generated at the end of the life cycle of the targeted products.

[8] Examples of regulations that explicitly target trade in specific pollutants and commodities include international environmental agreements (IEAs) such as the Basel Convention, whose limited impact on reducing hazardous waste has been demonstrated by Kellenberg and Levinson (2014), and the Rotterdam and Stockholm Conventions, which have been more effective in reducing trade in hazardous chemicals and persistent organic pollutants (Núñez-Rocha and Martínez-Zarzoso, 2019).

On the one hand, the adoption of an EPR policy can lead to a reduction in exports of the relevant waste from the country adopting the regulation. This could happen for at least three reasons. *First*, the *physical* responsibility increases the collection rates and, consequently, the demand for treatment of the targeted waste streams (Kaffine and O'Reilly, 2015; Massarutto, 2014). To achieve this objective, EPR regulations typically set waste collection targets (Gupt and Sahay, 2015).[9] In the case of Europe, an increase in the collection rates after the introduction of EPR has been observed for several products (Dubois and Eyckmans, 2015), including WB (Perchards and SagisEPR, 2018, 2022).[10] If, in the face of higher collection rates, the home country strategically invests and innovates in its own waste management system and recycling sector, developing more facilities and eventually better technologies[11] (Atasu, 2019; Favot et al., 2022; Massarutto, 2014; Nicolli et al., 2012), EPR can reduce the relative cost of domestic waste treatment and thus the need for exports, by encouraging an increase in waste management capacity (Kellenberg, 2015; Latorre et al., 2021). In short, stable national waste management systems would develop (Tian et al., 2020). *Second*, on the upstream side of the product life cycle, the general objective of EPR is to prevent the generation of waste (Compagnoni, 2022). The *economic* and *physical* responsibilities of producers for waste management can support ecodesign practices such as dematerialization and product life extension (Kinokuni et al., 2019). In principle, these phenomena may well contribute to reducing the mass of waste generated and subsequently collected domestically, with a negative impact on waste export flows. However, despite some success of EPR in the packaging sector in preventing waste generation (Joltreau, 2022), the upstream effectiveness of the policy has been considered limited (Compagnoni, 2022). *Third*, and finally, *informative* responsibility increases the non-monetary costs for domestic producers of reporting and informing about treatment and disposal procedures of targeted products. These costs may be exacerbated when waste is exported due to logistics, foreign bureaucracy and customs controls. All this is particularly relevant in the case of hazardous waste, which is highly regulated (Moïsé and Rubínová, 2023). Therefore, the adoption of an EPR approach may encourage the home producers to manage waste domestically, thereby reducing the incentives to export.

On the other hand, the implementation of EPR can lead to an increase in the export of waste related to the targeted products for several reasons. *First*, and symmetrically to what has been put forward above, the increase in collection rates of waste, pushed by the *physical* responsibility on producers of targeted products, can lead to an increase in export of waste if a proper waste management system and recycling capacity is not established at home. Once the waste is collected, the EPR imposes an obligation on producers to recycle

[9] For example, in the the case of WB, the EU Waste Batteries Directive (2006/66/EC) initially set a collection target of 25% of the average weight of batteries sold, which was increased to 45% from 2016 (see Appendix B.4). The EPR mechanism has allowed to overcome an increasing quantity of WB to end up in urban unsorted waste streams (European Commission, 2019a).

[10] Three main operational EPR models for batteries have been implemented at the state level in Europe: the single organization model, the state fund model and the competing organizations model (Perchards and SagisEPR, 2018). All of them have been shown to be potentially effective in increasing collection rates (Perchards and SagisEPR, 2018).

[11] Encouraging innovation in the recycling of WB is one of the objectives specifically stated in the EU Batteries Directive (see Appendix B.4)

or dispose of it at home or abroad. Therefore, if collected waste cannot be treated domestically due to a disadvantage (higher relative costs) in terms of waste management system and recycling facilities, exports in waste may well increase. Instead, where EPR is not implemented, separate WB collection rates remain low, with limited masses to be treated (Levänen et al., 2018). *Second*, and similarly to the previous argumentation, the *economic* responsibility of producers encourages them to treat waste for recycling or disposal in the country where it is more efficient to do so (Bernard, 2015; Joltreau, 2021; Premalatha et al., 2014). Again, this would induce those countries with a disadvantage (higher relative costs) to increase exports of waste rather than treat it domestically. Also notice that countries are increasingly building specialized capacity for specific hazardous materials operations (Yang, 2020). Hence, a relatively high waste treatment cost may arise not only from generally scarce physical capital (i.e. facilities and technologies) and human capital endowments, but also from the specialization of the exporting country in some waste categories and not others. *Third*, the *informative* responsibility of producers (Gerrard and Kandlikar, 2007; Lifset, 1993) should lead to more accurate tracking of waste streams as they move from domestic borders to foreign destinations. Thus, an indirect effect of the introduction of EPR could be the emergence of international trade flows in the waste generated by targeted products which, in the absence of EPR, might be lumped together with unsorted municipal or other waste flows and thus not show up in trade data.

In summary, EPR could indirectly affect trade in waste generated by the targeted products through a variety of simultaneous and opposing effects. Thus, quantifying a *net* effect is ultimately an empirical question. In this paper, we assess the impact of a country's (exporter's) adoption of EPR on trade (exports) of waste generated by batteries. In particular, we consider those regulations that are clearly attributable to the EPR concept and that affect battery producers. To our knowledge, this paper is the first to empirically test the relationship between EPR and exports in waste products in general and for the specific case of waste batteries in particular.

## 2.3 Contributions to the literature

Within the framework outlined above, our paper contributes to the existing literature in several directions. *First*, we make a twofold contribution to the literature on EPR. We develop a link between EPR policies and waste exports, both by describing the possible channels influencing this relationship and by testing this relationship empirically for the first time. In a broader perspective, we contribute to the debate on trade and circular economy policies. In addition, we provide the most comprehensive review of EPR regulations on waste batteries adopted worldwide.

*Second*, we contribute to the policy debate on how WB -as hazardous waste- should be managed in modern and sustainable economies. Indeed, trade in hazardous waste may be dangerous for health and environmental reasons, and there may be reasons to restrict transboundary movements of such waste[12].

[12] In this sense, several IEAs do not seem to be sufficiently effective in restricting trade in hazardous waste, and specific

Following this argument, a reduction in the export of trade in WB may be a desirable outcome of the adoption of an EPR policy. However, WB also contain a high concentration of critical raw materials that are essential inputs for strategic value chains and are subject to high supply risks. Therefore, national and supranational institutions are interested in controlling the flows of these materials. In this sense, trade in such wastes may be a desirable outcome of an EPR policy, if the flows of WB are directed to countries with a comparative advantage in managing, treating and recycling such waste, in the spirit of a circular economy approach. We find that, the adoption of EPR by the exporting country leads to an increase of exports in WB with respect to other waste products not targeted by the EPR policy. In this sense, and with reference to Section 2.2, the factors favoring exports in WB overcome the forces that should reduce them.

*Third*, we provide some tentative explanations for the observed increase in exports of WB. The existing literature so far has shown that trade in waste depends on two main factors, which affect the relative cost of its treatment at home versus abroad: (i) the difference in natural and technological endowments related to waste treatment between trading partners, and (ii) the difference in the stringency of environmental policies.[13] In this paper, we examine whether changes in the export of WB following the adoption of EPR by the exporter are affected by the technological endowment for waste treatment or by the stringency of environmental regulations in the trading partner. This is a relevant issue, because if exports of WB accrue in countries with a superior technological endowment for recycling, this would indicate an improvement in the supranational waste management system in the sense of a circular economy. Conversely, if WB flows are directed to countries with looser environmental regulations, this would indicate the lack of effectiveness of EPR policies, providing evidence in favor of a possible waste heaven hypothesis. Furthermore, we examine how the availability of the most common minerals used in batteries production affects import flows of WB. In fact, countries with significant endowments of those natural resources might find little convenience in importing WB for the purpose of materials recovery.

*Fourth*, from a methodological point of view, by focusing on the impact of EPR on trade in WB, we can identify waste that is directly and unambiguously linked to the products targeted by the EPR policy. WB can be properly tracked with a specific HS6 code in bilateral trade flows. In this way, we minimize the risk of considering a noisy measure of indirectly targeted waste. Moreover, by using highly disaggregated data at the product level, we are able to control for multiple sources of unobserved heterogeneity, ranging from multilateral resistance terms to product attributes such as quality or technological characteristics, and time-invariant differences across origin-destination-product triplets. This allows us to minimize the risk of omitted variable bias.

rules for producers, such as EPR, may well be a complementary tool.

[13] A third relevant factor in explaining waste trade patterns relates to corruption and organized crime (Cesi et al., 2019; Kellenberg, 2015). However, these mainly affect the illegal trade of waste, while in this paper we focus on the legal shipment of waste. Thus, in the empirical specification, organized crime at the country level (as well as other time-varying country characteristics) is accounted for by a vector of country-year fixed effects.

# 3 Data and descriptive statistics

To address the research questions outlined above, we base our empirical analysis on bilateral trade flows for the period 1996-2019. The data are taken from BACI, a detailed international trade database that contains annual product-level information on imports and exports for over 200 countries. The BACI dataset reconciles trade declarations from importers and exporters, as they appear in the COMTRADE (Commodities Trade Statistics) database[14], but fill gaps and corrects for data incongruencies (Gaulier and Zignago, 2010). Products are identified according to the Harmonized System (HS) classification, which is the standard nomenclature for international trade used by most customs, and they are reported at the 6-digit level (HS6), which is the finest product classification at the international level. Because the Harmonized System, which has been employed progressively from 1989, has been importantly revised over time (in 1996, 2002, 2007, 2012 and 2017), it is important to harmonize the classifications to a single version. The BACI dataset harmonizes the different HS classifications using UN conversion tables to HS-1996. Bilateral trade flows between countries are reported both in values and quantities. Whereas values are reported in thousands of US dollars, quantities can be registered in different units of measure (tons, meters, square meters), although 85% of transactions are reported in tons. To standardize the other 15% of flows, the BACI estimates conversion rates from other units to tons (Gaulier and Zignago, 2010).

Overall, the BACI dataset include approximately exporter ($e$) -importer ($i$) -product ($p$) -year ($y$) transactions for more than 200 countries and approximately 5,000 products, between 1996 and 2019, ending up with more than 200 million observations over the sample period. Because our analysis focuses on waste products ($\text{WP}_p$), we restrict the BACI dataset to all 6-digit HS codes identifying these products. Following Kellenberg and Levinson (2014)'s approach, we select the HS code containing the words "waste", "scrap", "slag", "residue" or "ash" in their product description[15], ending up with 114 6-digit products. Table B1 in the Appendix provides the list of the HS6 codes identified as waste products, along with a brief description of each product. By restricting the BACI dataset to this subset, we obtain a total of about 2 million observations over the period under analysis. On average, each year waste products account approximately for 1.2% of the total trade flows among countries.[16]

The importance of trade in waste products is observed in the data, as the amount of waste traded in recent years has been noteworthy. Panel A of Figure 2 shows that waste is increasingly moving across borders, even if with lower growth rates compared to the early 2000s. This holds for waste shipments originating from

[14] UN Comtrade provides bilateral goods trade flows in US dollar value and quantity, at annual frequency and broken down by commodities according to various classifications (BEC, HS, SITC). COMTRADE accounts for more than 95% of the world trade.

[15] See https://unstats.un.org/unsd/classifications/econ/ for the complete description of product categories.

[16] Note that the BACI dataset do not include null bilateral trade flows, i.e. exporter-importer-product-year observations equal to zero. The zero trade flows are therefore not included in the main estimation sample. In order to account for the presence of zero trade flows, in a robustness check we estimate a gravity model in multiplicative form instead of logarithmic form, by applying a Poisson Psuedo Maximum Likelihood (PPML) estimator.

EU-28 countries as well as for transfers of waste from non-EU-28 countries; currently, the former amounts to about half of the latter. The phenomenon of waste trade, as previously outlined, is due to the presence of comparative advantages of various nature among countries in disposing or recycling waste, and it is fuelled by the growing mass of waste generated worldwide.

The same growth dynamic is observed for hazardous waste ($HW_p$), that is waste that has substantial or potential threats to public health or the environment in terms of toxicity, corrosivity ignitability and reactivity. The data presented in Panel B of Figure 2 are based on the HS6 codes related to hazardous waste products proposed by Kellenberg and Levinson (2014).[17] Differently from $HW_p$ exports originating in non-EU-28 countries, that grew exponentially in the early 2000 and slowed down since 2006, $HW_p$ exports of EU-28 countries continued growing roughly in a linear way. Most of these shipments are directed towards other EU countries. This is in line with the previous evidence provided by (EEA, 2012) and European Commission (2015).[18] According to our estimates, over the period 1996-2019, $HW_p$ intra-EU exports has roughly quadrupled, as for $HW_p$ exports originating in the rest of the world, while exports from the EU to the rest of the world has doubled.

Among $HW_p$ products, we are particularly interested in waste batteries and accumulators ($WB_p$), which represents the "treatment" group in our empirical analysis. As shown in Panel C of Figure 2 there has been a general increase in exports of $WB_p$, both from EU-28 members and from non-EU-28 countries. In particular, a clear change in the trend is observable around 2010-2011 with a peak in the exports of $WB_p$, which has not returned to previous levels. This waste product covers, on average, 0.35% of the total trade value in waste products. USA, France, Netherlands, United Arab Emirates are the top four largest exporters of $WB_p$ (Panel A of Figure 3), while South Korea is by far the world's largest importer of $WB_p$ and it is the home of three of the world's 10 biggest battery makers, LG Energy Solution, Samsung SDI and SK On. Among importers of $WB_p$, two European countries, i.e. Germany and Spain rank second and third, followed by India, and by other European countries (Panel B of Figure 3). Focusing on the EU, intra-EU exports of WB dominate (Panel C of Figure 3). In fact, as represented in Figure 3 Panel D, EU exports towards non-EU countries represents a very small share of total EU WB exports, while exports following the opposite direction are about seven times higher. Among European importers, Germany and Spain are by far the largest European importers of WB (Figure B1 in Appendix B.1). It is interesting to read this descriptive evidence in consideration of the generalized adoption of EPR on WB in the EU, but also of other two factors characterizing European waste management systems, namely strict environmental regulations and the advanced level of treatment facilities and technologies.

[17] Table B1 in the Appendix clarifies which waste products are classified as hazardous. Since no official correspondence tables between the HS classification and hazardous waste (Basel Convention) codes are available, our figures for hazardous waste exports should be considered as a proxy of the actual ones.

