



Extended Producer Responsibility and Trade Flows in Waste: The Case of Batteries

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Accepted: 22 July 2024
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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”. WB are a hazardous waste that also contain a high concentration of critical raw materials. As such, they are of strategic importance for the recovery of critical resources, while at the same time requiring proper environmental management. Therefore, it is crucial to understand where WB are treated and how this is affected by related policies. Our results, based on difference-in-difference models in a gravity framework, show a consistent increase in WB exports after EPR implementation compared to the trend for other wastes. This result is likely to be an indirect consequence of the ability of EPR to support growth in waste collection rates, more accurate tracking of transboundary waste flows, and specialization of national waste management systems. In particular, WB exports appear to be directed to countries with more advanced waste management systems, more stringent environmental regulations, and limited endowments of the mineral resources typically contained in batteries.

Keywords Extended producer responsibility · Batteries · Trade · Critical raw materials · Circular economy

JEL Classification K32 · Q51 · Q53 · Q56

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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; Martins et al. 2021). 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 (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 (Mrozik et al. 2021; Tanaka et al. 2022). 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 (International Energy Agency 2021) - and they are exposed to high supply risks since their extraction and refining is concentrated in few countries (Eggert et al. 2016; Liu et al. 2022; Carrara et al. 2023).¹

Strikingly, research on the current management of waste batteries is extremely limited. In particular, while a rich literature has investigated the economic potential of battery recycling or reuse (Wang et al. 2014; Innocenzi et al. 2017; Liu et al. 2019), 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 international flows of WB.

In recent years, it is increasingly seen as strategic to strengthen national or international circular economy systems aimed at recovering (critical) resources (Rosendahl and Rubiano 2019; Kojima 2020; Pommeret et al. 2022). The aim is to reduce supply risks and to alleviate the pressure on resource extraction.² Indeed, the growing demand for minerals for global battery production is responsible for undeniable impacts on local populations and ecosystems around mining sites, typically located in developing countries (Luckeneder et al. 2021; Agusdinata and Liu 2023). Within this framework, attempts have been done to implement a strategy of “marketization of waste” (Gregson et al. 2013; Reis 2016), referring to the process of treating waste as a commodity that can be bought, sold, and traded on ad-hoc markets. This has been especially true for CRM-rich waste flows (Theis 2021).³

In such a context, 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

¹ This concentration of CRM-related value chains rises the concerns for supply and price volatility in dependent countries (Kowalski and Legendre 2023). As proof of this, prices of battery metals increased dramatically in early 2022, posing a significant challenge to the electric vehicles industry (IEA 2022).

² From a life cycle perspective, recycling batteries reduces energy consumption and greenhouse gas emissions, over and above saving natural resources, when compared to landfilling (Boyden et al. 2016). For this reason, high levels of recycling are also necessary to achieve net zero targets (International Energy Agency 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.

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 costs associated with waste management, and for informing public stakeholders about the amount 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).

This paper attempts to empirically investigate the impact of EPR regulations in general, and EPR on WB in particular, on the trade of the affected waste flows. Why EPR should be expected to have an impact on waste trade flows? What is the expected sign of the relationship between EPR and waste trade? As discussed in the conceptual framework presented in Sect. 2.2, EPR could have indirect impacts on waste trade networks, both discouraging or boosting exports. 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 raw 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 Harmonized System classification for traded goods.⁴

The rest of the paper is organized as follows. In Sect. 2, we introduce the concept of EPR, describe the regulations related to it, and outline the main theoretical channels through which EPR affects waste trade. In Sect. 3, we describe the data and variables used to test the relationship between EPR and trade in WB. Section 4 presents the econometric analysis and results, first outlining the direction of the relationship under investigation and then exploring the possible mechanisms at play. Section 5 provides policy recommendations and concluding remarks.

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

⁴ Addressing the same research questions using other categories of waste would not necessarily ensure the same level of precision. 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.

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 regulations⁷, 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 (Lindhqvist and Lifset 1998; Gupt and Sahay 2015). The former refers to the design and production phase of products, where the aim of EPR is for producers to internalise the post-sale environmental costs of the equipment they put on the market, leading to waste prevention strategies (e.g. lifetime extension) or eco-design (e.g. design for recycling). The latter refers to the waste management phase of the product life cycle; from this perspective, EPR aims to increase the (separate) collection of the targeted waste streams and their recycling rates, thus supporting the development of the recycling sector and the market for recycled materials. An overview of the general expectations originally addressed by EPR policies is provided in Compagnoni (2022).

The relevance of EPR regulations and the support the principle has received from policy makers to promote 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 Fig. 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 (Fig. 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.

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

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 through a variety of instruments, ranging from deposit/refund schemes, to upstream combined tax/subsidies, to advanced disposal fee, 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 Sect. 2.2 can be considered as generalizable.

