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## Summary

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, EPR exports appear to be directed to countries with more advanced waste management systems rather than to developing countries.

**Keywords:** Extended producer responsibility, batteries, trade, recycling, circular economy

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

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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

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# 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 (Bianchini et al., 2023; IEA, 2021) - and they are exposed to high supply risks since their extraction and refining is concentrated in few countries (EU CRM sectors 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; Valero et al., 2018). 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., 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

[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.

[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

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 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]

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 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.

# 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, 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-

[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.

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.

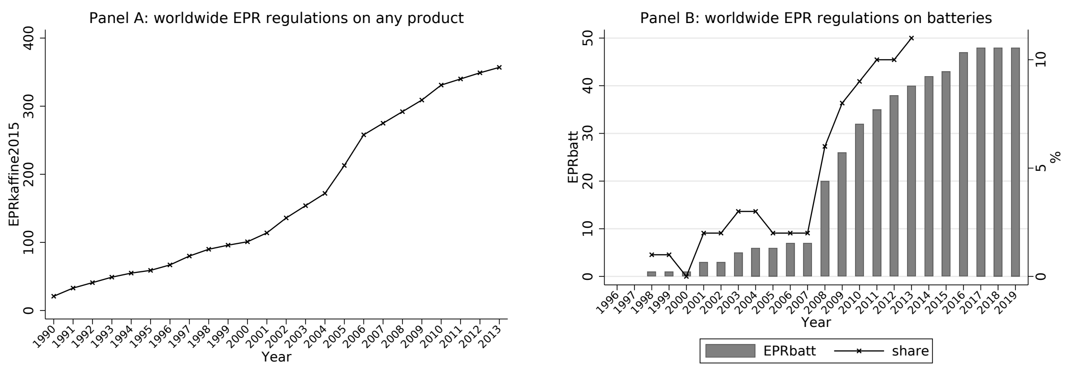


Figure 1: Panel A: overall worldwide number of EPR regulations, 1990-2013; our elaboration on Kaffine and O'Reilly (2015) data. Panel B: worldwide number of EPR regulations on batteries (bars, left scale) and share of EPR regulations on batteries over the overall number of EPR regulations (line), 1996-2019;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

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

[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).

[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 A.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

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 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. *Second*, and similarly, 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

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 A.4)

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]. Following this argument, a reduction in the export of trade in WB may be a desirable

[12] In this sense, several IEAs do not seem to be sufficiently effective in restricting trade in hazardous waste, and specific rules for producers, such as EPR, may well be a complementary tool.

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 related to differences in the technological endowment for waste treatment and/or differences in the stringency of environmental regulations within pairs of trading countries. 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.

*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.

[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 5 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]

[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] The complete description of product categories is available at `https://unstats.un.org/unsd/classifications/econ/`.

[16] Note that the BACI dataset do not include null bilateral trade flows, i.e. exporter-importer-

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 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,

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.

[17] Table 5 in Appendix A.2 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.

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, such as Czech Republic, Slovenia, Sweden, Bulgaria (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 4 in Appendix A.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.