[18] Data on hazardous waste in these reports are slightly different from those reported here as they are drawn on official data, based on country reporting to the European Commission (Eurostat). Nonetheless, according to the European Commission (2015), hazardous waste exports within EU countries grew by 127% in the period 2001-2012, while exports from the EU to non-EU countries amounted to five million tonnes in 2012.

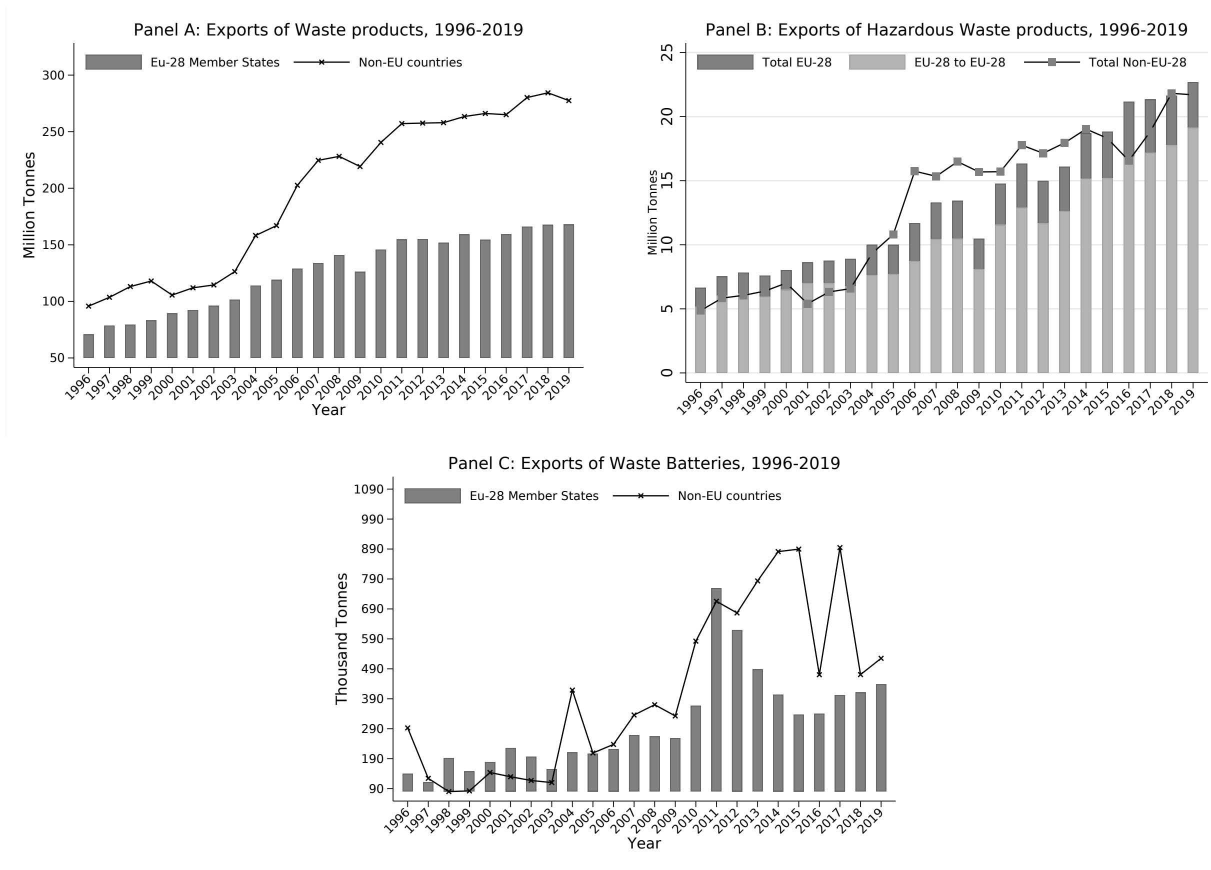


Figure 2: Export Quantity of Waste, Hazardous Waste and Waste Batteries, 1996-2019. Our elaboration on BACI data.

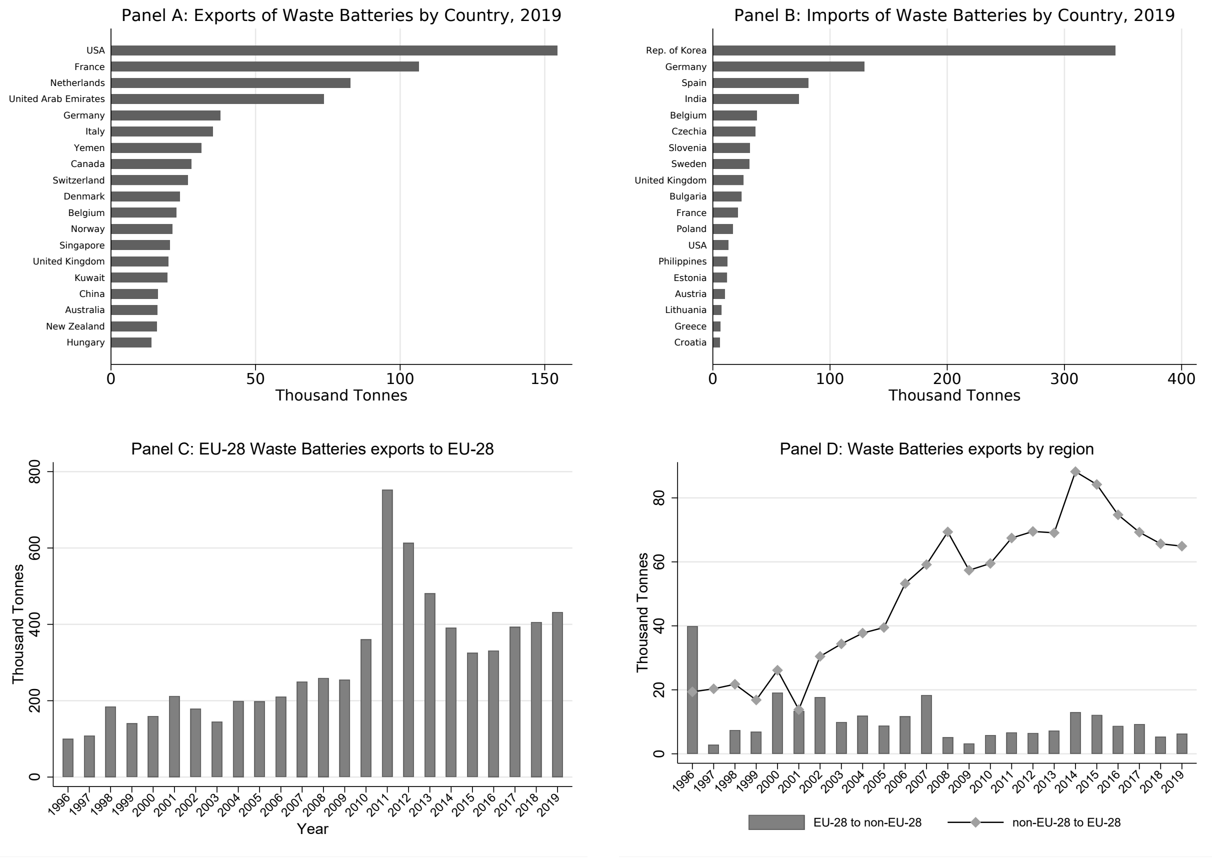


Figure 3: Exports and Imports Quantity of Waste Batteries by Country or Region (EU/non-EU), 1996-2019. Our elaboration on BACI data.

The second source of information that we employ in the empirical analysis, concerns the implementation of EPR regulations on batteries at the country level. This information has been extracted from a variety of sources, including technical reports published by the European Portable Battery Association, the European Commission, the OECD, the United Nations, and national governments. The information is quite relevant as for the first time it provides a common framework to analyze the phenomenon. It covers 89 countries, of which 48 implemented the EPR policy during the period under study and 41 did not. Table B2 in Appendix B.3 reports: the countries for which it was possible to collect information on the implementation of EPR on WB; the year and name of the regulation introducing the policy, in the case of adopting countries; the sources of the information. The year of adoption of EPR varies from country to country: while for EU countries the reference regulation is the national law transposing the EU Directive 2006/66/EC, which in most cases took place between 2008 and 2010, for non-EU countries the national regulations apply.

Since we expect that national regulations mostly affect the activities of domestic firms, we focus on the effects of EPR adoption taking the perspective of the exporting country. Therefore, our variable $\text{EPR}_{ey}$ is a dummy that takes value one from the year the EPR policy was adopted in the exporting country onward, and zero otherwise.

We complement the analysis with other standard gravity variables obtained from the Cepii Gravity dataset developed by Conte et al. (2022), which gathers a wide range of potential determinants of trade flows such as geographic distances, indicators of cultural proximity and trade facilitation measures.[19] Following the gravity theory, we include bilateral distances between exporting and importing countries, $\text{Dist}_{ei}$, where the distance is measured between the most populated city in each country. Among the bilateral variables, we include some cultural characteristics. Specifically, we consider whether exporter and importer share a common border by including a dummy that equals to one if countries are contiguous ($\text{Contig}_{ei}$); whether the country pair was ever in colonial relationship ($\text{Colony}_{ei}$); and whether countries have the same official language ($\text{Language}_{ei}$). Among the variables capturing characteristics of the single country, we consider the level income, proxied by the $\text{GDP}_{ey}$ and $\text{GDP}_{iy}$, respectively. Table 1 describes the variables included in the analysis, together with some descriptive statistics which also allow to grasp the relevance of the phenomenon under investigation. For instance, $\text{WB}_p$ represents the share of trade flows in our dataset which are related to WB and $\text{EPR}_{ey}$ the share of observations for which the exporting country is an EPR adopter.

Merging the three source of information, BACI, EPR and Cepii Gravity, gives us a final sample of about 1.65 million observations, covering 89 exporting countries and all destinations, for 114 waste products.

[19] This information are sourced from different institutions such as the World Bank, the WTO and the IMF.

Table 1: Variables' names, definitions and sources

| | Variable | Description | Mean | Median | Std. Dev | Data source |
|---|---|---|---|---|---|---|
| Dependent variable | $\text{Quantity}_{eipy}$ | Quantity of waste product exported (tonnes) | 5343.223 | 18.133 | 62208.592 | BACI |
| Treated group and Policy | $\text{WB}_p$ | Dummy for Waste Batteries | .011 | 0 | .105 | BACI |
| | $\text{EPR}_{ey}$ | Dummy for Extended Product Responsibility policy | 0.397 | 0 | 0.489 | Several (see App.5) |
| Gravity Variables | $\text{Dist}_{ei}$ | Simple distance between most populated cities (km) | 4550.128 | 2495.000 | 4323.265 | Cepii Gravity |
| | $\text{Contig}_{ei}$ | 1 if countries are contiguous | 0.145 | 0 | 0.352 | Cepii Gravity |
| | $\text{Colony}_{ei}$ | 1 if pair ever was in colonial or dependency relationship | 0.067 | 0 | 0.249 | Cepii Gravity |
| | $\text{Language}_{ei}$ | 1 if countries share common official or primary language | 0.138 | 0 | 0.345 | Cepii Gravity |
| | $\text{GDP}_{ey}$ | GDP Exporter (current thousands US\$, log) | 20.527 | 20.595 | 1.702 | Cepii Gravity |
| | $\text{GDP}_{iy}$ | GDP Importer (current thousands US\$, log) | 19.303 | 19.475 | 1.997 | Cepii Gravity |

*Notes*: The subscripts $e$ , $i$, $p$, and $y$ (if applicable) denote exporter, importer, HS-6 digit product and year, respectively.

# 4 Empirical Analysis

## 4.1 Empirical model and identification strategy

### 4.1.1 The 'augmented' gravity equation

In international economics, the gravity model of trade (Anderson, 1979; Bergstrand, 1985) has long been the default choice for explaining bilateral trade flows. The model has been initially conceptualized by Tinbergen (1962), and later on reformulated and extended by Eaton and Kortum (2002), Anderson and van Wincoop (2003, 2004), and Redding and Venables (2004). As the name suggests, it is based on the principle of gravity in which the volume of trade between two countries is directly proportional to their scale (measured by GDP or population) and inversely proportional to the distance between them (measured by geographical, cultural, or linguistic factors). The gravity equation has been used as a workhorse for analyzing the determinants of bilateral trade flows for 50 years, making it one of the most stable empirical relationships in economics (Leamer and Levinsohn, 1995; Head and Mayer, 2014).

Since the inception of the gravity model, one of the primary objectives has been to examine the effectiveness of different policies in influencing trade. In this perspective the primary focus is on estimating the coefficient of policy impact. Following this tradition, we apply an 'augmented' version of the gravity model to estimate the indirect effect of the EPR policy adoption by the exporting country trading WB. To investigate this effect, we rely on a difference-in-differences (DiD) approach that compares the trade flow of WB (the "treated" group) with the trade flow of other waste products not targeted by the policy, before and after its implementation by the exporter.

The dependent variable in the gravity equation is the bilateral trade flow, and the relevant independent variable is the EPR policy interacted with the WB dummy, which identifies the product subject to the policy, i.e. WB. We include, as control variables, the standard determinants of bilateral trade flows, such as GDP, distance, and a set of dummies to capture the common border effect, common language, historical and political links between partners.

The general log-linear specification takes the following form:

$$\begin{aligned}\ln \text{Quantity}_{eipy} =& \alpha + \beta_1 \text{EPR}_{ey} + \beta_2 \text{WB}_p + \beta_3 \text{EPR}_{ey} \times \text{WB}_p + \gamma_1 \ln \text{GDP}_{ey} + \gamma_2 \ln \text{GDP}_{iy} \\ &+ \gamma_3 \ln \text{Dist}_{ei} + \gamma_4 \text{Contig}_{ei} + \gamma_5 \text{Language}_{ei} + \gamma_6 \text{Colony}_{ei} + \epsilon_{eipt},\end{aligned} \tag{1}$$

where $\ln \text{Quantity}_{eipy}$ is the logarithm of the quantity (weight) [20] of waste products $p$ traded from exporter $e$ to importer $i$ in year $y$; $EPR_{ey}$ is a dummy capturing the adoption by the exporting country $e$ of the EPR policy in year $y$, and $\text{WB}_p$ is a dummy identifying waste batteries. Our interest lies in the estimation of $\beta_3$, which gives us the difference in the impact of the EPR policy on the exports flows of $\text{WB}_p$ relative to the control group made up by all the other waste products.