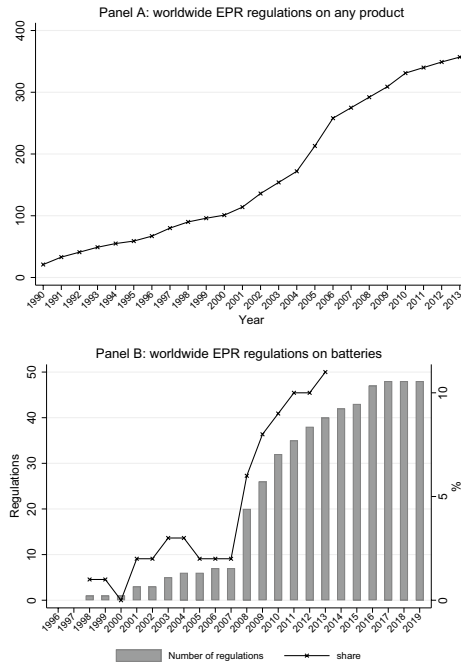


Fig. 1 Panel A: overall worldwide number of EPR regulations, 1990–2013; our elaboration on Kaffine and O’Reilly (2015) data. Panel B: worldwide number of EPR regulations on batteries (bars, left scale) and share of EPR regulations on batteries over the overall number of EPR regulations (line), 1996–2019;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

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; Kellenberg 2012; Higashida and Managi 2014; Falkowska 2020). 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

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).

(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 (Massarutto 2014; Kaffine and O'Reilly 2015). To achieve this objective, EPR regulations typically set waste collection targets (Gupt and Sahay 2015).⁹ In the case of Europe, an increase in the collection rates after the introduction of EPR has been observed for several products (Dubois and Eyckmans 2015), including WB (Perchards and SagisEPR 2018, 2022).¹⁰ If, in the face of higher collection rates, the home country strategically invests and innovates in its own waste management system and recycling sector, developing more facilities and eventually better technologies (Nicolli et al. 2012; Massarutto 2014; Atasu 2019; Favot et al. 2022),¹¹ 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, the *informative* responsibility increases the non-monetary costs to 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 (Moisé 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

⁹ For example, in 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 Online 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 2019).

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

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

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. Instead, where EPR is not implemented, separate WB collection rates remain low, with limited masses to be treated (Levänen et al. 2018). *Second*, and similarly to the previous argumentation, the *economic* responsibility of producers encourages them to treat waste for recycling or disposal in the country where it is more efficient to do so (Premalatha et al. 2014; Bernard 2015; Joltreau 2021). Again, this would induce those countries with a disadvantage (higher relative costs) to increase exports of waste rather than treat it domestically. Also notice that countries are increasingly building specialized capacity for specific hazardous materials operations (Yang 2020). Hence, a relatively high waste treatment cost may arise not only from generally scarce physical capital (i.e. facilities and technologies) and human capital endowments, but also from the specialization of the exporting country in some waste categories and not others. *Third*, the *informative* responsibility of producers (Lifset 1993; Gerrard and Kandlikar 2007) 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 outlining the various channels influencing this relationship and by testing this relationship empirically for the first time. In addition, we provide the most comprehensive review of EPR regulations on waste batteries adopted worldwide. In a broader perspective, we contribute to the debate on waste trade and circular economy policies.

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 outcome of the adoption of an EPR policy. However, WB also contain a high concentration of critical raw materials that are essential inputs for

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.

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 Sect. 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 WB exports after the adoption of the EPR, by examining the role played by some characteristics of the importing country. The existing literature so far has shown that trade in waste depends on two main factors, which affect the relative cost of its treatment at home versus abroad: (i) the difference in natural and technological endowments related to waste treatment between trading partners, and (ii) the difference in the stringency of environmental policies.¹³ In this paper, we examine whether changes in the export of WB following the adoption of EPR by the exporter are affected by the technological endowment for waste treatment or by the stringency of environmental regulations in the importing country. 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. We also examine whether greater availability in the importing country of the most common minerals used in battery production affects the intensity of WB trade flows. Indeed, countries with significant endowments of these natural resources may find it less convenient to import WB for the purpose of materials' recovery, nor may they have developed the necessary recycling technologies.

Fourth, from a methodological point of view, by focusing on 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.

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

¹³ A third relevant factor in explaining waste trade patterns relates to corruption and organized crime (Kellenberg 2015; Cesi et al. 2019). 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.

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 dataset estimates conversion rates from other units to tons (Gaulier and Zignago 2010).

Overall, the BACI dataset includes 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 (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 A1 in the Online Appendix provides the list of the HS6 codes identified as waste products, along with a brief description of each product. By restricting the BACI dataset to this subset, we obtain a total of about 2 million observations over the period under analysis. On average, each year waste products account approximately for 1.2% of the total trade flows among countries.¹⁶

The importance of trade in waste products is observed in the data, as the amount of waste traded in recent years has been noteworthy. Panel A of Fig. 2 shows that waste is increasingly moving across borders, albeit at lower growth rates than in 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 Fig. 2 are based on the HS6 codes related to hazardous waste products proposed by Kellenberg and