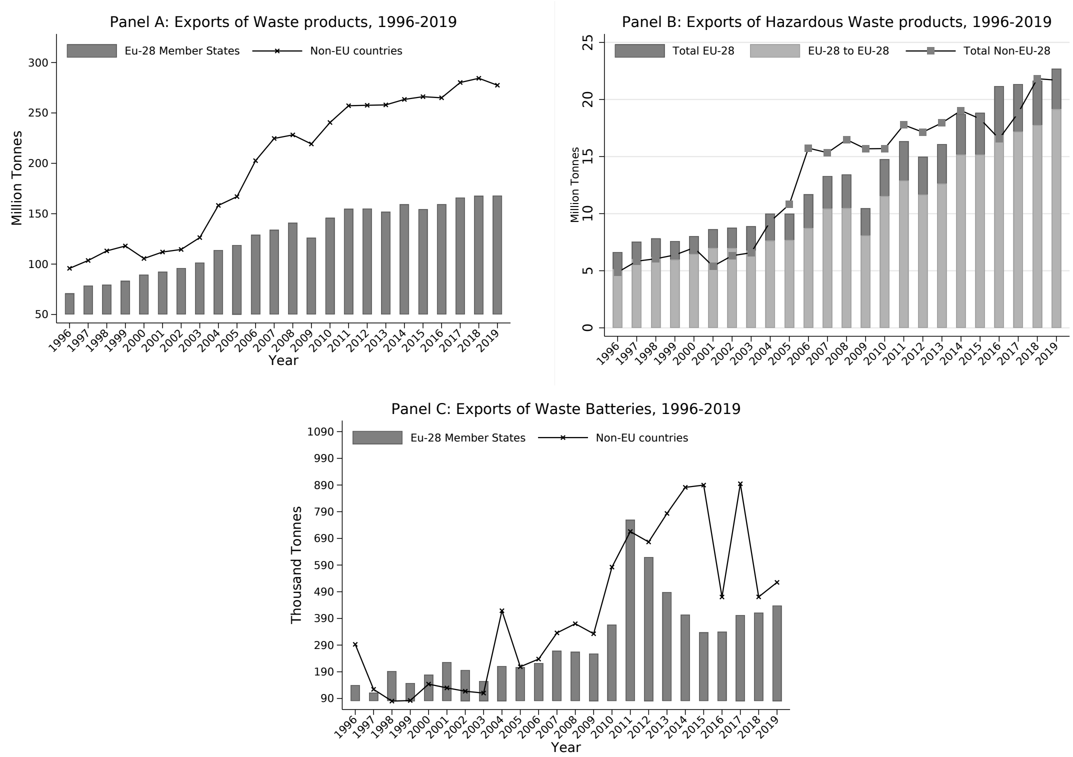


Figure 2: Export Quantity of Waste, Hazardous Waste and Waste Batteries, 1996-2019. Our elaboration on BACI data.

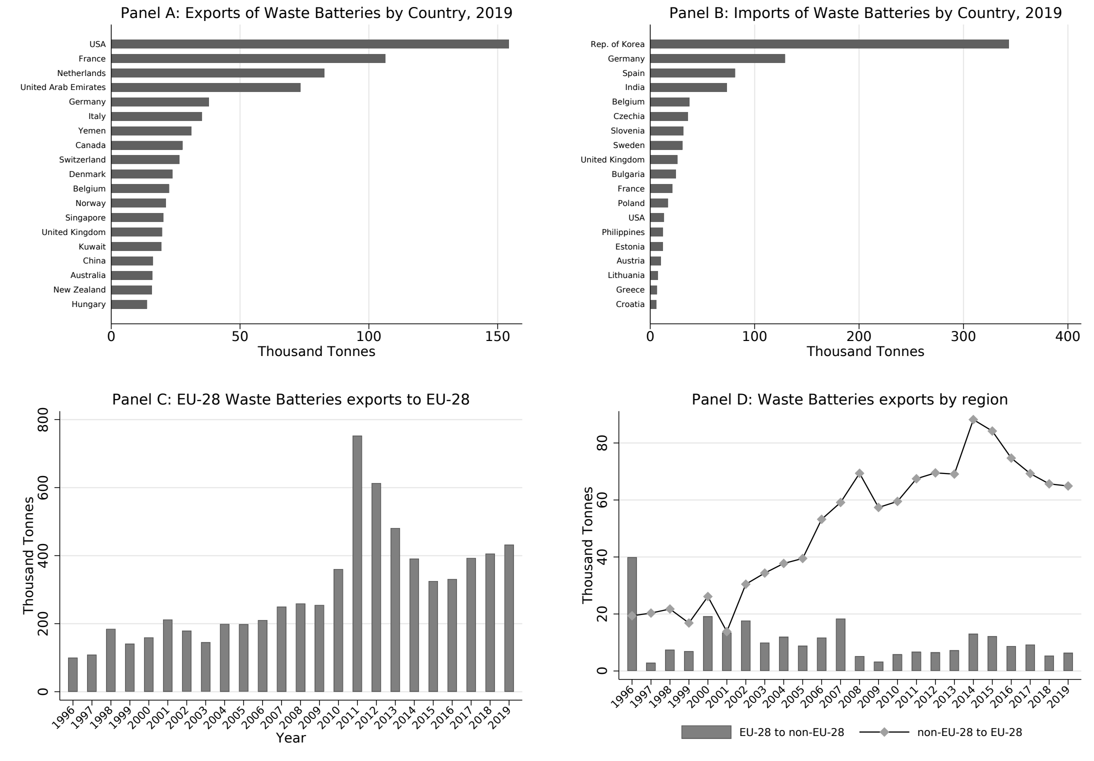


Figure 3: Exports and Imports Quantity of Waste Batteries by Country or Region (EU/non-EU), 1996-2019. Our elaboration on BACI data.

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 6 in Appendix A.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.[20] 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.

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.A.3) |
| 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.

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

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

[20] The Cepii Gravity dataset provides alternative measures of geographical distances, including the distance between the capital city in each country, or the weighted distances that takes into account the geographical distribution of population within each country. See Conte et al. (2022) for more details.

and all destinations, for 114 waste products.