There are several econometric issues that may arise when estimating the gravity model.[21] The first problem lies in the area of omitted variable bias, since there are some variables, such as the multilateral resistance terms, which are unobservable. Indeed, in attempting to provide a theoretical underpinning to the gravity equation based on a constant elasticity of substitution (CES) demand function, Anderson and van Wincoop (2003) have shown that the volume of bilateral trade is affected by trade impediments at the bilateral level (referred to as bilateral resistance), as well as the relative impact of these impediments compared to those of other countries (referred to as multilateral resistance).[22] Since this contribution, failing to include a multilateral resistance term in the gravity equation is seen as a significant source of bias and a crucial issue that researchers must address in their estimations.[23] As is standard in the literature, we use exporter-time fixed effects ($\omega_{ey}$) and importer-time fixed effects ($\omega_{iy}$) to control *inter alia* for unobservable exporter and importer multilateral resistances. These fixed effects will also control for any other country-time-specific characteristics that may impact bilateral trade on the exporter and importer sides. The introduction of these fixed effects absorb the proxies for the scale of the exporter ($\ln \text{GDP}_{ey}$) and importer ($\ln \text{GDP}_{iy}$) economy in Eq. (1), as well as other observable and unobservable country-year specific characteristics which vary across these dimensions, including various national policies (such as $\text{EPR}_{ey}$), institutions, and exchange rates. While the inclusion of time varying exporter and importer fixed effects allows to account for the multilateral dimension of the gravity model, another source of bias could arise due to time-invariant bilateral trade costs, both observable and unobservable. For instance, trade policy variable, such as Regional Trade Agreement, $\text{RTA}_{eiy}$, may suffer from reverse causality, because, other things being equal, a given country

[20] Research on international trade usually measures flows in either monetary or quantity terms. Following the rest of the trade literature on waste, we measure our dependent variable in terms of quantity, as this is better suited to give an idea of the potential pollution from waste trade, as well as the potential mass of materials to be recycled; moreover, non-recyclable waste can be exported at a negative price (Kellenberg and Levinson, 2014).

[21] See Head and Mayer (2014) for a exhaustive analysis on the estimation and interpretation of the gravity equation for bilateral trade.

[22] Kellenberg and Levinson (2014) clarify that these may include time-varying importer- and exporter-specific price indexes and multilateral price terms, environmental regulations and recycling costs, capital-labor ratios, political environments, or firm-level heterogeneity due to the fixed costs of exporting.

[23] Baldwin and Taglioni (2007) refer to the omission of the multilateral resistance term as the "gold medal mistake" of gravity equations, characterizing all the papers appearing before Anderson and van Wincoop (2003).

is more likely to liberalise its trade with another country that is already a significant trading partner. As suggested by Baier and Bergstrand (2007) a possible solution is to include a vector of country-pair ($\omega_{ei}$) fixed effects, which control for all time-invariant bilateral trade costs and will mitigate this endogeneity concern. The inclusion of the set of pair fixed effect absorb all bilateral time-invariant covariates in Eq. 1, but has the advantage of accounting for any unobservable time-invariant trade cost components.[24]

Taking into account all the caveats associated with estimating a bilateral trade equation, we use several specifications of our DiD model, gradually addressing the challenges posed by the empirical literature. In Table 2, we first estimate a simple regression including all the gravity variables. Because data are disaggregated at the HS6 product level, in this first model we account for product ($\omega_p$) and year ($\omega_y$) fixed effects, which allow us to control for product attributes, such as quality or technological features, as well as yearly macroeconomic shocks. Second, the role played by the multilateral dimension of trade is controlled for by means of time-varying country fixed effects that are included in a second empirical model, together with product fixed effects. Third, we further reduce the risk of biased results due to omitted variables by estimating an empirical model that includes the exporter-time, importer-time fixed effects together with the country-pair product specific fixed effects ($\omega_{eip}$). The inclusion of $\omega_{eip}$ allows us to control not only for time-invariant bilateral trade costs, but more precisely for any unobservable time-invariant differences in export volumes across origin-destination-product triplets.

As we are dealing with several dimensions, simply utilizing the conventional robust standard errors method is insufficient to rectify the error structure and can result in biased estimation errors and flawed statistical conclusions. Indeed, incorporating multi-level clustering has a significant impact, regardless of whether gravity models include fixed effects for country and time or for country-pair and time (Egger and Tarlea, 2015). As errors are likely to be correlated by country-pair in the context of the gravity model, we control for such interdependence in all specifications by reporting standard errors clustered at the exporter-importer level, together with standard errors clustered at the time level.

### 4.1.2 The difference-in-difference-in-difference (DDD) specification

The DiD approach can be a powerful tool in measuring the average effect of the treatment on the treated. However, identification of the effect using DiD relies on the parallel trend assumption which assumes that the trend in the outcome variable for the treated group would have followed the same path as the trend in the outcome variable for the control group in the absence of the treatment. In other words, the parallel trend assumption asserts that the treatment and control groups had similar trends in their outcomes before the treatment occurred, and that any differences in outcomes after the treatment can be attributed to the treatment itself rather than pre-existing differences between the groups. If the assumption is violated, the estimated treatment effect may be biased and unreliable. In what follows we discuss the robustness of our estimation

[24] Egger and Nigai (2015) argue that pair-fixed effects provide a more accurate measure of bilateral trade costs than the traditional set of gravity variables.

strategy.

Following Angrist and Pischke (2009), in order to increase the reliability of the parallel trends assumption between WB and other types of waste products, we estimate a DDD specification that exploits a triple difference and aims at addressing possible concerns associated with a more classical DiD model, which would be prone to either selection bias or the presence of confounding factors. In particular, a standard DiD approach would compare products subject to the policy (in our case WB) exported by a given country with products exported by the same country but not subject to the policy, with changes over time being the first source of variation exploited. In this case, there might be a selection problem if the product hit by the measure has significantly different characteristics from the control group; in other words, the common trend assumption may not hold. To address such a concern, an alternative specification would be to compare exports of waste batteries from a country with an EPR policy with exports of the same product from another country without an EPR policy. While this approach would address concerns about a possible selection bias, it opens the door to other unaccounted for confounders due to country-specific factors. A DDD approach allows us to exploit all sources of variation. Exports of WB before/after the imposition of the EPR policy are compared with the performance of the same product exported by countries not imposing the policy, and with different products exported by the same country that adopts the policy (all other waste products in our case). The DDD equation takes the following form:

$$\ln \text{Quantity}_{eipy} = \alpha + \beta_3 \text{EPR}_{ey} \times \text{WB}_p + \omega_{ey} + \omega_{iy} + \omega_{eip} + \omega_{py} + \epsilon_{eipt}, \tag{2}$$

where, with respect to the previous models, we also add product-year fixed effects ($\omega_{py}$). The inclusion of exporter-importer-product, exporter-year, importer-year and product-year fixed effects allows us to estimate a DDD model by exploiting the variability over time before and after the EPR measure is imposed, the within-country-pair across products variation between targeted and unaffected products, and the variation within HS6 product category across countries imposing and not-imposing the EPR policy. In particular, $\omega_{eip}$ captures the average export performance of each product in a given country-pair (so that the interaction captures variation over time), $\omega_{ey}$ and $\omega_{iy}$ refers to average origin and destination-time effects (thus exploiting variation across products within the same country), while $\omega_{py}$ controls for product-time effects and thus lets us compare the same good traded by different countries. This complete set of fixed effects is meant to saturate all possible sources of variation unrelated to the policy.

The DDD estimation strategy is adopted in the econometric models presented in columns (4) - (6) of Table 2.

Table 2: The effect of EPR policy on the exports of Waste Battery: baseline results

| Dep. Var. | ln Quantity$_{eipy}$ | | | | | |
|---|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) | (5) | (6) |
| EPR$_{ey}$ | $-0.065$*** | | | | | |
| | (0.011) | | | | | |
| $\times$ WB$_p$ | 0.304*** | 0.269*** | 0.623*** | 0.694*** | 0.456** | 0.449** |
| | (0.063) | (0.062) | (0.145) | (0.209) | (0.198) | (0.198) |
| EPR$_{iy}$ $\times$ WB$_p$ | | | | | 0.568*** | 0.549*** |
| | | | | | (0.170) | (0.203) |
| EPR$_{ey}$ $\times$ EPR$_{iy}$ $\times$ WB$_p$ | | | | | | 0.020 |
| | | | | | | (0.240) |
| ln GDP$_{ey}$ | 0.100*** | | | | | |
| | (0.013) | | | | | |
| ln GDP$_{iy}$ | 0.170*** | | | | | |
| | (0.012) | | | | | |
| ln Dist$_{ei}$ | $-0.694$*** | $-0.692$*** | | | | |
| | (0.004) | (0.004) | | | | |
| Contig$_{ei}$ | 1.063*** | 1.045*** | | | | |
| | (0.011) | (0.011) | | | | |
| Language$_{ei}$ | 0.078*** | 0.075*** | | | | |
| | (0.010) | (0.009) | | | | |
| Colony$_{ei}$ | 0.176*** | 0.177*** | | | | |
| | (0.013) | (0.013) | | | | |
| $\omega_y$ | Yes | No | No | No | No | No |
| $\omega_p$ | Yes | Yes | No | No | No | No |
| $\omega_e$ | Yes | No | No | No | No | No |
| $\omega_i$ | Yes | No | No | No | No | No |
| $\omega_{ey}$ | No | Yes | Yes | Yes | Yes | Yes |
| $\omega_{iy}$ | No | Yes | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | No | No | Yes | Yes | Yes | Yes |
| $\omega_{py}$ | No | No | No | Yes | Yes | Yes |
| Adj. $R^2$ | 0.410 | 0.417 | 0.734 | 0.740 | 0.742 | 0.742 |
| No. of Obs | 1,401,055 | 1,429,644 | 1,568,988 | 1,568,959 | 1,301,971 | 1,301,971 |

*Notes*: Observations are at the exporter-importer-product-year level. The coefficients appear together with standard errors clustered at the country-pair-product and year level *** significant at the 1% level, ** significant at the 5% level and * significant at the 10% level.

## 4.2 Econometric results

### 4.2.1 Baseline results

We present in Table 2 the estimates of the empirical model specified in Eq. 1.

Col. (1) shows the estimates of the 'augmented' gravity equation estimated by OLS, after controlling for annual common shocks, time-invariant unobserved heterogeneity at the product level, and vectors of exporter and importer fixed effects. While the coefficient on the adoption of the EPR policy by the exporting country (EPR$_{ey}$) shows that bilateral trade in all other waste products decreases after the adoption of the policy, this effect is counteracted and even reversed for WB, given the magnitude and sign of the estimated coefficient of the interaction term (EPR$_{ey}$ $\times$ WB$_p$). After the adoption of the EPR policy by the exporter country, the flow of trade in WB has increased more than the flow of the other waste products, *ceteris paribus*. This is an interesting result, which suggests that the responsibility on producers leads to an increase in export of WB.

The standard determinants of bilateral trade flows show the expected signs. In particular, the positive coefficient of ln GDP$_{ey}$ shows that larger economies produce more waste and have more to dispose of, which increases the quantity exported. As for the positive coefficient of ln $\text{GDP}_{iy}$, a scale effect plays a role here too. Indeed, larger economies are characterized by more disposal capacity, which for hazardous waste such as WB implies investments in treatment and recycling facilities. Consistent with Baggs (2009), the coefficient on importer GDP is larger than the one on exporter GDP, suggesting that as scale increases, disposal capacity may increase more than production capacity. The coefficient on ln $\text{Dist}_{ei}$ shows that as the geographical distance between the pair of trading countries increases, the trade flow of WB between them decreases. On average, a 1% increase in the distance between the two trading countries reduces trade by 0.69% . Contiguity is relevant too. If the two trading countries share a common land border, trade in WB increases by about 190%.[25]. Putting the evidence on distance and contiguity into perspective, this shows that transportation costs for waste products in general (including WB) are not negligible, as suggested by (Kellenberg, 2012). Pairs of trading countries that share the same language or are linked by colonial history trade more waste on average, as indicated by the respective coefficient estimates.

In col. (2), the vectors of exporter-year and importer-year fixed effects are included (together with product fixed effects) to take multilateral resistance terms into account. Due to the inclusion of these effects, the coefficients of the country time-variant characteristics, including the dummy capturing the adoption by the exporter country of the EPR policy, cannot be identified. The DiD coefficient can nonetheless be identified and shows, consistently with col. (1), a positive sign. In col. (3), product fixed effects are replaced by exporter-importer-product fixed effects to additionally control for any unobservable time-invariant differences in export volumes across origin-destination-product triplets. All control variables that are specific to country pairs cannot be identified. However, the DiD coefficient is larger in magnitude and more precisely estimated than those in col. (1) and col. (2). In particular, the DiD coefficient suggest that the volume of WB shipped increases by about 86% more than the volume of other waste products after the adoption of the EPR policy by the exporting country.

In the last three columns of Table 2, we add product-year fixed effects and estimate a DDD coefficient (as explained in Section 4.1.2) to saturate the empirical model for all possible sources of variation unrelated to the policy. The DDD coefficient of the model in col. (4), which controls for product-specific trends, is practically identical to the DiD coefficient shown in col. (3), and this reassures that results are not driven by uncontrolled differences between WB and other waste products not affected by the policy. Lastly, we introduce the role of the adoption of EPR on WB in the importing country. By controlling for the interaction between $\text{EPR}_{iy}$ and WB, we want to exclude the fact that our DDD coefficient is erroneously capturing an effect on WB exports that is actually driven by the importer's adoption of EPR on WB. Col. (5) shows that export flows of WB increase significantly more than other waste flows after EPR adoption, even when the importing country is also choosing to adopt the same kind of policy[26]. The triple interaction among $\text{EPR}_{ey}$,

[25] The percentage change is calculated as 100*(exp(1.063)-1), where 1.063 is the estimate of the $\text{Contig}_{ei}$ coefficient in col. (1).

[26] In this article, we focus on the relationship between EPR and exports of waste, accounting for EPR adoption in

$\text{EPR}_{iy}$ and WB, introduced in model (6), points out that once both trading partners have implemented the policy WB flows stabilize on the same trend of other wastes. The role of policies in the importing country is further explored in the Section 4.2.3, both to control for further possible confounding factors for the effect of EPR on WB exports as well as to explore the characteristics of WB importing countries.

Overall, the econometric evidence suggests that the adoption of the EPR policy by the exporting country leads to an increase in the export volume of WB compared to other waste products not targeted by the regulation. In this sense, and with reference to the indirect effects of the EPR policy (see to Section 2.2 above) on the trade in waste batteries, the trade-enhancing factors outweigh the trade-decreasing forces. This result of increased WB exports is even more relevant in the light of the efforts and regulations aimed at restricting waste imports implemented in various countries (Balkevicius et al., 2020; D'Amato et al., 2023; Tian et al., 2021).