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

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

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

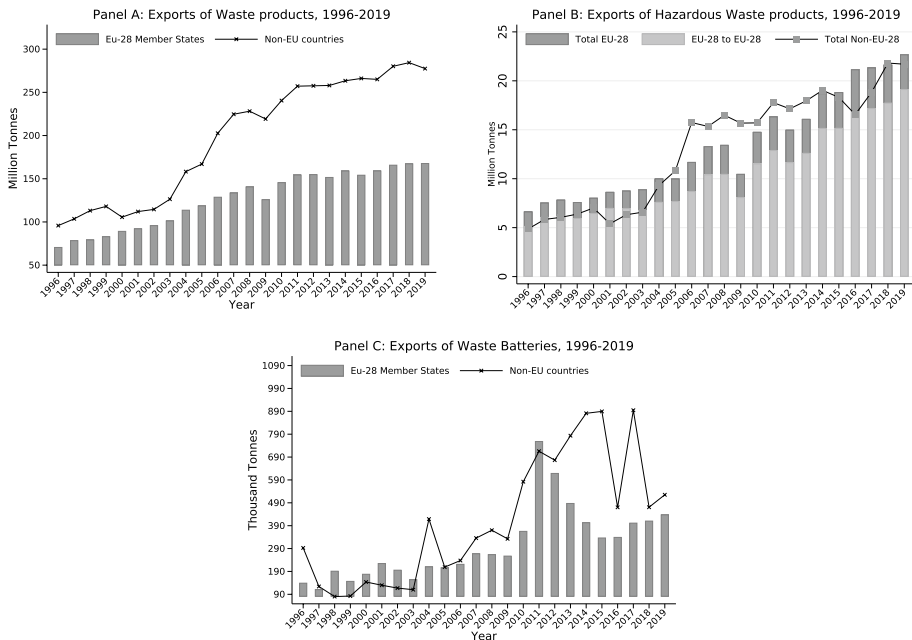


Fig. 2 Export quantity of waste, hazardous waste and waste batteries, 1996–2019. Our elaboration on BACI data

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 the European Environmental Agency (2012) and European Commission (2015).¹⁸ 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. According to our estimates, over the period 1996–2019, HW_p intra-EU exports have roughly quadrupled, as have exports of HW_p originating in the rest of the world, while exports from the EU to the rest of the world have doubled.

Among HW_p products, we are particularly interested in waste batteries and accumulators (WB_p), which represents the “treated” group in our empirical analysis. As shown in Panel C of Fig. 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

¹⁷ Table A1 in Online Appendix A.2 clarifies which waste products are classified as hazardous. Since no official correspondence tables between the HS classification and hazardous waste as defined in the Basel Convention) codes are available, our figures for hazardous waste exports should be considered as a proxy of the actual ones.

¹⁸ Hazardous waste data in these documents are slightly different from those presented here, as they are based on official data reported by countries to the European Commission (Eurostat).

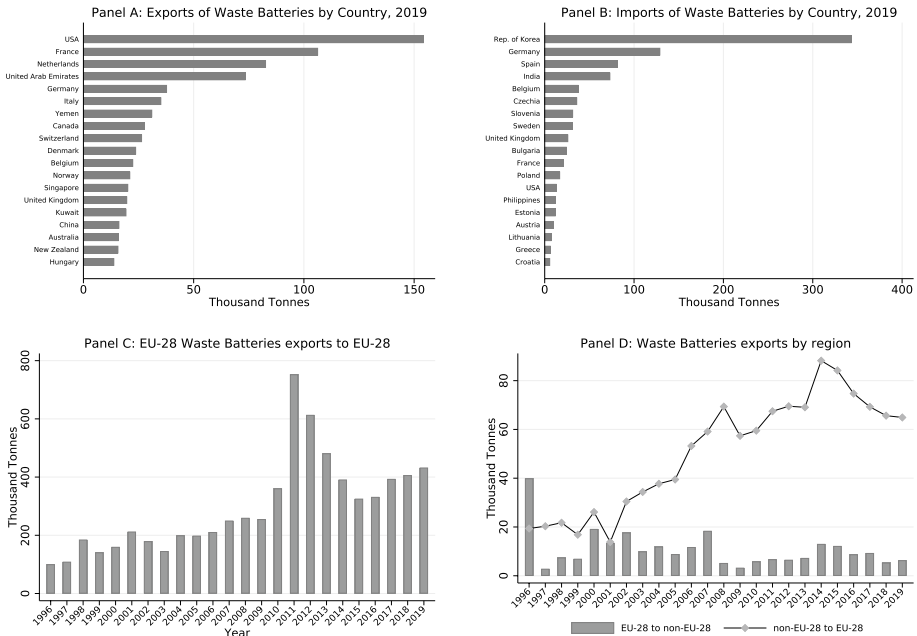


Fig. 3 Exports and imports quantity of waste batteries by country or region (EU/non-EU), 1996–2019. Our elaboration on BACI data

in waste products. In 2019, the USA, France, the Netherlands, and the United Arab Emirates are the four largest exporters of WB_p (Panel A of Fig. 3), while South Korea is by far the world’s largest importer of WB_p in the same year and is home to three of the world’s 10 largest battery manufacturers, namely LG Energy Solution, Samsung SDI and SK On. Among the importers of WB_p , two European countries, i.e. Germany and Spain rank second and third, followed by India, and by other European countries (Panel B of Fig. 3 and Figure A1 in Online Appendix A.1). Focusing on the EU, intra-EU exports of WB dominate (Panel C of Fig. 3). In fact, as represented in Fig. 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. It is interesting to read this descriptive evidence in the light of the generalized adoption of EPR on WB in the EU, but also in the light of two other factors that characterize European waste management systems, namely strict environmental regulations and the advanced level of treatment facilities and technologies.

