PLASTIC DUMPING GROUNDS: THE INTERNATIONAL INCIDENCE OF ENVIRONMENTAL REGULATION

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Abstract. Can environmental regulation reduce pollution globally, or does it merely shift environmental burdens across countries? We study China's 2017 ban on plastic waste imports, which abruptly closed the world's largest destination market for plastic waste recycled into manufacturing inputs. The ban triggered a major reallocation of global waste flows and provides a rare opportunity to study a central question in the pollution haven literature: when one country tightens environmental policy, where does the displaced environmental burden go? Displaced waste did not flow to the world's weakest regulators. Instead, it was redirected disproportionately towards Turkiye, a country with weaker waste-management outcomes than China and sufficient capacity to absorb part of the displaced trade. Using newly assembled data linking global trade flows, firm-to-firm production networks, waste-management practices, and regional air pollution, we trace the consequences of the ban from international trade diversion to firms responses and local environmental outcomes. The dominant environmental mechanism differs from the textbook pollution haven narrative. Cheaper imported plastic waste displaced domestically generated waste. Domestic waste suppliers lost buyers and increasingly disposed of unsold waste through dumping and open burning. Cities more exposed to displaced domestic waste experienced significantly larger increases in particulate air pollution. Embedding these mechanisms in a quantitative model with environmental externalities, we find that economic gains from cheaper imported inputs are modest while estimated damages from local air pollution are substantial, implying negative net welfare effects for Turkiye. Our findings show that environmental regulation can export environmental costs through market displacement, shifting pollution burdens toward countries less equipped to manage them.

JEL Codes: F18, F64, Q56

Date: May 31, 2026.

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We are grateful to Clare Balboni, Antonio Cabrales, Vasco Carvalho, Meredith Crowley, Kamala Dawar, Peter Egger, Helene Ollivier, Mathieu Parenti, Giacomo Ponzetto, Milan Qunetel, Caterina Soto Vieira, Daniel Xu, and various seminar participants for their comments. The usual disclaimer applies.

1. Introduction

Do trade and environmental policies reduce or shift pollution across international borders? A longstanding concern, formalized in the pollution haven hypothesis, is that stricter environmental policy in one country may relocate pollution-intensive activity to countries with weaker regulation or enforcement. While empirical work finds that differences in environmental regulation affect trade patterns and production location decisions, much less is known about where environmental burdens relocate following environmental policy changes and through what mechanisms they are transmitted (Copeland et al., 2022). Understanding these mechanisms is critical for evaluating whether environmental regulation reduces pollution or merely redistributes it internationally.

This paper studies China's 2017 ban on plastic waste imports as a natural experiment to examine how a major environmental policy shock reshapes global trade and environmental outcomes. The challenges posed by various types of plastic waste to the environment, human health and biodiversity have led to their inclusion in the materials covered by the Basel Convention since 2021. China's ban intensified concerns over creation of waste havens in lower-income economies, that are now front and center in negotiations over a Global Plastics Treaty.[1] Prior to the ban, China absorbed more than half of globally traded plastic waste, using recyclable plastic material as an input into manufacturing production. The ban abruptly closed the world's largest destination market and triggered huge trade reallocations. This provides a rare opportunity to study a central question: when a country tightens environmental policy, where does the displaced environmental burden go?We show that the answer differs from the textbook pollution haven hypothesis in two important respects. First, displaced waste did not flow to the countries with the weakest environmental management. Second, the largest environmental damages arose not from the relocated waste itself, but from the response of domestic firms displaced by import competition.

Total global volumes declined but a sizable share was redirected to countries with weaker environmental management. Displaced waste trade was disproportionately diverted towards

[1]See Brooks et al. (2018); Martin et al. (2021b); Bergmann et al. (2022).

Turkiye, which rapidly became the world's largest importer of plastic waste. Although Turkiye was not the weakest regulator, it had substantially higher rates of waste mismanagement than China and sufficient processing capacity to absorb the displaced trade.[2] Standard North-South pollution haven theory predicts that tightening environmental regulation in the North shifts pollution-intensive activity to countries in the South with weaker regulation. With more than two potential destinations, the displaced activity flows to the marginal absorber—the country whose willingness to accept the activity is just below that of the country that tightened, conditional on having the productive capacity to absorb meaningful volumes. Countries with even weaker regulation but limited capacity to process the activity at scale remain peripheral. This distinction is important because it implies that environmental regulation need not relocate pollution to the weakest regulator; instead, environmental burdens are shaped jointly by regulatory differences and absorptive capacity, in line with multi-country versions of the pollution haven hypothesis (Copeland and Taylor, 2003, 2004).

The idea that pollution-intensive activity relocates to jurisdictions with weaker environmental standards is well established through the pollution haven hypothesis but it has been empirically difficult to study (Taylor, 2005). Waste trade provides an unusually clean setting because countries directly trade the environmentally harmful material itself. This allows environmental regulation to be linked more directly to the location of pollution than in most pollution-intensive industries, where production, location, and pollution decisions are jointly determined. At the same time, waste differs from conventional pollutants because recycling can generate environmental benefits by substituting for virgin resource extraction. Understanding the consequences of waste trade therefore requires tracing both economic gains and different environmental costs.

Using newly assembled data on global trade flows, Turkish customs transactions, firm-to-firm production linkages, waste management practices, and regional air pollution, we follow the adjustment to China's ban from global trade diversion to firm responses nationally and environmental outcomes locally. At the aggregate level, we document a dramatic reallocation

[2] Turkiye mismanaged 47 percent of its plastic waste compared to China's 25 percent.

of plastic waste imports away from China and towards Turkiye. At the microeconomic level, however, the dominant environmental mechanism differs sharply from the textbook pollution haven narrative.

The standard channel operates as expected: cheaper imported plastic waste is redirected towards a country with weaker environmental management. Yet the largest environmental damages arise through a different mechanism. Imported waste displaced domestically generated waste. Domestic waste suppliers lost buyers and increasingly disposed of unsold waste through dumping and open burning. Cities more exposed to displaced domestic waste experienced significantly larger increases in air pollution. Environmental regulation therefore transmitted pollution internationally not only through relocation of waste-related activity, but also through the market displacement of domestic firms whose environmental management deteriorated following import competition.

To quantify the overall consequences of these adjustments, we embed our empirical estimates within a quantitative framework that aggregates firm-level responses into economy-wide welfare effects. The framework combines observed firm-level exposure to imported recyclable plastics and domestic waste generation with market-clearing conditions that recover the absolute scale of economic and environmental adjustment. Firms more exposed to imported plastic waste gained access to cheaper recyclables, bought more of them and expanded their own production following the ban. Firms generating plastic waste by-products experienced reduced waste sales but showed no systematic change to their main production. The model generates these difference-in-differences responses and embeds them in market-clearing conditions to pin down the absolute adjustments. Aggregating the difference-in-differences findings through the framework, we quantify the economic gains and the effects on pollution-relevant outcomes—imports of China-banned recyclables, virgin plastic inputs and waste disposal.

We find that economic gains from cheaper imported inputs are positive but modest. In contrast, environmental damages are substantial. Cities more exposed to displaced domestic waste experience roughly twenty percent larger increases in particulate pollution after the ban. The missing intercept of the pollution effects in cities that were not directly exposed to domestic waste displacement is determined through pollution impact factors from the scientific literature. These cities could have been affected through pollution generated from recycling and manufacturing with recycled plastic waste, that are obtained from the estimated increases in import diversion after the ban. Combining these pollution effects with estimates from the health economics literature implies damages ranging from $558 million to $655 million. By comparison, gains from lower input costs and increased production amount to no more than $116 million. The net welfare effects for Turkiye are therefore negative. The international incidence of environmental regulation therefore differed sharply from its domestic incidence. China's environmental improvements and the reduced waste burden faced by advanced economies were partly achieved through the relocation of environmental costs to countries less equipped to manage them.

Our findings contribute to three strands of literature. First, we study the international incidence of environmental regulation. While much of the pollution haven literature focuses on North-South trade patterns or unbundling the sources of comparative advantage, we study how environmental regulation in one country affects welfare in third countries. This setting also avoids many confounding policy changes that accompany domestic environmental reforms. We answer two key textbook questions about pollution havens—are pollution haven effects really an important factor and do they deserve to be a subject of international negotiation (Krugman et al., 2012, p. 289). In our setting, the pollution burden of China's ban in Turkiye is quantitatively meaningful and arises through international trade diversion that is currently the subject of several global negotiations. The quantitative analysis necessitates new methods to model environmental externalities in domestic supply and to depart from standard sufficient statistics that are inadequate in the presence of recycling. Global trade statistics do not distinguish between virgin and recyclable plastic imports, even though they differ markedly in their environmental burdens. Consequently, standard import penetration ratios of the literature are not sufficient to quantify welfare through environmental damages.

Second, we uncover a new mechanism through which pollution haven forces operate. Environmental regulation in one country can increase pollution abroad not only through relocation of pollution-intensive activity but also through displacement of domestic suppliers whose environmental management deteriorates following import competition. To our knowledge, this provides the first direct evidence for the "distressed and dirty industry" mechanism emphasized by recent work (Cherniwchan et al., 2017; Forslid et al., 2018). Relatedly, we contribute to the literature on trade and firm adjustment by showing that firms negatively affected by import competition reduce environmental compliance, complementing work that has focused primarily on pollution reductions from access to cleaner imported inputs and better export performance (Barrows and Ollivier, 2018; Gutiérrez and Teshima, 2018).

Third, we contribute to the literature on waste trade and environmental management by linking global waste shocks to local environmental outcomes and welfare. A longstanding concern in the waste-trade literature is that economic gains from importing waste may be offset by environmental costs in receiving countries.[3] We provide direct evidence on the key margin through which these costs arise—waste mismanagement—and quantify host country welfare effects.[4] This is closest in spirit to Tanaka et al. (2022), that studies how tightening US lead standards affected health outcomes in Mexico which starts to host the dirty activity.

The remainder of the paper proceeds as follows. Section 2 documents the global reallocation of plastic waste trade following China's import ban and quantifies its trade diversion impacts. Section 3 presents product/firm-level evidence from Turkiye on imports, production and waste management. Section 4 develops the quantitative framework to map the empirical findings to welfare gains through economic effects and environmental damage estimates. Section 5 concludes.

[3]See, for example, Copeland (1991); Levinson (1999); Lee et al. (2020); Bunn and Blaney (1997); Baggs (2009); Kellenberg (2012); Kellenberg and Levinson (2014); Thakur (2022) for waste trade. For a review and broader discussion, see Kellenberg (2015), Ray (2008).

[4]See The World Bank (2018a); Chong et al. (2015); Dhingra and Machin (2025) for a growing literature on waste in emerging economies.

2. Plastic Waste Trade and China's Import Ban

2.1. **Context.** Global plastic production increased dramatically, from 2 million metric tons in 1950 to 322 million in 2015 (Worm et al., 2017).[5] This expansion has generated large volumes of plastic waste that degrades slowly and persists in the environment, necessitating treatment (Brooks et al., 2018; Geyer et al., 2017; MacLeod et al., 2021; Worm et al., 2017). Managing waste in a way that is environmentally friendly—through recycling or energy recovery—is costly (Kellenberg, 2012), whereas improper management—through open dumping, uncontrolled burning, or inadequate recycling—may be cheap but generates substantial environmental and health damages.[6]

While high-income countries generate a disproportionate share of global waste, they are able to reduce their domestic plastic waste burden by exporting it to lower-income countries, where disposal and processing costs are lower but environmental standards are often weaker. This practice has been a source of controversy at least as far back as the 1980s (Copeland, 1991; Baggs, 2009; Kellenberg, 2012). Despite this, global waste trade grew rapidly. Between 1993 and 2016, global plastic waste exports increased by 817 percent, with 87 percent flowing from high-income to developing countries (Brooks et al., 2018; Li and Mu, 2024).

China emerged as the main player in the global waste market. Imported waste provided cheap inputs for manufacturing, but when it arrived contaminated—mixed with food, garbage, and other pollutants—a substantial share had to be rendered unrecyclable and needed landfilling. Often, the facilities handling waste lacked adequate infrastructure to manage it safely. As a result, the process of importing waste generated pollution, groundwater contamination, and health risks for workers. Following a severe winter haze in 2013, these

[5] By 2015, annual plastic production had reached a scale comparable to the total weight of the human population (Worm et al., 2017).

[6] Plastics are penetrating deep into the environment, and a growing body of work is evidencing its detrimental consequences for human health, biodiversity and the environment. See for example, Marfella et al. (2024); Trasande et al. (2022); Landrigan et al. (2025); Law (2017); Worm et al. (2017); OECD (2022b); United Nations Environment Programme (2025) for overviews. More broadly, waste-related activities account for a substantial share of air pollution in many urban areas, the majority of marine plastic leakage, and material shares of global greenhouse gas emissions—comparable in magnitude to emissions from aviation or shipping (The World Bank, 2018a; Jambeck et al., 2015; Brooks et al., 2018; IPCC, 2021; OECD, 2022a). See overviews in (The World Bank, 2018a; OECD, 2022a)

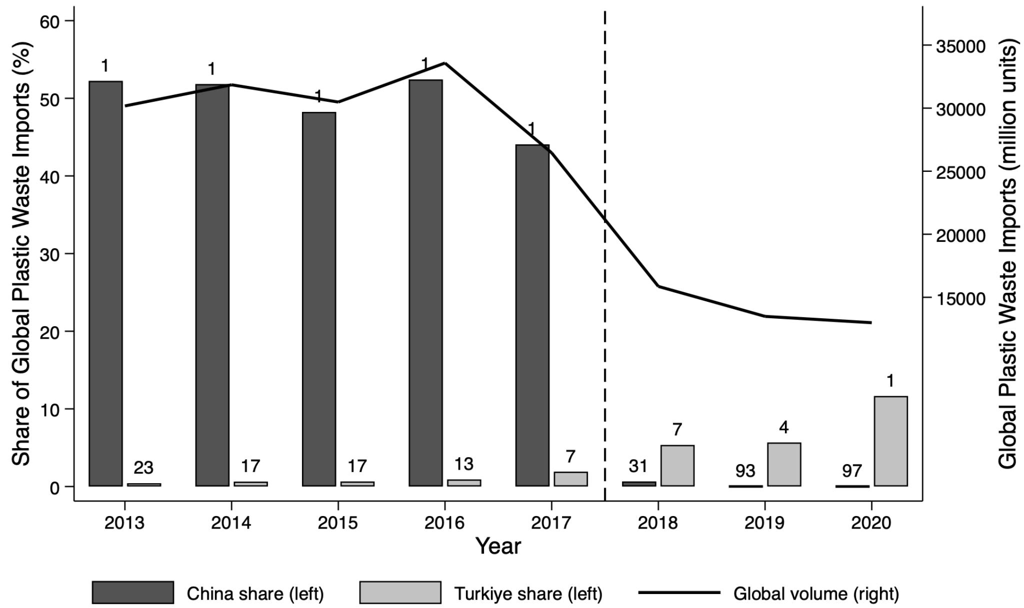
risks prompted the introduction of "Operation Green Fence", an inspection campaign aimed at reducing contaminated waste imports and tightening permits for polluting activities (Li and Takeuchi, 2023). Despite this campaign, China remained the dominant destination for plastic waste, accounting for 56% of global imports in 2016 (Brooks et al., 2018). A more drastic policy response was implemented in 2017, after public concern over plastic waste intensified, reportedly in part due to the 2016 documentary "Plastic China". This policy, known as "Operation National Sword" (ONS), imposed an outright ban on imports of 24 types of solid waste, including certain plastics and paper, while imposing very strict quality standards on others.[7] The ONS policy proved to be highly effective, triggering a sharp decline in waste imports—particularly of plastic waste, as shown in Figure 2.1.

Given China's dominant role as the world's largest importer of plastic waste, the abrupt collapse in its imports reshaped global waste flows, raising a central question: where did the displaced waste go, and did it flow toward countries better equipped to manage it, or toward those facing similar or greater environmental risks? We examine this next.

2.2. **Global Plastic Waste Trade after ONS.** Before turning to the formal analysis, Figure 2.1 summarizes the global response to ONS along two dimensions. The bars show the share of China and Turkiye in worldwide imports of ONS-banned plastic waste; the line shows total global imports. Three patterns stand out. First, prior to ONS, China was the dominant importer, accounting for over half of global imports of these products in 2015 and 2016. After 2017, China's share collapses, falling to near zero by 2019. Second, Turkiye's share, negligible before the ban, rises sharply after 2017 and reaches its post-ban peak by 2020, making Turkiye the largest single national destination for ONS-banned plastic waste in the world. Third, the global line shows that the reduction in Chinese imports was not fully offset by increases elsewhere: total global trade in these products contracts following the ban and remains below its pre-ONS level through the end of the sample. Appendix

[7]After July 2017, China progressively introduced additional restrictions on waste trade. Notably, by December 31, 2017, a new contamination standard was set, rejecting waste imports with contamination rates exceeding 0.5%. China also successively banned 16 categories of waste products by the end of 2018, with plans to ban another 16 by the close of 2019. See 99% Invisible (2021) and Mattioli (2020) for commentary.

Figure 2.1. Global Plastic Waste Trade and Importer Shares



Note: The bars show the share of global plastic waste imports accounted for by China (dark grey) and Turkiye (light grey), measured on the left axis. The solid line shows total global plastic waste imports in million units, measured on the right axis. The number above each bar reports the country's world rank in plastic waste imports for that year. The vertical dashed line marks the start of China's Operation National Sword policy in September 2017. Source of data is BACI.

Figure A.1 shows the rise of Turkiye alongside the contemporaneous reallocations to other destinations. The combined patterns are consistent with both trade destruction (a net global contraction) and trade diversion (a reallocation toward new destinations), with Turkiye absorbing a disproportionate share and being referred to as a "dumping ground" for foreign waste.[8]

To more formally quantify how global trade in plastic waste evolved after ONS, we estimate a difference-in-differences specification that exploits variation across exporter-importer pairs and product categories. We categorize countries into four groups: (i) China, which implemented the ONS policy; (ii) the largest exporters of plastic waste;[9] (iii) Turkiye, which

[8]https://www.theguardian.com/news/2025/feb/18/turkey-said-it-would-become-a-zero-waste-nation-instead-it-became-a-dumping-ground-for-europes-rubbish

[9]We refer to these countries as *Top Exporters*. They are the top 20 exporters to China of banned waste products before the ban (that have greater than average market share in exports to China in 2015 and 2016). These countries are: Australia, Belgium, Canada, Germany, Spain, France, United Kingdom, Hong Kong,

emerged as a major destination for plastic waste post-ONS, based on descriptive trends in the data; and (iv) the rest of the world (RoW). This classification allows us to capture bilateral trade dynamics between key country groups before and after the implementation of the ONS policy.

In addition, we distinguish exporter–importer pairs according to whether the origin country runs a bilateral trade surplus or deficit vis-à-vis the destination country. We compute this using pre-ONS trade flows. This split is motivated by a backhaul-shipping mechanism: industry sources note that when trade flows are imbalanced—particularly when ships travel full from Asia to Europe but return with many empty containers—shipping companies often offer discounted freight rates on the return leg in order to avoid repositioning empty containers. Plastic waste traders therefore find it cheaper to ship waste along bilateral links characterized by systematic trade imbalances, taking advantage of these lower backhaul shipping rates.[10]

If this backhaul-shipping mechanism is empirically relevant, then following ONS, we would expect to observe stronger post-policy increases in plastic waste shipments along surplus versus deficit relationships. Specifically, we estimate the following specification:

$$
\begin{aligned}
\ln x_{odht} = {} & \sum_{g\in\mathcal{G}}\sum_{g'\in\mathcal{G}}\sum_{T\in\{S,D\}} \beta^{\mathrm{HS6}}_{gg'T}\,\mathrm{Post}_t \times \mathrm{Banned}^{\mathrm{HS6}}_h \times \mathbf{1}[o\in g] \times \mathbf{1}[d\in g'] \times \mathrm{Trade}^T_{od} \\
& + \sum_{g\in\mathcal{G}}\sum_{g'\in\mathcal{G}}\sum_{T\in\{S,D\}} \beta^{\mathrm{HS2}}_{gg'T}\,\mathrm{Post}_t \times \mathrm{Banned}^{\mathrm{HS2}}_h \times \mathbf{1}[o\in g] \times \mathbf{1}[d\in g'] \times \mathrm{Trade}^T_{od} \\
& + \mathrm{Post}_t \times \text{Tariff Rate}_{odh,\,t=2016} + \alpha_{odt} + \alpha_{odh} + \epsilon_{odht},
\end{aligned}
\tag{2.1}
$$

where the dependent variable is the value of trade from origin country o to destination country d of product h in year t. The variable Post_t equals 1 for years 2017 and later, and 0 otherwise. $\mathrm{Banned}^{\mathrm{HS6}}_h$ identifies the set of plastic waste products banned under the ONS policy. $\mathbf{1}[o\in g]$ and $\mathbf{1}[d\in g]$ indicate origin and destination country groups, with groups $\mathcal{G} =$

Indonesia, Japan, South Korea, Macao, Mexico, Malaysia, Netherlands, Philippines, Slovenia, Thailand, United States and Vietnam.

[10]See Tran et al. (2021) for China and Kellenberg (2010) for a broader discussion of backhaul rates in waste trade.

$\{\text{China}, \text{Top Exporters}, \text{Turkiye}, \text{Rest of the World}\}$. Trade^S_{od} equals 1 if destination country d runs a bilateral trade surplus vis-à-vis origin country o based on pre-policy trade flows, and 0 otherwise. Trade^D_{od} equals 1 if d runs a bilateral trade deficit vis-à-vis o, and 0 otherwise. The two dummies are therefore mutually exclusive and partition origin-destination pairs by the direction of the bilateral imbalance. We also control for potential spillovers to closely related products (in the same HS 2-digit category) and for tariff changes, as well as origin–destination–year and origin–destination–product fixed effects.

The estimated changes in plastic waste trade from Equation 2.1 should be interpreted relative to the change in net trade of non-plastic products within each country group. The corresponding gravity estimates are reported in Table A.1. These relative estimates may differ from absolute effects due to general equilibrium adjustments. To recover absolute changes, we impose a balance-of-trade (BoT) condition and solve for the implied log change in non-plastic trade flows. Specifically, we combine the estimated coefficients from the gravity specification with the BoT adjustment, ensuring that the implied bilateral trade changes are consistent with aggregate trade balances across country groups.[11] The resulting estimates allow us to recover the absolute changes in plastic waste trade flows across bilateral country groups.

The recovered absolute trade flows confirm that the ONS policy triggered a large reallocation of global plastic waste trade.[12] Top Exporters (both those running a trade deficit with China and those running a trade surplus) reduced exports to China by around 91–93%.[13] In contrast, we observe a large increase in plastic waste exports from Top Exporters to Turkiye: countries with which Turkiye runs a trade surplus rose by 2,091%, while exports from countries with which Turkiye runs a trade deficit increased by 381%. A similar pattern, though

[11]Balanced trade in country k is $\sum_{o\neq k}\sum_h \Delta x_{okht} - \sum_{d\neq k}\sum_h \Delta x_{kdht} = 0$ where $\Delta x_{okht} = x_{okh,t=0}\left(\exp\left(\alpha_k + \beta^h_{ok,t=T}\right) - 1\right)$ at the inferred absolute effect α_k.

[12]Figure A.2 presents similar results using aggregate trade data. It presents an event-study where global trade in 6-digit HS products is regressed on the interaction of a dummy for China-banned products with a post-ONS dummy, controlling for product- and year-level fixed effects. Results show a sharp post-policy drop in banned plastic products and a lack of systematic pre-trends.

[13]Table A.2 provides complementary evidence by examining the share of exports directed to China. It shows that this share declined by 17.3% following the ONS policy, consistent with a reallocation of trade away from China.

of smaller magnitude, is observed for the Rest of the World. The empirical findings are consistent with the hypothesis that empty shipping containers facilitate the trade of plastic waste.

Table 1. Change in Plastic Waste Imports by Destination

Destination	**Change in imports, model implied (%)**	**Mismanagement Rate (%)**
China	−86.47	25.25
Other Destinations	+13.39	9.87
Top Exporters	+8.83	5.96
RoW	+11.66	21.55
Turkiye	+517.79	47.00
World	−38.11	14.63

Note: "Change in imports, model implied" is defined as $100 \times (X_{d,1} - X_{d,0})/X_{d,0}$, where $X_{d,1}$ is the model-implied level of imports after the policy and $X_{d,0}$ is baseline imports. The model-implied trade flows are recovered by combining the bilateral gravity estimates reported in Table A.1 with the balance-of-trade condition described in the text. "Mismanagement rate" is the share of plastic waste that is inadequately managed in each country.

To assess whether plastic waste was redirected toward countries with weaker environmental management, we aggregate the bilateral trade reallocations to the destination-group level. Table 1 reports the estimated change in imports for each destination group, along with the plastic waste mismanagement rate in each group as reported by The World Bank (2018b).[14] China, the dominant importer of ONS-banned plastic waste prior to 2017, experienced a near collapse in imports following the policy, with imports falling by 86.47% relative to the pre-ONS baseline. At the same time, imports increased across all other destinations. Taken together, non-China destinations experienced a 13.39% increase in imports relative to their baseline levels. This increase, however, was highly uneven across destinations. Turkiye experienced the largest relative increase, with imports of ONS-banned plastic waste rising by 517.79% over the baseline, making it the world's largest destination for these products.[15]

[14]The World Bank (2018b) reports waste by management practice. We follow Law et al. (2020) in classifying inadequately managed and littered waste as mismanaged.

[15]Turkiye's market share rose even further after our sample period. However, our empirical analysis is restricted to 2013–2019 to avoid confounding effects from the COVID-19 pandemic and policy changes implemented in Turkiye beginning in 2020.

Notably, Turkiye's plastic waste mismanagement rate was 47 percent, substantially higher than China's 25.25 percent. Other destinations saw more modest increases: imports into the Rest of the World rose by 11.66%, while imports into Top Exporters increased by 8.83%.[16]

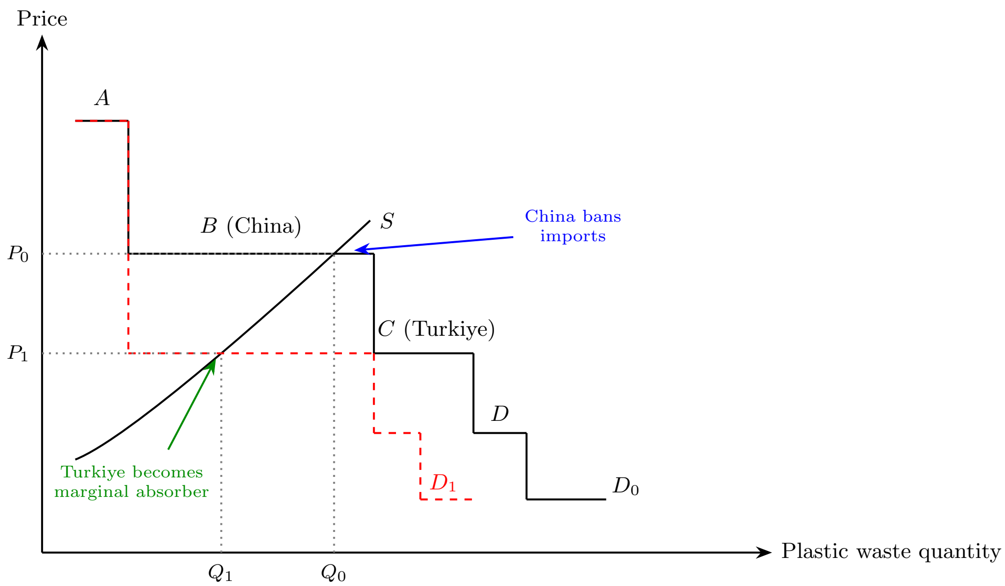
2.3. **Trade Destruction and Diversion.** Overall, global plastic waste trade declined, as the sharp reduction in China's imports was not fully offset by increases in imports elsewhere. However, the reallocated trade was disproportionately directed toward destinations with weaker waste management systems. In particular, according to The World Bank (2018b), Turkiye exhibits a substantially higher mismanagement rate than China—nearly twice as large—suggesting that the environmental consequences of the reallocation may be significant. This pattern of trade diversion, however, does not fully align with the textbook Pollution Haven mechanism in which trade would have been redirected to the countries with the weakest environmental capacity if only regulations were a source of comparative advantage. According to The World Bank (2018b), several countries—including, for example, Ethiopia, Mozambique, and Myanmar—exhibit some of the highest mismanagement rates in the world, yet did not emerge as major destinations following ONS.