# 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 EPR policy, before and after the implementation of the policy 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) [21] 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.[22] 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).[23] 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.[24] 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

[21] 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).

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

[23] 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.

[24] 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).

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 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.[25]

The second 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.

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. First, we 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,

[25] 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.

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. In a fourth model we employ a Poisson pseudo-maximum likelihood (PPML) estimator.

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 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 is different 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:

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

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.

## 4.2 Econometric results

### 4.2.1 Baseline results

We show 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 ($\text{EPR}_{ey}$) 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 ($\text{EPR}_{ey} \times \text{WB}_p$). 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

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

| Dep. Var. | ln $Quantity_{eipy}$ | | | | $Quantity_{eipy}$ |
|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) | (5) |
| $EPR_{ey}$ | $-0.065$*** | | | | |
| | (0.011) | | | | |
| $\times$ $WB_p$ | 0.304*** | 0.269*** | 0.623*** | 0.694*** | 0.395** |
| | (0.063) | (0.062) | (0.145) | (0.209) | (0.195) |
| 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 |
| $\omega_p$ | Yes | Yes | No | No | No |
| $\omega_e$ | Yes | No | No | No | No |
| $\omega_i$ | Yes | No | No | No | No |
| $\omega_{ey}$ | No | Yes | Yes | Yes | Yes |
| $\omega_{iy}$ | No | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | No | No | Yes | Yes | Yes |
| $\omega_{py}$ | No | No | No | Yes | No |
| Adj. $R^2$ | 0.410 | 0.417 | 0.734 | 0.740 | |
| No. of Obs | 1,401,055 | 1,429,644 | 1,568,988 | 1,568,959 | |

*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.

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 $\text{GDP}_{ey}$ 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 by 0.69% . Contiguity is relevant too. If the two trading countries share a common land border, trade in WB increases by about 190%.[26]. 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 col. (4), 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, which controls for product-specific trends, is practically identical to the DiD coefficient shown in

[26] 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).

col. (3), and this reassures that results are not driven by uncontrolled differences between WB and other waste products not affected by the policy.

Finally, in col. (5), according to Santos Silva and Tenreyro (2006), we estimate the model using the Poisson pseudo maximum likelihood (PPML) estimator to address the omitted zeros problem. The DiD coefficient is positive and significant, in line with the result shown in col. (3). This result reassures us regarding our main results not being driven by a selection issue, possibly due to unobserved choices by exporter countries not to export specific products to specific markets.

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 EPR regulations. 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.

#### 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 ($HW_p$) under the Basel Convention, and exclude the HS6 code referring to WB from the list. The 13 HS6 codes defined in

[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.

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.

this way are described in Appendix A.2 Table 5. 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 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 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.

### 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. It is clear that the data used do not allow for a deeper and more precise identification of the mechanisms responsible, 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

[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.

source of comparative advantage for countries with lower levels of regulation in terms of attracting flows of waste. The literature refers to this phenomenon as "waste haven hypothesis". To test for this possibility, 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 ($D_{iy}^{Dev}$), i.e. it ranks in the top half distribution of countries by GDP per capita.[29] As shown in col. (1) of Table 4, 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. Admittedly, by using GDP per capita, we rely on an indirect measure of environmental regulation stringency (Kellenberg, 2015; Brunel and Levinson, 2016). 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 that the export flow of WB has increased more towards EU-28 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".[30] 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) explanation. Larger economies have more advanced recycling programs 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

[29] 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. Moreover, this categorisation by GDP allows to avoid a further reduction of the sample, with respect to the introduction of ad hoc variables accounting for environmental regulation stringency.

[30] 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.

since long the most common type of battery on the market, are economically recycled (and manufactured) in Europe (European Commission, 2019a)[31]. Thus, larger economies may have greater demand for recyclable wastes, despite their stricter environmental regulations when comparing to developing countries.

Table 4: The effect of EPR policy on the exports of Waste Battery: possible mechanisms