The second source of information we use 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 systematization of this heterogeneous information from legislative and technical sources is a relevant contribution *per se*, as it provides a common framework to quantitatively analyze the adoption of EPR on waste trade in a cross-country perspective. This information covers 89 countries, of which 48 implemented the EPR policy during the period under study and 41 did not. Table A2 in the Online Appendix A.3 reports: (i) the countries for which it was possible to collect information on the adoption

of EPR on WB; (ii) the year and the name of the regulation introducing the policy, in the case of adopting countries; and (iii) 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 from the perspective of the exporting country. Therefore, our variable EPR_{ey} is a dummy that takes a value of 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, $Dist_{ei}$, where the distance is measured between the most populated city in each country. Among the bilateral variables, we consider whether exporter and importer share a common border by including a dummy that equals to one if countries are contiguous ($Contig_{ei}$); whether the country pair was ever in colonial relationship ($Colony_{ei}$); and whether countries have the same official language ($Language_{ei}$). Among the variables capturing characteristics of the single country, we consider the level income, proxied by the GDP_{ey} and GDP_{iy} , respectively. Table 1 describes the variables included in the analysis, along with some descriptive statistics that also allow us to grasp the relevance of the phenomenon under study. For instance, WB_p represents the share of trade flows in our dataset that are related to WB, and EPR_{ey} represents the share of observations for which the exporting country is an EPR adopter. By merging the three sources of information, BACI, EPR and Cepii Gravity, we obtain a final sample of about 1.65 million observations, covering 89 exporting countries 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

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

Table 1 Variables' names, definitions and sources

	Variable	Description	Mean	Median	SD	Data source
Dependent variable	Quantity _{ep_y}	Quantity of waste product exported (tonnes)	5343.223	18.133	62,208.592	BACI
Treated group and policy	WB _p	Dummy for waste batteries	.011	0	.105	BACI
	EPR _{ey}	Dummy for extended producer responsibility policy	0.397	0	0.489	Several (see online app.A.3)
Gravity variables	Dist _{e_i}	Simple distance between most populated cities (km) in <i>e</i> and <i>i</i>	4550.128	2495.000	4323.265	Cepii gravity
	Contig _{e_i}	1 if countries <i>e</i> and <i>i</i> are contiguous	0.145	0	0.352	Cepii gravity
	Colony _{e_i}	1 if countries <i>e</i> and <i>i</i> ever were in colonial or dependency relationship	0.067	0	0.249	Cepii Gravity
	Language _{e_i}	1 if countries <i>e</i> and <i>i</i> share a common official or primary language	0.138	0	0.345	Cepii gravity
	GDP _{ey}	GDP exporter (current thousands US\$, log)	20.527	20.595	1.702	Cepii gravity
	GDP _{iy}	GDP importer (current thousands US\$, log)	19.303	19.475	1.997	Cepii gravity

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

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 main 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 that trades WB. To investigate this effect, we rely on a difference-in-differences (DiD) approach that compares the trade flow of WB (the "treated" group) with the trade flow of other waste products not targeted by the policy, before and after its adoption 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. 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:

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

where $\ln \text{Quantity}_{eipy}$ is the logarithm of the quantity²⁰ 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 WB_p is a dummy identifying waste batteries. Our interest lies in the estimation of β_3 , which gives us the difference in the impact of the EPR policy on the exports flows of WB_p relative to the control group made up by all the other waste products.

There are several econometric issues that may arise when estimating the gravity model.²¹ The first problem lies in the area of omitted variable bias, since there are some variables, such as the multilateral resistance terms, which are unobservable. Indeed, in attempting to provide a theoretical underpinning to the gravity equation based on a constant elasticity of substitution (CES) demand function, Anderson and van Wincoop (2003) have shown that the volume of bilateral trade is affected by trade impediments at the bilateral level (referred to as bilateral resistance), as well as the relative impact of these impediments compared to those of other countries (referred to as multilateral resistance).²² Since this contribution, failing to include a multilateral resistance term in the gravity equation is seen as a significant source of bias and a crucial issue that researchers must address

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

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

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

in their estimations.²³ As is standard in the literature, we use exporter-time fixed effects (ω_{ey}) and importer-time fixed effects (ω_{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 EPR_{ey}), institutions, and exchange rates. While the inclusion of time varying exporter and importer fixed effects allows to account for the multilateral dimension of the gravity model, another source of bias could arise due to time-invariant bilateral trade costs, both observable and unobservable. For instance, trade policy variable, such as Regional Trade Agreement, 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 (ω_{ei}) fixed effects, which control for all time-invariant bilateral trade costs and will mitigate this endogeneity concern. The inclusion of the set of pair fixed effect absorb all bilateral time-invariant covariates in Eq. 1, but has the advantage of accounting for any unobservable time-invariant trade cost components.²⁴

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

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

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

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

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

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

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

4.1.2 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 has significantly different characteristics from the control group; in other words, the common trend assumption may not hold. To address such a concern, an alternative specification would be to compare exports of waste batteries from a country with an EPR policy with exports of the same product from another country without an EPR policy. While this approach would address concerns about a possible selection bias, it opens the door to other unaccounted for confounders due to country-specific factors. A DDD approach allows us to exploit all sources of variation. Exports of WB before/after the imposition of the EPR policy are compared with the performance of the same product exported by countries not imposing the policy, and with different products exported by the same country that adopts the policy (all other waste products in our case). The DDD equation takes the following form:

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

where, with respect to the previous models, we also add product-year fixed effects (ω_{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, ω_{eip} captures the average export performance of each product in a given country-pair (so that the interaction captures variation over time), ω_{ey} and ω_{iy} refers to average origin- and destination-time effects (thus exploiting variation across products within the same country), while ω_{py} controls for product-time effects and thus lets us compare the same good traded by different countries. This complete set of fixed effects is meant to saturate all possible sources of variation unrelated to the policy.

