Resources, Conservation & Recycling 223 (2025) 108527

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

journal homepage: www.sciencedirect.com/journal/resources-conservation-and-recycling

Full length article

Shift to intra-EU-OECD trade enhanced environmental benefits after Basel Convention Plastic Waste Amendments

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ARTICLE INFO

Dataset link: https://doi.org/10.5281/zenodo.10810163

Keywords:
Plastic waste trade
Basel Convention Plastic Waste Amendments
Transboundary movement of waste
Life cycle assessment
Supply chain management

ABSTRACT

The Basel Convention Plastic Waste Amendments, implemented in 2021, have the potential to reshape traditional 'North-to-South' plastic waste trade patterns and their environmental impacts. We analyze plastic waste trade among 21 countries before (2019–2020) and after (2021–2022) the amendments, quantifying environmental impacts from transport and waste treatment using life cycle assessment. We find that post-amendment trade among selected EU and non-EU OECD countries increased to 71 %, up 12 percentage points from pre-amendment period, when half of the trade flowed to non-OECD Asian countries. This shift yielded modest increases of 2 % in climate and 5 % in energy benefits. Further expanding intra-EU-OECD trade could boost climate benefits by up to 12 %, mainly by reducing open burning in non-OECD Asian countries. These findings offer environmental insights into the EU's upcoming ban on plastic waste exports to non-OECD countries, suggesting future trade will likely concentrate among countries with aligned waste shipment rules.

1. Introduction

Managing plastic waste is increasingly challenging due to its widespread use across sectors and products, low virgin material costs, growing polymer complexity, significant sorting and contamination issues, and low landfilling costs in some areas (Geyer et al., 2017). This issue is intricately woven into the dynamics of the global market, with developed countries often seeking to export the plastic waste they cannot recycle or treat economically (Subramanian, 2022). Such traded plastic waste has predominantly been shipped from the Global North to the Global South (Wen et al., 2021), driven by differences such as gate fees at treatment facilities, recycling labour costs, environmental taxes, and policy stringency (European Environment Agency, 2023). In 2020, nearly 90 % of the 6.4 Mt (million tonnes) of traded plastic waste originated from the Organization for Economic Cooperation and Development (OECD) countries, with the European Union (EU) countries contributing to half of this volume. Conversely, the non-OECD Asian countries accounted for half of the total imports (Brown et al., 2022). However, the actual quality of imported plastic waste often fell short of expectations due to the presence of waste mixtures, contamination, impurities, and other factors (Meijer et al., 2021). Consequently, an inevitable portion of the imported plastic waste ended up in landfills, was burned in the open air, or ended up in oceans, raising substantial environmental concerns (Wen et al., 2021). Such mismanagement can lead to detrimental impacts on ecosystems (Meijer et al., 2021), human health (Geyer et al., 2017), and natural resources (Nava et al., 2023).

Addressing this critical issue, the Basel Convention of 1989 (Basel Convention, 1989), a global agreement regulating the transboundary movement of hazardous waste, took a significant step forward in May 2019, with specific amendments targeting plastic waste (further referred to as 'the amendments') (Basel Convention, 2019). The amendments clarified the scope of the plastic wastes subject to the prior informed consent (PIC) procedure under the Basel Convention. The key difference lay in the mandatory export requirement for 'mixed' plastic waste (entry 'Y48′) under the PIC procedure, whereas exceptions were granted for plastic waste that is 'sorted by polymer,' 'destined for recycling,' and 'almost free from contamination' (entry 'B3011′). Supported by 186 countries, the amendments triggered stringent rules governing plastic waste shipment within organizations such as the EU and OECD. In response, the EU revised its waste shipment regulation and applied a full ban on the export of mixed plastic waste (entry 'Y48′) to non-OECD countries from 2021 (European Union 2020). Similarly, to align with

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https://doi.org/10.1016/j.resconrec.2025.108527
Received 3 April 2025; Received in revised form 27 June 2025; Accepted 27 July 2025
Available online 2 August 2025
0921-3449/

the PIC procedure by the amendments, the OECD enhanced its registration system for pre-consented waste details in 2021, integrating contact details of competent authorities and pre-consented recovery facilities (OECD, 2021).

China's 2018 ban on plastic waste imports significantly reshaped the global market, leading to a 30 % to 41 % reduction in market size, as evidenced by discrepancies between reported imports and exports (see Fig. 1A). In contrast, the Basel Convention Plastic Waste Amendments and related regulations are expected to transform market dynamics. Since 2021, as shown in Fig. 1B, the 27 EU countries have substantially increased their plastic imports, surpassing the 16 non-EU-OECD countries that previously dominated the market. This shift is likely to boost plastic waste trade among Global North countries, while reducing exports to other Asian countries. Such changes may reverse the long-standing trade routes from the Global North to the Global South, which have raised concerns about environmental justice (Douglass and Cooper, 2020; Subramanian, 2022). These evolving trade patterns could also impact the environmental consequences of plastic waste transport and treatment (Wen et al., 2021), though these impacts remain largely unexplored.

Here we explore shifts in global plastic waste trade patterns from the pre-amendment period (2019–2020) to the post-amendment period (2021–2022) and their associated environmental impacts among the top 21 trading countries. Using a unit price-weighted approach, we balance bilateral trade data from the UN Comtrade database and assess the environmental impacts of international transport and waste treatment for six plastic waste types (HDPE, LDPE, PS, PVC, PET, and PP) under four scenarios through a life cycle assessment (LCA), focusing on climate change and energy resource use. The following section provides a detailed description of our methodology.

2. Methods

2.1. Research scope

We identified 21 countries that consistently ranked within the top 70 % of global plastic waste importers or exporters between 2019 and 2022, relying on data sourced from the UN Comtrade database (UN Comtrade, 2019). Trade flows among selected countries alone account for 60 % of the global trade in plastic waste during 2019–2022. The countries were categorized into three groups based on affiliation: the eight EU countries (the Netherlands, Germany, Austria, Belgium, Spain, France, Italy, and Poland); the six non-EU OECD countries (the USA, Canada, Mexico, Türkiye, the UK, and Japan); and the seven non-OECD Asian countries or regions (Malaysia, Indonesia, Vietnam, Thailand, Taiwan, Hong Kong, and China). These groups collectively accounted for substantial portions of global plastic waste imports and exports during 2019–2022, representing 72 % of imports and 77 % of exports for all EU countries, 87 % of imports and 84 % of exports for all non-EU OECD countries, and 83 % of imports and 54 % of exports for all non-OECD Asian countries.