The reallocation of trade was more subtle—in line with a multi-country variant where differences in locations are not purely determined by environmental regulations. Figure 2.2 illustrates this logic for the case of China's plastic waste import ban. The horizontal axis represents the world quantity of plastic waste imports; the vertical axis is the world price. World supply S is upward-sloping. World demand is a step function across importer types, ordered from highest to lowest willingness to pay and the width of the steps denote their absorptive capacity. B denotes China prior to its ban of plastic imports, C Turkiye, and D countries with the weakest regulation and limited processing capacity. When China bans imports, B's segment is removed from world demand. The figure illustrates how a unilateral tightening of environmental policy by China diverts pollution-intensive trade to Turkiye (the marginal absorber with relatively weaker regulation), rather than to country

[16]Table A.3 reports the change in imports by destination group without imposing the balance-of-trade (BoT) condition, i.e., using the relative estimates directly. The results are highly similar to the ones with the BoT condition, indicating that the ONS policy had negligible spillovers on non-plastic trade.

D (the weakest regulator, with limited absorptive capacity). Post-policy demand D_1 now has supply intersecting it on C's segment at the lower price $P_1 < P_0$ and lower quantity $Q_1 < Q_0$.

FIGURE 2.2. Trade Under the Multi-Country Pollution Haven Hypothesis



Note: Stylized world market for plastic waste imports, adapted from the pollution-haven supply-demand framework in Copeland and Taylor (2003) and based on Brian Copeland's lecture notes available at https://www.nottingham.ac.uk/gep/documents/lectures/nottm-lectures-in-int-economics/2013/lecture-1.pdf. The horizontal axis shows the world quantity of plastic waste; the vertical axis shows its world price.

The framework rationalizes the aggregate and bilateral trade patterns, but leaves open the channel through which environmental damage materializes in the receiving country. In standard treatments, the main channel is that pollution rises in the destination country because the dirty activity locates in the weaker regulation country. Our microdata-based analysis revisits this assumption directly and finds a different channel dominates the standard channel: dirtier processing of imported waste by importer firms is smaller than the pollution arising from displacement of domestic waste suppliers exposed to import competition.

To assess the environmental consequences of the reallocation of plastic waste caused by ONS, a natural approach could be to combine these trade flows with country-level mismanagement rates to infer the environmental impact of ONS. However, such an exercise would be incomplete, as it misses a key margin: how changes in plastic waste imports affect the management of domestically generated waste and virgin resource use. Understanding these margins requires recognizing how imported plastic waste is used. Imported plastic waste is not purchased for disposal but as an input into downstream production. It therefore has economic value but can also come with displacement effects when it substitutes for virgin plastic and domestically generated plastic waste. Domestic waste management and virgin resource extraction have different pollution intensities, and the substitution mechanisms therefore have important implications for how we measure environmental impacts. Trade flows do not distinguish between virgin and recyclable plastic, and domestic waste is rarely available in cross-country production statistics. Because these substitution effects are not captured by bilateral trade flows, a quantitative evaluation based on trade data alone will miss out on the overall environmental impacts.

To address this limitation, we focus on Turkiye, that emerged as the top destination for displaced waste and where disaggregated administrative data allow us to study these mechanisms directly. In the next section, we examine the environmental consequences of ONS in Turkiye through changes in domestic waste management while also assessing the economic benefits of it to downstream firms.

3. Microeconomic Impacts in Turkiye

We study whether China's ban on plastic waste imports induced a redirection of these products to Turkiye and how this shock affected Turkish firms. Section 3.1 documents the diversion of banned products into Turkiye using detailed customs data. Sections 3.2 and 3.3 study the consequences for domestic waste generators and for importing firms using firm registry, VAT and waste management survey data.

Let i index importing firms and m index domestic waste-generating firms. Plastic waste in our sample is predominantly imported by manufacturing firms that recycle these materials as intermediate inputs. More than 70% of firms importing China-banned products are in manufacturing and only about 6% are firms classified as waste management companies; fewer than 10% of importers are suppliers to waste management firms; the rest are trading firms (see Figure B.1 in the Appendix). Hence, most imported waste is used directly as an input by manufacturers rather than being processed by specialized waste firms. We exploit this feature to analyze how access to imported waste affected input costs, sales and profitability of importing firms, and the extent of displacement among domestic suppliers of plastic waste.

Our empirical analysis combines three linked administrative datasets from Turkiye with a separate firm-level waste survey. Turkish customs records report annual firm-level imports and exports by partner country and 8-digit HS product code, allowing us to identify the China-banned waste products. Firm registry and corporate financial statements report firm-level annual gross sales, material and wage costs, employment, as well as location (city) and 4-digit NACE industry code (the Statistical Classification of Economic Activities in the European Community). VAT declarations allow for the construction of firm-to-firm input-output flows and the identification of domestic suppliers. We complement these administrative sources with the Manufacturing Industry Waste Statistics survey (biennial), which covers all formal firms with more than 50 employees and a representative sample of smaller firms. The survey records firm-level waste volumes and disposal methods (e.g. selling, reuse, dumping, burning) by EWC-Stat product category. We map EWC-Stat (European Waste Classification for Statistics) codes to HS6 to identify China-banned plastic waste products.[17]

3.1. **Imports of Banned Plastic Waste.** Building on the aggregate diversion pattern documented in Section 2, we now turn to Turkish customs data. To control for time-invariant product-origin-level trends and for origin-year-level shocks, we estimate an event-study specification at the product-origin-year level:

$$\ln(\text{Imports}_{oht}) \;=\; \sum_{l=2013}^{2019} \beta_l \,(D_t^l \times \text{Banned}_h^{HS6}) \;+\; \alpha_{oh} \;+\; \alpha_{ot} \;+\; e_{oht}, \tag{3.1}$$

[17]China's restrictions are formally imposed on specific HS codes, but these codes collectively encompass the relevant universe of traded plastic waste. Because EWC-Stat classifies waste at a broader material level, any EWC-Stat category corresponding to plastic waste is treated as exposed. The purpose of the crosswalk is therefore not to establish exact concordance between individual EWC-Stat and HS6 subcategories, but to determine whether a firm's waste falls within the set of plastic waste products covered by China's ban. Since the banned HS codes span essentially all plastic waste categories, this approach provides comprehensive identification of treated plastic waste within the survey data.

where Imports_{oht} is the quantity (kg) of HS8-digit product h imported from origin country o in year t. As before, Banned_h^{HS6} is an indicator for products banned by the Chinese ONS policy. D_t^l are year dummies, α_{oh} are product×origin fixed effects, and α_{ot} are origin×year fixed effects. The sample runs from 2013 to 2019 and 2016 is omitted as the base year. Coefficients β_l measure year-by-year deviations in imports of banned products from a given origin relative to non-banned products imported from the same origin in the same year.

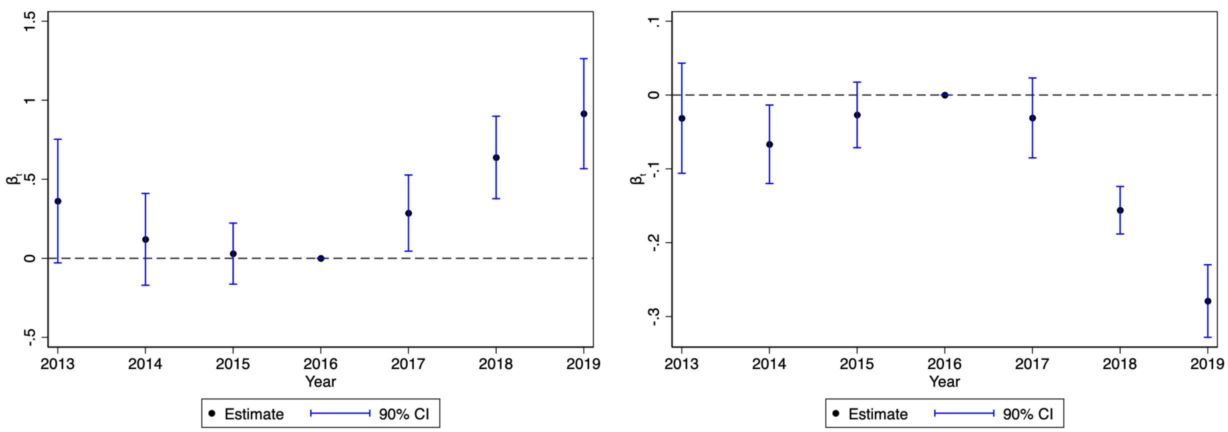
Figure 3.2 plots the estimated β_l coefficients with their 90% confidence intervals. Turkish imports of China-banned plastic products increase significantly after 2016 and remain high through 2019. Figure B.2 replicates this exercise at the firm-product-origin-year level and confirms that individual firms raised imports of banned products, ruling out the possibility that aggregate patterns are driven by changes in the composition of active importers. In complementary analysis, Figure 3.1 shows the decomposition into quantities and unit values: quantities of treated products increase (Panel (A)) while unit values decline (Panel (B)), suggesting that the policy induced a global contraction in demand for these items. Figure B.3 reinforces this interpretation using quality-adjusted prices: even after controlling for observable product characteristics, ONS-affected products experienced a statistically significant decline in world prices.

As a falsification exercise, and to address concerns that our results might be influenced by misclassification of products, we randomly reassign "treatment" (banned status) across 8-digit HS products within each 4-digit HS group and re-estimate Equation (3.1) 250 times. The resulting distribution of placebo estimates is centered tightly around zero (Appendix Figure B.5), compared to our baseline estimate that is slightly higher than unity. This exercise lends credibility to our baseline findings, demonstrating that the estimated post-ONS increase in imports of banned products is not an artifact of product misclassification, but instead reflects a true response to the policy shock.

Overall, the evidence points to a key displacement mechanism. The ONS policy reduced China's demand for banned waste products, generating excess supply in international markets and driving down their world prices. These lower prices, in turn, stimulated import

FIGURE 3.1. Event Study: Decomposition of Turkish Imports into Quantities and Unit Values

(A) Quantity (B) Unit Prices



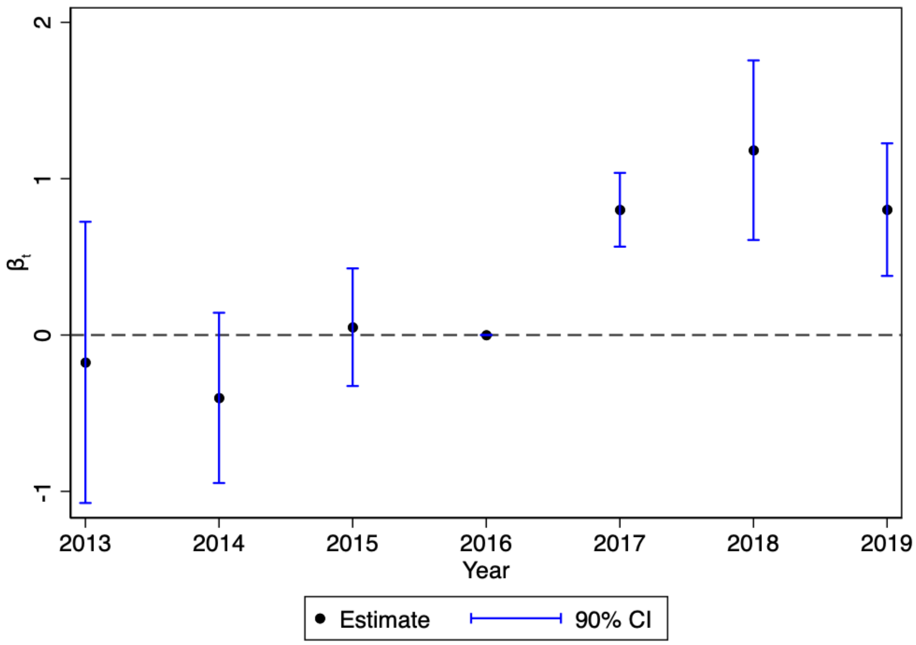
Note: These figures plot the estimates of β_l, together with 90% confidence intervals, obtained from estimating Equation (3.1). The dependent variable in Panel (A) is import quantities (measured in kilograms) at the 8-digit HS level, while Panel (B) reports unit values (measured in dollars per kilogram). The interaction with year 2016 is omitted to serve as the reference year. The sample covers the period 2013–2019.

demand in countries not subject to the ban, with Turkiye emerging as a particularly important destination of the displaced flows.[18] The estimates obtained from detailed Turkish customs data reveal a discrete and persistent jump in imports of ONS-banned products beginning in 2017, with no evidence of differential pre-trends relative to unaffected products.

3.2. **Waste Sales of Domestic Firms.** We next study how domestic waste generators responded to increased import competition. The waste survey records for each firm the annual quantity of each EWC-Stat waste type generated and the fraction sold, reused, or mismanaged (dumped, uncontrolled burning, discharge). We map EWC-Stat codes to HS6 to identify waste types that correspond to China-banned products, as discussed earlier.

[18]We also test whether Turkiye redirected these inflows to third countries. Figure B.4 in the Appendix shows no discernible increase in exports of China-banned plastic waste products, indicating that the majority of diverted waste was processed domestically rather than transshipped.

Figure 3.2. Event Study: Value of Imports



Note: The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in (3.1) in addition to the 90% confidence intervals. The dependent variable is the (log) quantity of Turkish imports of product h from origin o at year t. Coefficients on the interaction between year dummies D_t^l and an indicator Banned_h^{HS6} for waste products banned by China are plotted. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

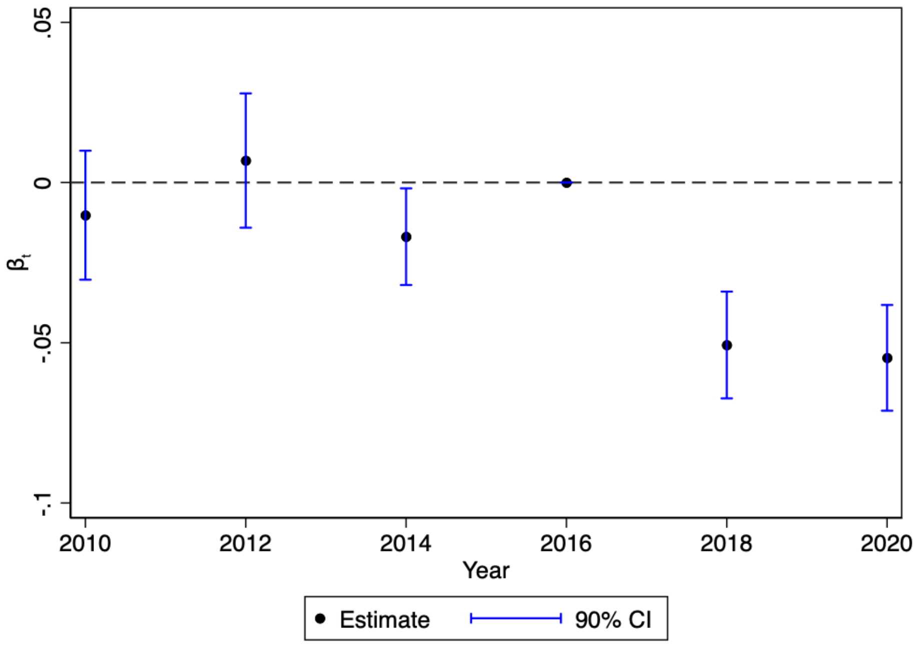
To test whether the increase in import competition induced by the Chinese policy led domestic firms to reduce the volume of waste they sold, we estimate the following specification:

$$\left(\frac{\text{Volume of Waste Sold}}{\text{Total Waste Volume}}\right)_{mht} = \sum_{l=2010}^{2020} \beta_l \left(D_t^l \times \text{Banned}_h\right) + \alpha_{mt} + \alpha_{mh} + \varepsilon_{mht}, \tag{3.2}$$

where the dependent variable is the share of firm m's quantity of waste product h sold (rather than reused or mismanaged) in year t, α_{mt} are firm×year fixed effects and α_{mh} are firm-product fixed effects. Identification exploits within-firm changes in sales shares for banned versus non-banned waste types. Figure 3.3 reports the event-study estimates: the share sold declines by about 5 percentage points for banned waste types after 2016.

Domestic firms sold a smaller share of their ONS-banned waste products following the policy shock, raising the question of how the unsold waste was subsequently handled. To investigate this, we replace the dependent variable in Equation (3.2) with the share of waste

FIGURE 3.3. Event study: Waste Sales of Firms in Turkiye

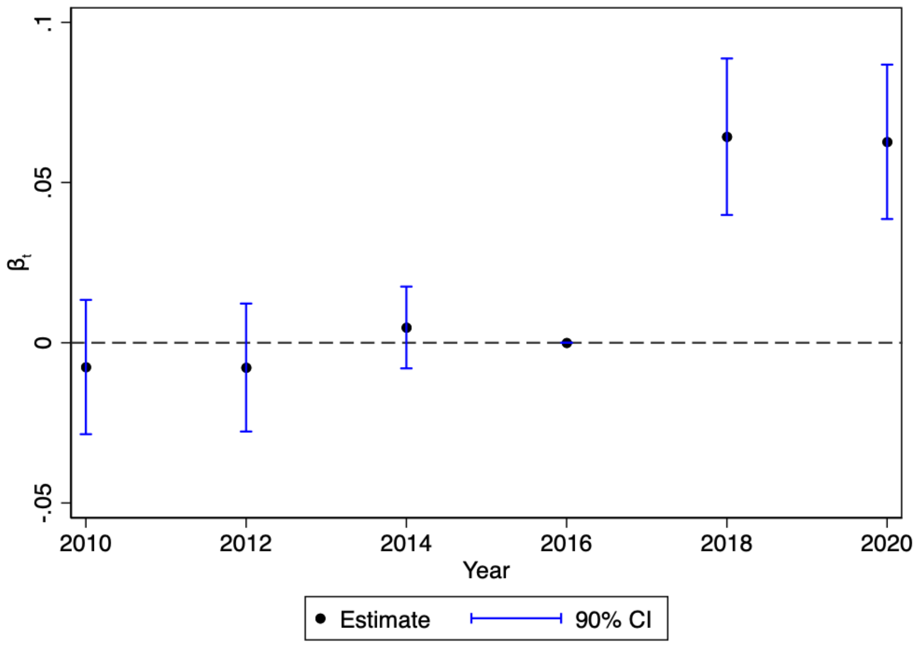


Note: This figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in 3.2. The dependent variable is the share of waste that firm m sells of waste product h at year t. Coefficients on the interaction between year dummies D_t^l and an indicator $Banned_h$ for waste products banned by China are plotted. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2010 to 2020.

that is *mismanaged*. Figure 3.4 shows that mismanagement increases for China-banned waste types after 2016, with the largest effects observed among smaller firms (see Panel (A) of Appendix Figure B.6). Appendix Figure B.7 shows that there is no detectable change in waste-management practices among the importers themselves.

Overall, the evidence suggests that imported plastic waste displaced domestically generated waste in Turkiye. The most plausible mechanism is a form of quality-biased substitution: Turkish manufacturers shifted towards imported waste because it is cleaner, better sorted, and substantially less costly to prepare for use as a production input. As documented in Appendix B.2, even before the ONS policy, waste exported by the countries that China historically relied on—*Top Exporters*—was systematically higher quality than waste exported

Figure 3.4. Event study: Management of Waste by Firms in Turkiye



Note: This figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in (3.2). Each observation is at the firm-product-year level. The dependent variable is the share of waste that is mismanaged. Coefficients on the interaction between year dummies D_t^l and an indicator $Banned_h$ for waste products banned by China are plotted. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2010 to 2020.

by Turkiye.[19] This quality difference is important because the early stages of sorting, separating, and cleaning are technologically irreversible and largely determine the mechanical properties of the final recycled polymer (Awaja and Pavel, 2005). Errors or contamination introduced at these stages cannot be corrected during downstream processing such as shredding, melting, or molding.[20] When manufacturers purchase domestic waste, they undertake

[19]Figure B.9 in the Appendix shows that the distribution of estimated product quality for Top Exporters first-order stochastically dominates that of Turkiye for both all waste and plastic waste. Table B.3 further shows that Top Exporters supply plastic waste of approximately 35% higher average quality relative to Turkiye, a difference that is statistically significant at the 1 percent level.

[20]Mixed polymer streams have incompatible melt temperatures and viscosities, producing weak or brittle recycled materials (Awaja and Pavel, 2005); organic or particulate contaminants induce polymer degradation during melting, resulting in discoloration, brittleness, and gel formation (Al-Salem et al., 2009). International regulations governing cross-border plastic waste shipments—including the Basel Convention and OECD Council Decision C(2001)107—require exporters to pre-sort and pre-clean waste before shipment, a requirement emphasized in country-level analyses such as Karasik (2022). Imported waste therefore arrives in substantially better condition than domestically generated waste, which typically receives little or no preprocessing.

the early processing stages themselves. The early stages are typically labor-intensive activities. By contrast, imported waste arrives already sorted, cleaned, and partially pre-processed, meaning manufacturers can proceed directly to the final processing stages (shredding and molding).

The quality-biased substitution mechanism is consistent with the higher unit values of imported waste relative to domestically exported Turkish waste that we discussed earlier. It can also be examined more directly. A direct measure of quality is the intensity of water used in the processing of waste. Water is a key input in the early stages of plastic waste recycling—particularly washing and cleaning—required to remove contaminants before it can be reused. Imported plastic waste, by contrast, arrives already sorted and cleaned due to international shipment requirements, as most advanced economies are signatories to international conventions that proscribe international shipping of various low quality mixed waste. To examine this channel, we exploit firm-level data on water consumption, though it is available only for the period 2016-2018. Specifically, we regress the change in water consumption divided by sales between 2016 (pre-ONS) and 2018 (post-ONS) on firm's pre-ONS intensity of the use of China-banned plastic waste, controlling for industry and region fixed effects as well as initial firm size. Firms with higher pre-ONS share of China-banned plastic waste in their total input costs show a decline in water consumption per unit of sales (Table B.1 in the Appendix shows the full results).[21] This pattern is consistent with firms reducing their reliance on water-intensive preprocessing stages by substituting toward higher-quality imported waste inputs. Together with the evidence on domestic waste sales and mismanagement, these findings provide additional support for our interpretation that domestic waste mismanagement increased because lower-quality domestic waste was displaced by cleaner imported inputs.

When the ONS policy generated excess supply and drove down world prices for the banned products, the total cost—including the purchase price and additional processing costs—to Turkish manufacturers of using imported waste fell below that of using domestic waste (with

[21]Total input costs are constructed as the sum of wage payments, domestic purchases based on VAT data, and imports.

its lower purchase price but higher processing costs). This relative price-quality combination made imported waste distinctly more attractive. As a result, manufacturers substituted toward higher-quality imported inputs, leaving domestic waste generators with a lower-quality by-product that could not be sold. Waste sales represent a tiny share of the revenues of domestic waste generators; waste is an unavoidable by-product, not a primary revenue-generating activity. Lacking incentives to upgrade or recycle this unsold fraction, firms disposed of it through dumping or burning, leading to a rise in domestic mismanagement.

Scale of Waste Generation. Our results so far indicate that following China's ban, domestic firms in Turkiye mismanaged a larger share of the waste they generated and re-used a smaller share. A natural concern is whether these changes reflect shifts in waste-management behavior per se, or instead are driven by changes in the overall scale of waste generation—for example, due to upstream changes in plastic production or demand. To address this concern, we first examine whether firms that were more exposed to ONS-banned products experienced differential changes in total waste generation.

To test whether the observed increase in mismanagement reflects changes in the quantity of waste generated rather than its disposal, we estimate the following event-study specification:

$$\text{(3.3)} \qquad \text{Total Waste}_{mt} = \sum_{l=2010}^{2020} \beta_l \, D_t^l \times \text{Exposure}_m + \alpha_m + \alpha_{s(m)t} + \alpha_{r(m)t} + \epsilon_{mt},$$

where the dependent variable is the total amount of waste generated by firm m in year t. Exposure_m is defined as the share of China-banned waste products in the total waste generated by firm m prior to the ONS policy. We include firm-level fixed effects α_m, industry-year fixed effects $\alpha_{s(m)t}$, and region (city)-year fixed effects $\alpha_{r(m)t}$ to control for time-invariant firm characteristics as well as sectoral and regional shocks. None of the estimated coefficients β_l is statistically different from zero, and the point estimates do not exhibit a systematic post-2016 trend (Figure B.8 in the Appendix plots the estimated coefficients). The estimates show no economically or statistically significant changes in total waste generation for firms more intensive in ONS-banned waste products. This finding indicates that the increase in mismanagement documented above does not reflect a change in the scale of waste generation, but rather a change in how waste is handled. The surge in import competition affected the allocation of waste across disposal methods, not the total quantity of waste generated.

Firm-to-Firm Sales of Domestic Waste. We next examine whether the decline in waste sales by domestic generators reflects reduced demand from downstream buyers. This is the mirror image of sales by domestic waste generators, but recorded from the buyer side. Using firm-to-firm sales data constructed from Turkish VAT declarations, we test directly whether buyers substituted away from domestic waste suppliers following the ONS policy. To do so, we estimate:

$$\begin{aligned}\ln(\text{Purchases}_{i,s',t}) &= \sum_{l=2013}^{2019} \beta_l \, D_t^l \times \text{Exposure}_i \times \text{Exposure}_{s'} + \alpha_{it} + \alpha_{s(i),s',t} \\ &\quad + \alpha_{is'} + e_{i,s',t}, \end{aligned} \tag{3.4}$$

where $\text{Purchases}_{i,s',t}$ denotes the value of purchases by buyer firm i in 4-digit NACE industry s from supplying industry s' in year t. Exposure_i measures the pre-ONS intensity with which firm i relied on China-banned plastic waste in its input mix, defined as the share of ONS-banned imports in total input costs (wages, domestic purchases, and imports) in 2016.[22] $\text{Exposure}_{s'}$ captures the propensity of supplying industry s' to sell plastic waste, measured as the share of plastic waste sold by industry s' in total plastic waste generated in Turkiye in 2016.

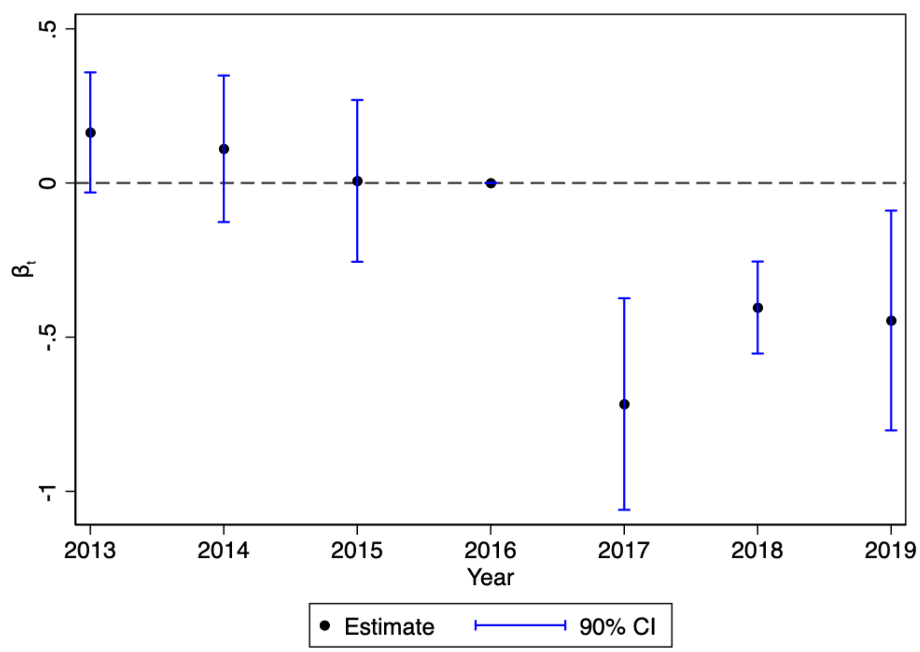
The specification includes buyer-year fixed effects α_{it}, which absorb changes in buyer firm's scale and overall input demand, as well as time varying source-destination industry pair fixed effects $\alpha_{s(i),s',t}$ and time-invariant buyer firm and supplying industry fixed effects $\alpha_{is'}$. Identification therefore comes from within-buyer-year variation across supplying industries with different exposure to plastic waste sales.[23] Figure 3.5 presents the estimated β_l coefficients

[22]This exposure measure captures the intensity of imported plastic waste use among existing importers and does not incorporate new entrants after 2017. While the number of plastic waste importers increased following the ONS policy, these entrants are small relative to incumbent importers.