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

4.2 Econometric Results

4.2.1 Baseline Results

We present in Table 2 the estimates of the empirical model specified in Eq. 1. Col. (1) shows the estimates of the ‘augmented’ gravity equation estimated by OLS, after controlling for annual common shocks, time-invariant unobserved heterogeneity at the product

level, and vectors of exporter and importer fixed effects. While the coefficient on the adoption of the EPR policy by the exporting country (EPR_{ey}) shows that bilateral trade in all other waste products decreases after the adoption of the policy, this effect is counteracted and even reversed for WB, given the magnitude and sign of the estimated coefficient of the interaction term ($EPR_{ey} \times WB_p$). After the adoption of the EPR policy by the exporter country, the flow of trade in WB has increased more than the flow of the other waste products, *ceteris paribus*. This is an interesting result, which suggests that the responsibility on producers leads to an increase in export of WB. The standard determinants of bilateral trade flows show the expected signs. In particular, the positive coefficient of $\ln GDP_{ey}$ shows that larger economies produce more waste and have more to dispose of, which increases the quantity exported. As for the positive coefficient of $\ln GDP_{iy}$, 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 $\ln Dist_{ei}$ shows that as the geographical distance between the pair of trading countries increases, the trade flow of WB between them decreases. On average, a 1% increase in the distance between the two trading countries reduces trade by 0.69%. Contiguity is relevant too. If the two trading countries share a common land border, trade in WB increases by about 190%.²⁵ Putting the evidence on distance and contiguity into perspective, this shows that transportation costs for waste products in general (including WB) are not negligible, as suggested by (Kellenberg 2012). Pairs of trading countries that share the same language or are linked by colonial history trade more waste on average, as indicated by the respective coefficient estimates.

In col. (2), the vectors of exporter-year and importer-year fixed effects are included (together with product fixed effects) to take multilateral resistance terms into account. Due to the inclusion of these effects, the coefficients of the country time-variant characteristics, including the dummy capturing the adoption by the exporter country of the EPR policy, cannot be identified. The coefficient on the interaction term 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 than those in col. (1) and col. (2). In particular, the DiD coefficient suggest that the volume of WB shipped increases by about 86% more than the volume of other waste products after the adoption of the EPR policy by the exporting country.

In the last three columns (col.4–6) of Table 2, we add a vector of product-year fixed effects to saturate the empirical model for all possible sources of variation at the product-time level unrelated to the policy. The coefficient of the interaction term in col. (4) is still positive and similar to that in col. (3), which reassures that the results are not driven by uncontrolled time-varying differences between WB and other waste products not affected by the policy. In cols. (5) and (6) we examine the possible additional effect that the adoption of the EPR policy by the importing country may have on the main relationship of

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

interest, i.e. the impact of the EPR policy on the export flows of waste batteries.²⁶ By controlling for the interaction between EPR_{iy} and WB, we want to exclude the fact that our main coefficient of interest, i.e. β_3 , is erroneously capturing an effect on WB exports that is actually driven by the importer's adoption of EPR on WB. Col. (5) shows that export flows of WB increase significantly more than other waste flows after EPR adoption, even when the importing country also chooses to adopt the same kind of policy. The coefficient of the triple interaction between EPR_{ey} , EPR_{iy} and WB_p , shown in col. (6), indicates that there is no additional complementary effect played by the joint adoption of the EPR policy by both the exporting and importing country. We further explore the role of different relevant policies adopted by the importing country in Sect. 4.2.3 to examine the characteristics that make it easier for a country to import WB.

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

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 Sect. 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) in Table 3, 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 coefficient on the interaction term ($EPR_{ey} \times WB_p$), 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

²⁶ We thank an anonymous reviewer for bringing this issue to our attention. In this article, we focus on the relationship between EPR and WB exports, considering the adoption of EPR in importing countries only as a control factor. Nevertheless, our results, which suggest a positive correlation between WB imports and the country's decision to adopt EPR, leave room for further investigation of this other relationship.

²⁷ The difference in the number of products considered as waste is due to the fact that Kellenberg and Levinson (2014) use 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 the EPR policy on the export of Waste Battery: robustness checks

Dep. var. In Quantity _{<i>eipy</i>}	Waste products as in Kellenberg and Levinson (2014)													
	(1)	(2)	(3)	Fake treat	(4)	Fake treat	HW _{<i>p</i>}	(5)	Only WB Products	(6)	Only countries with EPR	(7)	Restricted sample	(8)
EPR _{<i>cy</i>}									0.525***	(0.179)				
EPR _{<i>cy</i>} × WB _{<i>p</i>}	0.342***	0.703***						0.756***			0.619***		0.274*	(0.143)
EPR _{<i>cy</i>} × HW _{<i>p</i>}	(0.136)	(0.212)	0.095	-0.099	(0.061)									
ω_{ey}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	Yes	Yes	Yes	Yes
ω_{iy}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
ω_{eip}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
ω_{py}	No	Yes	No	Yes	Yes	Yes	Yes	Yes	No	No	No	No	No	No
Adj. R ²	0.709	0.713	0.710	0.714	0.714	0.714	0.760	0.760	0.618	0.618	0.737	0.737	0.787	0.787
No. of Obs	879,862	879,851	862,813	862,802	862,802	862,802	245,287	245,287	15,756	15,756	1,207,329	1,207,329	603,707	603,707

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

Convention, and exclude the HS6 code referring to WB from the list. The 13 HS6 codes defined in this way are described in Online Appendix A.2, Table A1. Col. (3) and col. (4) in 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 col. (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 Sect. 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) 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) 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 on the treated (ATT) 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 de Chaisemartin and D’Haultfœuille 2020; Goodman-Bacon 2021; Callaway and Sant’Anna 2021; Athey and Imbens 2022, among others). Moreover, according to recent econometric 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 this possible source of bias by restricting the analysis to a sample of observations that should be less affected by this problem. 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 the interaction terms