### 4.2.2 Robustness checks

A first concern relates to the decision taken so far to consider all 114 (HS6) waste products as a control group in the analysis. In fact, waste commodities are heterogeneous in terms of hazardousness, recyclability, composition and, ultimately, value. Hence, WB (the target of the EPR policy) may be significantly different from many of the waste products in the control group. Although, as discussed in Section 4.1.2, the inclusion of product-time fixed effects mitigates this concern, we test the robustness of our results by repeating the estimation with a different control group and using the list of HS6 codes provided by Kellenberg and Levinson (2014) to define the category of waste products. This sample amounts to 51 waste products instead of 114.[27] Col. (1) and col. (2), which show the two most demanding specifications (corresponding to those in col. (3) and col. (4) of Table 2), show that the results are virtually unchanged, both in terms of the magnitude and significance of the estimated DiD coefficient, suggesting that there is no major bias associated with the use of different control groups.

Second, we conduct a *placebo* test with the aim of testing whether, by considering a group of products *not* targeted by EPR regulations on batteries, one still finds a significant effect of the policy. Obviously, should this be the case, one would conclude for a misspecification of the research design. To identify the 'fake' treated group, we use a sub-list of HS6 products classified by Kellenberg and Levinson (2014) as hazardous waste ($\text{HW}_p$) under the Basel Convention, and exclude the HS6 code referring to WB from the list. The 13 HS6 codes defined in this way are described in Table B1in Appendix. Col. (3) and col. (4) of Table 3, report a non-significant impact of the EPR policy on the 'fake' treated group. This fact reassures us about the role of EPR in affecting the export of WB, instead of simultaneous and different events, which

import countries only as a control factor. Nonetheless, our results, hinting to a positive correlation between imports of WB and the country's choice to introduce EPR, open room for further investigations on this other relationship

[27] The difference in the number of products considered as waste is due to Kellenberg and Levinson (2014) using the 2002 HS definition. Instead, we convert their codes to the 1996 HS classification using the conversion tables provided by UNCTAD.

may have affected a similar (in terms of hazardousness) set of waste products. The sub-list of hazardous waste products identified by Kellenberg and Levinson (2014) may be nonetheless a finer control group than the entire list of waste commodities as control group. In this respect column (5) shows the results when only hazardous waste products, as previously defined, are considered as control group. The main result is confirmed.

We then verify the robustness of our main results by reducing one dimension of variation in the main independent variable at a time. For this purpose, it is useful to recall that, as explained in Section 4.1.2, in the DDD model we have exploited a triple source of variation, since we have both exporting countries that impose the policy and those that do not, products that are treated and those that are not, and two time periods, namely before and after the implementation of the policy. First, in col. (6) of Table 3, we select only WB and compare the exports of countries that implement the policy (treated group) with those that do not implement the EPR regulation (control group). Second, in col. (7) of the same table, we keep all waste products, but select only those countries that adopt the policy (sooner or later). All estimates confirm our main results.

Finally, we explicitly account for the staggered nature of the EPR policy. Staggered adoptions do not pose a problem for estimating the average treatment effect if the effects are homogeneous across countries and time periods (Baker et al., 2022) (i.e., no dynamic changes in the effects of treatment). When this is not the case, the resulting staggered DiD estimates are likely to be biased (see Athey and Imbens, 2022; Callaway and Sant'Anna, 2021; Goodman-Bacon, 2021; de Chaisemartin and D'Haultfœuille, 2020, among others). Moreover, according to recent research, these biases are not eliminated by implementing an event study estimator (Sun and Abraham, 2021).[28] Since we cannot exclude that the treatment effects of EPR are dynamic, we try to mitigate the sources of these possible biases by restricting the sample. Indeed, although the econometric literature has not agreed on a standard alternative approach, all the different solutions show that the presence of always-treated units exacerbates the ATT bias, while the presence of never-treated units mitigates it. To this end, we rely on two main considerations. First, given that the vast majority of EU countries implemented the waste battery directive between 2008 and 2010, we limit the time span of our regressions to 2004 to 2014 to have sufficient pre- and post-treatment periods, while excluding observations that are too far from the policy adoption. Second, we drop observations referring to (non-EU) countries that adopted EPR before 2005. This procedure allows to remove 'always treated' countries from the estimation. In addition, countries that adopt EPR after 2014 are treated as 'never treated.' Given these sample restrictions, the majority of countries, and especially the largest economies, adopt EPR within a period of only three years, potentially mitigating the risk of bias due to the possible dynamic nature of treatment effects. The results of this robustness check are reported in col. (8) of Table 3. Clearly the sample has shrunk in terms

[28] Sun and Abraham (2021) have shown that in the presence of staggered treatment timing and treatment effect heterogeneity, the dynamic effect estimates obtained by an event-study estimator may be contaminated by the causal effects of other relative time periods in the estimation sample, affecting the accuracy of the estimates.

of observations and, accordingly, the coefficient is less precisely estimated, but the sign of DiD coefficient is in line with the main results shown in Table 2. The magnitude of the impact of the policy remains large in this model, predicting an increase of the volume of WB exported after EPR implementation about 32% higher than that of other types of waste. [29]

Table 3: The effect of EPR policy on the exports of Waste Battery: robustness checks

| Dep. Var. ln Quantity$_{eipy}$ | Waste Products as in Kellenberg and Levinson (2014) | | | | | Only WB products | Only countries with EPR | Restricted sample |
|---|---|---|---|---|---|---|---|---|
| | | | Fake Treat. | Fake Treat. | HW$_p$ | | | |
| | (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) |
| EPR$_{ey}$ | | | | | | 0.525*** | | |
| | | | | | | (0.179) | | |
| EPR$_{ey}$ × WB$_p$ | 0.342*** | 0.703*** | | | 0.756*** | | 0.619*** | 0.274* |
| | (0.136) | (0.212) | | | (0.219) | | (0.145) | (0.143) |
| EPR$_{ey}$ × HW$_p$ | | | 0.095 | −0.099 | | | | |
| | | | (0.055) | (0.061) | | | | |
| $\omega_{ey}$ | Yes | Yes | Yes | Yes | Yes | No | Yes | Yes |
| $\omega_{iy}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{py}$ | No | Yes | No | Yes | Yes | No | No | No |
| Adj. $R^2$ | 0.709 | 0.713 | 0.710 | 0.714 | 0.760 | 0.618 | 0.737 | 0.787 |
| No. of Obs | 879,862 | 879,851 | 862,813 | 862,802 | 245,287 | 15,756 | 1,207,329 | 603,707 |

*Notes*: Observations are at the exporter-importer-product-year level. HW$_p$ is a dummy that takes value 1 for products defined as hazardous waste by Kellenberg and Levinson (2014), excluding the HS6 code '854810', i.e. WB. The coefficients appear together with standard errors clustered at the country-pair-product and year level *** significant at the 1% level, ** significant at the 5% level and * significant at the 10% level.

A final methodological concern is related to the use of an appropriate estimation strategy that takes into account the large numbers of zero trade flows. The gravity model (Anderson and van Wincoop, 2003), which Eq. 1 is based on, expresses trade as the multiple of strictly positive variables, and it does not take into account the information contained in the zero trade flows because these observations are simply dropped from the estimation sample when the value of trade is transformed into a logarithmic form. Thus, this specification focuses on explaining changes in the quantity traded in the product under consideration, i.e. the intensive margin of trade. However, there are significant portions of zero values, which become even more prevalent as the data becomes more finely disaggregated at the product level. The presence of trade flows with a bilateral value of zero carries significant implications for the gravity equation since it may indicate a selection issue. If these zero entries arise from countries choosing not to sell specific products to specific markets or being unable to do so, the standard ordinary least squares estimation (OLS) of Eq. 1 would be inappropriate and yield biased results. Following Santos Silva and Tenreyro (2006), we also estimate the model using the Poisson pseudo maximum likelihood (PPML) estimator. This approach considers the heteroscedasticity in trade data and makes use of the information available in zero trade flows.

Table 4 present the estimates obtained with the PPML estimator. The first column provides our baseline DiD coefficient, which, once again, is positive and significant. This result reassures us regarding our main results

[29] To our knowledge, at the time of writing staggered DiD estimation strategies have not yet been integrated in gravity approaches, given the specificity and variety of fixed effects typically included in gravity models. Nonetheless, we performed a preliminary attempt to estimate our DiD coefficient exploiting the method proposed by Callaway and Sant'Anna (2021) and not-yet-treated countries as control group. The results, available on request, show non-significant average pre-treatment trends and significant coefficients at the 1% level for at least four post-treatment periods, averaging on +0.6.

not being driven by a selection issue, possibly due to unobserved choices/possibility of exporter countries not to export specific products to specific markets. PPML is also used for two further robustness check chosen among the ones presented in Table 3 and explained above.

Table 4: The effect of EPR policy on the exports of Waste Battery: robustness checks - PPML

| Dep. Var. Quantity$_{eipy}$ | Baseline | Waste Products as in Kellenberg and Levinson (2014) | Only countries with EPR |
|---|---|---|---|
| | (1) | (2) | (3) |
| $\text{EPR}_{ey} \times \text{WB}_p$ | 0.392** | 0.349** | 0.380** |
| | (0.195) | (0.195) | (0.156) |
| $\omega_{ey}$ | Yes | Yes | Yes |
| $\omega_{iy}$ | Yes | Yes | Yes |
| $\omega_{eip}$ | Yes | Yes | Yes |
| No. of Obs | 6,064,484 | 3,167,121 | 4,460,999 |

*Notes*: Observations are at the exporter-importer-product-year level. The coefficients appear together with standard errors clustered at the country-pair-product and year level *** significant at the 1% level, ** significant at the 5% level and * significant at the 10% level.

### 4.2.3 Possible mechanisms

Sections 4.2.1 and 4.2.2 have shown that the adoption of the EPR policy by the exporting country leads to an increase in the export volume of WB with respect to other waste products not targeted by this specific EPR policy. A series of ad hoc robustness checks then confirmed this finding. Clearly, trade data do not allow for a deeper and more precise identification of the mechanisms responsible for the identified phenomenon, but it is possible to explore the possible channels that could be associated with such a change or that are likely to be excluded.

Differences in environmental regulation across countries may constitute a source of comparative advantage, i.e. lower costs, for countries with lower levels of regulation in terms of attracting flows of waste (Kellenberg, 2015). The literature refers to this phenomenon as “waste haven hypothesis”. We test this possibility rely, first, on indirect measures environmental policy stringency base on countries classifications and, subsequently, on direct indicators.

To begin, we include in the model a double interaction term that is the dummy for EPR on WB adoption relative to the exporting country multiplied by the WB dummy and an indicator variable that takes value equal to 1 when the importing country is a developed country ($\text{D}_{iy}^{Dev}$), i.e. it ranks in the top half distribution of countries by GDP per capita.[30] As shown in col. (1) of Table 5, the coefficient of the double interaction is positive and significant: the increase in WB export volumes after EPR adoption has been higher towards developed economies rather than developing ones. Accordingly, a waste-haven type effect can be ruled out by the possible mechanisms. Along the same line of the exercise performed in Col (1), we also add the double interaction of $\text{EPR}_{ey} \times \text{WB}_p$ with a dummy equal to 1 when the importing country belongs to EU-28. Results in Col. (2) show that the coefficient associated with the double interaction is positive, confirming

[30] To test a possible waste haven effect, we follow here Baggs (2009) and the geography framed by the Basel Convention, contrasting developed (Annex VII) and developing countries.

that the export flow of WB has increased more towards EU countries, and that this result is robust to controls for omitted variable bias, reverse causality, and products' time trends.

These two results are in line with our expectations in consideration of a few facts, partially discussed by Theis (2021) also. First, exports of hazardous waste are generally forbidden by enforcing international environmental agreements, like the "Basel Convention's Ban Amendment".[31] Second, the EU Waste Shipment Regulation (2006) prohibits exports from the EU to non-EU countries of waste for disposal. In parallel, it can be noted that the majority of EPR adopters are EU countries. Actually, as also shown in Figure 3 Panel D, not only the EU is basically self-sufficient in terms of WB and, in general, hazardous waste treatment, but it is also a net importer of these types of waste (European Commission, 2015; Giosuè et al., 2021). The results of col. (1) and (2) can be interpreted in the light of Kellenberg (2015) and Falkowska (2018) argumentations. Larger economies have more advanced recycling programs, waste management markets, and technologies to recover materials even from hazardous waste. In fact, not all hazardous waste is intended for disposal. Waste containing lead, for example, is considered hazardous under the Basel Convention, yet lead is a highly recyclable waste product. It turns out indeed that lead-acid batteries, which are since long the most common type of battery on the market, are economically recycled (and manufactured) in Europe (European Commission, 2019a)[32]. Thus, larger economies may have greater demand for recyclable wastes, despite their stricter environmental regulations when comparing to developing countries.

To test further the occurrence of a waste haven phenomenon, we revert now to two direct measures of environmental policy. The Environmental Policy Stringency index (EPS) developed by the OECD (Kruse et al., 2022) has become a widely used tool for policy analysis, covering three decades from 1990 to 2020, and 40 countries[33]. As a composite index (Brunel and Levinson, 2016), OECD EPS compresses the multidimensionality of environmental regulations (in this case, 13 policy instruments focusing on climate change and air pollution) in a single indicator. Instead, the waste management indicator (WMG) provided within the Environmental Performance Index (Wendling et al., 2020) specifically represents a proxy for the effectiveness of waste management policies in a country[34]. Columns (3) and (4) in Table 5 show positive and significant coefficients for the double interactions $\text{EPR}_{ey} \times \text{WB}_p \times \text{EPS}_{iy}$ and $\text{EPR}_{ey} \times \text{WB}_p \times \text{WMG-EPI}_i$ respectively. Hence, we do not find evidence of a waste haven effect. Rather, WB exports after EPR adoption appear to be mainly directed towards countries with relatively strict environmental and waste management regulations, reinforcing the thesis that these countries retain a comparative advantage in WB treatment thanks to their

[31] Although the Basel Convention is currently ratified by 191 countries, only 103 have ratified the Ban Amendment. These include the EU, but not, for example, the US, Canada, Japan, South Korea, India, Australia and Russia.

[32] The situation is different for lithium-ion batteries (LIB). Currently, China accounts for 73% of global recycling capacity of LIB (Moïsé and Rubínová, 2023), but this is expected to decline to about 50% already around 2025. Indeed, as battery production capacity expands outside China, more recycling capacity is also being built elsewhere, driven by foreign direct investment in Europe and North America. The available LIB waste stock is still too limited to make LIB recycling profitable.