[23]It is important to note that the magnitudes of the estimated event study coefficients from Equations (3.2) and (3.4) are not directly comparable for several reasons. First, the waste management survey reports quantities while the VAT dataset reports sales values. Second, the two datasets differ in their levels of disaggregation. The waste management survey aggregates the volume of waste at the supplier-product level, while the VAT dataset aggregates sales at the buyer-supplier level. For identification purposes, we aggregate the latter at the level of buyer and supplying industry when estimating Equation (3.4). Since each supplier

with their 90% confidence intervals. The results show a large decline in purchases from domestic plastic waste-supplying industries by firms that were more exposed to ONS-banned imports after 2017. This pattern complements the results obtained from the waste management survey and provides additional evidence that downstream buyers reduced their input purchases from domestic waste suppliers following the ONS policy.

FIGURE 3.5. Purchases from Domestic Waste Producers



Note: The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in (3.4). Each observation is at the firm-supplying sector-year level. Coefficients on the interaction between year dummies D_t^l and the product of exposures are plotted in the figure. $Exposure_i$ is a continuous variable indicating the share of firm i's usage of China banned plastic imports in its inputs (wages + domestic purchases + imports) in 2016. $Exposure_{s'}$ represents the share of plastic waste sold by industry s' in total plastic waste generated in Turkiye in 2016. The sample covers the years from 2013 to 2019.

These robustness checks reinforce the interpretation of our main results. The increase in domestic waste mismanagement is not driven by changes in the scale of waste generation, but rather by a demand-side reallocation away from domestic waste. Firms that gained access to cheaper and higher-quality imported waste reduced their purchases from domestic waste

can have multiple buyers and each buyer multiple suppliers within an industry, a direct comparison of the magnitude of estimates is not feasible. However, the qualitative results obtained from estimating the two equations should point in the same direction.

generators, leaving the latter with unsold waste that was increasingly disposed of through mismanagement. The consistency between the waste survey evidence and the firm-to-firm transaction data strengthens the conclusion that import competition induced by the ONS policy led to quality-biased displacement of domestic plastic waste.

3.3. **Firm Performance of Importers.** We next examine whether firms that imported China-banned plastic waste benefited in terms of increased sales and competitiveness. As documented above, the ONS policy generated a substantial reduction in world prices of banned plastic waste products and redirected global supply toward alternative destinations, including Turkiye. Because the majority of imported plastic waste is directly used as an intermediate input by manufacturing firms, improved access to cheaper and higher-quality imported waste may have translated into lower production costs and improved firm performance.

To test these effects, we estimate the following event-study specification:

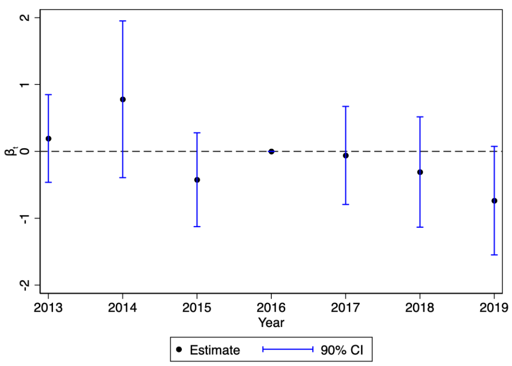
$$X_{it} = \sum_{l=2013}^{2019} \gamma_l \, D_t^l \times \text{Exposure}_i + \sum_{l=2013}^{2019} \delta_l \, D_t^l \times \text{Employment}_i + \alpha_i + \alpha_t + \epsilon_{it}, \tag{3.5}$$

where X_{it} denotes an outcome of interest for firm i in year t, including measures of input costs, sales, and industry market shares. Exposure_i is a continuous measure capturing firm i's pre-ONS reliance on China-banned plastic waste, defined as the share of such products in total input costs in 2016, and Employment_i denotes firm size in 2016. The specification includes firm fixed effects and year fixed effects, thereby exploiting within-firm variation over time. The control group consists of firms importing other products within the same 4-digit NACE industry as importers of banned 8-digit HS plastic waste products.

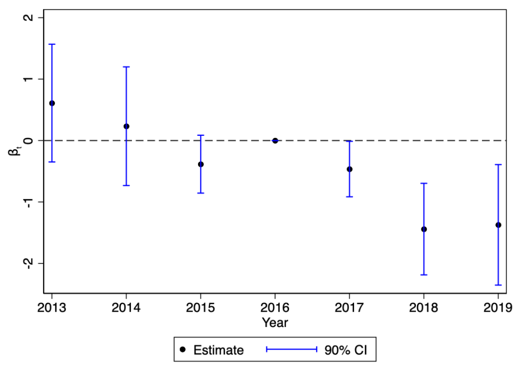
Input Costs. We first ask whether the decline in world prices for imported plastic waste translated into lower production costs for exposed firms. Figure 3.6 reports event-study estimates for material costs and wages, each expressed relative to sales. Firms with greater exposure to China-banned imports experienced a modest decline in material costs relative to sales following the ONS policy, although the estimates are imprecisely estimated. More noticeably, these firms exhibit a statistically significant decline in their wage share. This pattern is consistent with the mechanism discussed earlier: imported plastic waste arrives already sorted and cleaned, reducing the labor-intensive pre-processing required when using domestically generated waste.[24]

FIGURE 3.6. Effects of ONS on Importer-level Costs

(A) Material costs over sales



(B) Wages over sales



Note: These figures plot the estimates of γ_l, together with 90% confidence intervals, obtained from estimating the specification in (3.5). Each observation is at the firm-year level. The dependent variable changes across sub-figures as stated in the title. The coefficient of interest is on an interaction term of year dummies D_t^l and Exposure$_i$, where Exposure$_i$ is the share of firm i's usage of banned plastic products in its inputs in year 2016. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

Sales, Market Shares, and Competitiveness. We next test whether improved access to imported waste affected firm-level revenues and competitiveness. Using domestic sales and industry-level market shares as outcome variables, we re-estimate Equation (3.5).[25] The left and middle panels of Figure 3.7 show that firms with higher exposure to ONS-banned imports experienced relatively faster growth in sales and industry market shares after the policy. The right panel examines exporting probabilities as a proxy for international competitiveness. While these estimates are less precisely estimated, they point to some muted increase in export participation among more exposed firms on the margin.

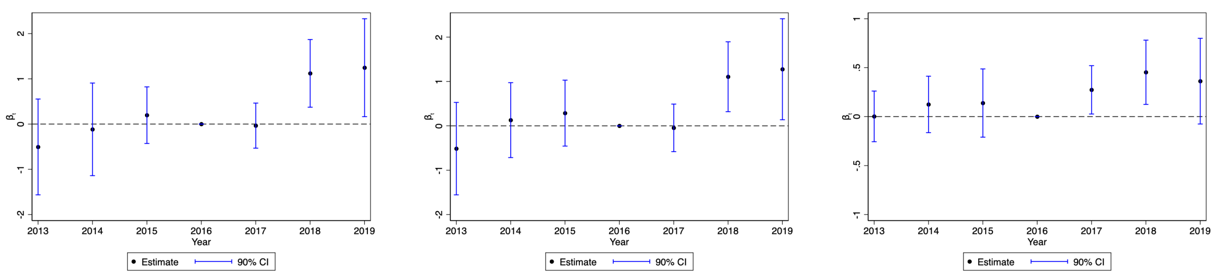
[24]This finding is consistent with evidence in Castro Vincenzi and Kleinman (2020), who show that lower input prices reduce the labor share, and Hummels et al. (2014), who document that offshoring and input sourcing reduce demand for low-skilled labor.

[25]Market share is defined as the ratio of a firm's gross revenues to total revenues in its 4-digit NACE industry.

These results indicate that firms in Turkiye that relied more heavily on imported plastic waste benefited from the ONS policy through improved access to cheaper and higher-quality inputs. These gains manifested primarily through reduced labor intensity and higher sales, rather than through large reductions in material costs alone. Importantly, the benefits accrued to downstream manufacturing firms contrast with the adverse effects experienced by domestic waste generators documented in the previous subsection. This divergence highlights the role of quality-biased substitution: while manufacturers gained from cheaper, cleaner imported inputs, domestic waste suppliers—unable to compete on quality and prices—faced reduced demand and increased mismanagement of unsold waste.

Figure 3.7. Effects of ONS on Importer Performance

(A) Domestic Sales | (B) Output Market Share | (C) Exporting Probability



Note: These figures plot the estimates of γ_l, together with 90% confidence intervals, obtained from estimating the specification in (3.5). Each observation is at the firm-year level. The dependent variable changes across sub-figures as stated in the title. The coefficient of interest is on an interaction term of year dummies D_t^l and Exposure$_i$, where Exposure$_i$ is the share of firm i's usage of banned plastic products in its inputs in year 2016. The interaction with year 2016 is removed from the equation to serve as a reference year. The sample covers the years from 2013 to 2019.

To sum up, three key empirical facts emerge from the analysis in this section. First, the ONS policy generated a large and persistent increase in Turkish imports of banned plastic waste, accompanied by a decline in world prices. Second, these import surges reshaped domestic waste markets: as foreign plastic displaced local waste, domestic waste generators lost buyers, sold a smaller share of their waste, and increasingly diverted the remainder towards dumping and burning. Third, importing manufacturers benefited from cheaper imported inputs through lower input costs, higher sales, and declining wage shares as cleaner imported materials reduced the need for labor-intensive preprocessing. Taken together, the estimates imply that China's ban created both economic gains and environmental losses in Turkiye, and we next turn to quantifying the overall magnitude of either side of this trade-off.

4. Quantifying Welfare Impacts of ONS in Turkiye

Section 3 established reduced-form evidence of the economic and environmental effects in Turkiye, that emerged as the top destination for displaced plastic waste following China's import ban. Three steps are needed to move from the reduced-form estimates to quantification of the welfare effects of the ban. First, the environmental consequences of domestic waste displacement must be aggregated and valued in monetary terms, requiring a framework that links firm-level increases in waste mismanagement to pollution damages. Second, firm-level economic responses must be aggregated across the economy, which requires structure on how production gains translate into welfare. Third, because the DiD estimates are inherently effects among more exposed firms relative to less-exposed firms, market-clearing conditions are needed to recover the absolute scale of both environmental and economic adjustment.

The model that follows provides the minimal structure required for each step. Its key advantage is that the sufficient statistics for welfare are firm-level exposure shares that can be directly measured from the observed data—specifically, the pre-policy share of imported plastic waste in total input costs of domestic plastic waste-using firms, and the share of plastic waste in total waste generated for domestic waste supplying firms. As we show below, these coincide exactly with the treatment-intensity measures already used in the DiD regressions of Section 3. Combined with external damage valuations from the environmental literature, these sufficient statistics allow us to place environmental damages and economic gains on a common monetary scale.

4.1. **Welfare.** We study welfare in a setting where economic gains from consumption must be weighed against the environmental damages generated by production and waste disposal. This framework captures the central trade-off created by the ONS policy: cheaper imported plastic waste can lower production costs and raise consumption, but it can also increase the scale of dirty activity in the host country along with displacing domestic waste toward more environmentally harmful disposal.

We conceptualize ONS as a large negative shock to global demand for traded recyclable plastic waste. By prohibiting plastic waste imports into China, ONS eliminated a major destination for global recyclables, generating excess supply in exporting countries, depressing world recyclable-waste prices, and reshaping production, consumption, and pollution outcomes worldwide. Formally, we model ONS as a prohibitive increase in iceberg trade costs on recyclable plastic waste shipments into China. For any origin country $o \neq \text{CHN}$, bilateral trade costs satisfy $\tau_{o,\text{CHN}} \to \infty$. As a result, China's imports of recyclable plastic waste collapse to zero.

We examine the change in welfare induced by ONS in country k, with particular emphasis on Turkiye ($k = \text{TUR}$). The full model is developed in Appendix C. Here we present only the minimal structure needed for welfare quantification. Let Δ denote the change in equilibrium outcomes induced by the ONS policy. We define the change in welfare in country k as

$$\Delta W_k = \Delta Q_k - \Delta Z_k \equiv \Delta Q_k - \xi_{bk}\Delta B_k - \xi_{xk}\Delta X_k - \xi_{vk}\Delta V_k, \tag{4.1}$$

where Q_k denotes aggregate final consumption and Z_k denotes damages from plastic pollution that consist of environmentally consequential changes, denoted by B_k, X_k, and V_k, arising from waste disposal, from recycling and re-use, and from virgin plastic extraction, respectively. The parameters ξ_{bk}, ξ_{xk}, and ξ_{vk} denote the corresponding country-specific marginal social damages associated with these activities.

Aggregate consumption Q_k is a constant elasticity of substitution (CES) composite of sectoral consumption indices Q_k^j, where $j \in \{s, u, n\}$ indexes the *Supplying*, *Using*, and *Neither* sectors, described in detail later. Each Q_k^j is itself a CES aggregate of firm-level varieties q_{fok}^j produced by firm f in origin country o and consumed in country k. The model features firms that produce differentiated final goods, that have plastic recyclables as inputs or by-products. To characterize the welfare effects of the ONS policy, we next describe the production structure and firm behavior that determine consumption and environmental outcomes.

4.2. **Sectors and Firms.** Firms are partitioned into sectors $j \in \{s, u, n\}$—*Supplying*, *Using*, and *Neither*—according to whether they generate and supply recyclable plastic waste, use recycled plastic waste as an input, or neither.

Supplying firms generate plastic waste as a by-product of production at varying intensities. Let $\zeta^s_{fk} > 0$ denote this firm-specific plastic intensity in total by-products generated. Firms choose how much of their plastic waste to sell for re-use, versus how much to dispose of themselves. These choices satisfy the material balance constraint:

$$\zeta^s_{fk} W^s_{fk} = b^s_{fk} + \sum_{g,d} r^s_{f(k)g(d)}, \tag{4.2}$$

where W^s_{fk} denotes total waste generated (as an increasing function of total final output production), $r^s_{f(k)g(d)}$ denotes the plastic waste sold to firm g in country d, and b^s_{fk} denotes plastic waste disposed of by firm f itself.

Using firms purchase recyclable plastic from *Supplying* firms to use as an input alongside labor and virgin material. These firms differ in how effectively they recycle plastic in production. Let $\chi^u_{fk} > 0$ denote firm-specific recycled-input efficiency, which governs how effectively firm f converts recyclable plastic into final output.[26]

Neither firms neither generate recyclable plastic waste nor use recycled plastic as an input. They are affected by the ONS policy only through general equilibrium adjustments.[27]

Each firm $f \in \mathcal{F}^j_k$ chooses destination-specific output quantities $\{q^j_{fkd}\}_d$, labor input l^j_{fk}, and virgin material input v^j_{fk}. *Using* firms additionally choose recycled input x^j_{fk}, while

[26]More precisely, χ^u_{fk} governs firm-specific efficiency in the use of imported recyclable plastic inputs, rather than recyclable inputs more broadly (see Appendix C.3).

[27]The sector classification is a modeling device, not a hard partition: firm-level exposure to the policy enters our analysis through continuous shares S^u_f and S^s_f, so a firm with both positive shares is treated as exposed along both margins. Empirically, the overlap is small. The waste-management survey contains a small number of firms that are also importers of banned plastic waste, and in any case our domestic-supplier analyses (Section 3.2) exclude importers from the sample, isolating the supply-side response to import competition.

Supplying firms choose recycling shipments $\{r^s_{f(k)g(d)}\}_{g,d}$ and disposal b^s_{fk}. Firms are monopolistically competitive in each input and output market and their choices yield the following profit function:

$$\begin{aligned}\pi^j_{fk}(\mathbf{p}_k) =& \sum_d \left(p^j_{fk} - c^j_{fk}\Big(\mathbf{p}_k, \mathbb{1}\{j=u\}\chi^u_{fk}\Big)\right) q^j_{fkd}/\phi^j_{fk} \\ &+ \mathbb{1}\{j=s\}\left[\sum_d \sum_{g\in\mathcal{F}^u_d}\left(\frac{\rho^s_{fkd}}{\tau_{kd}} - w^r_k A_k(\zeta^s_{fk})\right) r^s_{f(k)g(d)} - w^r_k D_k b^s_{fk}\right].\end{aligned} \tag{4.3}$$

The first line captures profits from final-good production, where $c^j_{fk}(\cdot)$ denotes unit production costs. Unit costs depend on Hicks-neutral productivity ϕ^j_{fk} and the vector of input prices $\mathbf{p}_k$, which includes the wage, the price of virgin material, and, for *Using* firms, access to recycled inputs. The parameter $\chi^u_{fk} > 0$ captures the efficiency with which firm f recycles China-banned plastic waste.

The second line, relevant only for *Supplying* firms, captures profits from selling recyclables. Firms generate waste as a byproduct of effective production units, $W^s_{fk} = \theta^s_k q^s_{fk}/\phi^s_{fk}$ where $\theta^s_k < 1$ is the sectoral by-product conversion factor. A fraction ζ^s_{fk} of the total by-products W^s_{fk} contain recyclable plastic material. Firms sell recyclable plastic at price ρ^s_{fkd} subject to iceberg trade costs τ_{kd} and per-unit costs $A_k(\zeta^s_{fk})$ that are incurred to convert by-products into recyclable plastic waste, such as primary sorting of plastic recyclables from other material. Firms sell $r^s_{f(k)g(d)}$ units of the recyclable plastic waste and dispose of the rest of the by-products by paying disposal costs D_k for unsold waste. The dependence of $A_k(\zeta^s_{fk})$ on ζ^s_{fk} allows recycling costs to vary with the plastic content of waste.

To characterise firm choices and the effects of the ONS policy, we make the standard assumption below that Hotelling elasticities with respect to prices are constant across firms. In quantitative trade models, this is typically applied to bilateral trade flows between countries to get a constant trade elasticity across firms within a market.

Assumption. *Firms' profit function* ${\pi^j_{fk}}^*$ *exists. Inputs are normal and Hotelling's lemma applies, with constant Hotelling elasticities across firms within each product-country market.*

A full discussion and first-order conditions of the profit maximization are provided in Appendix C.3, and we next summarise the key results for quantification of welfare effects.

4.3. **Firm Exposure, Sufficient Statistics and General Equilibrium Effects of the ONS Policy.** In this subsection, we use the model to characterize which firms are directly exposed to the ONS policy and to derive sufficient statistics that summarize firm-level heterogeneity in exposure. These theoretically grounded exposure measures allow us to bring the model to the data by empirically testing how directly affected firms respond along key outcomes. We then use market clearing conditions to interpret how these firm-level adjustments propagate through the broader economy, generating general equilibrium effects for indirectly exposed firms. Together, this framework links theory to empirics in order to identify the firm-level effects of the ONS policy.

The model implies that two types of firms are directly affected by the ONS policy: (i) *Using* firms, which rely on imported recycled plastic waste as an input, and (ii) *Supplying* firms, which recycle and sell the plastic waste they generate during production. Firms in the *Neither* sector are affected only through general equilibrium effects.

Using firms are heterogeneous along two dimensions: Hicks-neutral productivity, ϕ^u_{fk}, and factor-biased productivity in the use of recycled plastic inputs, χ^u_{fk}. While χ^u_{fk} is not directly observed, under standard technologies the model implies that it maps monotonically into the firm's expenditure share on imported banned waste in total costs, S^u_{fk}, which is strictly increasing in χ^u_{fk}. Hence, conditional on initial firm size or productivity, S^u_{fk} serves as an observable sufficient statistic for a *Using* firm's exposure to the ONS ban.

Similarly, *Supplying* firms differ along two dimensions: Hicks-neutral productivity, ϕ^s_{fk}, and the share of waste that is plastic, ζ^s_{fk}. Conditional on initial firm size or productivity—which absorbs variation in Hicks-neutral productivity—ζ^s_{fk} determines the heterogeneity in differential exposure to the ONS policy. Under constant Hotelling elasticities, the model therefore implies that $S^s_{fk} \equiv \zeta^s_{fk}$ serves as an observable sufficient statistic for a *Supplying* firm's exposure to the policy.

Proposition 1 (Sufficient Statistics)**.** *Differences across firms' responses to changes in recycled plastic waste import prices in outputs and input demands can be summarized by the share of imported plastic waste expenditure in total costs, S^u_f, and the share of plastic waste in total waste, $S^s_f \equiv \zeta^s_f$.*

Additionally, differences across supplying firms' responses in total waste, as well as banned and non-banned waste sold, can be summarized by S^s_f. Differences in total revenue and total exports (including revenues from waste sales) can be approximated by (S^s_f, L_f), where L_f denotes initial firm size.

Proof. See Appendix C.5.

The intuition is simple.[28] Within each sector, all firms share a common production technology and common substitution elasticities, so the heterogeneity in their responses to the ONS price shock arises from how intensively they relied on the affected inputs before the ban. For firms that import and use plastic waste as an input, this is captured by S^u_f, the pre-policy share of imported plastic waste in total input costs; for domestic waste generators, by S^s_f, the share of plastic waste in total waste generated. Firm-level productivities affect the level of outcomes but not the differential response to a common price change, conditional on initial employment and up to a first order approximation. Both measures are constructed from the customs records and waste survey data described in Section 3 and enter directly as treatment intensity variables in the difference-in-differences specification introduced below:

$$\ln Y_{ft} = \beta^Y_u \, Post_t \, S^u_f + \beta^Y_s \, Post_t \, S^s_f + \beta^Y_l \, Post_t \, l_f + \alpha_f + \alpha_{i(f)t} + \varepsilon_{ft}, \tag{4.4}$$

[28]The formal derivation of sufficient statistics is provided in Appendix C.5, which shows that under common production technologies and substitution elasticities within sectors, cross-firm differences in responses to the ONS price shock depend on pre-policy exposure shares. The mapping from these relative differences to absolute responses for exposed firms relies on labor market clearing (Assumption 3) and is established in Appendix D.1.

where $Post_t$ indicates the period after the ONS policy. Firm fixed effects α_f absorb time-invariant differences across firms. Industry-time fixed effects α_{it} absorb aggregate sectoral shocks, including the differential impact of macroeconomic shocks across industries. The interaction $Post_t \times l_f$ further controls for size-related channels, including export orientation, credit access, and import diversification. The coefficients β_j^Y, for $j \in \{u, s\}$, measure the relative difference-in-differences response of exposed firms compared to non-exposed firms within the same industry, holding firm size fixed. These estimates therefore identify relative treatment effects, not absolute changes. If the ONS policy generated meaningful general equilibrium spillovers onto non-exposed firms through labor markets, input demand, or equilibrium price adjustments, then the estimated coefficients would understate or overstate the true absolute responses relevant for welfare quantification.

To map relative firm-level estimates into absolute policy effects, we use standard market-clearing conditions to characterize the equilibrium responses of indirectly exposed firms. As shown formally in Appendix D.1, the magnitude of these general equilibrium effects depends on the economic size of directly exposed sectors relative to the broader economy.

In the Turkish context, these general equilibrium effects turn out to be negligible because direct exposure is too small relative to the broader economy to generate meaningful spillovers. Pre-ONS, *Using* firms—the primary beneficiaries of lower recycled-input prices—account for only \$1.7 billion in annual sales and 13,136 workers (Table D.1), representing less than one percent of economy-wide employment. By contrast, *Supplying* firms account for \$172.5 billion in sales and 852,057 workers, but their reliance on by-product sales of plastic waste are too small to create employment effects (see null results of employment for *Supplying* firms in Column (1) of Table 2), and the *Neither* sector comprises the vast majority of remaining economic activity.[29] As we formally show in Appendix D.1, this sharp asymmetry in sectoral scale, combined with standard labor market clearing, implies that equilibrium effects on *Neither* firms are approximately zero. Relative difference-in-differences estimates therefore closely approximate absolute policy impacts, allowing us to interpret the estimated responses

[29] Column (2) of Table 2 shows an expansion in sales for *Using* firms and no systematic revenue response for *Supplying* firms.

of exposed firms as sufficient statistics for the welfare-relevant environmental and economic consequences of ONS in Turkiye.

We next evaluate the environmental consequences of the ONS policy

4.4. **Environmental Impacts of the ONS Policy.** The theoretical framework identifies three primary environmental margins through which ONS can affect welfare: changes in plastic waste disposal (ΔB_k), changes in recycling and re-use of plastic waste (ΔX_k), and changes in virgin plastic production (ΔV_k). Together, these margins capture how the policy reshapes the full lifecycle of plastics, from downstream waste management to upstream production responses.

We use the theory-guided difference-in-differences specification in Equation (4.4) to estimate how the ONS policy affected firm-level outcomes that map directly into the components of welfare, with results reported in Table 2. As established above, general equilibrium spillovers to non-exposed (*Neither*) firms are negligible in the Turkish context. Because labor inputs of non-exposed firms show negligible effects, the standard assumption of monotonicity of inputs and output with respect to labor would imply that non-exposed firms also see negligible changes in their output sales and environmentally-relevant waste and virgin import outcomes (Assumption 3 in Appendix C provides specific details). We therefore interpret the estimated responses in Table 2—identified relative to non-exposed firms—as closely approximating the absolute firm-level effects of the policy.

Columns (3)–(5) of Table 2 identify the environmentally-relevant margins through which ONS affects welfare in Turkiye. Column (3) shows that more exposed *Supplying* firms experience a significant decline in the share of waste that is formally managed, while Column (4) shows no corresponding change in total waste generation. Together, these results imply that ONS did not increase the amount of plastic waste produced, but instead altered its downstream allocation: a larger share of existing waste is diverted away from formal recycling and re-use towards dumping or burning.[30] In terms of the model, this implies an increase

[30]The Greenpeace Mediterranean ‘Game of Waste’ report documents illegal open burning of imported plastic waste at dumpsites in Turkiye’s Adana province, with soil and ash samples revealing hazardous chemical

Table 2. Welfare Relevant Margins of the ONS Policy

	Employment	Sales	Share of Managed Waste	Waste	Virgin Imports
	(1)	(2)	(3)	(4)	(5)
$Post_t \times S_f^u$	12.14^a	12.11^a	1.990	-13.01	595.3
	(4.641)	(3.460)	(3.555)	(19.67)	(2789.8)
$Post_t \times S_f^s$	-0.0429	-0.0556	-0.0862^a	0.256	-0.201
	(0.0698)	(0.0852)	(0.0276)	(0.505)	(0.183)
$Post_t \times \text{Employment}_{f,t=0}$	0.158^a	0.194^a	-0.0185^a	0.00940	-0.0581
	(0.0105)	(0.0132)	(0.00386)	(0.166)	(0.0532)
N	76465	76465	76465	76465	76465
R^2	0.777	0.830	0.538		
Fixed Effects:					
Sector × Year	Yes	Yes	Yes	Yes	Yes
Firm	Yes	Yes	Yes	Yes	Yes

Note: This table presents the production-relevant and environmentally-relevant margins of Equation (4.4). The dependent variables are employment, sales, the share of managed waste, total waste generation, and virgin imports. All outcome variables are estimated as in Equation 4.4, with OLS for logarithmic outcomes (employment, sales, and share of managed waste) and PPML for levels of waste and virgin imports. The coefficients of interest are on $Post_t \times S_f^u$ and $Post_t \times S_f^s$, where $Post_t$ equals one after 2017, S_f^u is the 2016 cost share of imported plastic waste, and S_f^s is the 2016 share of plastic waste in total waste. The sample covers the period from 2013 to 2019. Standard errors are clustered at the 4-digit NACE industry level. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

in environmentally harmful disposal, $\Delta B_k > 0$, alongside a decline in domestically managed recycling and re-use, $\Delta X_k < 0$.