The amendments became effective on January 1, 2021. It is worth noting that China's ban on plastic waste imports starting in 2018 may have influenced the subsequent trade market (Wen et al., 2021). As shown in Fig. 1, the China's ban significantly disrupted trade in 2018, while trade remained relatively stable afterwards. To minimize the

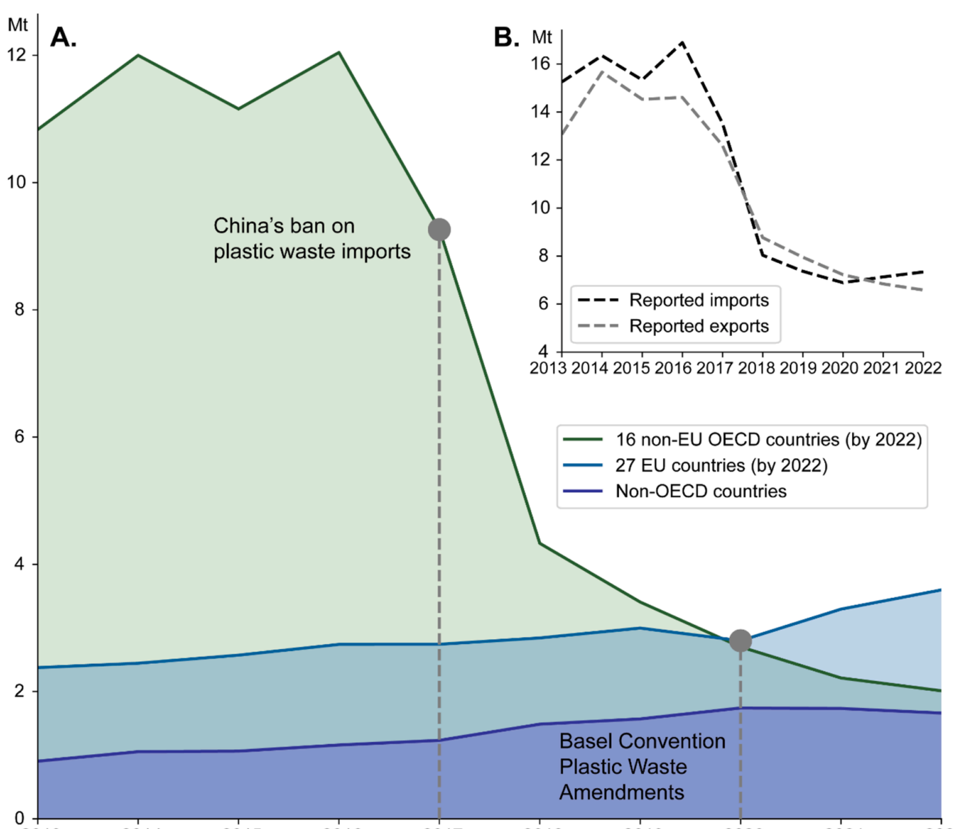

Fig. 1. Global plastic waste trade under Harmonized System code 3915 (2013–2022)
(A) Balanced global plastic waste trade by importer group, categorized as: 27 EU countries (as of 2022), 16 non-EU OECD countries (as of 2022), and non-OECD countries. Trade asymmetries between importers and exporters were addressed using a unit price reconciliation method (see Methods for details). Data are sourced from the UN Comtrade database and correspond to HS code 3915, which includes four categories of plastic waste (see Methods). To ensure consistency, country groupings reflect the 2022 membership of the EU (27 countries) and OECD (38 countries), regardless of historical membership changes. (B) Reported global plastic waste trade (original, unreconciled data) under HS 3915, also sourced from the UN Comtrade database. 'Mt' stands for million tonnes.

impact of China's ban, we selected the research year between 2019 and 2022, considering the two years before 2021 as the pre-amendment period and the two years starting from 2021 as the post-amendment period. We assume that ban-related regulations implemented in 2019, such as stringent national policies in Southeast Asia, as well as the impacts of the COVID-19 pandemic during 2019–2022, had a consistent effect across all four research years and thus served as a controlled factor when comparing periods before and after the amendments.

2.2. Unit price-weighted trade reconciliation

The United Nations Comtrade database, managed by the United Nations Statistics Division (UNSD), is one of the most comprehensive sources of global trade data, compiling statistics on over 5000 commodities from around 200 countries since 1962. It offers annual and monthly trade records detailing the reporting country, its role as either importer or exporter, the trading partner, commodity, trade volumes (primarily in kilograms, with others in units or liters (Zhang et al., 2022)), and trade values. Export trade values are typically reported as Free on Board (FOB), while import trade values are often given as Cost, Insurance, and Freight (CIF). For our study, we collected annual bilateral plastic waste trade data among 21 research countries for 2019–2022, including waste PE (HS code 391510), waste PS (HS code 391520), waste PVC (HS code 391530), and plastic waste categorized as 'Others' (HS code 391590) (UN Comtrade, 2024).

Trade discrepancies can occur where reported import volumes from one country may not match the export volumes reported by its trading partner. To address these discrepancies, we improved upon the unit price-weighted reconciliation method introduced by Chatham House for international trade in natural resources (Chatham House, 2024). This method is based on two assumptions: first, that reported trade values and volumes from both exporter and importer should be approximately the same, and second, that the calculated unit price relates to the world market prices. The 'distance' between the calculated unit price and the world average serves as the weighting factor in the reconciliation. We improved this method by eliminating the inherent trade value differences between CIF and FOB terms when measuring that distance.

We assume that the logarithm of the unit price for each commodity in a given year follows a normal distribution (Vining and Elwertowski, 1976), where the unit price is the trade value divided by the trade volume (Eq. (1)). Unlike Chatham House's method, which applies a single distribution for unit prices regardless of whether trade values are reported in CIF or FOB terms, our approach distinguishes between distributions for importers and exporters:

$$\begin{aligned} &ln\left(\frac{TV^{x}_{i,j,k,t}}{N^{x}_{i,j,k,\ t}}\right) \sim N\left(\mu^{x}_{k,\ t}, {\sigma^{x}_{k,t}}^{2}\right) \\ &ln\left(\frac{TV^{m}_{i,j,k,t}}{N^{m}_{i,j,k,\ t}}\right) \sim N\left(\mu^{m}_{k,\ t}, {\sigma^{m}_{k,t}}^{2}\right) \end{aligned} \tag{1}$$

where $TV^{x}_{i,j,k,t}$ and $TV^{m}_{i,j,k,t}$ indicate the reported trade value of commodity k in year t from the exporting country i and importing country j, respectively; $N^{x}_{i,j,k,\ t}$ and $N^{m}_{i,j,k,\ t}$ indicate the trade volumes reported by exporting country i and importing country j for commodity k in year t, respectively; $\mu^{x}_{k,\ t}$ and $\mu^{m}_{k,\ t}$ are the mean values of the unit price distributions for exporting and importing countries, respectively, for commodity k in year t; ${\sigma^{x}_{k,t}}^{2}$ and ${\sigma^{m}_{k,t}}^{2}$ are the corresponding variance.