| Dep. Var. | ln Quantity$_{eipy}$ | | | |
|---|---|---|---|---|
| | (1) | (2) | (3) | (4) |
| $\text{EPR}_{ey} \times \text{WB}_p$ | 0.168 | 0.246 | 0.703** | 0.191 |
| | (0.271) | (0.205) | (0.251) | (0.297) |
| $\text{EPR}_{ey} \times \text{WB}_p \times \text{D}_{iy}^{Dev}$ | 0.593** | | | |
| | (0.267) | | | |
| $\text{EPR}_{ey} \times \text{WB}_p \times \text{D}_{iy}^{EU28}$ | | 0.770*** | | |
| | | (0.213) | | |
| $\text{EPR}_{ey} \times \text{WB}_p \times$ Gradient Patent$_{iy}$ | | | 0.219*** | |
| | | | (0.070) | |
| $\text{EPR}_{ey} \times \text{WB}_p \times$ Gradient Facilities$_{iy}$ | | | | 0.264** |
| | | | | (0.119) |
| $\omega_{ey}$ | Yes | Yes | Yes | Yes |
| $\omega_{iy}$ | Yes | Yes | Yes | Yes |
| $\omega_{eip}$ | Yes | Yes | Yes | Yes |
| $\omega_{py}$ | Yes | Yes | Yes | Yes |
| Adj. $R^2$ | 0.740 | 0.740 | 0.743 | 0.757 |
| No. of Obs | 1,568,914 | 1,568,959 | 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 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

[31] 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.

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[32]. This figure is taken from Eurostat, so it covers European 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:

$$E_{iey} = \frac{E_i - E_j}{\frac{E_i + E_j}{2}}. \tag{3}$$

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 recycling WB – either proxied by a specialization in the patent domain of WB recycling, or by a higher number of general recycling facilities – is a relevant driver of the increase in trade in WB after the adoption of the EPR policy by the exporting country.[33]

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

[33] As a further robustness check, we introduce the interaction of the indicator measuring the adoption of EPR by the importing country with the categorical variable on waste batteries. The coefficient of our variable of interest ($\text{EPR}_{ey} \times \text{WB}_p$) remains relevant in magnitude and statistically significant. This result (available from the authors upon request) reassures us that our results are not determined by the choice of the importing country to adopt EPR.

# 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.

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, the policy does not seem to have promoted a waste haven effect. The EU in particular is not only basically self-sufficient in WB management, but also a net importer. Rather than environmental stringency, the level of sophistication of the waste management system, both in terms of patents and facilities, seems to play a stronger role. 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 of EPR as an effective tool to indirectly support "waste marketing strategies" (Kama, 2015; Theis, 2021), 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 impact 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 has been investigated in a rather exploratory way. Further research with this specific focus could consider complementing our approach with data on illegal waste exports and environmental policy stringency.

# A Appendix

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

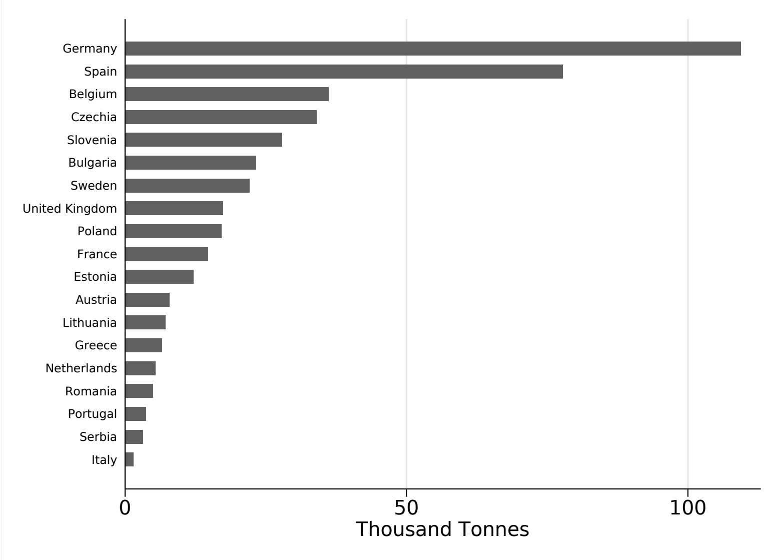


Figure 4: Imports quantity of waste batteries by EU-28 country, 2019.

## A.2 Waste products

Table 5 lists the 1996 HS-6 codes included in our analysis. In this table, "Hazardous" identifies the HS included in the hazardous waste set in this paper. It is important to note that our list of HS codes for hazardous waste should not be an exhaustive list. More precisely, all the HS that we have classified as hazardous waste are certainly hazardous waste types, but other 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 5: 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 |

## A.3 EPR regulations on batteries

Table 6 provides information on EPR regulations for batteries worldwide. In this table, "Year" refers to the date of implementation of the regulation in each country, where available; alternatively, the date of publication of the regulation has been used. For European countries, "Year" refers to the date of national implementation of EU Directive 2006/66/EC. Subnational regulations were not considered. Countries for which "Year" is left blank have not yet introduced EPR on WB according to the reported references. In the column "Regulation" we report the name of the reference regulation, if available.

Table 6: 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)[34] |
| 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.[35] |
| Turkey | 2004 | Batteries Regulation | Perchards and SagisEPR (2018) |
| Bulgaria | 2006[36] | 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) |
| 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) |

[34] 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).