At the same time, our previous import results in Table 1 show that Turkish firms increased their reliance on imported recycled plastic waste following ONS, as lower global recyclable-waste prices made foreign inputs cheaper. This implies that part of the decline in domestic recycling was offset by greater imported recyclable-input use. Thus, while ONS reduced domestic recycling and re-use, its net effect on ΔX_k reflects a compositional shift away from domestic towards more imported recyclable plastic rather than a pure collapse in recyclable-input use.

Column (5) shows no significant change in virgin plastic imports for the directly exposed sector, implying that the policy did not measurably affect upstream virgin plastic production

pollutants consistent with plastic combustion (Gündoğdu, 2022). This corroborates our finding that the rise in mismanaged waste in Turkiye by a large part includes open burning.

within Turkiye, so $\Delta V_k = 0$.[31] Turkiye does not have reserves of oil and natural gas for domestic production of virgin polymers, and must import them.[32]

Taken together, these results imply that the primary environmental cost of ONS in Turkiye operates through increased waste mismanagement ($\Delta B_k > 0$), while recycling and re-use (ΔX_k) are reshaped through substitution from domestic towards imported recycled inputs, and virgin plastic production remains largely unchanged ($\Delta V_k = 0$).

4.4.1. *Local Air Pollution.* To translate the environmental outcomes to welfare-equivalent changes, we first determine their effects on pollution and then examine various monetary valuations of pollution. We quantify the local environmental implications of ONS by focusing on particulate matter pollution, proxied by PM10 concentrations, which constitutes the primary local environmental concern in our context.[33] Plastic waste can generate multiple environmental externalities—including greenhouse gas emissions, toxic pollutants, and ecotoxicity—but in the Turkish context, particulate air pollution is particularly salient because open dumping and burning of displaced plastic waste directly release substantial fine particulate matter with immediate human health consequences. For this reason, our main analysis focuses on PM10 as the dominant local environmental margin, while broader environmental outcomes—including CO_2e emissions and additional pollutants—are examined in Appendix E.3 and Appendix E.4.

As highlighted above, ONS affected local pollution in Turkiye primarily through two environmental margins: changes in waste disposal (ΔB_k) and changes in recycling and re-use (ΔX_k). Both margins may contribute to pollution, but they differ substantially in expected severity. Increases in waste disposal—particularly through dumping or open burning—are

[31]Virgin plastics are defined as imports of crude petroleum, coal, palm oil, natural gas, and plastic in its primary form.
[32]Columns (1) and (2) show that more exposed *Using* firms experience significant employment and sales gains, consistent with downstream production expansion from cheaper imported recycled inputs. Notably, this growth is not accompanied by a corresponding increase in observed waste generation (in Column (4)). This suggests that the expansion of *Using* firms following ONS did not meaningfully increase their own waste output, even as they grew economically.
[33]PM10 is a standard indicator of air quality which refers to particulate matter with a diameter of 10 micrometres or less. Elevated PM10 concentrations pose significant health risks, as inhalation is associated with reduced lung function and increased blood pressure.

likely to generate the largest PM10 consequences, as these activities directly emit harmful particulate matter into the atmosphere. By contrast, recycling and re-use may also generate particulate emissions through sorting, shredding, and processing, but these effects are typically smaller in magnitude. In our empirical setting, these two margins map naturally to the two directly exposed sectors: changes in disposal primarily arise through *Supplying* firms, whose unsold waste is diverted away from formal management, while changes in recycling and re-use primarily operate through *Using* firms, whose access to cheaper imported recycled inputs expands.

This distinction provides a theory-guided framework for linking firm-level environmental responses to local pollution outcomes. As shown in Column (3) of Table 2, the clearest direct environmental response to ONS is the rise in waste mismanagement among more exposed *Supplying* firms, implying that the primary local pollution consequences of ONS are likely to operate through ΔB_k. Because open dumping and burning of plastic waste are expected to be the dominant sources of particulate emissions, we begin by quantifying this disposal channel as our baseline estimate of the local air-pollution costs of ONS. We then return to the secondary ΔX_k margin—changes in recycling and re-use among *Using* firms—to assess whether expanded recycled-input processing generated additional PM10 exposure beyond the primary disposal channel.

We estimate the impact of ONS on PM10 levels by exploiting regional variation in pre-policy plastic waste exposure. Because PM10 is a local pollutant, changes in disposal practices are expected to have geographically concentrated effects. We therefore test whether PM10 levels increased more in cities with greater pre-ONS plastic waste generation, proxied by the concentration of *Supplying* firms in each city, where waste displaced from formal management was most likely to be diverted, with open burning emerging as the primary method for mismanaged disposal after the ONS policy.

We combine two data sources. First, we use the Manufacturing Industry Waste Statistics survey to construct the geographic distribution of plastic waste generation across Turkish cities prior to the ONS policy. Specifically, for each city c, we compute the share of total national plastic waste generated in c in 2016 by firms in the *Supplying* sector, denoted Exposure_c.[34] Second, we obtain air quality data from the Ministry of Environment, Urbanization, and Climate Change, which operates a network of continuous monitoring stations reporting daily PM10 concentrations in micrograms per cubic metre (μg/m^3). Each station is identified by a location name, typically a district or neighbourhood (for example, Mamak within Ankara), and we map stations to Turkish cities. Most cities host multiple stations, particularly in large metropolitan areas. We keep the data at their original level of disaggregation: the estimation sample is a station-day panel of log daily PM10 readings over 2015-2019 across 81 cities.

We estimate the following difference-in-differences specification:

$$\begin{aligned}\ln PM10_{sct} =& \beta \, \text{Post}_t \times \text{Exposure}_c + \gamma \, \text{Post}_t \times \ln \text{Population Density}_{c,2015} \\ &+ \delta \, \text{Post}_t \times \ln \text{Area}_c + \phi \, \ln \text{Rainfall per km}^2{}_{ct} + \alpha_{sm} + \alpha_t + \alpha_{n(c)y} + \varepsilon_{sct},\end{aligned} \tag{4.5}$$

where $PM10_{sct}$ is the daily PM10 reading at monitoring station s located in city c on date t (with year $y(t)$ and calendar month $m(t)$). Post_t equals one for years after 2017, and zero otherwise. Exposure_c is the pre-policy share of national plastic waste generated in city c. Controls interacted with Post_t absorb differential trends in PM10 that are correlated with baseline population density ($\ln \text{Population Density}_{c,2015}$) and city land area ($\ln \text{Area}_c$). Contemporaneous log rainfall per unit area $\ln \text{Rainfall per km}^2{}_{ct}$ captures time-varying meteorological removal of airborne particles through precipitation.

The specification includes three sets of fixed effects, each of which plays a distinct identifying role. Station-by-month fixed effects (α_{sm}) absorb station-specific seasonality—each station's own winter baseline, summer baseline, and so on—as well as all time-invariant station characteristics (location, altitude, instrument type, surrounding land use). Date fixed effects (α_t) absorb common national shocks on any given day, including weather systems, holidays, and national-level economic shocks. Region-year (NUTS2–year) fixed effects ($\alpha_{n(c)y}$)

[34]Turkiye is divided into 81 cities (NUTS3 units); NUTS2 regions group these cities into 26 broader units.

absorb regional annual trends, including regional weather patterns and regional economic shocks. Standard errors are twoway clustered at the city-month and month-day levels, allowing for serial correlation in pollution within the same city-season and for spatial correlation across cities on the same calendar date. Observations are weighted by 2015 city population so that estimates reflect population-weighted exposure relevant for the welfare calculation.[35]

Identification of β therefore relies on within-station-calendar-month, across-year variation in daily PM10 readings, net of common national daily shocks and regional annual trends. Intuitively, for each station we compare readings in, say, March 2018 and March 2019 (post-ban) to readings at the same station in March 2015 and March 2016 (pre-ban), and ask whether the post-ban increase is larger at stations located in cities with greater pre-policy plastic-waste generation. Under the identifying assumption that, absent the ONS policy, high- and low-exposure cities would have experienced parallel trends in PM10 readings after accounting for station-seasonal patterns and regional-year shocks, β identifies the reduced-form causal effect of the shock on local air pollution operating through changes in domestic waste management.

The estimated coefficient is $\hat{\beta} = 0.925$, statistically significant at the 1% level. Evaluated at the mean exposure across affected cities ($\overline{\text{Exposure}_c} = 0.21$), this implies a post-ONS increase in daily PM10 of approximately $\exp(0.925 \times 0.21) - 1 \approx 21.4\%$ relative to cities that are not directly exposed to domestic plastic waste management (Table B.2 in the Appendix shows the full results). To assess the pre-trend assumption and trace out the dynamics of the pollution response, we also estimate an event-study version of Equation (4.5) in which $\text{Post}_t \times \text{Exposure}_c$ is replaced by a full set of year-specific interactions $\sum_{l=2015}^{2019} \beta_l \, D_y^l \times \text{Exposure}_c$, with 2016 omitted as the reference year. Figure 4.1 plots the estimated β_l coefficients with 90% confidence intervals. The 2015 coefficient is small and not statistically distinguishable from zero, consistent with parallel pre-trends. The data collection begins in 2015, which limits our ability to evaluate pre-trends over a longer pre-policy period. The coefficient rises modestly in 2017 (the transition year, when ONS restrictions

[35] Unweighted estimates are qualitatively similar but smaller in magnitude, consistent with pollution effects being concentrated in more populous industrial cities.

were progressively tightened), and increases sharply and significantly in 2018 and 2019—the years in which the full set of ONS restrictions were in force and the diverted waste inflows peaked. This dynamic pattern mirrors the firm-level responses in Sections 3.1-3.2: the import surge, the drop in domestic waste sales, and the rise in mismanagement all emerge after 2017 and build over time. We conclude that pollution increases more in cities where domestic waste *generators* are located, where displaced waste is dumped or burned.[36, 37]

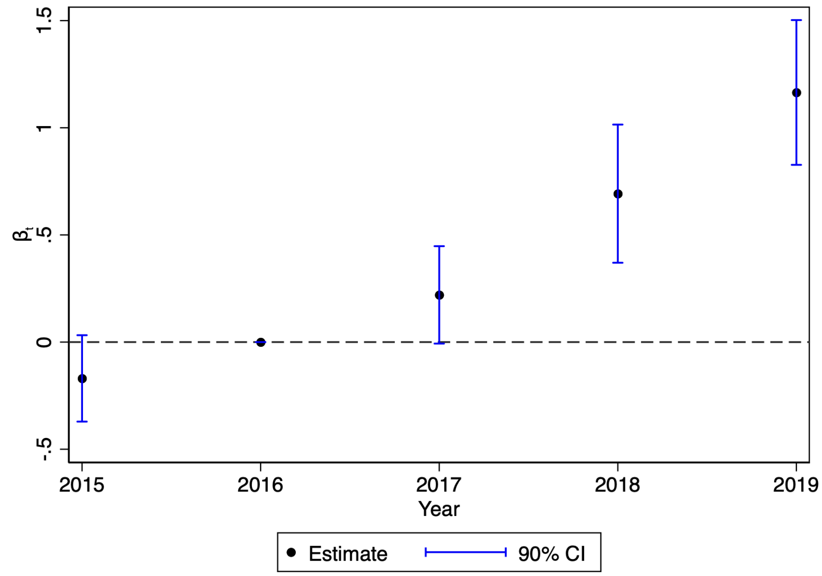
The results above capture relative differences across regions with greater exposure to *Supplying* firms compared to regions with little or no such exposure, rather than the full absolute environmental consequences of the policy. In this relative analysis, the estimates primarily identify how environmentally damaging activities associated with disposal and waste mismanagement changed differentially across regions. However, the model also implies that the ONS policy may affect pollution through additional channels that are not fully captured by these relative comparisons, including emissions generated by recycling activities themselves and by changes in recycled-input use among *Using* firms. To recover absolute effects, we therefore combine our reduced-form estimates with external evidence on emission intensities from the environmental science literature (Franklin Associates, 2010; Li et al., 2024; Kim et al., 2023); see Appendix E.1 for details. This broader accounting framework allows us to incorporate pollution arising not only from disposal through *Supplying* firms, but also from recycling and the use of recycled inputs through *Using* firms.

Panel (A) of Table 3 reports the resulting estimates. In the benchmark specification (Row (a)), where PM10 emissions arise only in regions with *Supplying* firms, we estimate an increase in PM10 of 21.4%. This estimate implies that a 293 thousand ton increase in imported plastic waste is associated with an increase of approximately 4,368 tonnes of PM10,

[36]To examine robustness, we restrict the exposure measure to small firms and find that the coefficient estimate is similar (Column (2) of Table B.2 in the Appendix). Panel (B) of Appendix Figure B.6 replicates the event study using city-level exposure through smaller firms, and shows an even larger post-ban response, consistent with small firms accounting for most of the increase in waste mismanagement documented in Figure B.6.

[37]Globally, imports get diverted to Turkiye that has higher waste mismanagement rates than China. But within Turkiye, recycling of waste does not become dirtier. This pattern is consistent with the mechanism documented in Section 3.2: importing firms recycle cleaner plastic waste, so their geographic concentration is not associated with higher PM10.

FIGURE 4.1. Air Pollution in Areas of Domestic Plastic Waste Generation



Note: This figure plots the event-study estimates of β_l together with 90% confidence intervals from a variant of Equation (4.5). Exposure$_c$ denotes the pre-policy share of national plastic waste generated in city c. Each observation is at the monitoring-station-day level. The dependent variable is the log daily PM10 reading (μg/m^3) at station s located in city c on date t. Controls include year-specific interactions with 2015 log population density and log city area, plus contemporaneous log rainfall per unit area. Fixed effects include station-by-calendar-month (absorbing station-specific seasonality and time-invariant station characteristics), day-by-month-by-year (absorbing national daily shocks), and NUTS2-by-year (absorbing regional annual trends). Observations are weighted by 2015 city population. Standard errors are clustered twoway at the city-month and month-day levels. The sample covers the years from 2015 to 2019.

a magnitude that falls within the range implied by emission factors from the literature on open waste burning.[38] Rows (b)–(d) extend this benchmark by incorporating additional pollution generated through recycling activity itself. Specifically, we first use our gravity estimates of ONS-induced changes in recyclable plastic imports to infer changes in recycling

[38]To interpret this magnitude, it is useful to benchmark it against pollution impact factors from the scientific literature on open waste burning. There is no certain conversion rate from plastic burning to PM10, as results depend on waste composition and combustion conditions, but existing estimates suggest that burning one kilogram of waste generates between roughly 8 and 30 grams of PM10 (Bond et al., 2004). Applying this range, burning 293 thousand tonnes of waste would generate between approximately 2,300 and 8,800 tonnes of PM10. Our estimate lies within this range. At the lower end of the range of pollution impact factors, reconciling our estimate would require that imported plastic waste displaces more than one unit of domestic waste burning, consistent with our evidence that imported waste is of higher quality than domestic waste. At the upper end, the same comparison would be consistent with less than one-for-one displacement, or with the fact that not all plastic waste is openly burned but instead managed through alternative disposal or recycling channels.

activity, and then translate those changes into emissions using a range of recycling emission factors from the environmental science literature. All three rows incorporate conversion-stage manufacturing impacts from Kim et al. (2023). Row (b) additionally incorporates recycling process emissions using lifecycle emission factors from Franklin Associates (2010), while rows (c) and (d) instead apply lower- and upper-bound PM2.5 recycling emissions estimates from Li et al. (2024).[39] Across all scenarios, these additional recycling-related emissions are quantitatively modest, increasing the overall PM10 effect by at most around 3.7 percentage points. This pattern indicates that the rise in air pollution following the ONS policy is driven primarily by disposal and mismanagement of waste, rather than by emissions generated through recycling or downstream production using recycled plastic.

The estimates of local air pollution effects are important not only because they are large, but also because they reveal a mechanism that would be largely invisible in standard trade analyses. In conventional models relying solely on trade flows or importer-level data, the focus would naturally fall on the direct effects of imported waste or on the firms that purchase recycled inputs. Such an approach would largely miss the domestic *Supplying* firms that generate, process, and potentially mismanage waste within the broader production network. Our findings show that this omission is first-order: the dominant environmental damages from the ONS policy arise not primarily from the imported waste itself or from importing firms, but from the induced changes in domestic suppliers' waste management and disposal behavior. In other words, understanding the true environmental consequences of trade policy requires tracing how shocks propagate through domestic production and waste networks, rather than focusing only on observed international trade reallocations.

To establish the monetary equivalent of the pollution increases generated by the ONS policy, we draw on the environmental health literature to translate the estimated rise in PM10 exposure into economically meaningful welfare costs. Specifically, we use a value-of-statistical-life (VSL) approach, which maps increased pollution exposure into mortality risk

[39]Plastic pollution research is an ongoing area of inquiry and we use published factors that are closest to our empirical channels, i.e. pertain to plastic types commonly traded and exclude emissions from virgin resource extraction that is often part of lifecycle assessment estimates (but is excluded here as Turkiye does not change its virgin resource production).

and the associated social cost of premature death. Specifically, we combine our estimated PM10 increase with established evidence on the relationship between particulate exposure and mortality risk, focusing on cardiorespiratory mortality, and monetize the implied excess mortality using VSL estimates adapted from Barwick et al. (2026). Intuitively, this approach measures the social welfare cost of the additional premature deaths associated with the pollution increase induced by the ONS policy. Under a range of conservative assumptions regarding exposure-response parameters and mortality incidence, Column (2) of Panel A in Table 3 reports mortality costs ranging from 557.78 million USD to 654.86 million USD across specifications.[40]

4.5. **Economic Gains from the ONS Policy.** Having quantified the environmental costs of the ONS policy, we next turn to its potential production gains and how they translate into consumption gains. The reduced-form evidence shows that more exposed *Using* firms expanded after ONS: sales and market shares rose, consistent with lower imported recycled-input prices increasing firm-level competitiveness (Figure 3.7). To compare these gains with the environmental damages estimated above, however, we must translate these firm-level responses into aggregate monetary values.

In this section, guided by the model, we evaluate how the potential consumption-side gains from the ONS policy compare with its environmental damages in Turkiye using two complementary approaches. First, we develop a minimal-structure break-even framework that asks how large *Using*-sector price declines would need to be for consumption gains to fully offset the environmental losses estimated in Section 4.4. Second, we use the full CES structure to derive the maximum plausible monetary welfare gains implied by the model under assumptions most favorable to consumer welfare. Together, these approaches allow us

[40]Mortality costs are calculated by combining our estimated ONS-induced increase in PM10 exposure with evidence that a 10 $\mu g/m^3$ increase in PM10 raises mortality risk by approximately 8% (Barwick et al., 2026). Total deaths are obtained from Turkish Statistical Institute's mortality statistics, while the share of pollution-relevant deaths (primarily cardiorespiratory disease and lung cancer) is based on WHO Global Health Estimates. We then estimate the implied increase in premature mortality attributable to higher PM10 exposure and monetize these excess deaths using value-of-statistical-life (VSL) estimates from Barwick et al. (2026), adjusted to Turkiye by scaling for GDP per capita differences. This approach provides an economically interpretable measure of the mortality burden associated with the policy-induced rise in local air pollution.

to assess both whether ONS could realistically have generated positive net welfare gains and how large such gains could be under optimistic assumptions. Full derivations are provided in Appendix Section D.3 and a summary of all terms in this analysis are summarized in Table D.5. .

Break-even price decline. We begin with a minimal-structure break-even framework that asks the question: how large would the decline in *Using*-sector prices need to be for the consumption-side gains from ONS to fully offset the environmental damages estimated above? ONS induced change in welfare in country k can be written as:

$$\Delta W_k = \Delta Q_k - \Delta Z_k,$$

where ΔQ_k denotes the change in real final consumption and ΔZ_k denotes total environmental damages. Since the empirical results imply that meaningful consumption-side responses occur only through the *Using* sector, aggregate consumption gains can be approximated by the nominal income gains generated by ONS, ΔI_k, net of the increase in expenditure required to purchase *Using*-sector goods at their post-policy prices:

$$\Delta Q_k = \frac{1}{P_k}\left(\Delta I_k - E_k^u \frac{\Delta P_k^u}{P_k^u}\right),$$

where E_k^u is total expenditure on *Using*-sector goods and $\Delta P_k^u / P_k^u$ is the aggregate price change in that sector. Rearranging the condition $\Delta W_k \geq 0$ yields a break-even threshold for prices:

$$-\frac{\Delta P_k^u}{P_k^u} \geq \frac{P_k \Delta Z_k - \Delta I_k}{E_k^u}.$$

This expression provides a direct benchmark: given observed income gains and environmental damages, it tells us how large *Using*-sector price declines would need to be for ONS to generate non-negative welfare. Operationalizing this condition requires three objects: total environmental damages ΔZ_k in monetary units, estimated in Section 4.4 and reported in Panel (A) of Table 3; the ONS-induced change in nominal income, ΔI_k; and total expenditure on *Using*-sector goods, E_k^u.

Using firm-level and product-level estimates on sales, imports, and exports, together with the national income identity, we recover the implied change in domestic income, ΔI_k. Specifically, revenues of domestic *Using* firms increase by \$195.8 million following ONS, while net exports of recycled plastic waste decline by \$92.2 million.[41] This implies:

$$\Delta I_k = 195.8 - 92.2 = \$103.6 \text{ million.}$$

The remaining object is total expenditure on *Using*-sector goods, E_k^u, which is not directly observable because foreign varieties cannot be assigned to sectors in the trade data. World trade data do not distinguish between goods produced using virgin versus recycled plastic, preventing direct observation of total *Using*-sector expenditure, E_k^u. To construct a conservative upper bound on potential consumption gains, we therefore impose a home-bias assumption: expenditure on imported *Using*-sector goods is at most equal to expenditure on domestic *Using*-sector goods. Since observed domestic *Using*-sector revenues are approximately \$1.7 billion (see Table D.1), this implies:

$$E_k^u \leq \$3.4 \text{ billion.}$$

Substituting this expenditure bound and our baseline mortality-based environmental damages from Table 3 ($Z_k^{VSL} = \$557.78$ million) into the break-even condition yields:

$$-\frac{\Delta P_k^u}{P_k^u} \geq \frac{557.78 - 103.6}{3400} = 13.36\%.$$

Thus, even under assumptions chosen to maximize the scope for consumption gains, ONS would need to reduce aggregate *Using*-sector prices by at least 13.36% for Turkiye to break even the pollution damages arising from China's ONS policy.

[41] Column (1) of Table 2 reports the effects of ONS on firm sales. The estimated coefficient implies that a Using firm with mean exposure, ($\bar{S}^u = 0.009$), increased sales by ($\exp(12.11 \times 0.009) - 1 = 11.5\%$). Multiplying this percentage increase by the pre-ONS domestic sales of Using firms, \$1.7 billion (see Table D.1), yields an increase in domestic revenues of approximately \$195.8 million. The trade effects are reported in Table A.1. The estimates imply that imports of plastic waste increased from \$24.35 million to \$121.10 million, while exports increased from \$12.24 million to \$16.78 million. As a result, net imports increased by approximately \$92.2 million.

We next assess whether such a price decline is plausible. Panel (B) of Figure 3.1 shows that ONS reduced imported recyclable plastic prices by roughly 25% in trade values. Let S_f^u denote the share of imported recycled plastic in total production costs for a *Using* firm. To give the consumption channel the greatest possible scope, suppose that imported input-cost reductions are passed through completely into final-good prices. Under this assumption, the magnitude of the decline in *Using*-sector output prices is bounded above by

$$-\frac{\Delta P_k^u}{P_k^u} \leq 25\% \times S_f^u.$$

Combining this expression with the break-even condition derived above implies that welfare neutrality requires

$$S_f^u \geq \frac{13.36\%}{25\%} = 53.4\%.$$

In other words, imported recyclable plastic would need to account for more than one-half of total production costs for ONS-induced price declines to fully offset the environmental damages generated by the policy. It is difficult to reconcile a recyclable-plastic-import cost share exceeding 50% because the observed average share was about 1% prior to ONS. At the upper end, Turkiye's imports of plastic recyclables rose by about 1200% by 2020 (Figure A.1). This would take the average cost share in the post-ONS period to about 12%, leaving even the post-ONS share far lower than the 50% threshold needed to achieve welfare neutrality.

Therefore, even under assumptions chosen to maximize potential consumption gains, the price effects required to offset environmental damages appear implausibly large. This minimal-structure exercise alone therefore implies that, once local pollution costs are incorporated, ONS generated negative net welfare in Turkiye.

Model-implied consumption gains. We next use the full CES structure of the model to quantify the monetary value of ONS-induced consumption gains. Because statistically significant sales responses arise only in the *Using* sector, the change in real consumption can be summarized by:

$$P_k \Delta Q_k = \Delta I_k - E_k^u \frac{\Delta P_k^u}{P_k^u} = \Delta I_k - E_k^u \left[\frac{1}{\sigma_u - 1} \left(\frac{\Delta R_{fkk}^u}{R_{fkk}^u} - \frac{\Delta E_k^u}{E_k^u} \right) + \frac{\Delta p_{fkk}^u}{p_{fkk}^u} \right] \tag{4.6}$$

Equation (4.6) decomposes ONS-induced consumption gains into two distinct components: first, the change in nominal income, ΔI_k, and second, compensating variation of the consumption gain arising from lower aggregate *Using*-sector prices. As shown above, the implied income gain is \$103.6 million. The compensating variation term is in turn disciplined by five sufficient statistics: (i) firm-level revenue responses, $\Delta R_{fkk}^u / R_{fkk}^u$; (ii) aggregate expenditure changes in the *Using* sector, ΔE_k^u; (iii) firm-level output price responses, $\Delta p_{fkk}^u / p_{fkk}^u$; (iv) the within-sector elasticity of substitution, σ_u; and (v) total sectoral expenditure on *Using* goods, E_k^u, which is not directly observed.

Each object in Equation (4.6) is disciplined by the model and data; full derivations and empirical details are provided in Appendix D.3. First, firm-level revenue changes, $\Delta R_{fkk}^u / R_{fkk}^u$, are estimated using the difference-in-differences specification above in equation 4.4. As shown earlier in Table 2, exposed *Using* firms experienced statistically significant revenue gains following the ONS policy. Second, aggregate expenditure changes in the *Using* sector, ΔE_k^u, are recovered by combining firm-level estimates with gravity results, balanced trade, and the national income identity. Because expenditure outside the *Using* sector remains unchanged, all expenditure-side adjustments occur through this sector, therefore the change in income ΔI_k will equal change in expenditure in Using sector ΔE_k^u and is approximately \$103.6 million.

Finally, firm-level price changes, $\Delta p_{fkk}^u / p_{fkk}^u$, and the within-sector elasticity of substitution, σ_u, are estimated using firm-product sales and prices from Prodcom data, instrumenting prices with policy exposure (Post $\times$ S_{fk}^u). Suppressing sector-country j, k scripts, the system of equations for domestic sales, domestic prices and the share of banned plastic waste imports of firms making product $\iota \in n, u$ at time t is:

$$\ln R_{f\iota t} = \beta^D \ln P_{f\iota t} + \alpha_{it} + \alpha_{f\iota} + \varepsilon_{f\iota t} \tag{4.7}$$

$$\ln P_{f\iota t} = \beta^P Post_t S_f^u + \alpha_{it} + \alpha_{f\iota} + \epsilon_{f\iota t} \tag{4.8}$$

where α_{it} are industry-time fixed effects and $\alpha_{f\iota}$ are firm-product fixed effects. The first stage coefficient β^P estimates the final relative price impact from exposure to banned plastic waste imported inputs after the ban, relative to non-exposed firms. It is estimated to be -0.243 (0.0265), implying that exposed *Using* firms also lowered output prices (relative to non-exposed firms and hence also in absolute terms). The second stage measures the structural relationship between expenditures and their relative prices. Earlier we had estimated the change in revenues with respect to the initial recyclable plastic exposure of firms in Column (5) of Table 2. The second stage now is at the level of a firm-product and estimates the slope of revenues with respect to prices to determine the welfare-relevant elasticity of substitution for *Using* firms as $\sigma_u \equiv 1 - \beta^D \approx 9.6$, implying a high degree of substitutability across varieties (full results are in Appendix D.2).