Outliers are identified by setting a bound of three standard deviations from the mean ($\mu_{k,t}\pm 3\sigma_{k,t}$). If no outliers are found from both trading countries, we use the Z-score to calculate weighting factors in Eq. (2). Unlike the Chatham House's method, which relies on absolute deviation from the global average, $Z^{\,x}_{i,j,k,t}$ and $Z^{\,m}_{i,j,k,t}$ measure how many standard deviations the unit prices of the importer or exporter are from their respective global means $\mu^{x}_{k,\ t}$ and $\mu^{m}_{k,\ t}$. This allows us to compare deviations across the unit price distributions of importing and exporting countries, which is essential for determining the weighting factor.

$$\begin{aligned} &\text{If } \left|ln\left(TV^{x}_{i,j,k,t}\middle/N^{x}_{i,j,k,\ t}\right)-\mu_{k,t}\right| \le 3\sigma_{k,t} \text{ and } \left|ln\left(TV^{m}_{i,j,k,t}\middle/N^{m}_{i,j,k,\ t}\right)-\mu_{k,t}\right| \le 3\sigma_{k,t}: \\ &RN_{i,j,k,\ t} = w_{i,j,k,t} * N^{x}_{i,j,k,\ t} + \left(1-w_{i,j,k,t}\right) * N^{m}_{i,j,k,\ t} \\ &w_{i,j,k,t} = \frac{1\Big/\left|Z^{\,x}_{i,j,k,t}\right|}{1\Big/\left|Z^{\,x}_{i,j,k,t}\right| + 1\Big/\left|Z^{\,m}_{i,j,k,t}\right|} \\ &Z^{\,x}_{i,j,k,t} = \frac{ln\left(\frac{TV^{x}_{i,j,k,t}}{N^{x}_{i,j,k,\ t}}\right)-\mu^{x}_{k,\ t}}{\sigma^{x}_{k,t}} \\ &Z^{\,m}_{i,j,k,t} = \frac{ln\left(\frac{TV^{m}_{i,j,k,t}}{N^{m}_{i,j,k,\ t}}\right)-\mu^{m}_{k,\ t}}{\sigma^{m}_{k,t}} \end{aligned} \tag{2}$$

where $RN_{i,j,k,\ t}$ is the final reconciled trade volume from country i to country j for commodity k in year t; $w_{i,j,k,t}$ is the weight assigned to $N^{x}_{i,j,k,\ t}$; Other variables remain the same as indicated by Eq. (1).

If the calculated unit price of a trading country, whether as an importer or exporter, is identified as an outlier, full weight is assigned to its trading partner, as shown in Eq. (3):

$$\begin{aligned} &\text{If } \left|ln\left(TV^{x}_{i,j,k,t}\middle/N^{x}_{i,j,k,\ t}\right)-\mu_{k,t}\right| > 3\sigma_{k,t}: \ w_{i,j,k,t} = 0 \\ &\text{If } \left|ln\left(TV^{m}_{i,j,k,t}\middle/N^{m}_{i,j,k,\ t}\right)-\mu_{k,t}\right| > 3\sigma_{k,t}: \ w_{i,j,k,t} = 1 \end{aligned} \tag{3}$$

where all variables remain the same as indicated by Eq. (2).

To capture the nuanced end-of-life environmental impacts of different plastic types in the LCA, we further distinguished HDPE and LDPE from the original PE category, and PET and PP from the original 'Others' category, based on the historical distribution of recyclate types in importing countries (as detailed in Table S3).

2.3. Required recycling rate and domestic plastic recycling rate

The recycling rate of imported plastic waste is a crucial indicator for assessing the environmental impacts of the plastic waste trade. Higher recycling rates indicate that a greater proportion of plastic waste is being recycled, leading to lower mismanagement rates and reduced environmental harm. In contrast, lower recycling rates imply a higher risk of mismanagement, meaning more imported plastic waste may end up incinerated, landfilled, dumped, or burned—activities that contribute to varying degrees of environmental damage.

Previous studies have often assumed that the recycling rate of imported plastic waste can be proxied by a country's domestic plastic recycling rate (Wen et al., 2021). However, this assumption is problematic for two key reasons: first, imported plastic waste is usually pre-sorted to a higher degree, making it a commodity with fewer impurities compared to domestically generated plastic waste. Second, as shown in the UN Comtrade database, importing countries pay for plastic waste, reflecting a sustained incentive to ensure profits through reliable and consistent recycling operations (Li et al., 2024).

To ensure profitability, the value of recycled plastics must exceed the costs associated with importing and recycling (e.g., labour, electricity, rental) and account for physical losses during processing. In our previous work, we developed a cost-benefit model to estimate the 'required recycling rate' (*RRR*) for four types of plastic waste, PE, PS, PVC, and 'Others', over the period from 2013 to 2022 (Li et al., 2024). The *RRR* for these waste types across 21 countries during 2019–2022 is presented in Table S4. Due to data limitations, we applied the same *RRR* for PE to both HDPE and LDPE and for 'Others' to both PET and PP without further differentiation.

2.4. Scenario setting

To analyse the impact of amendments and simulate future dynamics in plastic waste trade patterns, we developed four scenarios with varying treatment structures, all based on post-amendment trade volumes (2021–2022). The environmental impacts associated with each scenario are primarily driven by the differences in end-of-life treatment shares, including recycling, incineration, landfill, mismanagement, etc., which vary by destination country. These country-specific treatment profiles were incorporated into the analysis to accurately reflect the fate of traded plastic waste (Table 1).

Table 1
Scenarios for simulating changes in plastic waste trade patterns pre- (2019–2020) and post-amendment (2021–2022) using post-amendment trade volume as a reference.

Scenarios	Trade pattern before the amendments (TB)	Trade pattern after the amendments (TA)	Enhanced intra-EU-OECD trade pattern after the amendments (ETA)	Non-trade scenario after the amendments (NTA)
Modifications	Reference trade with pre-amendment distribution (2019–2020)	Reference trade	Reference trade with enhanced intra-EU-OECD trade distribution [a]	Assuming the traded plastic waste would be treated domestically with the country's historical treatment distribution [b]
International transport included or not	Yes	Yes	Yes	No
Waste-treating countries	Importing countries	Importing countries	Importing countries	Exporting countries
Share of recycling	*RRR* across countries and plastic waste types [c]	*RRR* across countries and plastic waste types	*RRR* across countries and plastic waste types	The same recycling rate as in the country's historical treatment share distribution
Share of other treatments [d]	Historical distribution of non-recycling treatments applied to new non-recycled waste flow	Historical distribution of non-recycling treatments applied to new non-recycled waste flow	Historical distribution of non-recycling treatments applied to new non-recycled waste flow	Same as in the country's historical treatment share distribution

Note:.
[a] trade flows originating from the eight EU and the six non-EU OECD countries to the seven non-OECD Asian countries in the post-amendment period are redirected to their existing EU and non-EU OECD partners while maintaining proportional trade shares.
[b] The country's historical treatment share distribution reflects the distribution of various plastic waste treatments within a country, based on domestically generated plastic waste.
[c] We implement a minimum required recycling rate (*RRR*) for recycling imported plastic waste in trade scenarios. The *RRR* is determined by the break-even point of importing and recycling costs and secondary plastics revenues across waste-importing countries, plastic waste types, and years. Please refer to the next section for details.
[d] Besides recycling, the other six treatments include incineration with or without energy recovery, sanitary and unsanitary landfills, open dumping, and open burning.