[33] The indicator ranges from 0 to 10. See Kruse et al. (2022) for the list of countries for which EPS is available.

[34] This indicator ranges from 0 to 100 and it was first provided in the 2020 version of the EPI. More information on the indicator and the list of the 181 covered countries see Wendling et al. (2020)

more developed waste management systems and technologies.

Table 5: The effect of EPR policy on the exports of Waste Battery: possible mechanisms.

| Dep. Var. | ln Quantity$_{eipy}$ | | | | | |
|---|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) | (5) | (6) |
| EPR$_{ey}$ × WB$_p$ | 0.168 | 0.246 | -0.226 | -0.195 | 0.703** | 0.191 |
| | (0.271) | (0.205) | (0.369) | (0.296) | (0.251) | (0.297) |
| EPR$_{ey}$ × WB$_p$ × D$_{iy}^{Dev}$ | 0.593** | | | | | |
| | (0.267) | | | | | |
| EPR$_{ey}$ × WB$_p$ × D$_i^{EU28}$ | | 0.770*** | | | | |
| | | (0.213) | | | | |
| EPR$_{ey}$ × WB$_p$ × EPS$_{iy}$ | | | 0.299*** | | | |
| | | | (0.138) | | | |
| EPR$_{ey}$ × WB$_p$ × WMG-EPI$_i$ | | | | 0.012*** | | |
| | | | | (0.003) | | |
| EPR$_{ey}$ × WB$_p$ × Gradient Patent$_{iy}$ | | | | | 0.219*** | |
| | | | | | (0.070) | |
| EPR$_{ey}$ × WB$_p$ × Gradient Facilities$_{iy}$ | | | | | | 0.264** |
| | | | | | | (0.119) |
| $\omega_{ey}$ | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{iy}$ | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{py}$ | Yes | Yes | Yes | Yes | Yes | Yes |
| Adj. $R^2$ | 0.740 | 0.740 | 0.743 | 0.738 | 0.743 | 0.757 |
| No. of Obs | 1,568,914 | 1,568,959 | 867,849 | 1,510,714 | 1,001,259 | 458,745 |

*Notes*: Observations are at the exporter-importer-product-year level. Note that all regressions include but do not report all the double interactions. The coefficients appear together with standard errors clustered at the country-pair-product and year level *** significant at the 1% level, ** significant at the 5% level and * significant at the 10% level.

Since, as anticipated, the location of treatment facilities and the availability of technology might be so crucial in determining the direction of WB flows, we investigate the role of these factors in conjunction with EPR adoption. We use two gradients as proxies for capturing differences in technological endowments between the importing ($i$) and exporting ($e$) country. First, we calculate the gradient in the share of patents that are specific to batteries recycling to the total number of patents related to recycling. These data are based on the OECD ENV-TECH classification (Haščič and Migotto, 2015), which is one of the most commonly used methodologies to identify green patents on the basis of their IPC and CPC codes (Bianchini et al., 2023; Favot et al., 2023). The share in consideration is represented by the ratio between the count of patents with at least one IPC or CPC code related to batteries recycling over the total number of patents with at least one IPC or CPC code related to recycling (excluding WB recycling). This share is calculated at a country-year level. Patent data with global coverage were taken from PATSTAT and they cover the full period of our trade data. More details on the procedure of collection of the data and on the IPC/CPC codes considered in this elaboration are provided in Priore et al. (2023). Second, we calculate the gradient in the number of general (i.e., not specific to WB) recycling facilities[35]. This figure is taken from Eurostat, so it covers European

[35] To the best of our knowledge, data on facilities specialising in battery recycling, covering several countries are not

countries only. To extend the series for recycling facilities, we calculate the mean number of facilities by country in the period 2010-2020, which is the information provided by Eurostat, and we impute it to the whole period covered by our trade data.

Following Kellenberg (2012) and Marin et al. (2017), gradients are calculated using the midpoints formula as:

$$E_{iey} = \frac{E_{iy} - E_{ey}}{\frac{E_{iy}+E_{ey}}{2}}. \tag{3}$$

Col. (3) and col. (4) show some interesting results. Indeed the increase in export of WB has been higher towards importing countries with a comparative advantage in terms of patent share for recycling batteries, and countries with a higher amount of general recycling facilities with respect to the exporting country. These results suggest that differences in the technological endowment in the domain of (WB) recycling is a relevant driver of the increase in trade in WB after the adoption of the EPR policy by the exporting country.

Table 6: The effect of EPR policy on the exports of Waste Battery: materials.

| Dep. Var. | lnQuantity$_{eipy}$ | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) |
| EPR$_{ey}$ × WB$_p$ | 0.726*** | 0.724*** | 0.744*** | 0.774*** | 0.708*** | 0.777*** | 0.782*** | 0.778*** |
| | (0.212) | (0.211) | (0.212) | (0.208) | (0.213) | (0.224) | (0.216) | (0.214) |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Antimony}$ | −0.030*** | | | | | | | |
| | (0.007) | | | | | | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Cobalto}$ | | −0.104* | | | | | | |
| | | (0.060) | | | | | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Graphite}$ | | | −0.034*** | | | | | |
| | | | (0.0.008) | | | | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Lead}$ | | | | −0.059*** | | | | |
| | | | | (0.015) | | | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Lithium}$ | | | | | −0.026*** | | | |
| | | | | | (0.008) | | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Manganese}$ | | | | | | −0.119** | | |
| | | | | | | (0.0.025) | | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Nickel}$ | | | | | | | −0.119*** | |
| | | | | | | | (0.039) | |
| EPR$_{ey}$ × WB$_p$ × Avg.Share$_i^{Zinc}$ | | | | | | | | −0.069*** |
| | | | | | | | | (0.021) |
| $\omega_{ey}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{iy}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| $\omega_{py}$ | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| Adj. $R^2$ | 0.740 | 0.740 | 0.740 | 0.740 | 0.740 | 0.740 | 0.740 | 0.740 |
| No. of Obs | 1,568,959 | 1,568,959 | 1,568,959 | 1,568,959 | 1,568,959 | 1,568,959 | 1,568,959 | 1,568,959 |

*Notes*: Observations are at the exporter-importer-product-year level. Note that all regressions include but do not report all the double interactions. The coefficients appear together with standard errors clustered at the country-pair-product and year level *** significant at the 1% level, ** significant at the 5% level and * significant at the 10% level.

We conclude the analysis of the mechanisms driving the direction of WB flows, following EPR adoption, by considering the possibility that WB are imported for recovering material resources of which the importing country is poor. Indeed, imports of waste could substitute those of virgin raw materials (Dussaux and Glachant, 2019). To explore this possible determinant of WB trade flows, as a first step we identified the most common types of batteries placed on the market in the last two decades and their material basis.

available.

Portable batteries are typically based on zinc and manganese, or on nickel (Stahl et al., 2019; Sayilgan et al., 2009); lead-acid are the most common battery chemistry in the automotive and industrial applications (Stahl et al., 2019); lithium-ion batteries, which often contain significant quantities of cobalt, are used in electric vehicles and as portable batteries (Alves Dias et al., 2018); antimony and graphite are also among the most common materials in batteries (Huisman et al., 2017). Subsequently, we collected information from the United States Geological Survey on countries' global share of production of the identified batteries minerals, in order to distinguish the countries which are abundant in those materials from those which are not. Table 6 show negative and significant coefficients for the interaction between our DiD coefficient and counties' share of global supply for a certain mineral[36]. This result indicates that, following EPR adoption in an exporting country, WB flows are generally directed towards countries which are not significant producers of batteries minerals. This might hint to the fact that WB are imported for the recovery of materials of which a country is not endowed or which are not being extracted. This interesting finding pairs with the observation that batteries minerals are mostly extracted in developing countries, while, as previously discussed, WB are mainly directed towards developed ones.

# 5 Conclusions, policy recommendations and future research

An effective global transition to a circular economy requires a better understanding of the potential interactions between international trade and waste policies to ensure that these two channels are mutually supportive (Yamaguchi, 2018, 2022). This transition aims to ease pressure on resource extraction, reduce supply risks and price volatility, while ensuring sound environmental management of waste flows.

In this context, this paper represents a first attempt to empirically investigate the impact of EPR legislation in general, and EPR on waste batteries in particular, on the exports of the affected waste flows. Our focus on WB is explained by both the hazardous nature of this waste and its high concentration of critical raw materials. To this end, we provide the most comprehensive review of EPR regulations on WB adopted worldwide currently available, and we discuss the possible indirect channels through which EPR may affect trade, through the physical, informative and economic responsibilities typically imposed on producers.

We outline how, in theory, EPR could affect exports in both directions, either boosting WB trade flows or, on the contrary, reducing them. It is then crucial to resort to empirical work. In this respect, our results, based on difference-in-difference models in a gravity setting, show that countries implementing EPR experienced an increase in the volume of WB exported, compared to the trend for other types of waste. It would then appear that the impact of channels supporting exports - such as increases in WB collection rates, more accurate tracking of WB trade flows, and the specialisation of countries in WB recycling and disposal - tend to outweigh those channels working in the opposite direction.

[36] The share of mineral supply is calculated as a country's average for the years 2001, 2010 and 2019.

Regarding the direction of WB exports in response to EPR implementation, our analysis integrates the literature emphasising the importance of technological endowments and economic structure in (hazardous) waste trade, beyond the level of environmental policy stringency (Kellenberg, 2015; Latorre et al., 2021; Lepawsky, 2015; Yang, 2020). In fact, as WB exports after EPR implementation are mainly directed to developed countries and to countries with relatively stringent environmental regulations, the policy does not seem to have promoted a waste haven effect. Rather than loosen environmental regulations, the level of sophistication of the waste management system, both in terms of patents and facilities, seems to provide a stronger comparative advantage in attracting WB flows. The EU, for instance, is not only basically self-sufficient in WB management, but also a net importer.

Lastly, countries' availability of the mineral resources typically contained in batteries was tested as a factor for WB imports. Following EPR adoption, WB flows are mainly attracted by countries which are not the major producers of batteries minerals. Indeed, these resources are mostly mined in developing countries. This finding hints to the fact that WB might be imported for the recovery of materials, possibly in substitution of virgin materials imports.

These results contribute to the scarce research on the management of WB and the impact of related policies, which is fundamental for evaluating and planning circular economy strategies and investments in this strategic sector. In conclusion, our paper presents the idea that EPR can indirectly support "waste marketization strategies" (Kama, 2015; Kronenberg and Winkler, 2009), i.e. to create the normative and economic conditions to turn waste into a resource, in the spirit of circular economy.

Finally, we also acknowledge some limitations of our work, which also helps to identify possible directions for future research on the relationship between EPR and trade. First, as we have pointed out, countries have implemented the EPR principle using different regulatory instruments. Future research could build on this and examine the varying effectiveness of different policies implementing the general principle of EPR and their impact on trade flows. Second, in order to maintain a homogeneous focus, we have limited the perspective throughout the paper to that of the exporting country. Clearly, an examination of the effects of the adoption of EPR on the import side would further enhance our understanding of the phenomenon. Third, this paper has only marginally addressed the analysis of the relationship between EPR on WB and innovation, and this is clearly an area where technical change and intellectual properties could play a strategic role. Fourth, the hypothesis of a waste haven effect fostered by the introduction of EPR in exporting countries could be further explored with the support of data on illegal waste exports.

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# Appendix A

*Table A1 UNU-keys with descriptions and classification within the Electronic waste (WEEE) Directive 2012/19/EU (Annex III). Source: Magalini et al. (2015).*

| UNU-key | Description | WEEE category |
|---|---|---|
| 0001 | Central Heating (household installed) | 4 |
| 0002 | Photovoltaic Panels (incl. inverters) | 4 |
| 0101 | Professional Heating & Ventilation (excl. cooling equipment) | 4 |
| 0102 | Dishwashers | 4 |
| 0103 | Kitchen (e.g. large furnaces, ovens, cooking equipment) | 4 |
| 0104 | Washing Machines (incl. combined dryers) | 4 |
| 0105 | Dryers (wash dryers, centrifuges) | 4 |
| 0106 | Household Heating & Ventilation (e.g. hoods, ventilators, space heaters) | 4 |
| 0108 | Fridges (incl. combi-fridges) | 1 |
| 0109 | Freezers | 1 |
| 0111 | Air Conditioners (household installed and portable) | 1 |
| 0112 | Other Cooling (e.g. dehumidifiers, heat pump dryers) | 1 |
| 0113 | Professional Cooling (e.g. large air conditioners, cooling displays) | 1 |
| 0114 | Microwaves (incl. combined, excl. grills) | 5 |
| 0201 | Other Small Household (e.g. small ventilators, irons, clocks, adapters) | 5 |
| 0202 | Food (e.g. toaster, grills, food processing, frying pans) | 5 |
| 0203 | Hot Water (e.g. coffee, tea, water cookers) | 5 |
| 0204 | Vacuum Cleaners (excl. professional) | 5 |
| 0205 | Personal Care (e.g. tooth brushes, hair dryers, razors) | 5 |
| 0301 | Small IT (e.g. routers, mice, keyboards, external drives & accessories) | 6 |
| 0302 | Desktop PCs (excl. monitors, accessoires) | 6 |
| 0303 | Laptops (incl. tablets) | 2 |
| 0304 | Printers (e.g. scanners, multi functionals, faxes) | 6 |
| 0305 | Telecom (e.g. (cordless) phones, answering machines) | 6 |
| 0306 | Mobile Phones (incl. smartphones, pagers) | 6 |
| 0307 | Professional IT (e.g. servers, routers, data storage, copiers) | 4 |
| 0308 | Cathode Ray Tube Monitors | 2 |
| 0309 | Flat Display Panel Monitors (LCD, LED) | 2 |
| 0401 | Small Consumer Electronics (e.g. headphones, remote controls) | 5 |
| 0402 | Portable Audio & Video (e.g. MP3, e-readers, car navigation) | 5 |
| 0403 | Music Instruments, Radio, Hi-Fi (incl. audio sets) | 5 |
| 0404 | Video (e.g. Video recorders, DVD, Blue Ray, set-top boxes) | 5 |
| 0405 | Speakers | 5 |
| 0406 | Cameras (e.g. camcorders, photo & digital still cameras) | 5 |
| 0407 | Cathode Ray Tube TVs | 2 |
| 0408 | Flat Display Panel TVs (LCD, LED, Plasma) | 2 |
| 0501 | Lamps (e.g. pocket, Christmas, excl. LED & incandescent) | 3 |
| 0502 | Compact Fluorescent Lamps (incl. retrofit & non-retrofit) | 3 |
| 0503 | Straight Tube Fluorescent Lamps | 3 |

| | | |
|---|---|---|
| 0504 | Special Lamps (e.g. professional mercury, high & low pressure sodium) | 3 |
| 0505 | LED Lamps (incl. retrofit LED lamps & household LED luminaires) | 3 |
| 0506 | Household Luminaires (incl. household incandescent fittings) | 5 |
| 0507 | Professional Luminaires (offices, public space, industry) | 5 |
| 0601 | Household Tools (e.g. drills, saws, high pressure cleaners, lawn mowers) | 5 |
| 0602 | Professional Tools (e.g. for welding, soldering, milling) | 4 |
| 0701 | Toys (e.g. car racing sets, electric trains, music toys, biking computers) | 5 |
| 0702 | Game Consoles | 6 |
| 0703 | Leisure (e.g. large exercise, sports equipment) | 4 |
| 0801 | Household Medical (e.g. thermometers, blood pressure meters) | 5 |
| 0802 | Professional Medical (e.g. hospital, dentist, diagnostics) | 4 |
| 0901 | Household Monitoring & Control (alarm, heat, smoke, excl. screens) | 5 |
| 0902 | Professional Monitoring & Control (e.g. laboratory, control panels) | 4 |
| 1001 | Non-Cooled Dispensers (e.g. for vending, hot drinks, tickets, money) | 4 |
| 1002 | Cooled Dispensers (e.g. for vending, cold drinks) | 1 |