As discussed earlier, the share of the *Using*-sector in national expenditure, E_k^u, is not directly observable because trade data do not distinguish plastic products made from recyclable and virgin plastic material. Compensating variation is increasing in the magnitude of sectoral expenditure, and we can therefore adopt a consumption-maximizing calibration by setting the share to zero. This yields an upper bound on the consumption gains that ONS could plausibly generate through lower *Using*-sector prices of

$$CV_k \leq \$12.05 \text{ million.}$$

Adding this upper bound to the estimated nominal income gain yields maximum plausible economic gains of:

$$P_k \Delta Q_k = \Delta I_k + CV_k \leq \$115.65 \text{ million.}$$

These income and compensating variation gains are reported in Panel (B) of Table 3. Even this upper bound remains substantially smaller than estimated mortality-based environmental damages of \$557.78–\$654.86 million. Therefore, even under assumptions chosen to maximize consumer gains, ONS generates strongly negative net welfare:

$$P_k \Delta W_k = P_k \Delta Q_k - P_k \Delta D_k \in [-\$539.21,\ -\$442.13] \text{ in million.}$$

Break-even PM10 valuation. As an alternative benchmark, we can also ask what social cost of PM10 would be required for ONS to be welfare-neutral. Using total PM10 increases of 4,368–5,128 tonnes, the implied break-even PM10 shadow price is approximately \$22.55–\$26.48 per kilogram as reported in the last column of Panel (A) of Table 3.[42] By contrast, our mortality-based estimates imply a PM10 cost of roughly \$127.68 per kilogram. In other words, PM10 damages would need to be valued at only about one-fifth of standard mortality-based estimates for ONS to break even.

Taken together, these results yield a sharp conclusion: while ONS generated meaningful private gains for a subset of Turkish *Using* firms through cheaper imported recycled inputs, these gains were quantitatively far too small to offset the much larger environmental damages generated by increased plastic waste processing and pollution exposure.

4.6. **Extensions.** We complement the baseline pollution quantification in Section 4.4 with four additional exercises that broaden the welfare interpretation of the ONS policy along both revealed-preference and broader environmental dimensions. Together, these extensions ask whether our baseline mortality-based estimates understate or overstate the true social costs of plastic waste reallocation by incorporating government behavior, global pollutants, broader toxicological externalities, and cross-country pollution spillovers.

4.6.1. *Waste Fines.* We endogenize environmental enforcement by allowing the Turkish government to choose inspections and fines on improperly disposed plastic waste. Observed fines provide a revealed-preference lower bound on the government's valuation of environmental damages: under standard Pigouvian logic, the expected fine on illegal disposal must

[42]We obtain the total PM10 increase by multiplying the percentage increase in PM10 reported in Panel (A) of Table 3 by Turkiye's pre-ONS PM10 levels.

Table 3. Economic Gains and Air Pollution Costs in Turkiye

Panel A: Environmental Costs	(1)	(2)	(3)
Pollution Source	Δ PM10 (%)	Mortality Cost (VSL) (million USD)	Maximum Break-even PM10 Price (USD/kg PM10)
(a) Disposal only	21.44	557.78	26.48
(b) (a) + recycling + Using firms	21.93	570.63	25.88
(c) (a) + PM2.5 from recycling (low) + Using firms	23.49	611.23	24.16
(d) (a) + PM2.5 from recycling (high) + Using firms	25.17	654.86	22.55
Panel B: Economic Gains			
Income change		103.6 million USD	
Compensating Variation of price changes		≤12.1 million USD	

Notes: All monetary values are reported in 2016 USD.

Panel A. Reports environmental damages from increased particulate pollution associated with expanded plastic waste activity following ONS. Column (1) reports the estimated percentage increase in particulate exposure under each pollution scenario. Column (2) reports associated mortality costs based on a Value of Statistical Life (VSL) approach following Barwick et al. (2026). Column (3) reports the maximum break-even PM10 price (USD/kg) required for environmental damages to exactly equal the maximum plausible economic gains reported in Panel B.

Row (a) includes disposal-related emissions only. Rows (b)–(d) additionally incorporate emissions from recycling processes and plastic conversion-stage manufacturing impacts from Kim et al. (2023). Row (b) uses recycling process emission factors from Franklin Associates (2010), while Rows (c) and (d) instead apply lower- and upper-bound PM2.5 recycling emissions estimates from Li et al. (2024). Because PM2.5 is typically considered more harmful to human health, a one-to-one conversion ratio between PM10 and PM2.5 is a conservative lower bound of the change in pollution reported in Column (1). Across all scenarios shown in the rows, estimated environmental damages substantially exceed maximum plausible economic gains, implying negative net welfare under standard pollution valuations.

Panel B. Reports model-implied economic gains in Turkiye from ONS. Income change reflects the estimated increase in domestic *Using*-sector revenues net of changes in recycled plastic trade flows from the national income identity. Compensating variation (CV) reports the upper bound on consumption gains from lower *Using*-sector prices under a consumption-maximizing calibration that minimizes sectoral expenditure. Together, these values provide an upper bound on total plausible economic gains from ONS.

be weakly below the true marginal social damage when enforcement is costly and political-economy pressures distort policy (details are in Appendix E.2). Following the surge in public scrutiny over imported waste, Turkiye substantially increased inspections and penalties afterwards, implying that even the government's own policy choices place meaningful value on reducing plastic-related environmental harm. This revealed-preference exercise therefore provides an independent benchmark that is directly observed from fine collections, that are almost twice the consumption gains. This reinforces our baseline conclusion that environmental damages were economically substantial relative to the private gains generated by increased plastic waste imports.

4.6.2. *Global Pollutants.* We consider global pollution, measured by CO_2e emissions in Appendix E.3. Unlike PM10, CO_2e is a global pollutant, so emissions generated abroad are relevant for welfare in Turkiye. We quantify two channels through which the ONS policy may affect global emissions: waste-site emissions (reflecting mismanagement and open burning) and emissions from virgin plastic production (reflecting substitution between recycled and virgin inputs).[43]

Using country-level data, we report a range of estimates from OLS and IV estimation (where waste imports to a destination are instrumented with the distance and deficit/surplus it has with source countries based on their initial export shares of bannned waste to China). The identification assumption, consistent with the global reallocation of waste trade documented in Section 2.2, is that destinations more accessible to former China-supplying origins, either through favorable shipping imbalances or through geographic proximity, face lower marginal costs of importing plastic waste after the ban.

Across specifications, increases in plastic waste imports raise emissions from waste sites but reduce virgin plastic production.[44] On net, we find that the ONS policy increases global CO_2e emissions, primarily driven by increased virgin plastic production in China. Aggregating across countries, this corresponds to an increase of between 1,256.86 and 3,772.53 million kg of CO_2e globally, with the lower bound reflecting OLS estimates and the upper bound reflecting IV estimates. Allocating this increase by population share implies an additional burden of between 12.27 and 36.83 million kg of CO_2e for Turkiye, equivalent to approximately 1.77 to 5.32 million USD under standard European carbon pricing. Taken together, these findings suggest that accounting for global pollutants would, if anything, amplify the environmental costs of the policy.

[43]While we do not detect substitution between recycled and virgin plastic within Turkiye, this channel is quantitatively relevant in the cross-country analysis.

[44]Our estimates from our IV and OLS regressions are broadly consistent with existing evidence from China. For example, Sun and Tabata (2021) document that China's 99.8% reduction in plastic waste imports following the ONS policy was accompanied by roughly a 10% increase in virgin resin production. Complementing this substitution channel, Shi and Zhang (2023) use satellite data on open burning to study the environmental consequences of the ONS policy within China and estimate a 14% decline in open-burning episodes in more exposed areas. Evaluated at their mean exposure of 38%, this implies approximately a 5.3% reduction in waste mismanagement, consistent with our finding that ONS reduced pollution within China while partially shifting environmental damages abroad.

4.6.3. *Other Pollutants.* Moving beyond PM10 and CO_2e, we also examine a broader range of environmental externalities by combining our estimated ONS-induced increase in domestic and mismanaged plastic waste in Turkiye with lifecycle impact intensities from Li et al. (2024) in Appendix E.4. Across these measures, conventional air pollution and climate-related impacts rise by 13% and 24% respectively. The air pollution estimates provide an independent cross-check on the observed PM10 rise that we have estimated. Air pollution from the lifecycle assessment intensities ranges from 13% for PM2.5 and 21% for smog. Though PM10 is not available, these different measures of air pollution are in a similar ballpark.[45]

By contrast, the largest increases arise in toxicity-related categories: human cancer toxicity rises by 23%, non-cancer human toxicity by 19%, and land-based ecotoxicity by 35%. These patterns suggest that the environmental consequences of plastic waste reallocation extend well beyond short-run air pollution, with especially large impacts stemming from persistent toxic exposures associated with waste burning, informal recycling, and environmentally harmful disposal practices. Overall, these results reinforce that the China ban's diversion of plastic waste toward Turkiye generated substantial broader environmental damages, particularly through toxicity and ecosystem-related channels.

4.6.4. *Global Pollution.* Our focus in this paper is on host country economic and environmental effects of pollution-intensive trade diversion. Because comparable micro-level data are not available outside Turkiye, we cannot conduct the same analysis for each relevant country group. However, we have estimated plastic waste import changes across country groups. We can therefore extrapolate what the import changes imply about PM10 changes in other countries. Using the PM10-import elasticity estimated for Turkiye, this exercise implies that PM10 declines by 3.35% in China, but increases by 0.34% among Top Exporters and 0.45% in the rest of the world.[46] These results suggest that the reduction in pollution

[45] PM10 is more directly related to open burning of plastic waste, than PM 2.5.

[46] The rise in Top Exporters is consistent with evidence for the United States. Sigman and Strow (2024) find that landfilling of waste increased while employment in recycling facilities temporarily declined in US states that were more exposed to the ONS ban, relative to those that were less exposed.

in China is partially offset by increases elsewhere, consistent with the global reallocation of waste trade. Admittedly, this exercise is merely suggestive but it is directionally consistent with commentary on trade and pollution during this episode.

5. Conclusion

This paper highlights the international effects of trade and environmental policy, focusing on China's waste import ban. The policy led to a large displacement of waste exports that previously flowed from advanced economies to China. A substantial part of these exports got diverted to Turkiye after China banned them, and just a few years after the ban, Turkiye had become the top importer of plastic waste in the world.

To understand the economic and environmental consequences of trade diversion, we incorporate waste trade in a gravity model of trade and the environment and study the welfare effects of the import ban. In addition to the usual economic gains from trade, changes in welfare depend on three main environmental effects: changes in waste mismanagement of domestic firms, changes in the environmental burden from recycling waste, and substitution possibilities between virgin resources and recyclables.

The theoretical framework provides market clearing conditions that enable a quantification of the aggregate national welfare impacts. Combining labour market clearing and balanced trade with a firm-level difference-in-differences analysis, we find that plastic using firms in Turkiye gained access to cheaper inputs due to China's policy, and experienced better outcomes such as increased sales and profitability. However, demand for locally generated waste in Turkiye dropped, resulting in higher levels of waste mismanagement by domestic plastic waste generators.

Turkiye also experienced worse environmental outcomes in terms of increased air pollution, that we can measure directly and through emission factors. Air pollution increased in cities where plastic waste generators were concentrated, with negligible offsetting effects from reduced virgin plastic production in the country. Our estimates show an increase in air pollution by 21.44-25.17% on average. While the ONS policy generated consumption gains in Turkiye and has been documented to have generated environmental benefits within China, these gains came with increased air pollution that was amplified through displacement of domestic waste by imports. A sizable share of the environmental burden was redistributed rather than eliminated. China's environmental improvements and the reduced waste burden of advanced economies have been partly achieved through the relocation of environmental costs to countries less equipped to manage them.

The central lesson of the paper is that environmental regulation can transmit pollution internationally through channels that are difficult to observe from trade flows alone. In our setting, the largest environmental damages did not arise because imported waste was processed more intensively in Turkiye. They arose because cheaper imported waste displaced domestic suppliers, causing a deterioration in domestic waste-management practices. Evaluating environmental regulation solely through the relocation of regulated activity may therefore substantially understate its international environmental consequences.

More broadly, our findings suggest that the international incidence of environmental regulation can differ sharply from its domestic incidence. Policies that generate environmental improvements in one country may create environmental burdens elsewhere, particularly in economies with weaker waste-management systems and limited environmental enforcement. As international negotiations increasingly focus on environmentally intensive activities, understanding these displacement channels and firm responses become crucial for evaluating the global consequences of environmental policy.

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Appendix A. Global Plastic Waste Trade After ONS

A.1. **Tables and Figures.**

Figure A.1. Imports of Plastic Waste

Note: Each bar represents the amount of imports of ONS-banned plastic waste (in million kg) by country in a given year.

Table A.1. Effect of China's ONS Policy on Bilateral Trade in Plastic Waste

$\ln(\text{value}_{odpt})$	(1)	(2)
$\text{Post}_t \times \text{Banned}_p^{\text{HS6}} \times$		
$\text{CHN}_o \times \text{Top Exporters}_d \times \text{Surplus}_{od}$	-0.460	-0.588
	(0.366)	(0.367)
$\text{CHN}_o \times \text{RoW}_d \times \text{Surplus}_{od}$	0.098	-0.027
	(0.374)	(0.375)
$\text{CHN}_o \times \text{Top Exporters}_d \times \text{Deficit}_{od}$	0.154	0.050
	(0.232)	(0.233)
$\text{CHN}_o \times \text{RoW}_d \times \text{Deficit}_{od}$	-0.242	-0.348[c]
	(0.211)	(0.212)
$\text{TUR}_o \times \text{Top Exporters}_d \times \text{Surplus}_{od}$	-0.125	-0.235
	(0.620)	(0.622)
$\text{TUR}_o \times \text{RoW}_d \times \text{Surplus}_{od}$	0.986[c]	0.886[c]
	(0.510)	(0.511)
$\text{TUR}_o \times \text{Top Exporters}_d \times \text{Deficit}_{od}$	-0.658[b]	-0.872[b]
	(0.334)	(0.344)
$\text{TUR}_o \times \text{RoW}_d \times \text{Deficit}_{od}$	-0.163	-0.286
	(0.572)	(0.573)
$\text{Top Exporters}_o \times \text{CHN}_d \times \text{Surplus}_{od}$	-2.199[a]	-2.376[a]
	(0.238)	(0.240)
$\text{Top Exporters}_o \times \text{Top Exporters}_d \times \text{Surplus}_{od}$	0.440[a]	0.403[a]
	(0.087)	(0.088)
$\text{Top Exporters}_o \times \text{TUR}_d \times \text{Surplus}_{od}$	3.196[a]	3.087[a]
	(0.400)	(0.407)
$\text{Top Exporters}_o \times \text{RoW}_d \times \text{Surplus}_{od}$	0.288[a]	0.236[b]
	(0.107)	(0.107)
$\text{Top Exporters}_o \times \text{CHN}_d \times \text{Deficit}_{od}$	-2.397[a]	-2.620[a]
	(0.458)	(0.464)
$\text{Top Exporters}_o \times \text{Top Exporters}_d \times \text{Deficit}_{od}$	-0.096	-0.151
	(0.096)	(0.096)
$\text{Top Exporters}_o \times \text{TUR}_d \times \text{Deficit}_{od}$	1.678[a]	1.578[a]
	(0.240)	(0.242)
$\text{Top Exporters}_o \times \text{RoW}_d \times \text{Deficit}_{od}$	0.202[b]	0.151
	(0.093)	(0.093)
$\text{RoW}_o \times \text{CHN}_d \times \text{Surplus}_{od}$	-0.849[a]	-0.986[a]
	(0.115)	(0.121)
$\text{RoW}_o \times \text{Top Exporters}_d \times \text{Surplus}_{od}$	-0.186[a]	-0.290[a]
	(0.067)	(0.069)

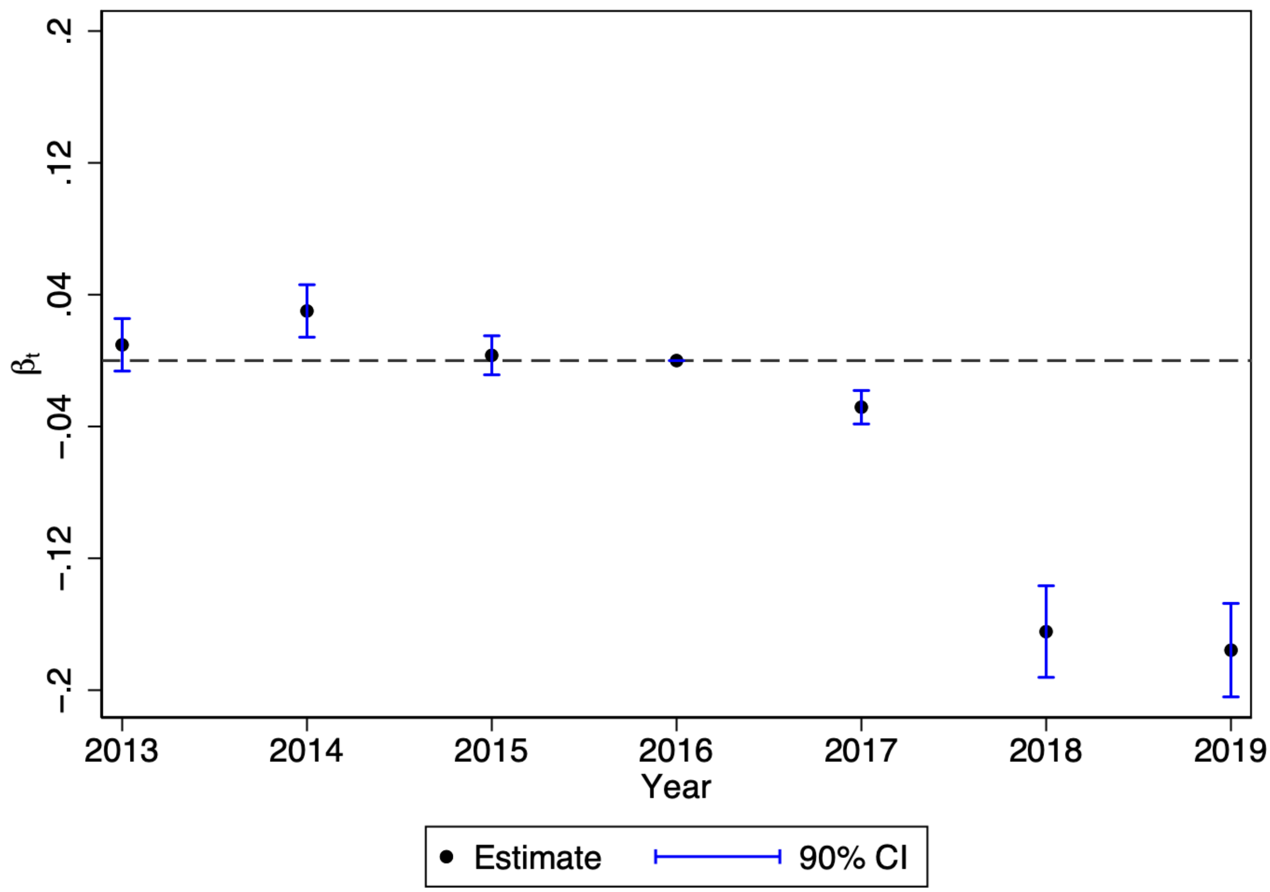
Continued on next page

Table A.1 (continued): Effect on Bilateral Trade

$\ln(\text{Trade}_{odpt})$	(1)	(2)
$\text{RoW}_o \times \text{TUR}_d \times \text{Surplus}_{od}$	0.714^b	0.587^b
	(0.290)	(0.295)
$\text{RoW}_o \times \text{RoW}_d \times \text{Surplus}_{od}$	0.074	0.008
	(0.082)	(0.083)
$\text{RoW}_o \times \text{CHN}_d \times \text{Deficit}_{od}$	-0.494^a	-0.647^a
	(0.133)	(0.162)
$\text{RoW}_o \times \text{Top Exporters}_d \times \text{Deficit}_{od}$	-0.308^a	-0.381^a
	(0.069)	(0.069)
$\text{RoW}_o \times \text{TUR}_d \times \text{Deficit}_{od}$	0.689^a	0.660^a
	(0.197)	(0.202)
$\text{RoW}_o \times \text{RoW}_d \times \text{Deficit}_{od}$	-0.107	-0.175^b
	(0.082)	(0.082)
$\text{CHN}_o \times \text{TUR}_d$	-0.591^a	-0.788^a
	(0.023)	(0.089)
$\text{Post}_t \times \text{Tariff}_{p,2016}$	0.139^a	0.137^a
	(0.014)	(0.014)
Fixed Effects:		
Origin × Destination × Year	Yes	Yes
Product × Origin × Destination	Yes	Yes
Observations	38,976,564	38,976,564
R^2	0.853	0.853

Note: This table presents the results obtained from estimating Equation (2.1). Robust standard errors in parentheses, clustered at the origin-destination-product level. Column (1) has the baseline results and (2) also includes the full set of interactions with $\text{Post}_t \times \text{Banned}_p^{HS2}$, omitted here for brevity. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

Figure A.2. Event Study: Global Trade after the Operation National Sword Policy



Note: This figure plots event-study estimates of the year-by-year coefficients β_l, together with 90% confidence intervals, from a regression of log global trade values for 6-digit HS products on interactions between year dummies D_t^l and $\text{Banned}_p^{\text{HS6}}$, an indicator for products banned by China under the ONS policy. The interaction with year 2016 is omitted as the reference year. The specification includes product and year fixed effects. The sample covers the period from 2013 to 2019.

Table A.2. Change in Trade in Plastic Waste

Dependent Variable: $\frac{Trade_{odpt}}{\sum_{d'} Trade_{pod't}}$	(1)	(2)
$Post_t$ × Banned$_{po}$ × CHN$_d$	-0.173a	-0.173a
	(0.014)	(0.014)
$Post_t$ × Banned$_{po}$	0.0011c	
	(0.00007)	
R^2	0.692	0.692
# observations	9689965	9689965
Fixed Effects:		
Destination×Product×Time	Yes	Yes
Origin×Destination×Time	Yes	Yes
Destination×Origin×Product	Yes	Yes
Origin×Product×Time	No	Yes

Note: This table shows the results from a difference-in-differences specification, where the dependent variable is the share of exports of product p by origin o to destination d at year t. The coefficient of interest is on the triple interaction term: $Post_t \times Banned_{po} \times CHN_d$. Where $Post_t$ is a dummy variable indicating 1 if year is greater than 2017, $Banned_{po}$ indicates the set of China-banned plastic waste products from initial exporter o, and CHN_d takes a value of 1 if the destination country is China. The sample covers the period from 2013-2019. Statistical significance is denoted by letters: a for p < 0.01, b for p < 0.05, and c for p < 0.10.

Table A.3. Change in Plastic Waste Imports by Destination: No Missing Intercepts

Destination	**Change in imports (no missing intercept) (%)**
China	−86.40
Other Destinations	+14.90
Top Exporters	+9.66
RoW	+15.43
Turkiye	+518.50
World	−37.34

Note: This table reports the implied change in imports by destination group using the difference-in-differences estimates, but without adding destination-specific missing intercepts. Reported values are percentage changes relative to baseline imports of China-banned plastic waste.

A.2. **Relative Changes to Absolute Changes in Plastic Waste Imports.** In Section 2.2, we quantify changes in plastic waste trade across country groups using the difference-in-differences specification in Equation (2.1). The corresponding regression results are reported in Table A.1. These estimates are relative to the log change in trade of non-plastic products within each country group.

To recover absolute changes in plastic waste trade flows, we impose a balance-of-trade (BoT) condition. This allows us to solve for the implied log change in non-plastic trade flows, denoted α_c, which plays the role of a missing intercept in the relative specification. Below we derive this adjustment step by step.

Assumption 1: Balanced Trade

For each country k, changes in total imports equal changes in total exports:

$$\sum_{o\neq k}\sum_{h}\Delta x_{okht} \;-\; \sum_{d\neq k}\sum_{h}\Delta x_{kdht} = 0.$$

Assumption 1 would follow from the standard assumption of balanced trade in goods, that is particularly suited to waste trade when it is exported to reduce processing costs by filling empty ships traveling to cheaper processing countries. More generally, when there are trade imbalances, Assumption 1 is the usual assumption of balanced trade in changes in quantitative models of gains from trade.