The TB scenario maintains pre-amendment trade patterns by scaling trade flows from 2019 to 2020 to match post-amendment volumes. The TA scenario reflects post-amendment trade patterns without further modifications, covering trade flows from 2021 to 2022. The ETA scenario builds on the TA setup but introduces an enhanced intra-EU-OECD trade pattern. Specifically, trade flows originating from eight EU and six non-EU OECD countries to seven non-OECD Asian countries in the TA scenario are redirected to their current EU and OECD partners while maintaining original trade proportions. If specific plastic waste types are not traded between the exporter and its EU-OECD partners, the flows are redistributed without differentiating between specific plastic types. The non-trade scenario (NTA) assumes that traded plastic waste is treated domestically, following historical treatment share distribution.

2.5. Life cycle assessment

We conduct an attributional LCA to evaluate the environmental impact of plastic waste trade pre- (2019–2020) and post-amendments (2021–2022), focusing on international transport and waste treatment. The functional unit for plastic waste treatment involves processing 1 kg of plastic waste in one of the 21 research countries, categorized by six plastic waste types and seven waste treatment methods: mechanical recycling, incineration with or without energy recovery, sanitary landfill, unsanitary landfill, open dumping, and open burning. Additionally, the functional unit for international transport is defined as the transportation of 1 kg of plastic waste for 1 km between trading countries, using one of four transport modes: sea, air, road, and railway. Transport distances between trading countries via sea, air, and road (including railway) were measured using the CERDI-sea distance database (Bertoli et al., 2016), great-circle distance calculation based on capital latitude and longitude (Chen et al., 2004), and the Google distance matrix API (Google Maps Platform, 2022), respectively.

The boundary for mechanical recycling treatment in this study begins with sorted plastic waste and ends with plastics in their primary forms, such as pellets, granules, or flakes. This process includes the handling of recycling residues and distinguishes between plastic types (Li et al., 2024b). Subsequent processes, for example, using these secondary plastics to produce semi-finished or final products, are not included, as they are not directly connected to the plastic waste trade flows under assessment. For incineration with energy recovery, the process begins with sorted plastic waste and ends with the recovery of heat or electricity, also including residue treatment. In the case of sanitary landfills, landfill gas is captured for flaring or utilization, and leachate is collected and treated. Unsanitary landfills, on the other hand, involve waste compaction and daily covering but lack systems for gas or leachate collection and do not have bottom liners. Open dumping and open burning are unmanaged, lacking containment or environmental controls (Doka, 2018). For further details, please refer to the life cycle inventory.

Regarding life cycle inventory (LCI) analysis, we have modelled 86 LCI activities, with country-specific settings for electricity mix, mechanical recycling, incineration with energy recovery, unsanitary landfill, and open dumping. The original LCI was sourced from the Ecoinvent 3.8 database (cut-off) and the LCA Commons database. The national electricity mix was assembled and distinguished by 21 research countries in activities with electricity input. The mechanical recycling LCI includes both lower and upper bounds for water, electricity, and waste disposal amounts for six types of plastic waste. Besides electricity, country-specific settings in recycling are applied to the recycling residue treatment and the avoided virgin plastic production. The recycling residue treatment is further tied to the country's historical treatment share, while the avoided virgin plastic production is divided between Europe and the USA.

The incineration LCI is distinguished by energy recovery and non-energy recovery practices. Country-specific settings for energy recovery are based on three parameters: the net energy generation from local incinerators (Table S5), the lower heating value (LHV) of specific plastic waste types (Table S6), and the LHV of general waste sent to local incinerators (Table S5). We maintain upper and lower bounds for each parameter for sensitivity analysis. Following Gabor Doka's methodology (Doka, 2018), we categorized the 21 research countries into five climate classes based on annual water infiltration, aligning with the five subcategories under unsanitary landfill and open dump in the Ecoinvent database. The detailed LCI with country-specific parameters is available in an Excel file on Zenodo (see Data and code availability section), with instructions provided. The substitution factors for six virgin plastics, considering the mechanical and non-mechanical properties of recyclates compared to their virgin counterparts, are shown in Table S7.

For impact assessment, we focused on impact categories of climate change (CO_2-Eq) and energy resource use (fossil fuels, MJ), considering key adjustments in the LCI related to electricity mix, recycling, and incineration with energy recovery by country. We used the Environmental Footprint (EF) v3.1 method, which includes updated climate change characterization factors aligned with the IPCC Sixth Assessment Report of 2022 (European Commission, 2022). The LCA results for the remaining impact categories under EF v3.1 can be found in Figs. S2–S3 in the supplementary materials.

The LCA with parameter scenarios was conducted using Activity Browser, an open-source software for life cycle assessment (LCA) built on the Brightway LCA software package (Steubing et al., 2020).