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

[36] 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)

| | | | |
|---|---|---|---|
| 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[37] | 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) |
| 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) |

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

| | | | |
|---|---|---|---|
| Ecuador | 2013[38] | | 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[39] |
| Israel | 2014 | Electrical and Electronic Equipment and Batteries (or e-waste) Law 2012 | see website of the Ministry for Environmental Protection[40] |
| 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) |
| 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 [41] |
| 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[42] |
| South Africa | 2023 | amendment No. 48283 to Extended Producer Responsibility Regulations 2020 | Iattoni et al. (2021), see also website of South African Government[43] |

[38] 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)

[39] https://www.pgrweb.go.cr/scij/Busqueda/Normativa/Normas/nrm_texto_completo.aspx?nValor1=1&nValor2=76879, Accessed 31 August 2023.

[40] https://www.gov.il/en/departments/guides/extended_producer_responsibility?chapterIndex=4, Accessed 31 August 2023

[41] 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

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

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

| | | | |
|---|---|---|---|
| New Zealand | 2024 | | see website of the Ministry for the environment[44] |
| Viet Nam | 2024 | Decree No. 08/2022/ND-CP Detailing a Number of Articles of the Law on Environmental Protection | see Enviliance Asia[45] |
| 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[46] |
| Bahrain | | | Iattoni et al. (2021) |
| Canada[47] | | | Perchards and SagisEPR (2018) |
| China, Hong Kong[48] | | | see Hong Kong Waste Reduction website[49] |
| 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[50] |
| Iraq | | | Iattoni et al. (2021) |
| Jordan | | | Iattoni et al. (2021) |
| Kuwait | | | Iattoni et al. (2021) |

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

[45] https://enviliance.com/regions/southeast-asia/vn/report_5407, Aceessed 31 August 2023.

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

[47] 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

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

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

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

| | | | |
|---|---|---|---|
| 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[51] |
| Philippines | | | World Wide Fund for Nature (WWF) Philippines (2022) |
| 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[52] |
| State of Palestine | | | Iattoni et al. (2021) |
| Tunisia | | | Iattoni et al. (2021) |
| Ukraine | | | Perchards and SagisEPR (2018) |
| USA[53] | | | Perchards and SagisEPR (2018) |
| Yemen | | | Iattoni et al. (2021) |

[51] `https://dicf.unepgrid.ch/paraguay/pollution`, Accessed 31 August 2023.

[52] `https://www.entec.or.th/knowledge-everything-you-need-to-know-about-batteries/`, Accessed 31 August 2023.

[53] 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.

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

In this section, we summarize the main provisions of the EU Batteries Directive of 2006, based on the Directive itself and the related implementation reports by European Commission (2019a) and Tsiarta et al. (2015). This Directive is of particular relevance to our paper, as it represents the basic regulation for the adoption of EPR on WB in EU countries. Moreover, the EU EPR regulations are recognized as a global reference point in this policy area (Corsini et al., 2017; Gerrard and Kandlikar, 2007).

The primary objective of the Directive is to minimize the negative impact of batteries and waste batteries on the environment in order to contribute to the protection, preservation and improvement of the quality of the environment. Second, it aims to ensure the proper functioning of the internal market and avoid 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. Battery classes include: portable batteries, automotive batteries, industrial batteries. In order to prevent the release of certain hazardous substances into the environment, the Directive prohibits the placing on the market of batteries containing mercury and cadmium above certain thresholds.

On the downstream side of the battery life cycle, the overall objective of the Directive is that Member States take the necessary measures to maximize the separate collection of waste batteries and to minimize 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 set targets for their collection rates, namely 25% in weight of the amount placed on the market by September 2012 and 45% by September 2016. For the other two types of batteries defined in 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. However, no targets are set for the collection of waste industrial or automotive batteries. All collected used batteries must be treated and recycled. The land-filling or incineration of industrial and automotive battery waste is prohibited. The Directive sets recycling targets for collected WB: 65% for lead-acid WB, 75% for nickel-cadmium WB and 50% for other types of WB. Note that this last class includes lithium-ion batteries used in electric vehicles, among others. 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 shipments of waste is respected. Therefore, as discussed above, exports of WB are not directly affected by the EPR Regulation.

The extended responsibility provisions make producers of batteries and producers of other products containing batteries responsible for the end-of-life management of the batteries they place on the market. The Directive specifies the national systems, tasks and targets, including financial aspects. Producers will have to finance the net costs of collection, treatment and recycling of all waste portable batteries and all waste industrial and automotive batteries, as well as any public information campaigns on the subject.

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

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