The balanced trade assumption enables determination of the missing intercept as α_k for each country k as follows:

$$\begin{aligned}
0 &= \sum_{o\neq k}\sum_{h}\Delta x_{okht} - \sum_{d\neq k}\sum_{h}\Delta x_{kdht} \\
&= \sum_{h}\left(\sum_{o\neq k}\Big(x_{okh,t=0}\times\Big(\exp(\alpha_k+\beta^h_{ok,t=T})-1\Big)\Big) - \sum_{d\neq k}\Big(x_{kdh,t=0}\times\Big(\exp(\alpha_k+\beta^h_{kd,t=T})-1\Big)\Big)\right) \\
&= \exp\alpha_k\sum_{h}\left(\sum_{o\neq k}x_{okh,t=0}\exp\beta^h_{ok,t=T} - \sum_{d\neq k}x_{kdh,t=0}\exp\beta^h_{kd,t=T}\right)
\end{aligned}$$

$$-\sum_h \left(\sum_{o\neq k} x_{okh,t=0} - \sum_{d\neq k} x_{kdh,t=0}\right)$$

The missing intercept, α_k therefore can be solved for as

$$\exp \alpha_k = \left[\sum_h \left(\sum_{o\neq k} x_{okh,t=0} - \sum_{d\neq k} x_{kdh,t=0}\right)\right] / \left[\sum_h \left(\sum_{o\neq k} x_{okh,t=0} \exp \beta^h_{ok,t=T} - \sum_{d\neq k} x_{kdh,t=0} \exp \beta^h_{kd,t=T}\right)\right]$$

Having determined the missing intercept α_k for each country, it is straightforward to determine the absolute change in exports from o to k as

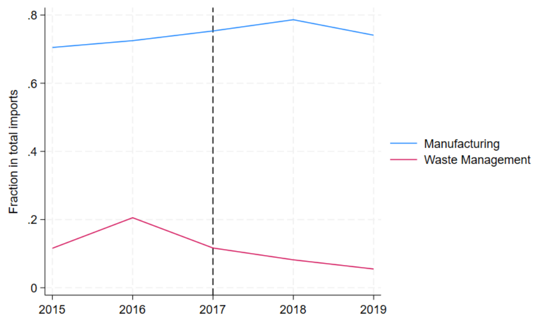
$$\Delta x_{okht} = x_{okh,t=0} \left(\exp\left(\alpha_k + \beta^h_{ok,t=T}\right) - 1\right).$$

Appendix B. Microeconomic Impacts of ONS in Turkiye

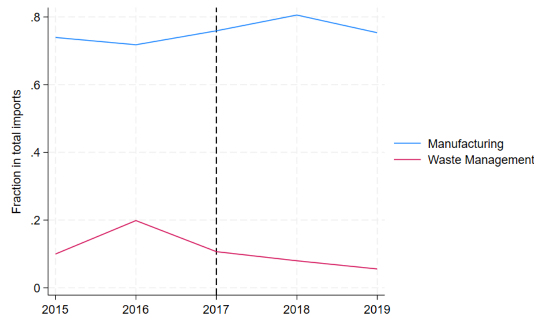
B.1. **Tables and Figures.**

Figure B.1. Share of Manufacturing Firms in Imports

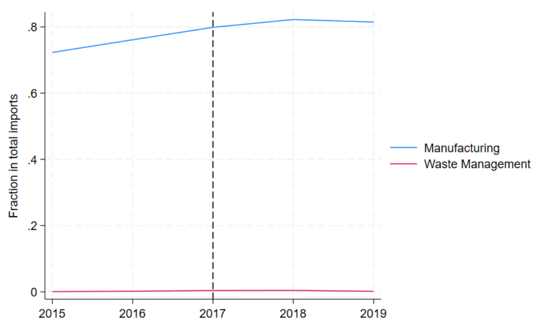
(a) Share of Imports by Quantity: Plastic Waste



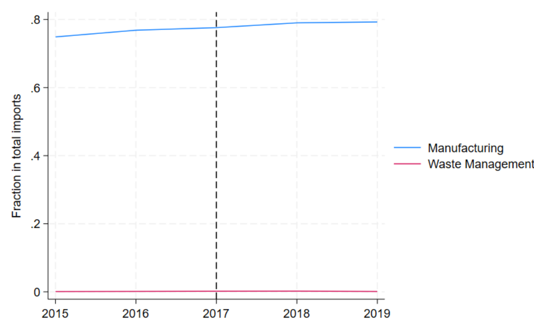
(b) Share of Imports by Value: Plastic Waste



(c) Share of Imports by Value: Other Goods

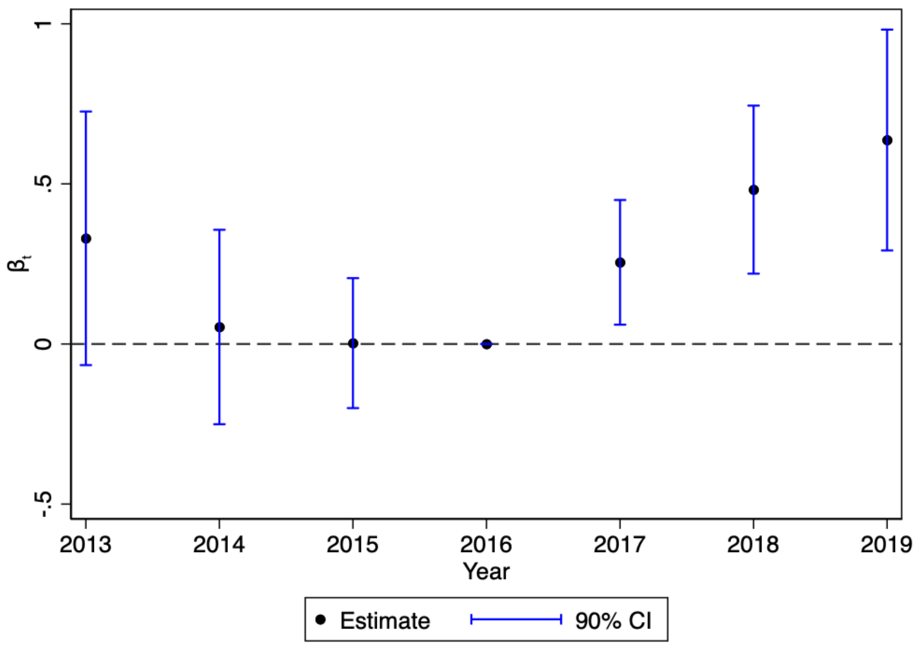


(d) Share of Imports by Value: Other Goods



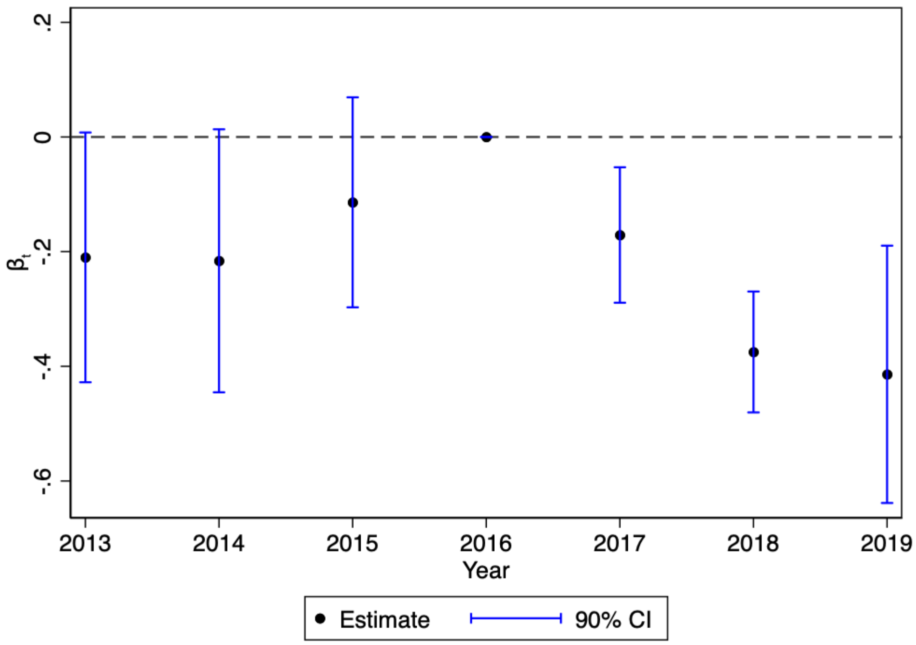
Note: These graphs show the share of imports of plastic waste and other goods by quantity and value over the years by manufacturing firms and waste management firms.

Figure B.2. Event Study: Firm-level Imports



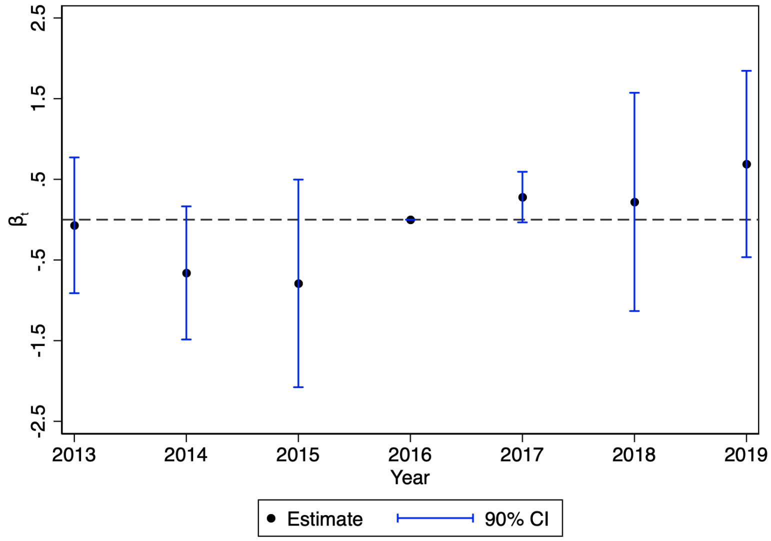
Note: The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from $\ln \text{Imports}_{iopt} = \sum_{l=2013}^{2019} \beta_l D_t^l * \text{Banned}_p^{HS6} + \alpha_{iop} + \alpha_{ot} + e_{iopt}$, where Imports_{iopt} denotes the value of imports of 8-digit HS product p by Turkish firm i from country o in year t, D_t^l are year dummies and Banned_p^{HS6} indicates the set of China-banned plastic waste products. The interaction with year 2016 is removed from the equation to serve as a reference year. The sample covers the years from 2013 to 2019.

Figure B.3. Event Study: Quality Adjusted Import Prices



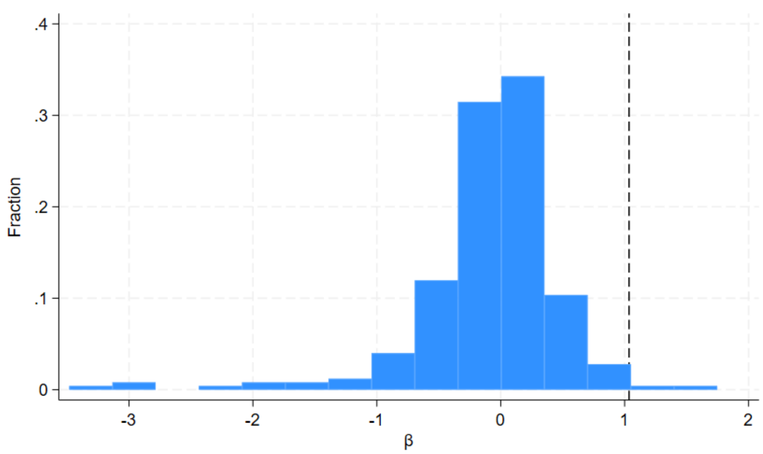
Note: This figure plots the estimates of β_l, together with the 90% confidence intervals, obtained from Equation (3.1), where the dependent variable is quality-adjusted import unit values. Quality of a 8-digit HS product is the residual from regressing the log quantity + 5*log unit value on country-year, product-year, and firm-year fixed effects. The quality adjusted price is log unit value - log quality. The sample covers the years from 2013 to 2019.

Figure B.4. Event Study: Exports



Note: The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from Equation (3.1), where the dependent variable is the quantity of Turkish exports of a 8-digit HS level product. The interaction with year 2016 is removed from the equation to serve as a reference year. The sample covers the years from 2013 to 2019.

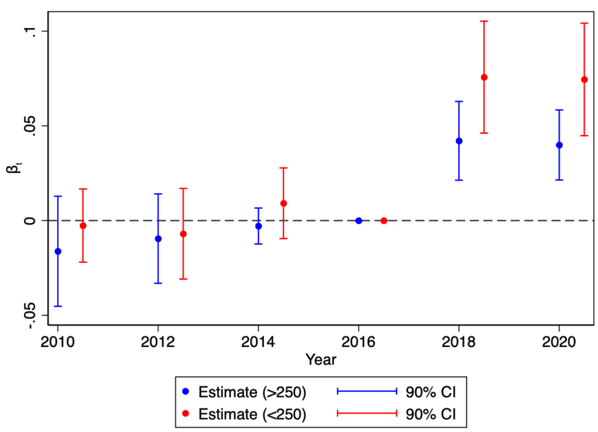
Figure B.5. Imports of Plastic Products Randomly Assigned Treatment



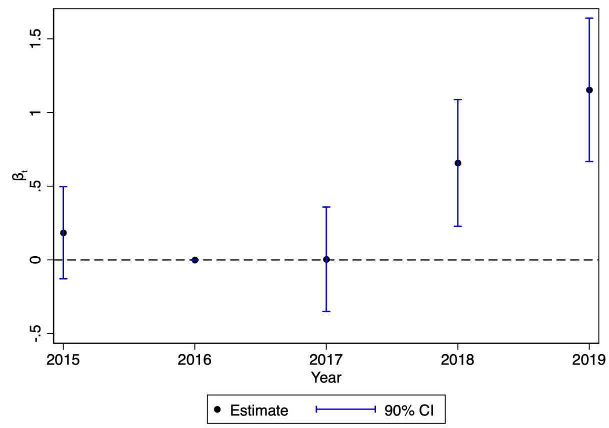
Note: This figure plots the β coefficients of Equation (3.1). The treatment variable randomly allocates treatment status to 8-digit HS products within their corresponding 4-digit HS codes and then re-estimates Equation (3.1). This procedure is replicated 250 times.

Figure B.6. Robustness by Firm Size: Waste Mismanagement and Local Air Pollution

(A) Management of Waste by Turkish Firms, By Firm Size

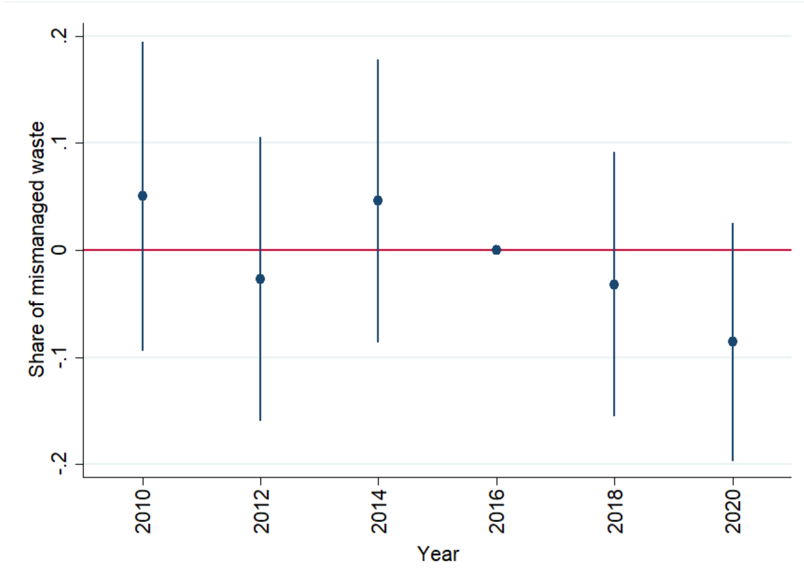


(B) Air Pollution in Areas of Domestic Plastic Waste Generation by Firms with less than 250 Employees



Note: These figures plot event-study estimates of β_l together with 90% confidence intervals. Panel (A) reports estimates from Equation (3.2), where each observation is at the firm-product-year level and the dependent variable is the share of waste that firm m mismanages of waste product p in year t. Coefficients on the interaction between year dummies D_t^l and an indicator for waste products banned by China, $Banned_h$, are plotted separately for two samples: firms with more than 250 employees (blue) and firms with fewer than 250 employees (red). The sample covers the years from 2010 to 2020. Panel (B) reports estimates from a variant of Equation (4.5) in which Exposure_c is the pre-policy share of national plastic waste generated in city c in 2016 by firms with fewer than 250 employees. Each observation is at the monitoring-station-day level, and the dependent variable is the log daily PM10 reading (μg/m^3) at station s located in city c on date t. Controls include year-specific interactions with 2015 log population density and log city area, plus contemporaneous log rainfall per unit area. Fixed effects include station-by-calendar-month, day-by-month-by-year, and NUTS2-by-year. Observations are weighted by 2015 city population, and standard errors are two-way clustered at the city-month and month-day levels. The sample covers the years from 2015 to 2019. In both panels, the interaction with year 2016 is excluded to serve as the reference year.

Figure B.7. Waste Mismanagement by Importers of Banned Plastic Waste



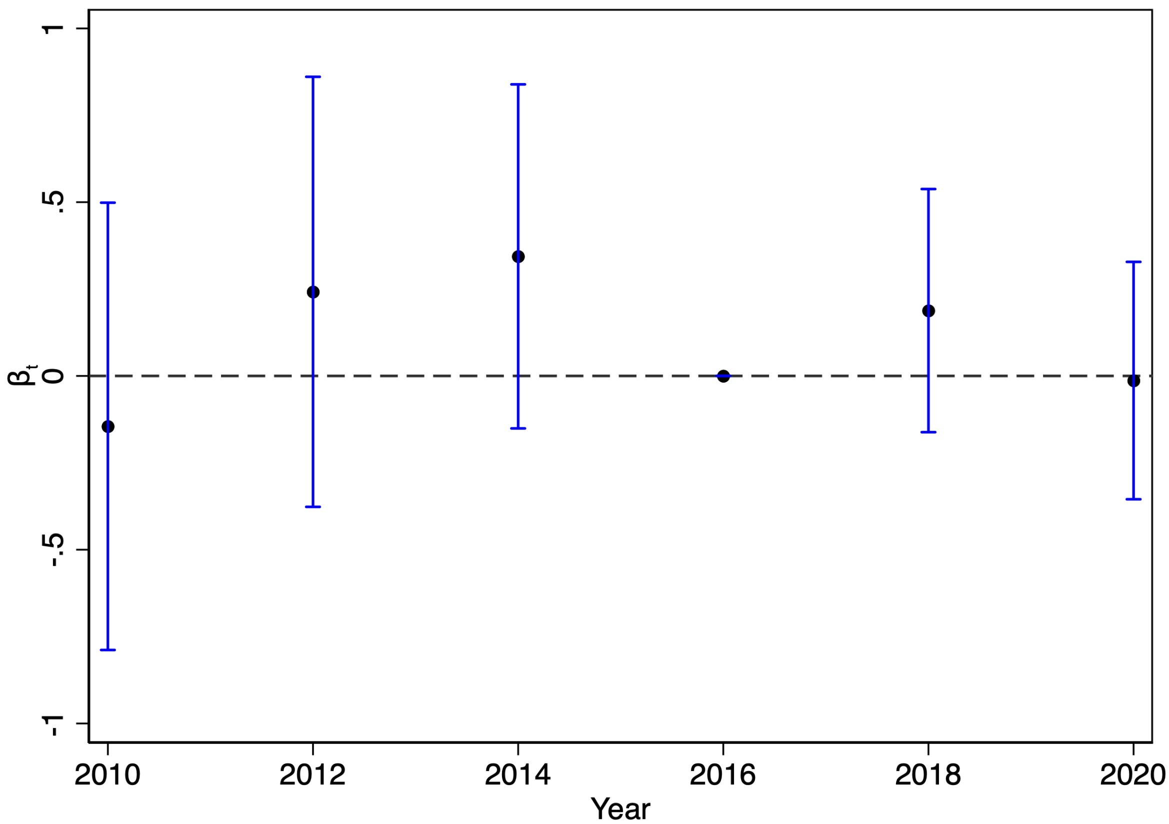
Note: This figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating Equation (3.2) on the sample of importers of plastic waste. Each observation is at the firm-product-year level. The dependent variable is the share of waste that firm m sells of waste product p at year t. Coefficients on the interaction between year dummies D_t^l and an indicator for waste products banned by China, $Banned_h$, are plotted in the figure. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2010 to 2020.

Table B.1. Water Consumption Intensity and Exposure to Imported Plastic Waste

Dependent Variable: $\Delta^{16-18}\left(\frac{\text{Water consumption}}{\text{Sales}}\right)$	(1)	(2)	(3)
Exposure$_f^u$	-1.665c		-1.665c
	(0.906)		(0.907)
Exposure$_f^s$		-0.017	-0.017
		(0.063)	(0.064)
Initial employment	0.053a	0.054a	0.053a
	(0.013)	(0.013)	(0.013)
N	8,215	8,215	8,215
R^2	0.032	0.032	0.032
Fixed Effects:			
Region	Yes	Yes	Yes
Sector	Yes	Yes	Yes

Notes: The dependent variable is the change in water consumption divided by sales between 2016 (pre-ONS) and 2018 (post-ONS). Water consumption data are only available for 2016-2018. Exposure$_f^u$ measures the pre-ONS intensity with which, firm i relied on China-banned imports in total input costs in 2016. Exposure$_f^s$ captures the share of China-banned waste products in total waste generated by firm in 2016. Standard errors are clustered at the 4-digit NACE industry level. Statistical significance is denoted by letters: a for p < 0.01, b for p < 0.05, and c for p < 0.10.

Figure B.8. Total Waste Production of Domestic Waste Generators



Note: This figure plots the PPML estimates of β_l, together with 90% confidence intervals, obtained from estimating Equation (3.3). Each observation is at the firm-year level. The dependent variable is the amount of waste that firm m produces at year t. Coefficients for interaction terms of year dummies D_t^l and $Exposure_m$, where $Exposure_m$ is the share of firm m's production of China banned waste types in its total waste. The interaction with year 2016 is removed from the equation to serve as a reference year. The sample covers the years 2010-2020.

Table B.2. Air Pollution in Areas of Domestic Plastic Waste Generation

Dependent Variable: $\ln(PM10)$	(1) All plastic waste	(2) Plastic waste by firms with less than 250 emp.
$Post_t \times \text{Exposure}_c^{\text{Waste}}$	0.925^a	
	(0.185)	
$Post_t \times \text{Exposure}_c^{\text{Waste, small}}$		0.907^a
		(0.259)
$Post_t \times \ln(\text{Population Density})_{c,t=2015}$	-0.101^a	-0.111^a
	(0.027)	(0.029)
$Post_t \times \ln(\text{Area})_c$	0.019	0.024^c
	(0.012)	(0.012)
$\ln(\text{Rainfall per km}^2)_{ct}$	0.026^b	0.025^b
	(0.012)	(0.012)
R^2	0.399	0.399
# observations	288,706	288,706
Fixed Effects:		
Station × Calendar Month	Yes	Yes
Date (Day × Month × Year)	Yes	Yes
NUTS2 × Year	Yes	Yes

Note: This table reports estimates of the specification in (4.5). The sample is a station-day panel covering 2015-2019, comprising 288,706 observations across monitoring stations in 81 Turkish cities. The dependent variable is the log of the daily PM10 reading (μg/m^3) at station s in city c on date t. $\text{Exposure}_c^{\text{Waste}}$ is the pre-policy share of national plastic waste generated by all firms in city c; $\text{Exposure}_c^{\text{Waste, small}}$ restricts this measure to firms with fewer than 250 employees. Observations are weighted by 2015 city population. Standard errors in parentheses are clustered twoway at the city-month and month-day levels. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

B.2. **Comparison of Product Quality across Countries.**

In subsection 3.2, we discussed that one potential explanation for the displacement of domestic plastic waste by imported plastic waste in the Turkish market is Turkish firms' shift toward higher-quality imported plastic waste, which became cheaper following the ONS policy. Although we lack data to directly compare the quality of imported and domestically produced plastic waste in Turkiye, we provide indirect evidence using detailed bilateral trade data from UNCOMTRADE.

Using bilateral trade data on values and quantities, we estimate a product-level quality measure following the methodology outlined by Khandelwal et al. (2013). In particular, we estimate the following regression:

$$\ln q_{podt} + \sigma ln \mathrm{uv}_{podt} = \alpha_{pt} + \alpha_{dt} + \epsilon_{podt} \tag{B.1}$$

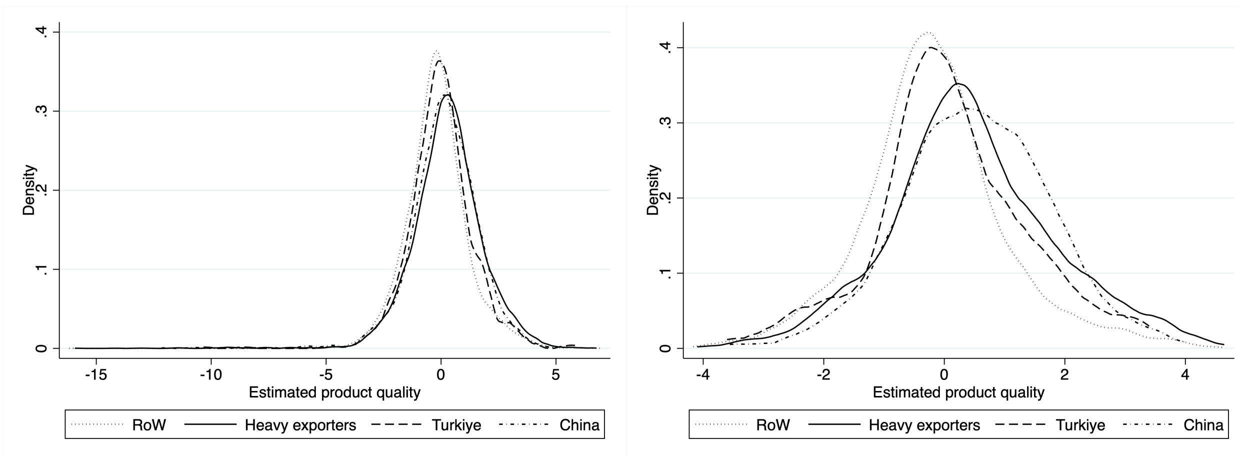
where q_{podt} is the quantity of exports of HS 6-digit product p by country o to country d in year t, and uv_{podt} is its unit value. We set the elasticity of substitution $\sigma = 5$. The estimated quality is $\hat{\epsilon}_{podt}/(\sigma - 1)$. The estimation sample includes all 2-digit HS codes containing at least one waste product and covers the pre-ONS period from 2012 to 2016.

Figure B.9 presents the distribution of estimated product quality, averaged over the 2012-2016 period, based on Equation (B.1). Panel (A) covers all waste products while Panel (B) restricts the sample to plastic waste. In both figures, the quality distribution of waste products exported by countries that China relied on heavily for plastic waste imports before the ONS policy, i.e. *Top Exporters*, first-order stochastically dominates the distribution of products exported by Turkiye.

In Table B.3, we regress the average estimated quality for waste products over the 2012-2016 period on dummy variables indicating the type of exporting countries, namely *China*, *Turkiye*, and *Top Exporters*, with the rest of the world serving as the base category. In both columns, the estimated average product quality for *Top Exporters* is higher than the one for *Turkiye*. In particular, for plastic waste products, the estimated difference is about

Figure B.9. Distribution of Estimated Product Quality by Country Groups

(A) All Waste Products (B) Plastic Waste Products



Note: These figures plot the distribution of estimated product quality, averaged over the 2012–2016 period, based on Equation (B.1). Panel (A) includes all waste products, while Panel (B) restricts the sample to plastic waste.

Table B.3. Estimated Product Quality by Country Group

Dependent Variable: Estimated product quality	(1) All waste products	(2) Plastic waste products
$China$	0.119a	0.404a
	(0.030)	(0.037)
$Turkiye$	0.186a	0.203a
	(0.030)	(0.037)
$TopExporters$	0.397a	0.500a
	(0.058)	(0.082)
R^2	0.012	0.026
# observations	45365	7854

Note: This table reports estimates from regressing the average product quality for waste products, recovered from Equation (B.1), on dummy variables indicating country groups (China, Turkiye and Top Exporters), with the rest of the world as the omitted category. Column (1) uses all waste products; Column (2) restricts to plastic waste. The sample covers the years from 2012 to 2016. Robust standard errors are clustered at the origin-country level. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

35% and is statistically significant at the 1% level. If Turkish firms export higher-quality plastic waste than they sell domestically, the quality gap between imported and domestically available plastic waste in Turkiye is likely even larger, in favor of imports.

Appendix C. Theoretical Framework

C.1. **Welfare.** We begin by defining welfare at the country level k. Welfare is increasing in the consumption index Q_k and decreasing in environmental damages Z_k (in real consumption units) associated with plastic waste disposal B_k, recycling and re-use activities X_k, and virgin plastic extraction V_k. The parameters ξ_{bk}, ξ_{xk}, and ξ_{vk} denote country-specific marginal disutilities from each source of environmental damage. Welfare in country k is given by

$$W_k \equiv Q_k - Z_k \equiv Q_k - \xi_{bk} B_k - \xi_{xk} X_k - \xi_{vk} V_k. \tag{C.1}$$

Aggregate consumption Q_k is a CES composite of sectoral consumption Q_k^j,

$$Q_k \equiv \left(\sum_{j \in \{u,s,n\}} \eta_j \, Q_k^{j\,\frac{\sigma-1}{\sigma}} \right)^{\frac{\sigma}{\sigma-1}}, \qquad Q_k^j \equiv \left(\sum_o \sum_{f \in \mathcal{F}_o^j} q_{fok}^{j\,\frac{\sigma_j - 1}{\sigma_j}} \right)^{\frac{\sigma_j}{\sigma_j - 1}},$$

where $j \in \{u, s, n\}$ indexes three sectors corresponding to *Using*, *Supplying*, and *Neither* (described below). The elasticity of substitution across sectors is $\sigma > 1$. Within each sector j, consumption Q_k^j is a CES aggregate of differentiated varieties q_{fok}^j, where q_{fok}^j denotes consumption in country k of the variety produced by firm $f \in \mathcal{F}_o^j$ in origin country o. The elasticity of substitution within sector j is $\sigma_j > \sigma > 1$, and $\eta_j > 0$ denotes sectoral preference weights (normalised as $\eta_u \equiv 1$).