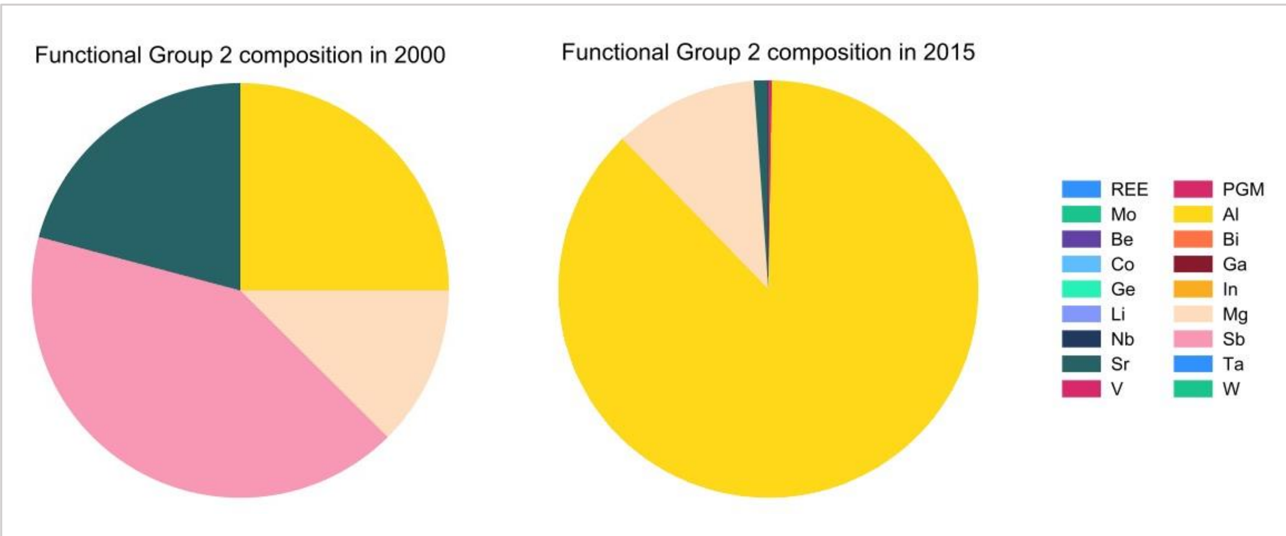


*Figure A1 Percentage composition of FG 2 by chemical element in 2000, left, and 2015, right. Note on the four elements with highest concentration: Sb Antimony, Sr Strontium, Al Aluminium, Mg Magnesium.*

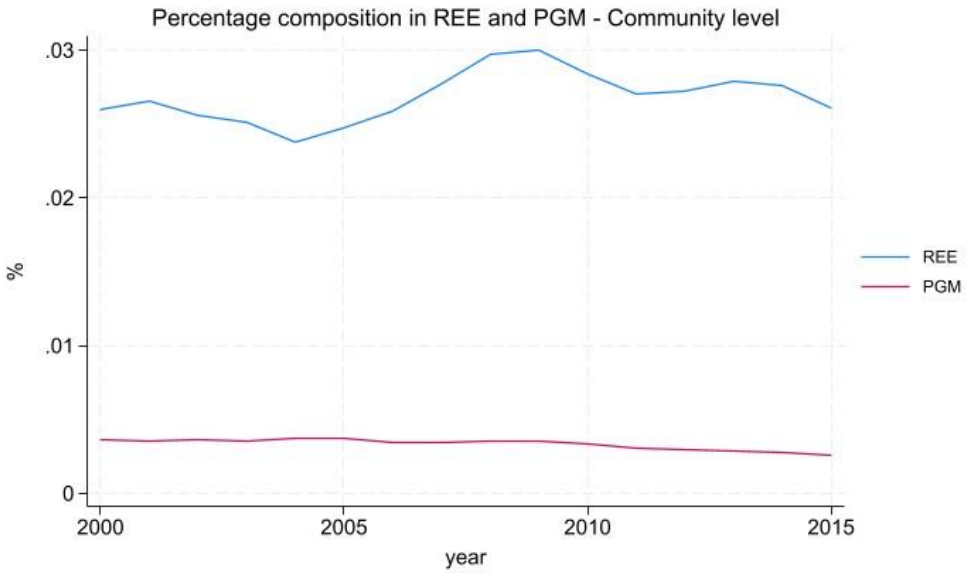


*Figure A2 Share of rare earths and platinum group metals over the total mass of materials at the community level.*

*Table A2 technological fields related to recycling and respective IPC and CPC codes according to OECD ENVTECH (2022 update).*

| **1. ENVIRONMENTAL MANAGEMENT**<br>**1.3 WASTE MANAGEMENT**<br>**1.3.2. Material recovery, recycling and re-use** | |
|---|---|
| Animal feeding-stuffs from waste material such as feathers, bones or skin; waste dairy products; hydrolysates of wood or straw; molasses; distillers' or brewers' waste | A23K10/26-28<br>A23K10/32-33<br>A23K10/37-38 |
| Footwear made of rubber waste | A43B1/12 |
| Separating solid materials; General arrangement of separating plant specially adapted for refuse | B03B9/06 |
| Manufacture of articles from scrap or waste metal particles | B22F8 |
| Preparing material; Recycling the material | B29B7/66 |
| Recovery of plastics or other constituents of waste material containing plastics | B29B17 |
| Presses specially adapted for consolidating scrap metal or for compacting used cars | B30B9/32 |
| Systematic disassembly of vehicles for recovery of salvageable components, e.g. for recycling | B62D67 |
| Stripping waste material from cores or formers of thin or filamentary material, e.g. to permit their re-use | B65H73 |
| Applications of disintegrable, dissolvable or edible materials. Packaging material. | B65D65/46 |
| Compacting the glass batches, e.g. pelletizing | C03B1/02 |
| Hydraulic cements from oil shales, residues or waste other than slag | C04B7/24-30 |

| | |
|---|---|
| Calcium sulfate cements starting from phosphogypsum or from waste, e.g. purification products of smoke | C04B11/26 |
| Use of agglomerated or waste materials or refuse as fillers for mortars, concrete or artificial stone; Waste materials or Refuse | C04B18/04-305 |
| Clay-wares; Waste materials or Refuse | C04B33/132 |
| Recovery or working-up of waste materials (plastics) | C08J11 |
| Luminescent, e.g. electroluminescent, chemiluminescent, materials; Recovery of luminescent materials | C09K11/01 |
| Working-up used lubricants to recover useful products | C10M175 |
| Working-up raw materials other than ores, e.g. scrap, to produce non-ferrous metals or compounds thereof | C22B7 |
| Obtaining zinc or zinc oxide; From muffle furnace residues; From metallic residues or scraps | C22B19/28-30 |
| Obtaining tin; From scrap, especially tin scrap | C22B25/06 |
| Textiles; Disintegrating fibre-containing articles to obtain fibres for re-use | D01G11 |
| Paper-making; Fibrous raw materials or their mechanical treatment - using waste paper | D21B1/08-10 |
| Paper-making; Fibrous raw materials or their mechanical treatment; Defibrating by other means - of waste paper | D21B1/32 |
| Paper-making; Other processes for obtaining cellulose; Working-up waste paper | D21C5/02 |
| Paper-making; Pulping; Non-fibrous material added to the pulp; Waste products | D21H17/01 |
| Apparatus or processes for salvaging material from electric cables | H01B 15/00 |
| Recovery of material from discharge tubes or lamps | H01J 9/52 |
| ***Reclaiming serviceable parts of waste cells or batteries*** | ***H01M 6/52*** |
| ***Reclaiming serviceable parts of waste accumulators*** | ***H01M 10/54*** |
| **6. CLIMATE CHANGE MITIGATION technologies related to WASTEWATER TREATMENT OR WASTE MANAGEMENT**<br><br>**6.2 SOLID WASTE MANAGEMENT**<br><br>**6.2.5 Reuse, recycling or recovery technologies** | **Y02W 30/50-91** |
| 6.2.5.1. Mechanical processing of waste for the recovery of materials, e.g. crushing, shredding, separation or disassembly | Y02W30/52 |
| 6.2.5.2. Waste management of vehicles | Y02W30/56 |
| 6.2.5.3. Construction or demolition [C&D] waste | Y02W30/58 |
| 6.2.5.4. Glass recycling | Y02W30/60 |
| 6.2.5.5. Plastics and rubber recycling | Y02W30/62 |
| 6.2.5.6. Paper recycling | Y02W30/64 |

| | |
|---|---|
| 6.2.5.7. Disintegrating fibre-containing textile articles to obtain fibres for re-use | Y02W30/66 |
| 6.2.5.8. Recovery of fats, fatty oils, fatty acids or other fatty substances, e.g. lanolin or waxes | Y02W30/74 |
| 6.2.5.9. Recycling of wood or furniture waste | Y02W30/78 |
| 6.2.5.10. Packaging reuse or recycling, e.g. of multilayer packaging | Y02W30/80 |
| **6.2.5.11. Recycling of waste of electrical or electronic equipment [WEEE]** | **Y02W30/82** |
| ***6.2.5.12. Recycling of batteries or fuel cells*** | ***Y02W30/84*** |
| 6.2.5.13. Use of waste materials as fillers for mortars or concrete | Y02W30/91 |

# Appendix B

## B.1 EU-28 imports of WB by country

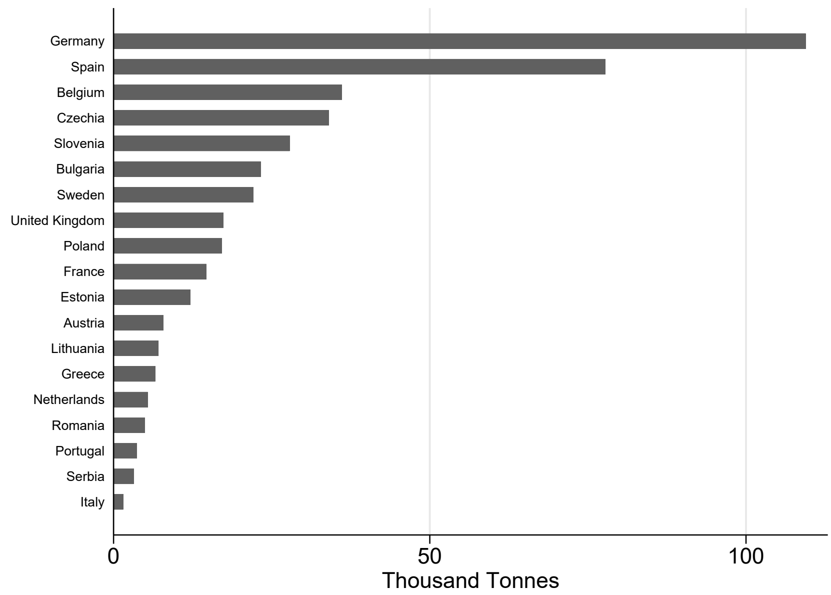


Figure B1: Imports quantity of waste batteries by EU-28 country, 2019.

## B.2 Waste products

Table B1 lists the 1996 HS-6 codes considered in our analysis. In this Table,"Hazardous" identifies the HS included in the set of hazardous waste in this paper. Importantly, our set of HS for hazardous waste must not be considered as an exhaustive list. More precisely, all of the HS that we classified as hazardous waste are certainly hazardous types of waste, but further HS may also identify hazardous waste, according to other criteria. Our list of hazardous HS is based on the work of Kellenberg and Levinson (2014), converted to the 1996 classification. There are no official concordances between (hazardous) waste classifications and HS codes.