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

is in line with the main results shown in Table 2. The magnitude of the impact of the policy remains large in this model, predicting an increase of the volume of WB exported after EPR implementation about 32% higher than that of other types of waste.²⁹

A final methodological concern is related to the use of an appropriate estimation strategy that takes into account the large numbers of zero trade flows. The gravity model (Anderson and van Wincoop 2003), which Eq. 1 is based on, expresses trade as the multiple of strictly positive variables, and it does not take into account the information contained in the zero trade flows because these observations are simply dropped from the estimation sample when quantities are 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 relevant 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 Tenreiro (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, thus considering both the *intensive* and *extensive* margins of trade. Table 4 presents the estimates obtained with the PPML estimator. The first column provides our baseline coefficient, which, once again, is positive and significant. This result reassures us regarding our main results not being driven by a selection issue, possibly due to unobserved choices by trading countries. The Poisson regression model is also used for two relevant robustness checks performed on the linear model, namely adopting a finer definition for the control group as suggested by Kellenberg and Levinson (2014) (col. 2), and restricting the sample of observations to those countries that adopt the policy sooner or later (col. 3). The sign and statistical significance of the PPML estimates are consistent with those shown in Table 3 for the linear model.

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 robustness checks then confirmed this finding. Clearly, trade data do not allow for a deeper and more precise identification of the mechanisms responsible for the identified phenomenon, but it is possible to explore the possible channels that could be associated with such a change or that are likely to be excluded.

Differences in environmental regulation across countries may constitute a source of comparative advantage, i.e. lower costs for countries with lower levels of regulation in

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

Table 4 The effect of the EPR policy on the export of waste battery: robustness checks - PPML

Dep. var. quantity _{eipy}	Baseline	Waste products as in Kellenberg and Levinson (2014)	Only countries with EPR
	(1)	(2)	(3)
$EPR_{ey} \times WB_p$	0.392** (0.195)	0.349** (0.195)	0.380** (0.156)
ω_{ey}	Yes	Yes	Yes
ω_{iy}	Yes	Yes	Yes
ω_{eip}	Yes	Yes	Yes
No. of Obs	6,064,484	3,167,121	4,460,999

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

terms of attracting flows of waste (Kellenberg 2015). The literature refers to this phenomenon as “waste haven hypothesis”, suggesting that stringent environmental regulations in one country may lead to the relocation of polluting industries to countries with less stringent regulations. In order to test this possibility, we explore whether the effect of the EPR policy on the exports of WB differs depending on the characteristics of the importing countries. First, based on the idea that countries with lower GDP per capita typically have less developed regulatory frameworks and enforcement mechanisms, we add to our most demanding specification with exporter-year, importer-year, exporter-importer-product, and product-year fixed effects, a triple interaction term ($EPR_{ey} \times WB_p \times D_{iy}^{Dev}$). The indicator variable D_{iy}^{Dev} takes value equal to 1 when the importing country is a developed country (D_{iy}^{Dev}), i.e. when it ranks in the top half distribution of countries by GDP per capita.³⁰ As shown in col. (1) of Table 5, the coefficient of the triple 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.³¹ Along the same line of the exercise performed in col (1), we also add the triple interaction of $EPR_{ey} \times WB_p$ 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 triple interaction is positive, confirming that the export flow of WB has increased more towards EU countries.

These two results are in line with our expectations in consideration of a few facts, partially discussed by Theis (2021), among others. First, exports of hazardous waste are generally forbidden by enforcing international environmental agreements, like the “Basel Convention’s Ban Amendment”.³² Second, the EU Waste Shipment Regulation (2006)

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

³¹ Note that in Table 5, the pairs’ interactions between the dummy for EPR_{ey} , the dummy for WB_p , and the indicator of interest are always included in each column, but these are not shown to save space. The full table is available from authors upon request.

³² 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.

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

Dep. Var	ln Quantity _{eipy}					
	(1)	(2)	(3)	(4)	(5)	(6)
$EPR_{ey} \times WB_p$	0.168 (0.271)	0.246 (0.205)	-0.226 (0.369)	-0.195 (0.296)	0.703** (0.251)	0.191 (0.297)
$EPR_{ey} \times WB_p \times D_{iy}^{Dev}$	0.593** (0.267)					
$EPR_{ey} \times WB_p \times D_i^{EU28}$		0.770*** (0.213)				
$EPR_{ey} \times WB_p \times EPS_{iy}$			0.299*** (0.138)			
$EPR_{ey} \times WB_p \times WMG-EPI_i$				0.012*** (0.003)		
$EPR_{ey} \times WB_p \times Gradient Patent_{iey}$					0.219*** (0.070)	
$EPR_{ey} \times WB_p \times Gradient Facilities_{ie}$						0.264** (0.119)
ω_{ey}	Yes	Yes	Yes	Yes	Yes	Yes
ω_{iy}	Yes	Yes	Yes	Yes	Yes	Yes
ω_{eip}	Yes	Yes	Yes	Yes	Yes	Yes
ω_{py}	Yes	Yes	Yes	Yes	Yes	Yes
Adj. R^2	0.740	0.740	0.743	0.738	0.743	0.757
No. of Obs	1,568,914	1,568,959	867,849	1,510,714	1,001,259	458,745

Observations are at the exporter-importer-product-year level. Note that all regressions include but do not report all the pairs' 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