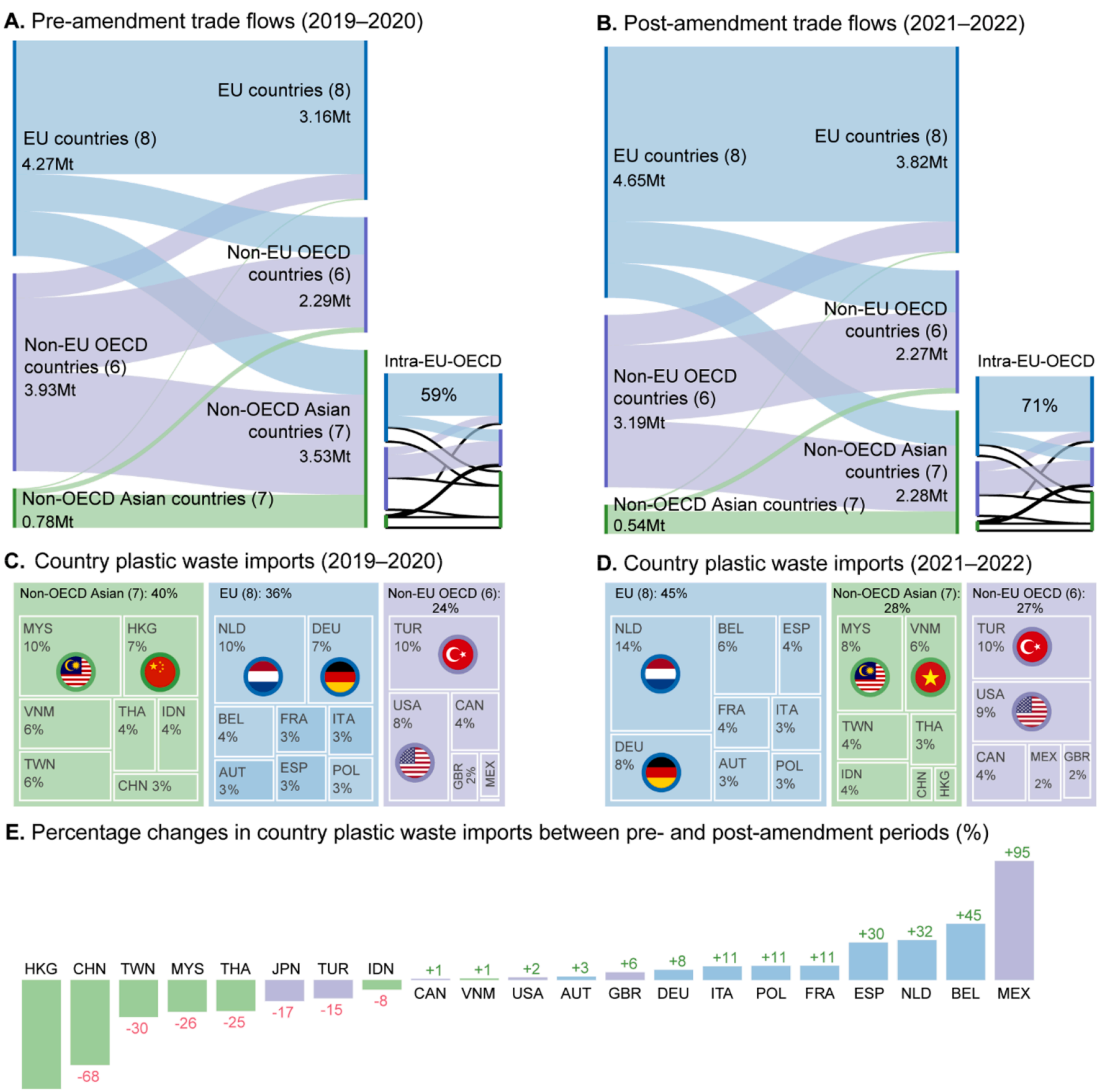

Fig. 2. Comparison of plastic waste trade flows and country imports pre- and post-amendments

Country names are represented by their alpha-3 ISO codes, with full names listed in Table S2. The top two countries are marked with their respective flags, as shown in panels (C) and (D); note that Hong Kong is represented by China's flag. For countries with an import share of 1 % or less, only the ISO codes are displayed in (C) and (D). Comparative exports for each country before and after the amendments are shown in Fig. S4. 'Mt' stands for million tonnes.

2.6. Sensitivity analysis

A one-at-a-time sensitivity analysis was conducted to determine how the alteration of seven key parameters affects the environmental impacts across impact categories and scenarios. When changing one parameter at a time, the fluctuation of environmental impacts (lower and upper bounds) is determined by two values associated with optimistic and pessimistic cases (Saltelli et al., 2005), which is defined in Table S1. The results of the sensitivity analysis are presented in Note S1 and illustrated in Figs. S1 and S2–S3.

3. Results

3.1. Growing intra-EU-OECD trade of plastic waste

Notable changes in the import market are observed following the amendments, compared with the relatively stable export market (Figs. 2A–2B). Before the amendments, the seven non-OECD Asian countries dominated plastic waste imports with a 40 % share, followed by the eight EU countries at 36 %. Post-amendment, the eight EU countries increased their share to 45 %, overtaking both the seven non-OECD Asian and the six non-EU OECD countries, with the seven non-OECD Asian countries' share falling to 28 %. This shift was also