The environmental components are given by

$$B_k \equiv \sum_{f \in \mathcal{F}_k^s} b_{fk}^s, \qquad X_k \equiv \sum_{f \in \mathcal{F}_k^u} \sum_o \sum_{g \in \mathcal{F}_o^s} x_{f(k)g(o)}^u, \qquad V_k \equiv \sum_{j \in \{u,s,n\}} \sum_{f \in \mathcal{F}_k^j} v_{fk}^j,$$

where, b_{fk}^s denotes waste disposed by firm f in country k, x_{fk}^u denotes the plastic waste recycled and re-used by firm f in country k, and v_{fk}^j denotes virgin plastic input use by firm f in sector j in country k.

C.2. **ONS-Policy.** We analyze the welfare consequences of the ONS policy for Turkiye by modeling the ban as a prohibitive iceberg trade cost on recycled plastic waste exports to China. Let $\tau_{o,d=CHN}$ denote the iceberg trade cost on recycled plastic waste produced in origin country $o \neq CHN$ and sold to firms in China. Under the ONS policy, $\tau_{o,d=CHN} \to \infty$, implying that bilateral recycled-waste flows to China collapse to zero.[47]

The elimination of Chinese import demand exerts downward pressure on the equilibrium price of recycled waste in the world market. Let Δ denote the change in equilibrium outcomes induced by the ONS policy. We define welfare in country k as final consumption net of environmental damages:

$$\Delta W_k = \Delta Q_k - \Delta Z_k = \Delta Q_k - \xi_{bk}\Delta B_k - \xi_{xk}\Delta X_k - \xi_{vk}\Delta V_k, \tag{C.2}$$

The three damage margins correspond directly to the empirical margins identified in Section 3. The environmental components entering welfare are defined as the aggregate changes in disposal, recycling, and virgin input use:

$$\Delta B_k \equiv \sum_{f\in\mathcal{F}_k^s} \Delta b_{fk}^s, \quad \Delta X_k \equiv \sum_{f\in\mathcal{F}_k^u}\sum_{o}\sum_{g\in\mathcal{F}_o^s} \Delta x_{f(k)g(o)}^u, \quad \Delta V_k \equiv \sum_{j\in\{u,s,n\}}\sum_{f\in\mathcal{F}_k^j} \Delta v_{fk}^j.$$

C.3. **Sectors and Firms.** Sectors differ in their production technologies and in the type of waste generated as a by-product of production.

Production. A firm $f \in \mathcal{F}_k^j$ produces output y_{fk}^j with Hicks-neutral productivity ϕ_{fk}^j according to

$$y_{fk}^j = \phi_{fk}^j \; F_j\Big(l_{fk}^j,\; v_{fk}^j,\; \mathbf{1}\{j=u\}\, x_{fk}^j\Big),$$

where l_{fk}^j, v_{fk}^j, and x_{fk}^j denote labor, virgin plastic, and recycled plastic inputs, respectively, and $\mathbf{1}\{j=u\}$ is an indicator equal to one for *Using* firms and zero otherwise.

Recycled plastic, x_{fk}^j, is used only by firms in the *Using* sector ($j=u$), such as producers of plastic goods that combine labor with both virgin and recycled materials. These firms source recyclable plastic from *Supplying* firms in both domestic and foreign markets. Let

[47] We can write the change in imports of recycled plastic waste into China as $\Delta x_{f(d=CHN),g(o)}^u = x_{f(d=CHN),g(o)}^u(\tau_{o,d=CHN}) - x_{f(d=CHN),g(o)}^u(\tau_{o,d=CHN,t=0}) = -x_{f(d=CHN),g(o)}^u(\tau_{o,d=CHN,t=0})$ where $x_{f(d=CHN),g(o)}^u$ denotes the recycled plastic waste imports of firm f in the *Using* sector located in China from firm g in the *Supplying* sector in origin country $o \neq CHN$. Here $\tau_{o,d=CHN,t=0}$ denotes the pre-policy iceberg trade cost, and under the ONS policy $\tau_{o,d=CHN} \to \infty$ for all $o \neq CHN$.

$x^u_{f(k)g(o)}$ denote purchases by *Using* firm f in country k from *Supplying* firm g located in country o. Domestic and foreign recyclable inputs are combined into the firm's recycled-input composite:

$$x^u_{fk} = G_u\Big(\{x^u_{f(k)g(k)}\}_{g\in\mathcal{F}^s_k},\ \chi^u_{fk}\{x^u_{f(k)g(o)}\}_{o\neq k,\, g\in\mathcal{F}^s_o}\Big)\,,$$

where $\chi^u_{fk} \geq 0$ is a firm-specific parameter governing the efficiency with which imported recycled inputs are used. A higher χ^u_{fk} reflects greater efficiency in the use of imported recycled inputs and generates cross-firm heterogeneity in import intensity.

Waste Generation. Production generates waste as a by-product. A firm f in sector j produces $\theta^j_k\, y^j_{fk}/\phi^j_{fk}$ units of total waste, where $\theta^j_k > 0$ is a sector-specific technological parameter and y^j_{fk}/ϕ^j_{fk} is the effective units of output produced by the firm. Defining in terms of effective units ensures that ϕ^j_{fk} is Hicks-neutral (because otherwise it would denote firm differences in output versus by-product productivity). A fraction $\zeta^j_{fk} \in [0,1]$of by-products contain recyclable plastic material, while the remaining consists of other materials. In the *Supplying* sector, all firms generate plastic waste, so $\zeta^s_{fk} > 0$ for all $f \in \mathcal{F}^s_k$. In the *Using* and *Neither* sectors, firms do not generate plastic by-products, so ζ^j_{fk} is set to zero for all $f \in \mathcal{F}^j_k$ and $j \in \{u, n\}$. Hence, only *Supplying* firms produce plastic waste that can potentially be recycled and sold to *Using* firms.

Recycling and Disposal. All firms $f \in \mathcal{F}^s_k$, where $\mathcal{F}^s_k$ denotes the set of firms in the *Supplying* sector in country k, generate plastic waste as a by-product of production equal to $\zeta^s_{fk}\,\theta^s_k\, y^s_{fk}/\phi^s_{fk}$. The firm chooses how much of this waste to sell as recyclables, denoted r^s_{fk}, with the remainder disposed of domestically, b^s_{fk}.[48]

These choices are subject to the material balance constraint that total waste produced must equal the sum of recyclable and disposed waste:

[48]For simplicity, we assume that all non-plastic waste is disposed of rather than sold and recycled. The model could be extended to allow recycling of other types of waste, but we abstract from this to reduce notation. Non-plastic waste equals $\tilde{b}^j_{fk} = (1-\zeta^j_{fk})\theta^j_k y^j_{fk}/\phi^j_{fk}$ and is disposed of at cost $\tilde{D}_k = 0$ per unit.

$$r^s_{fk} + b^s_{fk} = \zeta^s_{fk}\, \theta^s_k\, y^s_{fk}/\phi^s_{fk}. \tag{C.3}$$

Recyclable plastic waste is sold to *Using* firms across destinations. For example, a medical syringe producer can sell its plastic waste to a traffic-cone producer that uses it as an input in production. Let $r^s_{f(k)g(d)}$ denote this shipment of recyclable plastic from *Supplying* firm f in country k to *Using* firm g in country d. Total recyclable units sold by firm f is given by

$$r^s_{fk} = \sum_d \sum_{g \in \mathcal{F}^u_d} r^s_{f(k)g(d)}.$$

Under iceberg trade costs $\tau_{kd} \geq 1$, the quantity of recyclable input received by firm g in country d is

$$x^u_{g(d)f(k)} = \frac{r^s_{f(k)g(d)}}{\tau_{kd}}.$$

Because recyclable waste re-enters production as an input, it differs from standard pollutants that generate purely negative externalities. When virgin and recyclable plastic are substitutable in production, an increase in recycling reduces demand for virgin resource extraction and thereby lowers pollution associated with extraction activities as the latter tend to be much more polluting than the recycling activities. Recycling thus generates an indirect environmental benefit through input substitution.

Only *Using* firms demand recyclable plastic as an input, while only *Supplying* firms generate plastic waste. Firms in the *Neither* sector neither use nor generate plastic waste and are affected by any changes in plastic waste prices only through general equilibrium channels.

Firm's profit maximization. Having outlined the production, waste generation, and recycling environment, we now characterize the profit maximization problem of a firm $f \in \mathcal{F}^j_k$ in sector j and country k.

We denote $\mathbf{p}_k$ as the collection of all prices relevant for firm choices in country k, including output prices across destinations, input prices, and wages. Firm $f \in \mathcal{F}^j_k$ sets price p^j_{fk} for its differentiated final product, and has unit cost $c^j_{fk}(\mathbf{p}_k)$ implied by the firm-specific sectoral production technology and input prices.

Further, the Supplying firm $f \in \mathcal{F}^s_k$ sets its price of recyclable plastic that it sells to destination d as ρ^s_{fkd}, and is subject to iceberg trade costs when traded from country k to destination d with factor τ_{kd}.[49] Let $A_k(\zeta^s_{fk})$ denote the per-unit labor requirements for converting by-products to recyclable waste that can be sold and let D_k denote the per-unit labor requirements for disposal of by-products that remain unsold. The wage of waste-management workers is w^r_k (that is typically lower than the mean wage within countries).

The firm's profit therefore can be written as

$$\begin{aligned}\pi^j_{fk}(\mathbf{p}_k) \;=\; & \sum_d \left(p^j_{fk} - c^j_{fk}\left(\mathbf{p}_k, \mathbb{1}\{j=u\}\chi^j_{fk}\right)\right) q^j_{fkd}/\phi^j_{fk} \\ & + \mathbb{1}\{j=s\}\left[\sum_d \sum_{g\in\mathcal{F}^u_d} \left(\frac{\rho^s_{fkd}}{\tau_{kd}} - w^r_k A_k(\zeta^s_{fk})\right) r^s_{f(k)g(d)} \;-\; w^r_k D_k\, b^s_{fk}\right]\end{aligned}$$

subject to the material balance constraint. $\mathbb{1}\{j=u\}$ and $\mathbb{1}\{j=s\}$ are indicator functions equal to one for *Using* and *Supplying* firms, respectively, and zero otherwise. In addition to the non-negativity constraints, the *Supplying* firms are subject to the material balance constraint in Equation (C.3).

Firms are assumed to be monopolistically competitive in each market, and their maximized profit function is

$$\pi^{j\,*}_{fk}(\mathbf{p}_k) \equiv \max_{\{q^j_{fkd}\}_d,\, l^j_{fk},\, v^j_{fk},\, x^j_{fk},\, r^j_{fk},\, b^j_{fk}} \pi^j_{fk}(\mathbf{p}_k),$$

where r^j_{fk} and b^j_{fk} are only relevant for *Supplying* firms ($f \in \mathcal{F}^s_k$), and x^j_{fk} only for *Using* firms ($f \in \mathcal{F}^u_k$). This formulation is useful for empirical work because, under standard regularity conditions, Hotelling's lemma implies that equilibrium responses of firm activities can be expressed as functions of common price changes and firm-specific exposure shares, generating heterogeneous responses with common elasticities within sector-country. In particular, we assume the following structure on production:

[49]For simplicity, we introduce iceberg trade costs τ_{kd} only for recycled plastic waste. We have free trade for virgin plastic and final output, but positive trade costs could be incorporated without affecting the qualitative results.

Assumption 2 *The firm profit function ${\pi^j_{fk}}^*$ exists. Inputs are normal and Hotelling's lemma applies, with constant Hotelling elasticities across firms within each product-country market, $\varepsilon^j_{ii',fk} = \varepsilon^j_{ii',k}$ for all firms f in (j,k) and $p_i, p_{i'} \in \mathbf{p}$.*

Hotelling elasticities are defined in the standard way. For any activity $i \in \{l, v, x, q, r, b\}$ and any price component $p_{i'} \in \mathbf{p}$, the Hotelling elasticity is

$$\varepsilon^j_{ii',fk} \equiv \pi^* \pi^*_{ii'} / \pi^*_i \pi^*_{i'}$$

for $\pi^*_{ii'} \equiv \partial^2 {\pi^j_{fk}}^* / \partial p_i \partial p_{i'}$ and $\pi^*_{i'} \equiv \partial {\pi^j_{fk}}^* / \partial p_{i'}$, and the share of activity i' in profits is

$$s^j_{i'fk} \equiv |p_{i'} \pi^*_{i'}| / \pi^*.$$

We assume constant Hotelling elasticities and examine the case when the law of conservation of material constraints bind weakly (because firms typically do not sell all their waste).

C.4. **Market Clearing.** The model is closed with standard market-clearing conditions, that link firm-level outcomes to national income and ensure that general equilibrium adjustments are internally consistent.

Assumption 3: Market Clearing

(a) *National Income Identity. Income in country k equals revenues earned by domestic firms from sales of their final differentiated goods ($\sum_j \sum_f \sum_d R^j_{fkd}$)[50], plus net exports of tradable intermediate goods, namely virgin resources NX^v_k and recyclable waste NX^r_k.[51] Since all income is spent on final consumption, total expenditure on final goods is given by:*

$$R_k = I_k = \sum_j \sum_{f \in \mathcal{F}^j_k} \sum_d R^j_{fkd} + NX^v_k + NX^r_k, \tag{C.4}$$

[50]The revenue of firm $f \in \mathcal{F}^j_k$ from selling to destination d is given by $R^j_{fkd} = p^j_{fkd} q^j_{fkd}$.

[51]Net exports of recycled waste is defined as $NX^r_k = \sum_f \sum_{d \neq k} \sum_g \rho^s_{fkd} \quad r^s_{f(k)g(d)} / \tau_{kd} - \sum_f \sum_{o \neq k} \sum_h \rho^s_{hok} r^s_{h(o)f(k)}$. Net exports of virgin plastic is defined analogously.

and $R_k \equiv \sum_o \sum_j \sum_{f \in \mathcal{F}_o^j} R_{fok}^j$ *denotes total expenditure on final goods in country* k*, including both domestic and imported varieties.*[52]

(b) Labor Market Clearing Condition:

$$L_k = \sum_j \sum_{f \in \mathcal{F}_k^j} l_{fk}^j \, ; \quad L_k^r = \sum_{f \in \mathcal{F}_k^s} A_k(\zeta_{fk}^s) r_{fk}^s + D_k b_{fk}^s. \tag{C.5}$$

where l_{fk}^j *denotes labor demand in production and labor demand in waste-management activities* $A_k(\zeta_{fk}^s) r_{fk}^s$ *for generating plastic recyclables that can be sold for re-use and* $D_k b_{fk}^s$ *for disposing of unsold by-products. The equilibrium wage adjusts to equate total labor demand and labor supply in each labour market.*

(c) Virgin resource pricing. The world price of the virgin resource, p_v*, is taken as given, and remains unaffected by the China ban over the medium run considered here.*

Virgin plastic is derived primarily from oil or natural gas, and therefore the last Assumption above implies that we take the oil price as given for Turkiye in our welfare analysis (Turkiye does not have its own supplies of oil or gas, and is a net importer).

C.5. **Sufficient Statistics.** In this section, we show that various empirically-relevant firm outcomes can be summarized by sufficient statistics based on firms' exposure to imported recyclable plastic inputs and waste intensity.

Let

$$\widetilde{c}_{fk}^j \equiv c_{fk}^j(\mathbf{p}_k) + \mathbf{1}\{j = s\} \, \Omega_{fk}^s$$

denote firm f's effective unit cost in country k, sector j, where $c_{fk}^j(\mathbf{p}_k)$ is the production unit cost and Ω_{fk}^s captures net waste-management cost per unit of output for *Supplying* firms. Optimal pricing implies

$$p_{fkd}^j = \frac{\sigma_j}{\sigma_j - 1} \widetilde{c}_{fk}^j.$$

[52]Since all income is spent on final consumption, this identity implies that the sum of net exports across all tradable categories—final goods, virgin plastic, and recycled waste—must sum to zero, yielding the balanced trade condition, $0 = NX_k^{\text{final}} + NX_k^v + NX_k^r$. Net exports are defined as exports minus imports in each tradable category. For instance, net exports of final goods are $NX_k^{\text{final}} = \sum_{d \neq k} \sum_j \sum_{f \in \mathcal{F}_k^j} R_{fkd}^j - \sum_{o \neq k} \sum_j \sum_{f \in \mathcal{F}_o^j} R_{fok}^j$,.

Hence, firm-level responses of prices, quantities, and revenues are

$$
\begin{aligned}
d\ln p^j_{fkd} &= d\ln \widehat{c}^j_{fk}, \\
d\ln q^j_{fkd} &= -\sigma_j d\ln p^j_{fkd} + \left(\sigma_j d\ln P^j_d + d\ln Q^j_d\right), \\
d\ln R^j_{fkd} &= (1-\sigma_j) d\ln p^j_{fkd} + \left(\sigma_j d\ln P^j_d + d\ln Q^j_d\right).
\end{aligned}
$$

Summing across all destinations,

$$
d\ln Q^j_{fk} = -\sigma_j d\ln \widehat{c}^j_{fk} + \sum_d \left(q^j_{fkd}/Q^j_{fk}\right)\left(\sigma_j d\ln P^j_d + d\ln Q^j_d\right).
$$

Total revenues, TR^j_{fk}, including waste revenues WR^s_{fk} for supplying firms, change by

$$
d\ln TR^j_{fk} = \left(R^j_{fk}/TR^j_{fk}\right) d\ln R^j_{fk} + \mathbf{1}\{j=s\}\left(WR^s_{fk}/TR^s_{fk}\right) d\ln WR^s_{fk}.
$$

Let $m^j_{fk}, \tilde{m}^j_{fk} \in \{l^j_{fk}, v^j_{fk}, x^j_{D,fk}, x^j_{M,fk}\}$ index labor, virgin plastic, domestic recycled plastic, and imported recycled plastic inputs used by firm f in country k and sector j, with associated prices $p^m_k, p^{\tilde{m}}_k \in \{w_k, p^v_k, \rho^D_k, \rho^M_k\}$.[53] Domestic and imported recycled plastic inputs are relevant only for firms in the using sector $j=u$, and $x^j_{D,fk} = x^j_{M,fk} = 0$ for $j \neq u$. From Shephard's lemma, input demands satisfy

$$
\begin{aligned}
d\ln m^j_{fk} &= d\ln\left(Q^j_{fk}\frac{\partial c^j_{fk}}{\partial p^m_k}\right) \\
&= d\ln Q^j_{fk} + \sum_{\tilde{m}\in\{l,v,x_D,x_M\}} \left(p^{\tilde{m}}_k\, d\ln p^{\tilde{m}}_k\right)\frac{\partial^2 c^j_{fk}/\partial p^m_k \partial p^{\tilde{m}}_k}{\partial c^j_{fk}/\partial p^m_k} \\
&= d\ln Q^j_{fk} - \sum_{\tilde{m}\in\{l,v,x_D,x_M\}} \epsilon^j_{m\tilde{m}}\, s^j_{\tilde{m}fk}\, d\ln p^{\tilde{m}}_k.
\end{aligned}
$$

Here

$$
s^j_{\tilde{m}fk} \equiv \frac{p^{\tilde{m}}_k\, \tilde{m}^j_{fk}}{c^j_{fk}}
$$

[53]The recycled plastic inputs used for firm f could be written as $x^u_{fk} = G_u\left(x^u_{D,fk},\ \chi^u_{fk}x^u_{M,fk}\right)$, where $x^u_{D,fk}$, $x^u_{M,fk}$ are the domestic and foreign recycled inputs used by firm f in the *Using* sector in country k.

denotes the cost share of input $\tilde{m}$, and $\epsilon^j_{m\tilde{m}}$ is the elasticity of input demand m with respect to the price of input $\tilde{m}$ in sector j, defined by

$$-\epsilon^j_{m\tilde{m}} s^j_{\tilde{m}fk} = p^{\tilde{m}}_k \frac{\partial^2 c^j_{fk}/\partial p^m_k \partial p^{\tilde{m}}_k}{\partial c^j_{fk}/\partial p^m_k}.$$

Within each sector j and country k, firms share a common production function $F_j(\cdot)$ and common substitution elasticities. Firm heterogeneity enters through Hicks-neutral productivity ϕ^j_{fk}, and, for using firms, through the input-augmenting parameter χ^u_{fk} that scales the effective use of imported recycled plastic inputs. In particular, production takes the form

$$y^j_{fk} = \phi^j_{fk} F_j\left(l^j_{fk},\ v^j_{fk},\ x^j_{D,fk},\ \chi^j_{fk} x^j_{M,fk}\right),$$

where $\chi^j_{fk} = 0$ for $j \in \{s, n\}$ and $\chi^u_{fk} > 0$ for using firms. Hicks-neutral productivity ϕ^j_{fk} scales unit costs but does not affect conditional input shares, while χ^u_{fk} lowers the effective price of imported recycled plastic inputs and therefore generates cross-firm variation in imported recycled-input expenditure shares.

Firms' unit costs change by

$$\begin{aligned} d\ln c^j_{fk} = &\sum_{m\in\{l,v,x_D\}} s^j_{m0k}\left(1 - s^j_{x_M fk}\right) d\ln p^m_k \\ &+ \mathbf{1}\{j = u\}\, s^u_{x_M fk}\, d\ln\left(\frac{\rho^M_k}{\chi^u_{fk}}\right), \end{aligned}$$

where

$$s^j_{m0k} \equiv \frac{s^j_{mfk}}{1 - s^j_{x_M fk}}, \qquad m \in \{l, v, x_D\},$$

denotes the normalized expenditure share of input m within total spending on inputs other than imported recycled plastic. Thus,

$$s^j_{mfk} = s^j_{m0k}\left(1 - s^j_{x_M fk}\right).$$

We further assume that the composition of all inputs other than imported recycled plastic is common across firms within a sector-country. This implies that differences in cost shares across firms arise solely through variation in the expenditure share on imported recycled plastic inputs. By construction, s^j_{m0k} does not vary across firms within sector j, country k. Moreover, the imported recycled-input term appears only for firms in the using sector $j = u$, since $x^j_{M,fk} = 0$ for $j \neq u$.

Differences in unit-cost responses across firms f and f' are therefore

$$\begin{aligned}\Delta_{ff'} d\ln c^j_{fk} = &\sum_{m\in\{l,v,x_D\}} s^j_{m0k}\, d\ln p^m_k \cdot \Delta_{ff'}\left(1 - s^j_{x_M fk}\right) \\ &+ \mathbf{1}\{j = u\}\, d\ln\left(\frac{\rho^M_k}{\chi^u_{fk}}\right) \cdot \Delta_{ff'} s^u_{x_M fk}.\end{aligned}$$

Substituting into input-demand responses, employment changes by

$$\begin{aligned}d\ln l^j_{fk} &= d\ln Q^j_{fk} - \sum_{n\in\{l,v,x_D,x_M\}} \epsilon^j_{ln}\, s^j_{nfk}\, d\ln p^n_k \\ &= -\sigma_j d\ln \widehat{c}^j_{fk} + \sum_d \left(\frac{Q^j_{fkd}}{Q^j_{fk}}\right)\left(\sigma_j d\ln P^j_d + d\ln Q^j_d\right) - \sum_{n\in\{l,v,x_D,x_M\}} \epsilon^j_{ln}\, s^j_{nfk}\, d\ln p^n_k.\end{aligned}$$

For *Using* and *Neither* firms, $\widehat{c}^j_{fk} = c^j_{fk}$, so the expression above can be written directly in terms of production cost shares. For *Supplying* firms, effective unit cost includes an additional waste-management component, Ω^s_{fk}, so the corresponding response depends on $d\ln \widehat{c}^s_{fk}$ rather than $d\ln c^s_{fk}$.

For sectors with $\Omega^s_{fk} = 0$, using the unit-cost expression yields

$$\begin{aligned}d\ln l^j_{fk} = &-\sigma_j \left[\sum_{m\in\{l,v,x_D\}} s^j_{m0k}\left(1 - s^j_{x_M fk}\right) d\ln p^m_k + \mathbf{1}\{j = u\}\, s^u_{x_M fk}\, d\ln \rho^M_k\right] \\ &+ \sum_d \left(\frac{Q^j_{fkd}}{Q^j_{fk}}\right)\left(\sigma_j d\ln P^j_d + d\ln Q^j_d\right) - \sum_{n\in\{l,v,x_D,x_M\}} \epsilon^j_{ln}\, s^j_{nfk}\, d\ln p^n_k.\end{aligned}$$

An analogous expression holds for virgin-plastic demand v^j_{fk} and, for firms in the using sector $j = u$, for domestic and imported recycled-plastic demand, $x^u_{D,fk}$ and $x^u_{M,fk}$, obtained by replacing ϵ^j_{ln} with ϵ^j_{vn}, $\epsilon^u_{x_D n}$, and $\epsilon^u_{x_M n}$, respectively.

C.5.1. *Using Sector.* We now show that, within the *Using* sector, cross-firm heterogeneity in responses to input price changes is summarized by the firm-specific expenditure share on imported recycled plastic inputs.

For using firms, unit costs satisfy

$$\begin{aligned} d\ln c^u_{fk} &= \sum_{m\in\{l,v,x_D\}} s^u_{m0k}(1-s^u_{x_M fk})\, d\ln p^m_k + s^u_{x_M fk}\, d\ln \rho^M_k \\ &= \underbrace{\sum_{m\in\{l,v,x_D\}} s^u_{m0k} d\ln p^m_k}_{\equiv A^u_k} + s^u_{x_M fk} \underbrace{\left(d\ln \rho^M_k - \sum_{m\in\{l,v,x_D\}} s^u_{m0k} d\ln p^m_k \right)}_{\equiv B^u_k}. \end{aligned}$$

Both A^u_k and B^u_k are common across firms within country k and sector u. Hence,

$$\Delta_{ff'} d\ln c^u_{fk} = B^u_k\, \Delta_{ff'} s^u_{x_M fk}.$$

Thus, conditional on common input price changes, cross-firm differences in unit-cost responses are fully summarized by $s^u_{x_M fk}$, the share of imported recycled plastic inputs in total costs.

The same logic extends to input demands. Using

$$d\ln m^u_{fk} = d\ln Q^u_{fk} - \sum_{n\in\{l,v,x_D,x_M\}} \epsilon^u_{mn} s^u_{nfk} d\ln p^n_k,$$

together with

$$s^u_{lfk} = s^u_{l0k}(1-s^u_{x_M fk}), \qquad s^u_{vfk} = s^u_{v0k}(1-s^u_{x_M fk}), \qquad s^u_{x_D fk} = s^u_{x_D 0k}(1-s^u_{x_M fk}),$$

implies that the cost-share vector varies across firms only through $s^u_{x_M fk}$. Therefore, conditional on common sectoral elasticities, cross-firm differences in input-demand responses are also summarized by $s^u_{x_M fk}$.

More generally, any firm-level outcome whose response depends on firm heterogeneity only through unit costs and input cost shares can be written as a common sector-country component plus a firm-specific term proportional to $s^u_{x_M fk}$. Under the maintained assumptions of a common production technology, common substitution elasticities, and a common composition of all inputs other than imported recycled plastic, $s^u_{x_M fk}$ is therefore a sufficient statistic for heterogeneous responses of using firms to changes in input prices.

Sufficient Statistic for the Using Sector. In the data, we observe recycled plastic inputs separately by origin. Let x^u_{ofk} denote recycled inputs sourced from origin o, with associated cost shares s^u_{ofk}. Then the imported recycled-input share in total costs is given by

$$S^u_{fk} \equiv \sum_{o \neq k} s^u_{ofk}.$$

This corresponds directly to the model object $s^u_{x_M fk}$. When the shock operates through changes in the price of imported recycled plastic inputs, cross-firm heterogeneity in responses is therefore summarized by S^u_{fk}, the share of imported recycled plastic inputs in total costs.

C.5.2. *Supplying Firms.* An analogous argument applies to the *Supplying* sector, with one difference: the relevant source of heterogeneity is not efficiency in processing imported inputs but the share of plastic waste in total waste generated, ζ^s_{fk}. We show that this share summarises all cross-firm variation in responses to the ONS policy within the sector.