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 Fig. 3 Panel D, not only the EU is basically self-sufficient in terms of WB and, in general, hazardous waste treatment, but it is also a net importer of these types of waste (European Commission 2015; Giosuè et al. 2021). The results of col. (1) and (2) can be interpreted in the light of Kellenberg (2015) and Falkowska (2018) argumentations. Larger economies have more advanced recycling programs, waste management markets, and technologies to recover materials even from hazardous waste. In fact, not all hazardous waste is intended for disposal. Waste containing lead, for example, is considered hazardous under the Basel Convention, yet lead is a highly recyclable waste product. It turns out indeed that lead-acid batteries, which are since long the most common type of battery on the market, are economically recycled (and manufactured) in Europe (European Commission 2019).³³ Thus,

³³ The situation is different for lithium-ion batteries (LIB). Currently, China accounts for 73% of global recycling capacity of LIB (Moisé 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.

larger economies may have greater demand for recyclable wastes, despite their stricter environmental regulations when comparing to developing countries.

To test further the occurrence of a waste haven phenomenon, we revert now to two direct measures of environmental policy. The Environmental Policy Stringency index (EPS) developed by the OECD (Kruse et al. 2022) has become a widely used tool for policy analysis, covering three decades from 1990 to 2020, and 40 countries.³⁴ As a composite index (Brunel and Levinson 2016), OECD EPS compresses the multidimensionality of environmental regulations (in this case, 13 policy instruments focusing on climate change and air pollution) in a single indicator. The waste management indicator (WMG) provided within the Environmental Performance Index (Wendling et al. 2020) specifically represents a proxy for the effectiveness of waste management policies in a country.³⁵ Col. (3) and col. (4) in Table 5 show positive and significant coefficients for the triple interactions $EPR_{ey} \times WB_p \times EPS_{iy}$ and $EPR_{ey} \times WB_p \times WMG-EPI_i$, respectively. Hence, we do not find evidence of a waste haven effect. Rather, WB exports after EPR adoption appear to be mainly directed towards countries with relatively strict environmental and waste management regulations, reinforcing the thesis that these countries retain a comparative advantage in WB treatment thanks to their more developed waste management systems and technologies.

Since, as anticipated, the location of treatment facilities and the availability of technology might be 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 compute the gradient in the ratio of patents that are specific to batteries recycling to the total number of patents related to recycling. These data are based on the OECD ENV-TECH classification (Hašič 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 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) is calculated at 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.³⁶ This figure is taken from Eurostat, so it covers European countries only. In order 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:

³⁴ The indicator ranges from 0 to 10. See Kruse et al. (2022) for the list of countries for which EPS is available.

³⁵ This indicator ranges from 0 to 100 and it was first provided in the 2020 version of the EPI. More information on the indicator and the list of the 181 covered countries see Wendling et al. (2020)

³⁶ To the best of our knowledge, data on facilities specialising in battery recycling, covering several countries are not available.

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

Dep. Var	InQuantity _{expp}							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
$EPR_{ey} \times WB_p$	0.726*** (0.212)	0.724*** (0.211)	0.744*** (0.212)	0.774*** (0.208)	0.708*** (0.213)	0.777*** (0.224)	0.782*** (0.216)	0.778*** (0.214)
$EPR_{ey} \times WB_p \times Avg.Share_i^{Antimony}$	-0.030*** (0.007)							
$EPR_{ey} \times WB_p \times Avg.Share_i^{Cobalto}$		-0.104* (0.060)						
$EPR_{ey} \times WB_p \times Avg.Share_i^{Graphite}$			-0.034*** (0.0.008)					
$EPR_{ey} \times WB_p \times Avg.Share_i^{Lead}$				-0.059*** (0.015)				
$EPR_{ey} \times WB_p \times Avg.Share_i^{Lithium}$					-0.026*** (0.008)			
$EPR_{ey} \times WB_p \times Avg.Share_i^{Manganese}$						-0.119** (0.0.025)		
$EPR_{ey} \times WB_p \times Avg.Share_i^{Nickel}$							-0.119*** (0.039)	
$EPR_{ey} \times WB_p \times Avg.Share_i^{Zinc}$								-0.069*** (0.021)
ω_{ey}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
ω_{iy}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
ω_{expp}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
ω_{py}	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Adj. R ²	0.740	0.740	0.740	0.740	0.740	0.740	0.740	0.740
No. of Obs	1,568,959	1,568,959	1,568,959	1,568,959	1,568,959	1,568,959	1,568,959	1,568,959

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

$$\text{Gradient} = E_{ie} = \frac{E_i - E_e}{\frac{E_i + E_e}{2}} \quad (3)$$

Note that, as explained above, in the case of the gradient related to the ratio of patents with a domain in WB recycling, E_{iey} , has also a time (year) variability, while it does not (E_{ie}) in the case of the gradient related to the number of facilities.

Col. (5) and col. (6) show some interesting results. Indeed the increase in exports of WB has been higher towards importing countries with a comparative advantage in terms of patent ratio for recycling batteries, and countries with a higher amount of general recycling facilities with respect to the exporting country. These results suggest that differences in the technological endowment in the domain of (WB) recycling are a relevant driver of the increase in trade in WB after the adoption of the EPR policy by the exporting country.