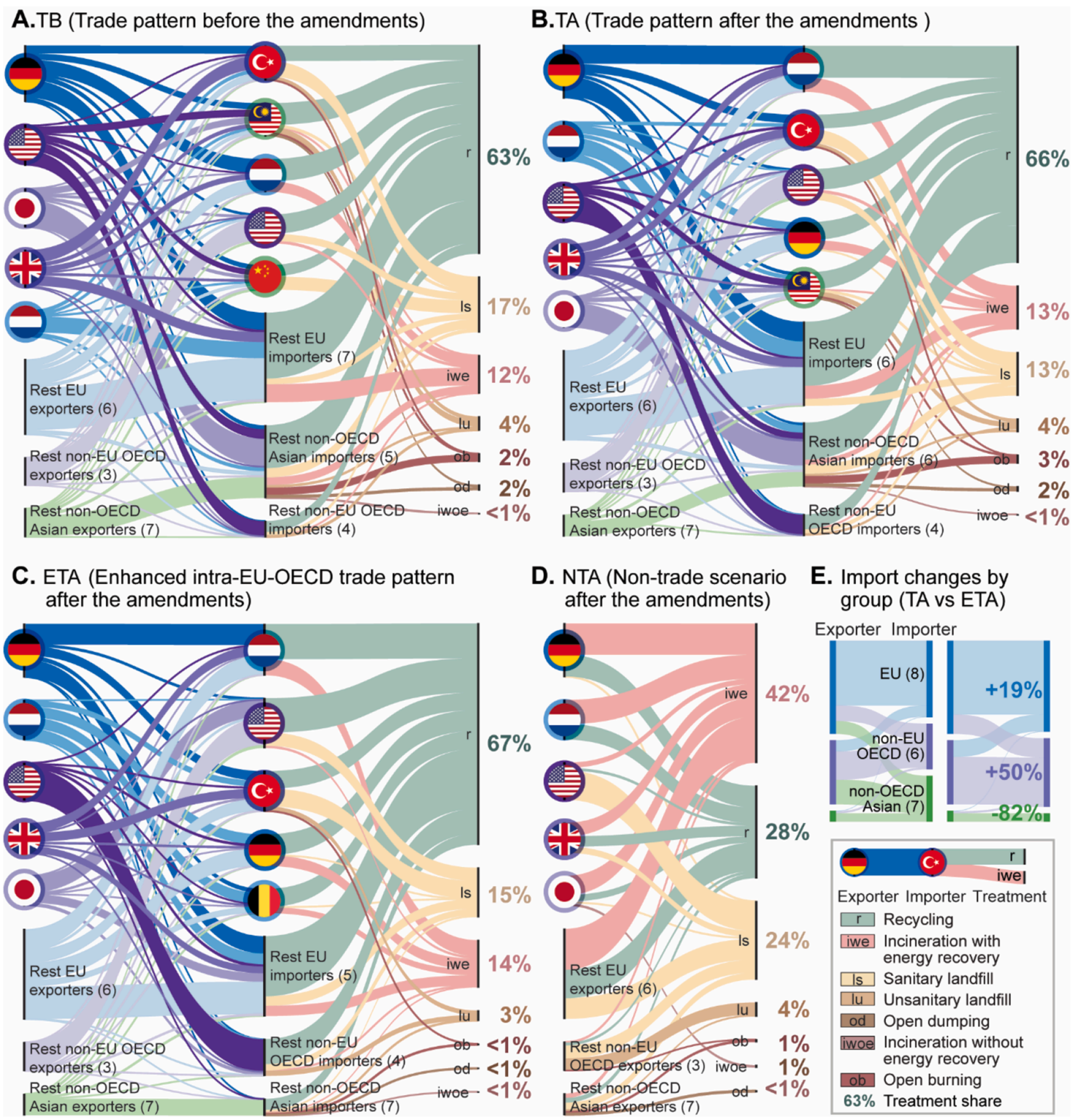

Fig. 3. Plastic waste trade flows across countries and treatment types under four scenarios
The top five importers and exporters are flagged in ascending order. The remaining research countries are grouped as the rest EU, the rest non-EU OECD, and the rest non-OECD Asian importers or exporters, with the number of countries in each group indicated in parentheses. Please note that the flag of China in (A) represents China's Hong Kong.

accompanied by a rise in intra-EU-OECD trade, compared to the traditional EU and non-EU OECD exports to the non-OECD Asian countries. In the pre-amendment period, intra-EU-OECD trade comprised 59 % of total trade, increasing to 71 % post-amendment. Meanwhile, exports from the eight EU countries and six non-EU OECD countries to the seven non-OECD Asian countries dropped significantly, decreasing by 39 % and 28 %, respectively.

The changes in trade patterns were also evident at the country level (Figs. 2C–2D). Following the amendments, significant increases in plastic waste imports were observed among the eight EU and the six non-EU OECD countries. Notably, the Netherlands emerged as the largest importer with a 14 % share in the post-amendment period (Fig. 2D), marking a substantial 32 % increase compared to the pre-amendment period (Fig. 2E). Similar import surges were noted in Mexico, Belgium, and Spain, showing significant increases of 95 %, 45 %, and 30 %, respectively, compared to the pre-amendment period. Conversely, the seven non-OECD Asian countries experienced significant reductions in imports. Most prominently in relative terms, Hong Kong saw an 87 % decrease in imports between the two periods (Fig. 2E), resulting in an import share of only 1 % (Fig. 2D), down from its previous 8 % share (Fig. 2C) as the second-largest importer among the seven non-OECD Asian countries (regions) before the amendments. Following Hong Kong, China, Taiwan, and Malaysia also experienced decreases in their plastic waste imports by 68 %, 30 %, and 26 %, respectively, compared to the pre-amendment period.

The trade flows under four scenarios are mapped to seven treatment types with varying shares (Fig. 3). As intra-EU-OECD trade increased from TB to TA and ETA scenarios, the shares of recycling rose from 63 % to 67 %, and incineration with energy recovery increased from 12 % to 14 %. While redirecting trade from the seven non-OECD Asian countries to the eight EU and the six non-EU OECD countries led to only a 1 % rise in both recycling and incineration with energy recovery (from TA to ETA), the share of mismanaged waste practices, such as open dumping and burning—more common in Asian countries—dropped sharply from 5 % to <1 %. Additionally, comparing the ETA to TA scenarios, trade flows to the six non-EU OECD countries increased by 50 %, and to the eight EU countries by 19 % (Fig. 3E). Türkiye and the USA absorbed most of the redirected trade flows that were originally destined for the