Table B1: List and description of HS6 waste products

| HS1996 Code | Hazardous | description |
|---|---|---|
| 050210 | no | animal products; hair and bristles, of pigs, hogs or boars, and waste thereof |
| 050290 | no | animal products; badger hair and other brush making hair and waste of such bristles or hair, n.e.s. in heading no. 0502 (excluding horsehair) |
| 050300 | no | animal products; horsehair and horsehair waste, whether or not put up as a layer with or without supporting material |
| 050590 | no | animal products; skins and other parts of birds, feathers and down (not for stuffing), powder and waste of such, not further worked than cleaned, disinfected or treated for preservation |
| 050690 | no | animal products; bones and horn-cores and powder or waste of such, unworked, defatted, simply prepared (not cut to shape), or treated with acid or degelatinised, n.e.s. in heading no. 0506 |
| 050710 | no | animal products; ivory, unworked or simply prepared but not cut to shape, ivory powder and waste |
| 050790 | no | animal products; tortoise-shell, whalebone and whalebone hair, horns, antlers, hooves, nails, claws and beaks, unworked or simply prepared but not cut to shape, waste and powder of these products |
| 050800 | no | animal products; coral and similar materials, shells of molluscs, crustaceans, echinoderms, cuttle-bone etc., unworked or simply prepared (but not cut to shape), and powder and waste thereof |
| 152200 | no | degras; residues resulting from the treatment of fatty substances or animal or vegetable waxes |
| 180200 | no | cocoa; shells, husks, skins and other cocoa waste |
| 230210 | no | bran, sharps and other residues; of maize (corn), whether or not in the form of pellets, derived from the sifting, milling or other workings thereof |
| 230220 | no | bran, sharps and other residues; of rice, whether or not in the form of pellets, derived from the sifting, milling or other workings thereof |

| | | |
|---|---|---|
| 230230 | no | bran, sharps and other residues; of wheat, whether or not in the form of pellets, derived from the sifting, milling or other workings thereof |
| 230240 | no | bran, sharps and other residues; of other cereals, whether or not in the form of pellets, derived from the sifting, milling or other workings thereof |
| 230250 | no | bran, sharps and other residues; of leguminous plants, whether or not in the form of pellets, derived from the sifting, milling or other workings thereof |
| 230310 | no | residues of starch manufacture and similar residues; whether or not in the form of pellets |
| 230320 | no | beet-pulp, bagasse and other waste of sugar manufacture; whether or not in the form of pellets |
| 230330 | no | brewing or distilling dregs and waste; whether or not in the form of pellets |
| 230400 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of soya-bean oil |
| 230500 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of ground-nut oil |
| 230610 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of cotton seed oils |
| 230620 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of linseed oils |
| 230630 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of sunflower seed oils |
| 230640 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of rape or colza seed oils |
| 230650 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of coconut or copra seed oils |

| | | |
|---|---|---|
| 230660 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of palm nuts or kernels oils |
| 230670 | no | oil cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of maize (corn) germ oils |
| 230690 | no | oil-cake and other solid residues; whether or not ground or in the form of pellets, resulting from the extraction of oils, n.e.s. in heading no. 2306 |
| 230810 | no | vegetable materials and vegetable waste, vegetable residues and bi-products; whether or not in the form of pellets, of a kind used in animal feeding, acorns and horse-chestnuts |
| 230890 | no | vegetable materials and vegetable waste, vegetable residues and bi-products; whether or not in the form of pellets, of a kind used in animal feeding, other than acorns or horse-chestnuts |
| 251720 | no | macadam of slag, dross or similar industrial waste; whether or not incorporating the materials in item no. 2517.10 |
| 252530 | no | mica; waste |
| 261800 | no | slag, granulated (slag sand); from the manufacture or iron or steel |
| 261900 | no | slag, dross; (other than granulated slag), scalings and other waste from the manufacture of iron or steel |
| 262011 | yes | ash and residues; (not from the manufacture of iron or steel), containing mainly zinc, hard zinc spelter |
| 262019 | yes | ash and residues; (not from the manufacture of iron or steel), containing mainly zinc, other than hard zinc spelter |
| 262020 | yes | ash and residues; (not from the manufacture of iron or steel), containing mainly lead |
| 262030 | yes | ash and residues; (not from the manufacture of iron or steel), containing mainly copper |
| 262040 | no | ash and residues; (not from the manufacture of iron or steel), containing mainly aluminium |
| 262050 | no | ash and residues; (not from the manufacture of iron or steel), containing mainly vanadium |

| | | |
|---|---|---|
| 262090 | no | ash and residues; (not from the manufacture or iron or steel), containing mainly metals or metal compounds n.e.s. in heading no. 2620 |
| 262100 | yes | slag and ash; including seaweed ash (kelp), n.e.s. in chapter 26 |
| 271390 | no | residues; of petroleum oils or of oils obtained from bituminous minerals |
| 284440 | no | radioactive elements, isotopes, compounds, n.e.s. in heading no. 2844 alloys, dispersions (including cermets), ceramic products and mixtures containing these elements, isotopes or compounds; radioactive residues |
| 380400 | no | lyes, residual; from the manufacture of wood pulp, whether or not concentrated, desugared or chemically treated, including lignin sulphonates but excluding tall oil of heading no. 3803 |
| 382490 | yes | chemical products, preparations and residual products of the chemical or allied industries, n.e.s. or included in heading no. 3824 |
| 391510 | no | ethylene polymers; waste, parings and scrap |
| 391520 | no | styrene polymers; waste, parings and scrap |
| 391530 | no | vinyl chloride polymers; waste, parings and scrap |
| 391590 | no | plastics n.e.s. in heading no. 3915; waste, parings and scrap |
| 400400 | no | rubber; waste, parings and scrap of rubber (other than hard rubber) and powders and granules obtained therefrom |
| 401700 | no | rubber; ebonite and other hard rubbers in all forms, including waste and scrap, and articles of hard rubber |
| 411000 | no | leather or composition leather; parings and other waste (not suitable for the manufacture of leather articles), leather dust, powder and flour |
| 440130 | no | wood; sawdust, waste and scrap, whether or not agglomerated in logs, briquettes, pellets or similar forms |
| 450190 | no | cork; waste cork, crushed, granulated or ground cork |
| 470620 | no | pulp; of fibres derived from recovered (waste and scrap) paper or paperboard |
| 470710 | no | paper or paperboard; waste and scrap, of unbleached kraft paper or paperboard or of corrugated paper or paperboard |

| | | |
|---|---|---|
| 470720 | no | paper or paperboard; waste and scrap, of paper or paperboard made mainly of bleached chemical pulp, not coloured in the mass |
| 470730 | no | paper or paperboard; waste and scrap, of paper or paperboard made mainly of mechanical pulp (eg newspapers, journals and similar printed matter) |
| 470790 | no | paper or paperboard; waste and scrap, of paper or paperboard n.e.s. in heading no. 4707 and of unsorted waste and scrap |
| 500310 | no | silk; waste, not carded or combed (including cocoons unsuitable for reeling, yarn waste and garnetted stock) |
| 500390 | no | silk; waste, carded or combed (including cocoons unsuitable for reeling, yarn waste and garnetted stock) |
| 500500 | no | silk; yarn spun from silk waste, not put up for retail sale |
| 500600 | no | silk yarn and yarn spun from silk waste; put up for retail sale, and silk-worm gut |
| 500720 | no | silk; woven fabrics, containing 85% or more by weight of silk or of silk waste other than noil silk |
| 510310 | no | wool and hair; noils of wool or of fine animal hair, including yarn waste, but excluding garnetted stock |
| 510320 | no | wool and hair; waste of wool or of fine animal hair, including yarn waste, but excluding garnetted stock and noils of wool or of fine animal hair |
| 510330 | no | wool and hair; waste of coarse animal hair, including yarn waste, but excluding garnetted stock |
| 520210 | no | cotton; yarn waste (including thread waste) |
| 520291 | no | cotton; garnetted stock waste |
| 520299 | no | cotton; waste other than garnetted stock and yarn (including thread) waste |
| 530130 | no | flax; tow and waste, including yarn waste and garnetted stock |
| 530290 | no | hemp (cannabis sativa l.); processed (other than retted) (but not spun), true hemp tow and waste (including yarn waste and garnetted stock) |
| 530390 | no | jute and other textile bast fibres; processed but not spun, tow and waste of these fibres, including yarn waste and garnetted stock (excluding flax, hemp (cannabis sativa l.), and ramie) |

| | | |
|---|---|---|
| 530490 | no | sisal and other textile fibres of the genus agave; processed (but not spun), tow and waste of these fibres, including yarn waste and garnetted stock |
| 530519 | no | coconut (coir); processed (but not spun), tow, noils and waste, including yarn waste and garnetted stock |
| 530529 | no | abaca (manila hemp or musa textilis nee); processed but not spun, tow, noils and waste, including yarn waste and garnetted stock |
| 530599 | no | ramie and other vegetable textile fibres; n.e.s. in chapter 53, processed (but not spun); tow, noils and waste of these fibres, including yarn waste and garnetted stock |
| 550510 | no | fibres; waste (including noils, yarn waste and garnetted stock), of synthetic fibres |
| 550520 | no | fibres; waste (including noils, yarn waste and garnetted stock), of artificial fibres |
| 631010 | no | rags; used or new, scrap twine, cordage, rope and cables and worn out articles of twine, cordage, rope or cables, of textile materials; sorted |
| 631090 | no | rags; used or new, scrap twine, cordage, rope and cables and worn out articles of twine, cordage, rope or cables, of textile materials; other than sorted |
| 680800 | no | panels, boards, tiles, blocks and the like; of vegetable fibre, of straw, shavings, chips, particles, sawdust or other waste, of wood, agglomerated with cement, plaster or other mineral binders |
| 700100 | no | glass; cullet and other waste and scrap of glass, glass in the mass |
| 711210 | no | metals; waste and scrap of gold, including metal clad with gold but excluding sweepings containing other precious metals |
| 711220 | no | metals; waste and scrap of platinum, including metal clad with platinum but excluding sweepings containing other precious metals |
| 711290 | no | metals; waste and scrap of precious metal other than gold or platinum but excluding sweepings containing other precious metals |
| 720410 | no | ferrous waste and scrap; of cast iron |
| 720421 | no | ferrous waste and scrap; of stainless steel |
| 720429 | no | ferrous waste and scrap; of alloy steel (excluding stainless) |

| | | |
|---|---|---|
| 720430 | no | ferrous waste and scrap; of tinned iron or steel |
| 720441 | no | ferrous waste and scrap; turnings, shavings, chips, milling waste, sawdust, fillings, trimmings and stampings, whether or not in bundles |
| 720449 | no | ferrous waste and scrap; n.e.s. in heading no. 7204 |
| 740400 | no | copper; waste and scrap |
| 750300 | no | nickel; waste and scrap |
| 760200 | no | aluminium; waste and scrap |
| 780200 | yes | lead; waste and scrap |
| 790200 | no | zinc; waste and scrap |
| 800200 | no | tin; waste and scrap |
| 810191 | no | tungsten (wolfram); unwrought, including bars and rods obtained simply by sintering, waste and scrap |
| 810291 | no | molybdenum; unwrought, including bars and rods obtained simply by sintering, waste and scrap |
| 810310 | no | tantalum; unwrought, including bars and rods obtained simply by sintering, waste and scrap, powders |
| 810420 | no | magnesium; waste and scrap |
| 810510 | no | cobalt; mattes and other intermediate products of cobalt metallurgy, unwrought cobalt, waste and scrap, powders |
| 810710 | yes | cadmium; unwrought, waste and scrap, powders |
| 810810 | no | titanium; unwrought, waste and scrap |
| 810910 | no | zirconium; unwrought, waste and scrap, powders |
| 811000 | yes | antimony; articles thereof, including waste and scrap |
| 811211 | yes | beryllium; unwrought, waste and scrap, powders |
| 811220 | yes | chromium; including waste and scrap |
| 811230 | no | germanium; including waste and scrap |
| 811240 | no | vanadium; including waste and scrap |
| 811291 | yes | gallium, hafnium, indium, niobium (columbium), rhenium and thallium; articles thereof, unwrought, waste and scrap, powders |
| 854810 | yes | waste and scrap of primary cells, primary batteries and electric accumulators; spent primary cells, spent primary batteries and spent electric accumulators |

## B.3 EPR regulations on batteries

Table B2 reports information on worldwide EPR regulations on batteries. In this table, "Year" refers to the date of enforcement of the regulation in each country, when available; alternatively, the date of publication of the regulation was considered. For European countries, "Year" refers to the date of national transposition of EU Directive 2006/66/EC. Sub-national regulations where not taken in consideration. Countries for which "Year" is left blank have not introduced EPR on WB yet, according to the references reported. In column "Regulation", we report the name of the reference regulation, when available.

Table B2: Regulations on EPR on waste batteries.

| **Country** | **Year** | **Regulation(s)** | **Source** |
|---|---|---|---|
| Taiwan | 1998 | | Perchards and SagisEPR (2018) |
| Japan | 2001 | Law for Promotion of Utilization of Recyclable Resources | Perchards and SagisEPR (2022, 2018); Tasaki (2014) |
| Switzerland | 2001 | Swiss Battery Ordinance | Perchards and SagisEPR (2022, 2018) |
| China | 2003 | Waste Battery Pollution Control Policy | Perchards and SagisEPR (2018); Kim et al. (2018)[37] |
| Uruguay | 2003 | Regulation of the management and disposal of lead batteries (Decree 373/003); Regulation on lead contamination (Law No. 17775 of 2004) | Acosta and Corallo (2021). See also website of Uruguay Ministry for the environment.[38] |
| Turkey | 2004 | Batteries Regulation | Perchards and SagisEPR (2018) |
| Bulgaria | 2006[39] | Waste Batteries Ordinance | Perchards and SagisEPR (2022, 2018); Tsiarta et al. (2015) |
| Austria | 2008 | Waste Management Law (Abfallwirtschaftsgesetz) | Perchards and SagisEPR (2022, 2018) |
| Denmark | 2008 | Amendment Act 509 of 2008 | Perchards and SagisEPR (2022, 2018) |
| Estonia | 2008 | Waste Act (Jäätmeseadus) | Perchards and SagisEPR (2022, 2018) |
| Finland | 2008 | 2008 amendment to the Waste Act; Ordinance on Batteries | Perchards and SagisEPR (2022, 2018) |
| Hungary | 2008 | Government Decree 181 (take-back) of July 2008 | Perchards and SagisEPR (2022, 2018) |

[37] The regulation establishes that battery industries are responsible for collecting waste batteries and for proper labeling (Bird et al., 2022; Sun et al., 2021). Notice that the "Technology Policy for the Recycling of Power Battery (2015 edition)", providing provisions on the recycling and utilization of waste electric vehicles batteries, and the "Implementation Plan of the Extended Producer Responsibility System, setting recycling targets to achieve a recovery rate of 40% for major waste products (including waste lithium-ion batteries) by 2020 and 50% by 2025, were launched in 2016 (Bird et al., 2022; Sun et al., 2021).

[38] `https://www.gub.uy/ministerio-ambiente/institucional/normativa/decreto-373003-regulacion-del-manejo-disposicion-baterias-plomo`. Accessed 31 August 2023.