We conclude the analysis of the mechanisms that determine the direction of WB flows after the adoption of the EPR, by considering the possibility that WB are imported for recovering material resources of which the importing country is poor. Indeed, imports of waste could substitute those of virgin raw materials (Dussaux and Glachant 2019). To explore this possible determinant of WB trade flows, as a first step we identified the most common types of batteries placed on the market in the last two decades and their material basis. Portable batteries are typically based on zinc and manganese, or on nickel (Stahl et al. 2019; Sayilgan et al. 2009); lead-acid are the most common battery chemistry in the automotive and industrial applications (Stahl et al. 2019); lithium-ion batteries, which often contain significant quantities of cobalt, are used in electric vehicles and as portable batteries (Alves Dias et al. 2018); antimony and graphite are also among the most common materials in batteries (Huisman et al. 2017). Subsequently, we collected information from the United States Geological Survey on countries' global share of production of the identified batteries minerals, in order to distinguish the countries which are abundant in those materials from those which are not. We then estimate a model that includes an interaction between $EPR_{ey} \times WB_p$ and countries' share of global supply for a certain mineral. Table 6 show negative and significant coefficients for this triple interaction coefficient.³⁷ This result indicates that, on average and *ceteris paribus*, following the adoption of the EPR regulation in an exporting country, higher WB trade flows (with respect to other waste products) are directed towards countries which are not intensive in the extraction of batteries minerals. This may indicate that WB are imported to recover materials that a country does not have or does not extract. This interesting finding is in line with the observation that battery minerals are mostly mined in developing countries, while, as discussed above, WB are mainly directed to developed countries.

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

³⁷ The share of mineral supply is calculated as a country's average for the years 2001, 2010 and 2019 to minimize the loss of observations due to missing values for specific minerals and countries in the Geological Survey, and to capture a structural natural resource characteristic of the country.

the 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 and to countries with relatively stringent environmental regulations, the policy does not seem to have promoted a waste haven effect. Rather than loosen environmental regulations, the level of sophistication of the waste management system, both in terms of patents and facilities, seems to provide a stronger comparative advantage in attracting WB flows. The EU, for instance, is not only basically self-sufficient in WB management, but also a net importer. Lastly, countries' availability of the mineral resources typically contained in batteries was tested as a factor for WB imports. Following the adoption of the EPR policy, WB flows are mainly attracted by countries which are not intensive in the extraction of batteries minerals. Indeed, these resources are mostly mined in developing countries. This finding hints to the fact that WB might be imported for the recovery of materials, possibly in substitution of virgin materials imports.

These results contribute to the scarce research on the management of WB and the impact of related policies, which is fundamental for evaluating and planning circular economy strategies and investments in this strategic sector. In conclusion, our paper presents the idea that EPR can indirectly support "waste marketization strategies" (Kama 2015; Kronenberg and Winkler 2009), i.e. creating the normative and economic conditions to turn waste into a resource, in the spirit of a circular economy. These strategies should be strengthened, especially in the case of waste flows rich in critical materials.³⁸

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 effectiveness of different policies implementing the general principle of EPR and their impact on trade

³⁸ Note that the EU, for instance, according to its CRM Act (European Commission 2023), is still projected to rely on virgin CRM, i.e. mining activities, for 85% of its demand by 2030.

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 EPR adoption on the import side would further enhance our understanding of the phenomenon. Third, this paper has only marginally addressed the analysis of the relationship between EPR on WB and innovation, and this is clearly an area where technical change and intellectual properties could play a strategic role. Fourth, the hypothesis of a waste haven effect fostered by the introduction of EPR in exporting countries could be further explored with the support of data on illegal waste exports.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s10640-024-00907-5>.

Acknowledgements This paper has benefited from the comments of participants in several seminars and conferences: Italian Association of Environmental and Resource Economists IAERE, Naples (February 2023); Bureau d'Economie Théorique et Appliquée BETA - CSI seminar, Strasbourg (February 2023); PhD SUSTEEMS seminar, Trento (March 2023); Augustin Cournot Doctoral Days, Strasbourg (May 2023); Società Italiana di Economia SIE (October 2023); workshop "Research Dialogues on the Complexity of the Energy Transition", University of Brescia and Fondazione Eni Enrico Mattei (December 2023); University of Notre Dame, Dept of Economics seminar (April 2024). We are also indebted to Stefano Bianchini, Alessio D'Amato, Marinella Favot, Gabrielle Gambuli, André Lorentz, Giovanni Marin, Camille Massié, Francesco Nicolli, Ilenia Romani, Erica Santini, Xi Sun, Marco Tomasi, and Mariangela Zoli for insightful comments. We are grateful to PatLib (Area Science Park) for valuable help with patent data, particularly, to Riccardo Priore. Marco Compagnoni acknowledges financial support from the Italian Ministry of University and Research, PRIN 2022CS5X8Y "Digital and green transitions in Italian businesses: an assessment of the main antecedents". Marco Grazzi gratefully acknowledges financial support from the Italian Ministry of University and Research (MUR) P2022LP43N - GREESCO - Green Specialization and Circularity: Constraints and Opportunities; from the Fulbright Commission and from the Nanovic Institute for European Studies at the Keough School of Global Affairs, and the Center for Italian Studies at the College of Arts and Letters, University of Notre Dame. Views and information presented are personal and do not represent the Fulbright Program, the U.S. or the Italian government. Fabio Pieri and Chiara Tomasi: the topics covered in this paper are further developed in the framework of the EU - Next Generation EU / PRIN 2022 project entitled "REWIND - Resilient Enterprises and Workers: leveraging INtangibles to address Disruptions". All remaining errors are our own.

Funding Open access funding provided by Università degli Studi di Milano - Bicocca within the CRUI-CARE Agreement.

Declarations

Conflict of interest No conflict of interest.

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