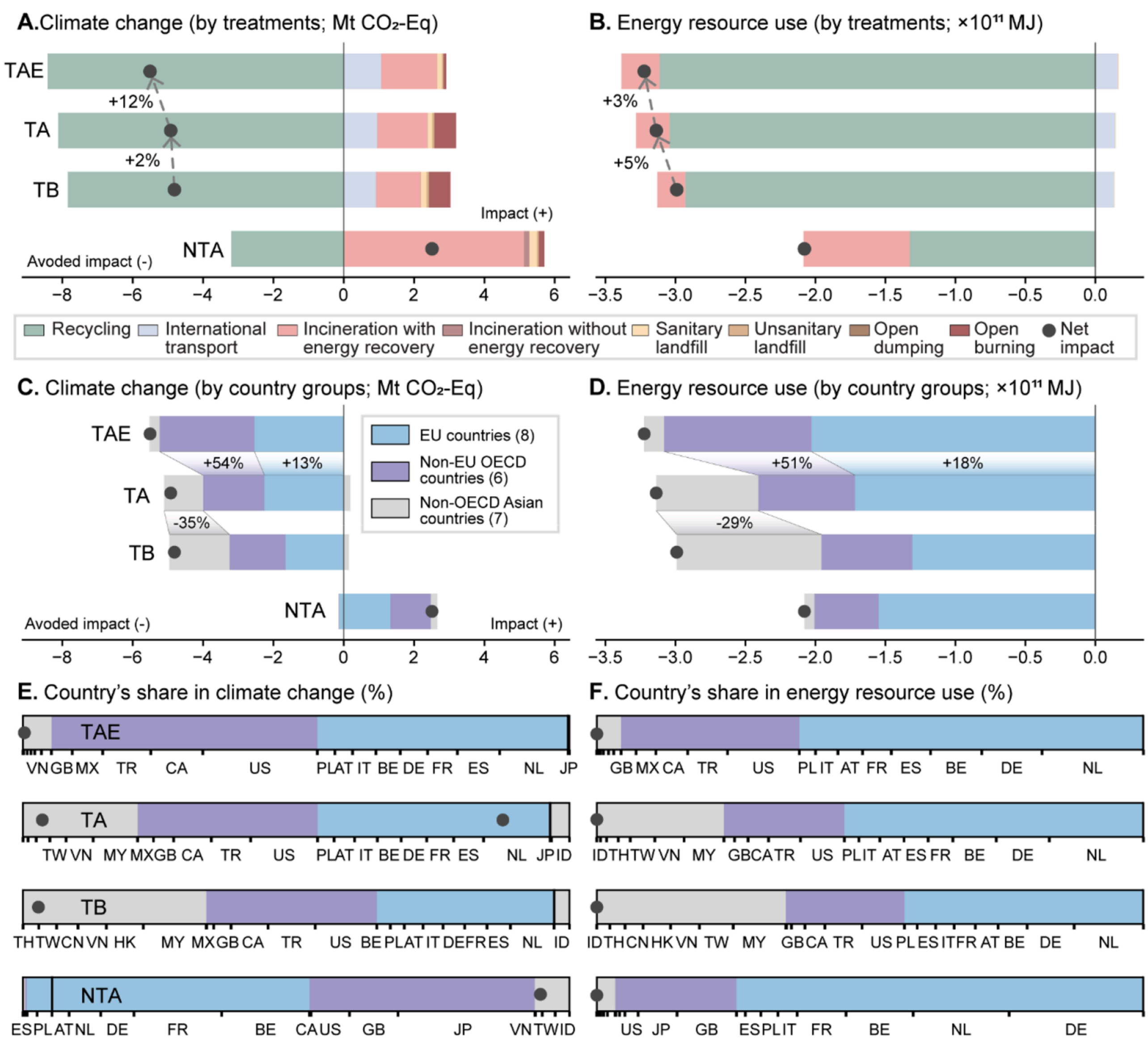

Fig. 4. Environmental impacts of climate change and energy resources under four scenarios
(A–D) Environmental impacts are illustrated using diverging bar charts, aggregating impacts by treatments or country (region) groups for both positive and negative values. (E–F) Detailed view of country shares, focusing on the data from (C–D). To avoid label overlap, labels for countries with smaller shares, which do not significantly impact the narrative, are excluded. Country names are displayed using ISO 3 codes, mapping to their full names as shown in Table S2.

seven non-OECD Asian countries from the eight EU countries and Japan in the TA scenario. In contrast, under the NTA scenario, which assumes domestic treatment of plastic waste using historical shares, the majority (42 %) would be incinerated with energy recovery, while only 28 % would be recycled (Fig. 3D).

3.2. Diverging environmental impact changes

Given the complexities of the global plastic waste supply chain and the variability in local impacts, our LCA analysis focuses on impact categories of climate change and energy resource use (Fig. 4), which is aligned with our country-specific settings for electricity mix, mechanical recycling, and incineration with energy recovery in life cycle inventory (as detailed in methods). Results for other impact categories are provided in Fig. S5 (treatment breakdown) and Fig. S6 (country or region group breakdown).

Comparing the TB and TA scenarios, we observed increased avoided environmental impacts due to trade pattern changes following the amendments. In the TA scenario, the avoided impacts on climate change and energy resources (indicated by the black dot in Fig. 4) increased by 2 % (0.11 Mt CO_2-Eq) and 5 % (14.6 gigajoules, GJ), respectively, compared to the TB scenario. This difference between these two lies in the varied sensitivity of these impact categories to the share of incineration with energy recovery. The increased share of incineration with energy recovery in the TA scenario contributes to energy conservation (negative values in Fig. 4B) but negatively affects climate change mitigation (positive values in Fig. 4A). Similar environmental benefits from the TB to TA shift were seen across fourteen other impact categories in Figure S5, except for human toxicity (carcinogenic). In this category, which is most affected by open burning, the TB scenario has higher impacts due to a 1 % higher open burning share than in TA (Fig. 3).

Redirecting the trade received by the seven non-OECD Asian countries to the eight EU and the six non-EU OECD countries in the ETA scenario further increased environmental benefits, though at varying rates across impact categories. For climate change, the avoided impact rose by 12 % (0.59 Mt CO_2-Eq) from TA to TAE, significantly higher than the 2 % increase from TB to TA. This accelerated mitigation is likely due to the reduced share of open burning, decreasing from 3 % in TA and <1 % in TAE. Despite a small 3 % share in TA (Fig. 3B), open burning contributes nearly 20 % of the total positive climate change impacts (Fig. 4A). However, the accelerated benefits from TA to TAE compared to TB to TA do not apply to all categories. For energy resources, sensitive to energy-related treatments like recycling and incineration with energy recovery, the avoided impacts increased by 3 % (8.6 GJ) from TA to TAE, slightly less than the 5 % increase from TB to TA.

Compared to the narrowing avoided environmental impacts observed among the seven non-OECD Asian countries from TB to TA and ETA scenarios, the six non-EU OECD countries are expected to see a significant increase in avoided impacts in climate change and energy resources in the ETA scenario. We allocated the environmental impacts by country (region) groups under four scenarios (Figs. 4C–4F), where the impacts of transport and treatment were assigned to each country handling the waste. As trade with the seven non-OECD Asian countries reduced following the amendments, the avoided environmental impacts for the seven non-OECD Asian countries dropped by 35 % (0.63 Mt CO_2-Eq) and 29 % (30 GJ) in climate change and energy resources, respectively. Focusing on Asia's decline in avoided environmental impacts, Hong Kong's share in the avoided impacts fell significantly, from 7 % (0.34 Mt CO_2-Eq) to <1 % (0.049 Mt CO_2-Eq) in climate change from TB to TA scenarios (Fig. 4E), in line with its plummeting trade imports (Fig. 2E).

Similarly, comparing the ETA to TA scenarios, the distinctive change in redirected imports for the eight EU and the six non-EU OECD countries (Fig. 3E) triggered varied environmental impact changes. Notably, the six non-EU OECD countries increased their avoided environmental impacts in climate change and energy resources by 54 % (0.95 Mt CO_2-Eq) and 51 % (36 GJ), respectively, significantly higher than the eight EU countries, which saw increases of 13 % (0.29 Mt CO_2-Eq) and 18 % (31 GJ). Among the six non-EU OECD countries, the USA's share in the avoided impacts grew significantly, increasing by 9 % (0.51 Mt CO_2-Eq) in climate change and by 5 % (16 GJ) in energy resources (Figs. 4E–4F).

4. Discussion and conclusion

The amendments to the Basel Convention, together with subsequent tightened regulations in the EU and non-EU OECD research countries, accelerated intra-EU-OECD trade and led to a decrease in plastic waste imports by the non-OECD Asian countries in our study. Comparing trade flows before and after the amendments, there was a significant decline of 39 % and 28 % in plastic waste trade from the eight EU countries and the six non-EU OECD countries to the seven non-OECD Asian countries, respectively. In contrast, intra-EU-OECD trade increased by 12 percentage points, maintaining a substantial 71 % share of the total plastic waste trade market after the amendments. This shift in trade patterns highlights the 'ripple effects' of the amendments, which triggered stringent waste shipment regulations within the EU and non-EU OECD countries, further restricting trade to non-member countries, including most Asian countries. Aligning with forecasts that future plastic waste trade would predominantly occur within regions rather than across them (Our World in Data, 2022), our findings suggest a nuanced change in plastic waste trade dynamics. There will be increasing trade among countries adhering to the same waste trading rules established through bilateral agreements (e.g., between the USA and Canada (Government of Canada, 2023)), regional networks (e.g., the Asian Network for Prevention of Illegal Transboundary Movement of Hazardous Wastes (Ministry of the Environment (Japan), 2003)), and international organizations (e.g., the EU Waste Shipment Regulation (European Union, 2020) and the OECD Control System for Waste Recovery (OECD, 2021)). Given that the global plastic treaty is still under negotiation (Ammendolia and Walker, 2022), we suggest that leveraging organizational waste shipment rules aligned with the international treaty could enhance its implementation in curbing global plastic pollution.