[39] The Regulation on the requirements for placing batteries and accumulators on the market and for treatment and transportation of waste batteries and accumulators, enforced in January 2006, is the main national law transposing the basic requirements of the Directive 2006/66/EC. Nonetheless, the above regulation was supplemented in 2008 for a full transposition of the Directive, achieving a full enforcement only in January 2009 (Perchards and SagisEPR, 2018; Tsiarta et al., 2015)

| | | | |
|---|---|---|---|
| Ireland | 2008 | Waste Batteries Regulations | Perchards and SagisEPR (2022, 2018) |
| Italy | 2008 | Decree 188/2008 | Perchards and SagisEPR (2022, 2018) |
| Lithuania | 2008 | 2008 amendment of Waste Act | Perchards and SagisEPR (2022, 2018) |
| Luxembourg | 2008 | Law on Batteries and Waste Batteries | Perchards and SagisEPR (2022, 2018) |
| Netherlands | 2008 | Batteries Regulation | Perchards and SagisEPR (2022, 2018) |
| South Korea | 2008 | Act on Resource Recirculation of Electrical and Electronic Waste and End of Life Vehicles | Perchards and SagisEPR (2018); Heo and Jung (2014); Kim (2010) |
| Spain | 2008 | Royal Decree 106/2008 | Perchards and SagisEPR (2022, 2018) |
| Sweden | 2008 | Batteries Ordinance 2008:834 | Perchards and SagisEPR (2022, 2018) |
| Cyprus | 2009[40] | Solid and Hazardous Waste Management (Batteries and Accumulators) Regulations 2009; 2012 amendment to 2011 Waste Act | Perchards and SagisEPR (2022, 2018) |
| France | 2009 | Decree 1139/2009 | Perchards and SagisEPR (2022, 2018) |
| Germany | 2009 | German Waste Batteries Act (Batteriegesetz) | Perchards and SagisEPR (2022, 2018) |
| Poland | 2009 | Batteries and Accumulators Act | Perchards and SagisEPR (2022, 2018) |
| Portugal | 2009 | Batteries Decree Law | Perchards and SagisEPR (2022, 2018) |
| United Kingdom | 2009 | Batteries and Accumulators Regulations S.I. 2164/2008 | Perchards and SagisEPR (2022, 2018) |
| Belgium | 2010 | Royal Decree of 27 March 2009 on the placing on the market and end-user information of batteries and accumulators | Perchards and SagisEPR (2022, 2018); Tsiarta et al. (2015) |
| Brazil | 2010 | Law No. 12,305 (2010) | Acosta and Corallo (2021); Perchards and SagisEPR (2018) |
| Czech Republic | 2010 | Act 297/2009 amending the batteries section of the Waste Act; Decree 170/2010 | Perchards and SagisEPR (2022, 2018) |
| Greece | 2010 | Ministerial Edict 41624 2057 E103 2010 | Perchards and SagisEPR (2022, 2018) |

[40] A full enforcement of Directive 2006/66/EC was not fulfilled earlier than 2012 (Perchards and SagisEPR, 2018)

| | | | |
|---|---|---|---|
| Malta | 2010 | Waste Management (Waste Batteries and Accumulators) Regulations | Perchards and SagisEPR (2022, 2018) |
| Slovenia | 2010 | Decree on the management of batteries and accumulators and waste batteries and accumulators | Perchards and SagisEPR (2022, 2018); Tsiarta et al. (2015) |
| Iceland | 2011 | amendment to the Waste Act (58/2011); Batteries Regulation (1020/2011) | Perchards and SagisEPR (2022, 2018) |
| Latvia | 2011 | amendment of the Waste Management Act in 2008 | Perchards and SagisEPR (2022, 2018) |
| Romania | 2011 | Decree No 1132/2008; Order 2743/2011 | Perchards and SagisEPR (2022, 2018) |
| Colombia | 2012 | | Perchards and SagisEPR (2018) |
| Macedonia | 2012 | | Perchards and SagisEPR (2018) |
| Norway | 2012 | amendment to Regulations on Waste Recycling | Perchards and SagisEPR (2022, 2018) |
| Croatia | 2013 | Waste Management Act | Perchards and SagisEPR (2022, 2018) |
| Ecuador | 2013[41] | | Perchards and SagisEPR (2018) |
| Costa Rica | 2014 | Regulation for the Declaration of Waste with Special Management Requirements No. 38272-S | Acosta and Corallo (2021); see also Costa Rica legal information system[42] |
| Israel | 2014 | Electrical and Electronic Equipment and Batteries (or e-waste) Law 2012 | see website of the Ministry for Environmental Protection[43] |
| Belarus | 2015 | | Perchards and SagisEPR (2018) |
| Bosnia-Herzegovina | 2016 | Law on Waste Management | Perchards and SagisEPR (2018) |
| Chile | 2016 | Law for Waste Management, Extended Producer Responsibility and Promotion of Recycling (Law N°20.920) | Acosta and Corallo (2021) |
| Russia | 2016 | 2015 amendment to the Federal Law on Waste Production and Consumption | Perchards and SagisEPR (2018) |
| Slovakia | 2016 | Waste Act; Decree on EPR and management of selected product waste streams | Perchards and SagisEPR (2022, 2018) |

[41] This only refers to manufacturers and importers of batteries of certain chemistries which can be removed from electrical and electronic devices (Perchards and SagisEPR, 2018)
[42] https://www.pgrweb.go.cr/scij/Busqueda/Normativa/Normas/nrm_texto_completo.aspx?nValor1=1&nValor2=76879. Accessed 31 August 2023.
[43] https://www.gov.il/en/departments/guides/extended_producer_responsibility?chapterIndex=4. Accessed 31 August 2023

| | | | |
|---|---|---|---|
| Kazakhstan | 2017 | Environmental Code (No. 212-III), Chapter 41-1 "Extended Obligations of Producers and Importers" | OECD (2019) |
| Singapore | 2021 | Resource Sustainability (Prescribed Regulated Products) Regulations 2019 | see website of Singapore National Environmental Agency [44] |
| United Arab Emirates | 2021 | Cabinet Decree No. 39 of 2021 | Iattoni et al. (2021) |
| India | 2022 | Battery Waste Management Rules, 2022 | see EPR Portal for Battery Waste Management[45] |
| South Africa | 2023 | amendment No. 48283 to Extended Producer Responsibility Regulations 2020 | Iattoni et al. (2021), see also website of South African Government[46] |
| New Zealand | 2024 | | see website of the Ministry for the environment[47] |
| Viet Nam | 2024 | Decree No. 08/2022/ND-CP Detailing a Number of Articles of the Law on Environmental Protection | see Enviliance Asia[48] |
| Algeria | | | Iattoni et al. (2021) |
| Argentina | | | Acosta and Corallo (2021); Perchards and SagisEPR (2018) |
| Australia | | | Battery Implementation Working Group (BIWG) (2014), see also website of Department of Climate Change, Energy, the Environment and Water[49] |
| Bahrain | | | Iattoni et al. (2021) |

[44] https://www.nea.gov.sg/our-services/waste-management/3r-programmes-and-resources/e-waste-management/extended-producer-responsibility-(epr)-system-for-e-waste-management-system. Accessed 31 August 2023

[45] http://www.eprbatterycpcb.in/. Accessed 31 August 2023

[46] https://www.gov.za/documents/national-environmental-management-waste-act-nemwa-extended-producer-responsibility-0. Accessed 31 August 2023.

[47] https://environment.govt.nz/what-government-is-doing/areas-of-work/waste/product-stewardship/regulated-product-stewardship/. Accessed 31 August 2023.

[48] https://enviliance.com/regions/southeast-asia/vn/report_5407. Acessed 31 August 2023.

[49] https://www.dcceew.gov.au/environment/protection/waste/publications/national-waste-reports/2013/product-stewardship#fact-sheets. Accessed 31 August 2023.

| | | | |
|---|---|---|---|
| Canada[50] | | | Perchards and SagisEPR (2018) |
| China, Hong Kong[51] | | | see Hong Kong Waste Reduction website[52] |
| Comoros | | | Iattoni et al. (2021) |
| Djibouti | | | Iattoni et al. (2021) |
| Egypt | | | Iattoni et al. (2021) |
| Honduras | | | Acosta and Corallo (2021) |
| Indonesia | | | see EPR Indonesia website[53] |
| Iraq | | | Iattoni et al. (2021) |
| Jordan | | | Iattoni et al. (2021) |
| Kuwait | | | Iattoni et al. (2021) |
| Lebanon | | | Iattoni et al. (2021) |
| Lybia | | | Iattoni et al. (2021) |
| Marocco | | | Iattoni et al. (2021) |
| Mauritania | | | Iattoni et al. (2021) |
| Mexico | | | Acosta and Corallo (2021) |
| Montenegro | | | Perchards and SagisEPR (2018) |
| Oman | | | Iattoni et al. (2021) |
| Paraguay | | | see UNEP website[54] |
| Philippines | | | World Wide Fund for Nature (WWF) Philippines (2022) |

[50] Canada has no federal legislation on EPR on WB (Perchards and SagisEPR, 2018). Only 4 out of 10 provinces have EPR regulations for portable batteries; in the other provinces, take-back schemes are implemented on a voluntary basis

[51] A compulsory EPR scheme is not set by law, but a voluntary program is in place.

[52] https://www.wastereduction.gov.hk/en/workplace/rechargebattery_intro.htm. Accessed 31 August 202.

[53] https://www.epr-indonesia.id/the-legal-framework-in-indonesia. Accessed 31 August 2023.

[54] https://dicf.unepgrid.ch/paraguay/pollution. Accessed 31 August 2023.

| | | | |
|---|---|---|---|
| Qatar | | | Iattoni et al. (2021) |
| Saudi Arabia | | | Iattoni et al. (2021) |
| Serbia | | | European Topic Centre on Waste Materials in a Green Economy (2021) |
| Somalia | | | Iattoni et al. (2021) |
| Sudan | | | Iattoni et al. (2021) |
| Syria | | | Iattoni et al. (2021) |
| Thailand | | | see National Energy Technology Center[55] |
| State of Palestine | | | Iattoni et al. (2021) |
| Tunisia | | | Iattoni et al. (2021) |
| Ukraine | | | Perchards and SagisEPR (2018) |
| USA[56] | | | Perchards and SagisEPR (2018) |
| Yemen | | | Iattoni et al. (2021) |

55 https://www.entec.or.th/knowledge-everything-you-need-to-know-about-batteries/. Accessed 31 August 2023.

56 There is no federal legislation requiring the take-back of waste batteries by retailers or producers. According to Perchards and SagisEPR (2018), just 9 of the 51 states have take-back requirements on some types of batteries in place, mostly on rechargeable batteries only.

## B.4 Main elements of EU Batteries Directive (2006/66/EC)

In this appendix we summarize the main provisions of the EU Batteries Directive of 2006, on the basis of the directive itself and of the related implementation reports by European Commission (2019a) and Tsiarta et al. (2015). This Directive has a specific relevance in our paper, since it represents the baseline regulation for the adoption of EPR on WB in EU countries. Moreover, EU EPR regulations are recognized as worldwide reference points in this policy field (Corsini et al., 2017; Gerrard and Kandlikar, 2007).

The directive's primary objective is to minimise the negative impact of batteries and waste batteries on the environment to help protect, preserve and improve the quality of the environment. Secondly, it aims to ensure the smooth functioning of the internal market and avoid the distortion of competition within the EU by regulating the placing of batteries on the market.

The directive applies to all batteries and classifies them according to their use. Classes of battery include: portable batteries, automotive batteries, industrial batteries.

To avoid the release of certain hazardous substances in the environment, the directive prohibits that batteries containing mercury and cadmium above certain thresholds are placed on market.

On the downstream side of batteries life cycle, the overarching objective of the directive is that Member States take the necessary measures to maximise the separate collection of waste batteries and to minimise the disposal of batteries as mixed municipal waste. Member States are required to ensure that appropriate collection schemes are in place for waste portable batteries and sets targets for their collection rates, namely 25% in weight of the amount placed on the market by September 2012 and 45% by September 2016. As for the other two types of batteries defined by the directive, the regulation requires Member States to set up collection schemes for waste automotive batteries and to ensure that producers of industrial batteries do not refuse to take back waste industrial batteries from end-users. Nonetheless, targets for the collection of waste industrial or automotive batteries are not set.

All spent batteries collected must undergo treatment and recycling. It is prohibited to landfill or incinerate waste from industrial and automotive batteries. The directive sets recycling targets for collected WB: 65% for lead-acid WB, 75% for nickel-cadmium ones and 50% for other types of WB. Notice that, this last class contains lithium-ion batteries, used in electric vehicles, among other scopes.

According to article 15 of the directive, treatment and recycling may take place outside the Member State concerned or even outside the EU, provided that EU legislation on the shipment of waste is respected. Hence, as we discussed, WB exports are not directly affected by the EPR regulation.

Provisions on extended responsibility give producers of batteries and producers of other products that incorporate batteries the responsibility for the end-of-life management of the batteries they place on the market. The directive specifies the national schemes, tasks and objectives, including financial aspects. Hence, producers must fund the net costs of collecting, treating and recycling all waste portable batteries and all waste industrial and automotive batteries as well as any public information campaigns on the topic.

The directive encourages Member States to support the development of new recycling and treatment technologies, and promote research into environmentally friendly and cost-effective recycling methods for all types of batteries and accumulators (article 13).

## B.5 Advantages and disadvantages of patent data measure (green) innovation

Patent data offer several advantages over alternative measures of innovation, including (Haščič and Migotto, 2015):

a) They are commensurable because patents rely on an objective standard. The type of invention eligible for a patent is well-defined and must satisfy three patentability criteria: novel, non-obvious (inventive step), and useful (with industrial application).

b) They gauge the intermediate outputs of the inventive process, in contrast to data on R&D expenditures that solely measure input or trade data that may not embody innovative technologies.

c) The data are quantitative, making them easily amenable to statistical analysis.

d) The data are widely accessible in the public domain, as opposed to proprietary information, such as licensing data.

e) The data can be disaggregated into specific technological fields, a crucial aspect for studying *"environmental"* innovation.

However, it's essential to acknowledge that patents alone cannot provide a comprehensive measure of innovation. The three commonly cited reasons are:

a) Not all innovations are patentable. Patents are designed to protect technological innovations meeting the three criteria mentioned above. This limitation excludes organizational, managerial, and non-technological innovations from measurement.

b) Not all patentable inventions are patented. Other intellectual property rights (IPR) regimes, such as copyrights and trademarks, exist to protect various innovations. Additionally, inventors may opt for informal strategies like industrial secrecy or purposefully complex technical specifications.

c) Patented inventions vary in quality due to the associated costs. The fees for patent application examination, grant, and renewal suggest optimism about commercialization and adoption. However, not all patented inventions achieve this, leading to variations in economic value.

There are three potential methods for identifying environment-related technologies through patent data:

i) Searches based on patent classifications, like the IPC or CPC, remain the most common approach, relying on the detailed knowledge of patent examiners.

ii) Searches based on keywords in titles or abstracts, often used when identifying relevant and *"clean"* patent classifications is challenging. However, a drawback is sensitivity to language, making it costly for cross-language search strategies.

iii) Manual selection, while limited by the time and expert knowledge required, is suitable for smaller-scale analyses involving specific countries and technological fields.

Historically, OECD work has predominantly relied on searches based on patent classifications.