The enhanced intra-EU-OECD trade scenario (ETA) underscores the potential environmental benefits of implementing a future ban on plastic waste exports to non-OECD countries. This proposed ban was adopted in the new EU Waste Shipments Regulation on April 11, 2024, and is set to be implemented on November 21, 2026 (European Union, 2024). Comparing environmental impacts before and after the amendments reveals larger avoided environmental impacts, primarily attributed to an increased recycling share of 3 %. Although the increase in recycling share may be marginal (1 %) during the transition to an enhanced intra-EU-OECD trade pattern, as illustrated from TA to ETA scenarios, the environmental benefits are further enhanced due to reduced mismanagement in the seven non-OECD Asian countries, such as a reduced open burning rate of <1 %. Prohibiting the export of plastic waste to non-OECD countries may not significantly improve energy resources, which are more influenced by the rates of recycling and incineration with energy recovery. However, it can more effectively curb climate change due to the reduction in open burning. Compared to a complete global ban on plastic waste, which could hamper the profits of recycling industries and impact the welfare of vulnerable working groups (World Economic Forum, 2022), shifting trade from non-OECD countries presents a balanced solution that addresses both market mechanisms and environmental concerns. In addition, as intra-EU-OECD trade increases, imported plastic waste is expected to compete with domestic plastic waste for limited recycling capacity. Adapting to the future recycling demand, considering both domestic and imported plastic waste, as well as differences among polymer types, will be a significant challenge for each EU and OECD country.

The declining exports of plastic waste to Asian countries are likely to optimize the allocation of their domestic recycling capacity for both imported and domestically generated waste. We observed consistent decreases in plastic waste exports to the seven non-OECD Asian countries, with reductions of 35 % and 82 % from the TB to TA and ETA scenarios, respectively. To maintain profitability, recycling companies in these countries, which previously relied on imported plastic waste, must now pivot towards local alternatives to ensure a steady supply of raw materials for secondary plastic production (Environmental Investigation Agency, 2021). This shift necessitates investments in upgrading domestic plastic waste collection and sorting systems to maintain consistent quality compared to imported plastic waste. Expanding recycling capacity is often advocated as a solution to mitigate the negative impacts of plastic waste trade on the Global South (Lau et al., 2020). However, the market preference for imported plastic waste can quickly overwhelm newly built recycling facilities, leaving limited capacity to handle domestically generated plastic waste, as seen in Malaysia, Vietnam, and Türkiye (Environment Investigation Agency, 2023). Thus, while the reduction in plastic waste imports may initially weaken the Asian recycling market and diminish environmental benefits from material and energy recovery, the increasing intra-EU-OECD trade patterns are supporting a shift in Asia from reliance on imported to domestically generated plastic waste. This transition is also expected to stimulate the development of domestic plastic waste collection and sorting systems in those Asian countries.

Several limitations should be acknowledged in this work. First, it should be noted that our analysis is based on a narrow definition of plastic waste trade, limited to HS codes under 3915. Other codes, such as HS 3825 (residues from the treatment of waste) and HS 5505 (waste of man-made fibres), may also contain plastic waste, particularly from synthetic fibres or mixed residues. However, these commodities are less commonly designated for plastic recycling than those classified under HS 3915. Overall, this represents a limitation that should be considered when interpreting our results. Second, potential over- or under-counting of trade flows and environmental impacts may arise due to limitations in the original trade data. Although UN Comtrade seeks to standardize the reporting of the country of origin and the final destination, and separately identifies the country of consignment (as the '2nd partner'), there remains a risk that transit countries are misclassified as final trading partners, potentially distorting the calculation of environmental impacts. Third, there is no universally accepted approach for allocating the environmental impacts associated with global plastic waste trade. In this study, both transport and treatment impacts are attributed to the importing country, reflecting its dual role as the initiator (buyer) of the trade and the operator of waste treatment processes. However, other allocation methods, such as assigning or sharing impacts with the exporting country, are also possible and warrant further investigation to ensure a fair and comprehensive assessment of environmental responsibilities. Fourth, although we assume that the impacts of the COVID-19 pandemic were evenly distributed across the pre- and post-amendment periods, the degree of pandemic impact may have varied between these years. Despite this limitation, we emphasize that the amendments implemented in 2021 remain the most significant disruption in the plastic waste trade market when comparing 2019–2020 to 2021–2022. Finally, the lack of detailed composition information for traded plastic waste introduces uncertainty into the assessment of environmental impacts from waste treatment. While our life cycle inventory, adapted from the Ecoinvent database, are varied to reflect possible differences in waste quality and composition, more comprehensive, real-world composition data would be needed to fully quantify these effects.

CRediT authorship contribution statement

Kai Li: Writing – original draft, Visualization, Software, Resources, Methodology, Formal analysis, Data curation, Conceptualization. **Hauke Ward:** Writing – review & editing, Supervision. **Hai Xiang Lin:** Writing – review & editing, Supervision. **Arnold Tukker:** Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data and code availability

All the data and original code, along with instructions, have been deposited at Zenodo under the DOI 10.5281/zenodo.10810163 and are publicly available as of the date of publication. The life cycle inventory from the Ecoinvent database requires a license (Wernet et al., 2016). This study also analyzes existing, publicly available data, including bilateral trade data from the UN Comtrade database (UN Comtrade, 2019) and life cycle inventory data from the LCA Commons data repository (USDA National Agricultural Library, 2015)

Acknowledgments

This paper is supported by the project "RESOURCE: Regional project development assistance for the uptake of an Aragonese circular economy" (Grant agreement ID: 101060142), funded by the European Union's Horizon Europe Research and Innovation Programme. K.L. also acknowledges support from the China Scholarship Council (No. 202006050026). We thank Sander van Nielen at Leiden University for his assistance with the software issue. Special thanks to Reid J. Lifset at Yale University for his insightful questions on recycling rates of imported plastic waste. We also appreciate Hauke Schlesier at ETH Zürich for his valuable reflections on visualizing the avoided environmental impacts in this work.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2025.108527.

Data availability

https://doi.org/10.5281/zenodo.10810163 (This code and data are available at)

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