Firm problem. A supplying firm produces final output and generates waste as a by-product of production. Total plastic waste (W^s_{fk}) is proportional to output,

$$W^s_{fk} = \zeta^s_{fk} \theta^s_k \frac{y^s_{fk}}{\phi^s_{fk}},$$

where ζ^s_{fk} denotes the fraction of waste that is plastic. This waste must be either recycled or disposed of. Let r^s_{fkd} denote recyclable plastic waste sold to destination d, and the residual unsold waste is disposed of.

The firm solves

$$\begin{aligned} \max_{q,r} \ \pi^s_{fk} = & \sum_d p^s_{fkd} q^s_{fkd} - \frac{\tilde{c}^s_k}{\phi^s_{fk}} y^s_{fk} \\ & + \sum_d \frac{\rho^s_{fkd}}{\tau_{kd}} r^s_{fkd} - w^r_k \sum_d A_k(\zeta^s_{fk}) r^s_{fkd} \end{aligned}$$

$$
- w_k^r D_k \left(\zeta_{fk}^s \theta_k^s \frac{y_{fk}^s}{\phi_{fk}^s} - \sum_d r_{fkd}^s \right),
$$

subject to

$$
\sum_d r_{fkd}^s \leq \zeta_{fk}^s \theta_k^s \frac{y_{fk}^s}{\phi_{fk}^s}.
$$

The final term captures disposal costs for non-recycled plastic waste. Converting by-products to recyclable plastic waste carries a per-unit labor cost $w_k^r A_k(\zeta_{fk}^s)$, but avoids the disposal cost $w_k^r D_k$, so that the net marginal cost of supplying recyclable plastic waste is $w_k^r \Big(A_k(\zeta_{fk}^s) - D_k \Big)$.

Pricing. Under CES demand with elasticity $\sigma_s > 1$ in the final-good market and $\varepsilon_r > 1$ in the recycled-input market, optimal prices are constant markups over marginal cost:

$$
p_{fkd}^s = \frac{\sigma_s}{\sigma_s - 1} \cdot \frac{\tilde{c}_k^s + w_k^r D_k \zeta_{fk}^s \theta_k^s}{\phi_{fk}^s},
$$

$$
\rho_{fkd}^s = \tau_{kd} \frac{\varepsilon_r}{\varepsilon_r - 1} w_k^r \Big(A_k(\zeta_{fk}^s) - D_k \Big).
$$

The price of the final good reflects both production costs and disposal costs associated with waste generation, while the price of recyclable plastic waste reflects the net marginal cost of drawing reyclables from by-products.

Price responses. Totally differentiating the pricing equations yields

$$
d \ln p_{fkd}^s = d \ln \tilde{c}_k^s + \Upsilon_k^s(\zeta_{fk}^s) \Big(d \ln w_k^r - d \ln \tilde{c}_k^s \Big),
$$

$$
d \ln \rho_{fkd}^s = d \ln \tau_{kd} + d \ln w_k^r + d \ln \Big(A_k(\zeta_{fk}^s) - D_k \Big),
$$

where

$$
\Upsilon_k^s(\zeta_{fk}^s) \equiv \frac{w_k^r D_k \zeta_{fk}^s \theta_k^s}{\tilde{c}_k^s + w_k^r D_k \zeta_{fk}^s \theta_k^s}.
$$

Taking differences across firms f and f' within the same country and period removes aggregate shocks:

$$
\Delta_{ff'} \, d \ln p_{fkd}^s = \Delta_{ff'} \, \Upsilon_k^s(\zeta_{fk}^s) \left(d \ln w_k^r - d \ln \tilde{c}_k^s \right),
$$

$$
\Delta_{ff'} \, d \ln \rho_{fkd}^s = \Delta_{ff'} \, d \ln \Big(A_k(\zeta_{fk}^s) - D_k \Big).
$$

These expressions show that cross-firm differences in price responses are governed by firm-specific exposure to disposal costs, summarized by ζ^s_{fk}.

Factor demand. Labor demand in final-good production satisfies

$$l^s_{fk} = Q^s_{fk} \cdot \frac{\partial(\tilde{c}^s_k/\phi^s_{fk})}{\partial w_k},$$

so that

$$d\ln l^s_{fk} = d\ln Q^s_{fk} + d\ln\left(\frac{\partial \tilde{c}^s_k}{\partial w_R}\right).$$

Under CES demand, output responds to own-price changes according to

$$d\ln Q^s_{fk} = -\sigma_s d\ln p^s_{fk} + \sum_d \left(\frac{R^s_{fkd}}{R^s_{fk}}\right)(\sigma_s d\ln P^s_d + d\ln Q^s_d).$$

Substituting for prices yields

$$\begin{aligned} d\ln l^s_{fk} = & -\sigma_s d\ln \tilde{c}^s_k + \sum_d \left(\frac{R^s_{fkd}}{R^s_{fk}}\right)(\sigma_s d\ln P^s_d + d\ln Q^s_d) \\ & - \sigma_s \Upsilon^s_k(\zeta^s_{fk})\Big(d\ln w^r_k - d\ln \tilde{c}^s_k\Big) + d\ln\left(\frac{\partial \tilde{c}^s_k}{\partial w_r}\right). \end{aligned}$$

The substitution term is common across firms and drops out when taking differences. Hence,

$$\Delta_{ff'}\, d\ln l^s_{fk} = -\sigma_s\, \Delta_{ff'}\, \Upsilon^s_k(\zeta^s_{fk})\,\Big(d\ln w^r_k - d\ln \tilde{c}^s_k\Big).$$

All suppliers of recyclable plastic waste selling into destination k face the same market-level demand shifter, since they compete in the same destination-k recyclable-input market and therefore share the same recyclable-input price index P^r_k. By Shephard's lemma, demand by using firm $f \in \mathcal{F}^u_k$ for recyclable plastic supplied by firm $g \in \mathcal{F}^s_o$ is

$$x^u_{f(k)g(o)} = Q^u_{fk} \cdot \frac{\partial \tilde{c}^u_{fk}}{\partial P^r_k} \cdot \frac{\partial P^r_k}{\partial \rho^s_{gok}}.$$

Aggregating across using firms in destination k, total demand faced by supplier g from origin o is

$$X^s_{gok} \equiv \sum_{f\in\mathcal{F}^u_k} x^u_{f(k)g(o)} = \sum_{f\in\mathcal{F}^u_k} Q^u_{fk} \cdot \frac{\partial \tilde{c}^u_{fk}}{\partial P^r_k} \cdot \frac{\partial P^r_k}{\partial \rho^s_{gok}}.$$

Define

$$\mathcal{M}_k^r \equiv \sum_{f \in \mathcal{F}_k^u} Q_{fk}^u \cdot \frac{\partial \tilde{c}_{fk}^u}{\partial P_k^r}.$$

Then derived demand can be written as

$$X_{gok}^s = \mathcal{M}_k^r \cdot \frac{\partial P_k^r}{\partial \rho_{gok}^s} \equiv \mathcal{M}_k^r \, \Gamma_k^r(P_k^r, \rho_{gok}^s),$$

where $\mathcal{M}_k^r$ is a destination-specific demand shifter common to all suppliers serving market k, and $\Gamma_k^r(\cdot)$ captures the sensitivity of demand to the supplier's own price.

Consequently, final-good and recycled-input revenues at the destination level satisfy

$$\begin{aligned} d\ln R_{fkd}^s &= (1-\sigma_s) d\ln p_{fkd}^s + \left(\sigma_s d\ln P_d^s + d\ln Q_d^s\right), \\ d\ln R_{gok}^{r,s} &= d\ln \rho_{gok}^s + d\ln X_{gok}^s \\ &= (1-\varepsilon_r) d\ln \rho_{gok}^s + \varepsilon_r d\ln P_k^r + d\ln \mathcal{M}_k^r. \end{aligned}$$

Substituting for optimal prices yields

$$d\ln R_{gok}^{r,s} = (1-\varepsilon_r)\left[d\ln \tau_{ok} + d\ln w_o^r + d\ln\Big(A_o(\zeta_{go}^s) - D_o\Big)\right] + \varepsilon_r d\ln P_k^r + d\ln \mathcal{M}_k^r.$$

Aggregating across destinations, final-good revenue changes by

$$\begin{aligned} d\ln R_{fk}^s = (1-\sigma_s)\left[d\ln \tilde{c}_k^s + \Upsilon_k^s(\zeta_{fk}^s)\Big(d\ln w_k^r - d\ln \tilde{c}_k^s\Big)\right] \\ + \sum_d \left(\frac{R_{fkd}^s}{R_{fk}^s}\right)\left(\sigma_s d\ln P_d^s + d\ln Q_d^s\right). \end{aligned}$$

Similarly, total recyclable-input revenue for supplier g in origin o is

$$\begin{aligned} d\ln R_{go}^{r,s} = (1-\varepsilon_r)\left[d\ln w_o^r + d\ln\Big(A_o(\zeta_{go}^s) - D_o\Big)\right] \\ + \sum_k \left(\frac{R_{gok}^{r,s}}{R_{go}^{r,s}}\right)\left[(1-\varepsilon_r) d\ln \tau_{ok} + \varepsilon_r d\ln P_k^r + d\ln \mathcal{M}_k^r\right]. \end{aligned}$$

Revenue responses also vary systematically with firm-specific plastic waste intensity. For final-good revenues, taking differences across firms within country k yields

$$\Delta_{ff'}\, d\ln R^s_{fk} = (1-\sigma_s)\, \Delta_{ff'}\, \Upsilon^s_k(\zeta^s_{fk}) \Big(d\ln w^r_k - d\ln \tilde{c}^s_k\Big),$$

where

$$\Upsilon^s_k(\zeta^s_{fk}) \equiv \frac{w^r_k D_k \zeta^s_{fk}\theta^s_k}{\tilde{c}^s_k + w^r_k D_k \zeta^s_{fk}\theta^s_k}.$$

For recyclable-input revenues, recall that

$$R^{r,s}_{gok} = \rho^s_{gok} X^s_{gok}.$$

Under CES demand with elasticity $\varepsilon_r > 1$,

$$d\ln R^{r,s}_{gok} = (1-\varepsilon_r) d\ln \rho^s_{gok} + \varepsilon_r d\ln P^r_k + d\ln \mathcal{M}^r_k.$$

Using the pricing equation

$$d\ln \rho^s_{gok} = d\ln \tau_{ok} + d\ln w^r_o + d\ln\Big(A_o(\zeta^s_{go}) - D_o\Big),$$

and taking differences across firms g and g' within origin o, common destination-level shifters drop out, so that

$$\Delta_{gg'}\, d\ln R^{r,s}_{go} = (1-\varepsilon_r)\, \Delta_{gg'}\, d\ln\Big(A_o(\zeta^s_{go}) - D_o\Big) = 0.$$

Quantity responses take a similar form. For final-good output,

$$\begin{aligned} d\ln Q^s_{fk} = &-\sigma_s \Big[d\ln \tilde{c}^s_k + \Upsilon^s_k(\zeta^s_{fk})\Big(d\ln w^r_k - d\ln \tilde{c}^s_k\Big)\Big] \\ &+ \sum_d \left(\frac{R^s_{fkd}}{R^s_{fk}}\right)(\sigma_s d\ln P^s_d + d\ln Q^s_d)\,, \end{aligned}$$

so that

$$\Delta_{ff'}\, d\ln Q^s_{fk} = -\sigma_s\, \Delta_{ff'}\, \Upsilon^s_k(\zeta^s_{fk}) \Big(d\ln w^r_k - d\ln \tilde{c}^s_k\Big).$$

For recycled-input quantities, CES demand implies

$$d \ln X^s_{gok} = -\varepsilon_r \, d \ln \rho^s_{gok} + \varepsilon_r \, d \ln P^r_k + d \ln \mathcal{M}^r_k.$$

Substituting for $d \ln \rho^s_{gok}$ and taking differences across firms within origin o gives

$$\Delta_{gg'} \, d \ln X^s_{go} = -\varepsilon_r \, \Delta_{gg'} \, d \ln\Big(A_o(\zeta^s_{go}) - D_o\Big).$$

Total firm revenue equals the sum of final-good and recyclable-input revenues:

$$TR^s_{fk} = R^s_{fk} + R^{r,s}_{fk}.$$

A first-order log approximation implies

$$d \ln TR^s_{fk} \approx \left(\frac{R^s_{fk}}{TR^s_{fk}}\right) d \ln R^s_{fk} + \left(\frac{R^{r,s}_{fk}}{TR^s_{fk}}\right) d \ln R^{r,s}_{fk}.$$

Hence, cross-firm differences in total revenue responses satisfy

$$\Delta_{ff'} \, d \ln TR^s_{fk} \approx \left(\frac{R^s_{fk}}{TR^s_{fk}}\right) (1 - \sigma_s) \, \Delta_{ff'} \, \Upsilon^s_k(\zeta^s_{fk}) \left(d \ln w^r_k - d \ln \tilde{c}^s_k\right).$$

Sufficient Statistic for the Supplying Sector. Differences across supplying firms in revenue and quantity responses are governed by the same underlying source of heterogeneity: the firm-specific plastic-waste share ζ^s_{fk}. Through $\Upsilon^s_k(\zeta^s_{fk})$, a higher plastic-waste share raises the sensitivity of final-good revenues and quantities to waste-management costs.

Appendix D. Economic Impacts of the ONS policy

D.1. **From Relative to Absolute Effects.** Table 2 shows the relative effects of the ONS-policy on varying outcomes. In these results, industry–time fixed effects subsume the average response of *Neither* firms, which we denote β^Y_n. The estimated coefficients β^Y_j therefore capture effects relative to this baseline. The implied absolute percentage change in outcome Y^j_{fk} for sector j is

$$\text{(D.1)} \qquad \frac{\Delta Y^j_{fk}}{Y^j_{fk}} = \exp\Big(\beta^Y_j S^j_{fk} + \beta^Y_n\Big) - 1.$$

for dependent variables that are expressed in logarithmic form. To recover the general equilibrium effects, we must determine the baseline impact β^Y_n. The model structure provides this through market-clearing conditions. In particular, market clearing for production labor in Assumption 3(b) requires that total employment changes across all firms sum to zero:

$$\sum_j \sum_f \Delta l^j_{fk} = 0.$$

Substituting the model-implied expression for employment changes,

$$\frac{\Delta l^j_{fk}}{l^j_{fk}} = \exp\Big(\beta^l_j S^j_{fk} + \beta^l_n\Big) - 1,$$

into Equation (D.1) allows us to solve for β^l_n.

The relative changes (β^l_j) are reported in Table 2, and the initial employment in each sector is summarized in Table D.1. These pin down the absolute employment response of non-exposed firms and, empirically, we find that employment change among non-exposed firms is tiny ($\beta^l_n = -0.00076$). Even though firms using plastic waste increase employment, their labor share in the economy is less than one percent. Consequently, their general equilibrium effect on employment turn out to be also negligible, and the relative changes in employment estimated in the difference-in-differences specification can be interpreted as the absolute change due tot the policy.

To extend this logic to changes in outcomes other than employment reported in Tables 2 and D.4, we impose regularity conditions that are standard in production theory below. Assumption 4 formalizes that labor has positive marginal product and that material inputs are used productively in equilibrium. Because *Neither* firms are not directly exposed to recyclable plastic waste prices (as their $\chi^n_{fk} = 0$, $\zeta^n_{fk} = 0$), their outcomes can change due to the ONS policy only through general equilibrium channels such as wages or other input prices. Empirically, however, we observe economically negligible employment changes among *Neither* firms, implying that these indirect general equilibrium price effects are likewise minimal in practice. Under Assumption 4, the economic environment facing *Neither* firms is effectively unchanged, so their output, revenues, input use, and waste generation also remain approximately constant. This is not a mechanical implication of monotonicity alone, but rather reflects the combination of no direct policy exposure and negligible equilibrium price movements. It follows that the relative difference-in-differences estimates can be interpreted as close approximations to absolute percentage changes for exposed firms.

Assumption 4: Monotonicity in Production

(i) Output and revenues are strictly increasing in labor inputs.

(ii) Virgin inputs, other material inputs, and by-products are strictly increasing in output.

Table D.1. Initial Sector-Level Aggregates (Pre-ONS)

Sector	**Sales**	**Employment**	**Net Imports**	**Virgin Imports**
n	127.3 billion	1,130,300	23.5 billion	0.78 billion
s	172.5 billion	852,057	57.6 billion	1.36 billion
u	1.70 billion	13,136	0.40 billion	0.05 billion

Note: This table reports aggregate annual values by sector for 2016, the pre-policy base year. Sector n denotes the *Neither* sector, s the *Supplying* sector, and u the *Using* sector. Sales and net imports of intermediates are expressed in 2016 USD. Virgin imports are defined as imports of crude petroleum, coal, palm oil, natural gas, and plastic in its primary form, also expressed in 2016 USD. Employment is measured in number of workers. The sample includes all manufacturing firms with more than 20 employees, as well as a representative sample of smaller firms.

Table D.2. National Income Accounting for Turkiye (Pre-ONS)

Component	**Billion USD**
Total firm revenues	301.5
Neither sector	127.3
Supplying sector	172.5
Using sector	1.7
Net exports of virgin plastic	−2.19
Net exports of recycled plastic waste	−0.01
Implied aggregate expenditure R_k	**299.3**

Note: Derived from the national income identity in Equation (C.4) under the balanced trade condition in Assumption 1. Annual pre-ONS averages, 2016 USD.

D.2. **Demand Elasticity.** Profit maximization implies that the log changes in output prices can also be re-written as $\ln P^j_{fkk} = \alpha^j_k + \beta^P S^j_{fk}$, as shown earlier in the derivations of firm's unit cost changes. From the CES structure, revenues of domestic firms are $\ln R^j_{fkk} = \ln A^j_k + (1-\sigma_j)\ln P^j_{fkk}$ where $A^j_k \equiv \left(I_k \left(Q^j_k/Q_k\right)^{\frac{\sigma-1}{\sigma}} \eta_j Q^{j\,\frac{1-\sigma_j}{\sigma_j}}_k\right)^{\sigma_j}$ is a sectoral shifter. Optimal final price of domestic firms is $P^j_{fkk} = \frac{\sigma_j}{\sigma_j - 1} c^j_{fk}$ for $j = n, u$ varieties. We observe domestic firm-product sales $R_{f\iota t}$ from firm-level data and prices $P_{f\iota t}$ from Prodcom data.

The results of Equations (4.7) and (4.8) are reported in Table D.3. Column (1) shows that exposure to banned plastic waste imports leads to a significant decline in output prices for *Using* firms, with no statistically significant effect for *Supplying* firms, consistent with the model in which only using firms benefit from access to imported waste inputs. Column (2) presents estimates of Equation (4.8), focusing on *Using* firms and confirms a strong and precisely estimated first-stage effect of exposure on prices. Column (3) reports IV estimates of Equation (4.7), instrumenting prices with exposure interacted with the post period. The second stage gives $\beta^D = -(\sigma_u - 1) \approx -8.6$. Defining β^P_n as the price impact of n firms, $\Delta P_{f\iota}/P_{f\iota} = \exp\left(\beta^P S^u_f + \beta^P_n\right) - 1$ where β^P_n is the sectoral price impact of n firms, that is subsumed in the industry-time fixed effects. The CES structure also implies that $\beta^P_n = 0$ when $\Delta l^n_{fk} \approx 0$ and $\Delta P_{f\iota}/P_{f\iota} = \exp\left(\beta^P S^u_f\right) - 1$.

D.3. **Consumption Gains from the ONS Policy.** Under CES preferences, aggregate final consumption satisfies

$$Q_k \equiv \left(\sum_j \eta^j (Q^j_k)^{\frac{\sigma-1}{\sigma}}\right)^{\frac{\sigma}{\sigma-1}},$$

which implies that changes in aggregate consumption can be written as expenditure-share-weighted changes in sectoral consumption:

$$\frac{\Delta Q_k}{Q_k} = \sum_j \left(\frac{E^j_k}{E_k}\right)\left(\frac{\Delta Q^j_k}{Q^j_k}\right).$$

As established above, ONS generates no meaningful consumption response in the *Supplying* or *Neither* sectors: revenues, prices, and general equilibrium spillovers are approximately unchanged. Column (5) of Table 2 shows no relative revenue response for *Supplying* firms,

Table D.3. Elasticity Estimation for Using Firms

	$\ln(\text{P}_{f\iota t})$	FS $\ln(\text{P}_{f\iota t})$	IV $\ln(\text{R}_{f\iota t})$
$Post_t \times S_f^u$	-0.246a	-0.243a	
	(0.0307)	(0.0265)	
$Post_t \times S_f^s$	-0.0300		
	(0.0335)		
$\ln(\text{Price}_{fpt})$			-8.596a
			(2.304)
$Post_t \times \text{Employment}_{f,t=0}$	0.00554	0.00981	0.119c
	(0.00525)	(0.00632)	(0.0655)
N	407168	407168	407168
Fixed Effects:			
Firm × Product	Yes	Yes	Yes
Year	Yes	Yes	Yes
KP Stat			84.20

Note: Robust standard errors clustered at the firm-product level in parentheses. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

and Column (1) of Table D.4 shows no relative export response across exposed sectors. Combined with the structural mapping from relative to absolute effects in Appendix D.1, this implies $\Delta Q_k^j = 0$ for $j \in \{s, n\}$. It follows that $\Delta Q_k^j = 0$ for $j \in \{s, n\}$, so all consumption-side gains operate through the *Using* varieties.

We recover ΔI_k from the national income identity:

$$\Delta I_k = \sum_{f \in \mathcal{F}_k^u} \Delta R_{fkk}^u + \Delta N X_k^r.$$

Using Column (1) of Table D.4, we estimate that revenues of domestic *Using* firms increase by \$195.8 million following ONS. Combining this with the ONS-induced decline in net exports of recycled plastic waste of \$92.2 million, recovered from Equation 2.1 and Table A.1, yields:

$$\Delta I_k = 195.8 - 92.2 = \$103.6 \text{ million}$$

.

Table D.4. Trade-Relevant Margins of the ONS Policy

	Exports	Imports excl. Virgin and Plastic Waste
	(1)	(2)
$Post_t \times S^u_f$	10.16	7.829
	(9.273)	(11.29)
$Post_t \times S^s_f$	0.152	-0.0477
	(0.149)	(0.133)
$Post_t \times \text{Employment}_{f,t=0}$	0.0310	0.0154
	(0.0202)	(0.0158)
N	76465	76465
R^2	0.820	0.853
Fixed Effects:		
Sector × Year	Yes	Yes
Firm	Yes	Yes

Note: This table presents trade-related margins as dependent variables in Equation (4.4). Outcomes are firm exports in Column (1) and firm imports of non-plastic/non-virgin products in Column (2). All outcome variables are in logarithms and estimated using OLS. The coefficients of interest are on $Post_t \times S^u_f$ and $Post_t \times S^s_f$, where $Post_t$ equals one after 2017, S^u_f is the 2016 cost share of imported plastic waste, and S^s_f is the 2016 share of plastic waste in total waste. The sample covers the years from 2013 to 2019. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

The remaining unknown is E^u_k. We cannot observe this directly. We therefore bound sectoral expenditure using observed domestic revenues, allowing imported expenditure to be at most equal to domestic expenditure, consistent with home bias in consumption:

$$E^j_k \leq 2 \sum_{f \in \mathcal{F}^j_k} R^j_{fkk}.$$

Home bias implies that expenditure on domestically produced varieties weakly exceeds expenditure on imported varieties. For the *Using* sector, this implies $E^u_k \leq$ 3.4 billion USD.[54] This assumption makes the break-even condition least demanding.

D.3.1. *Model Implied Welfare Cost using Compensating Variation.* Under CES demand within sector j, expenditure on domestic variety f satisfies:

[54]The corresponding statistics for each sector are reported in Table D.1.

$$\frac{R^j_{fkk}}{E^j_k} = \left(\frac{p^j_{fkk}}{P^j_k}\right)^{1-\sigma_j},$$

where R^j_{fkk} denotes expenditure on domestic variety f, E^j_k denotes total sectoral expenditure, p^j_{fkk} is the firm-level domestic price, P^j_k is the CES sectoral price index, and σ_j is the elasticity of substitution across varieties. Log-differentiating yields:

$$\frac{\Delta P^j_k}{P^j_k} = \frac{\Delta p^j_{fkk}}{p^j_{fkk}} + \frac{1}{\sigma_j - 1}\left(\frac{\Delta R^j_{fkk}}{R^j_{fkk}} - \frac{\Delta E^j_k}{E^j_k}\right).$$

For the *Using* sector, this expression maps observed firm-level revenue and price responses into the unobserved aggregate sectoral price index. Under CES preferences, the compensating variation associated with this price change is[55]:

$$CV_k = -E^u_k\left(\frac{\Delta P^u_k}{P^u_k}\right),$$

where positive values indicate welfare gains from lower prices. Substituting for the sectoral price index gives:

$$CV_k = \frac{1}{\sigma_u - 1}\Delta E^u_k - E^u_k\left(\frac{\Delta p^u_{fkk}}{p^u_{fkk}} + \frac{1}{\sigma_u - 1}\frac{\Delta R^u_{fkk}}{R^u_{fkk}}\right). \tag{D.2}$$

Identification of firm-level expenditure changes ($\Delta R^j_{fkk}/R^j_{fkk}$) relies on observed revenue responses following the ONS policy, as derived in Equation 4.4. Table 2 reports the empirical results of this specification in Column (1) and shows that statistically significant revenue adjustments arise exclusively in the *Using* sector.[56] The estimates indicate that the average exposed firm in the *Using* sector in Turkiye experienced an 11.5% increase in sales following

[55] There are no changes in the *Supplying* and *Neither* sectors in terms of sales, so they drop out of the CV equation.

[56] $\Delta R^j_{fkk}/R^j_{fkk}$ are changes in domestic sales. Because our data do not distinguish domestic from export sales, we cannot estimate this regression using domestic sales alone. Instead, we estimate the same specification using total and export revenues (Column 3) and find no significant changes in exports of any sector ($j \in \{u, s, n\}$). We therefore attribute the significant response in the *Using* sector to domestic sales.

the ONS policy, consistent with the model's prediction that recyclable plastic waste is redirected toward *Using* firms in Turkiye at lower prices.[57] On the other hand, the *Supplying* sector does not exhibit statistically significant changes in total revenue following the policy shock. The absence of a response among *Supplying* firms is consistent with the fact that plastic waste constitutes a by-product of their primary production activity, so changes in demand for this by-product do not materially affect the main output and hence total revenue responses are muted. Firms in the *Neither* sector, in turn, neither use nor supply recyclable plastic inputs, and the relatively small size of the *Using* sector implies that general equilibrium spillovers to these firms are negligible. (Appendix Section D.1 further shows that general equilibrium spillovers to the *Neither* sector are negligible.)

To get firm-level price changes ($\Delta p^j_{fkk}/p^j_{fkk}$), we use the output price of a firm-product pair as the dependent variable in Equation 4.8. Because firms often produce multiple products with distinct prices, identification exploits variation at the firm-product level. The results are reported in Column (1) of Table D.3. Consistent with the revenue results, statistically significant price adjustments arise only among *Using* firms. The estimates imply that the output price of the average exposed firm declines by approximately 0.2% following the ONS policy.[58] In contrast, we find no statistically significant price changes for firms in the *Supplying* sector, and firms in the *Neither* sector remain unaffected, through the absence of general equilibrium spillovers.

To determine the ONS-induced changes in sectoral expenditure ΔE^j_k, firm-level revenues in the *Supplying* and *Neither* sectors remain unchanged following the ONS policy ($\Delta R^j_{fkk} = 0$ for $j \in \{s, n\}$). Unchanged revenues imply unchanged equilibrium prices and quantities in these sectors under CES demand, and also unchanged expenditures on domestic and imported varieties ($\Delta E^j_k = 0$ for $j \in \{s, n\}$). Consequently, expenditure adjustments occur only in the *Using* sector:

$$\Delta E_k = \sum_j \Delta E^j_k = \Delta E^u_k.$$

[57]From $\exp(\beta^R \times \bar{S}^u_f) - 1$, where $\bar{S}^u_f = 0.009$ denotes the mean exposure of *Using* firms and $\beta^R = 12.11$.
[58]From $\exp(\beta^P \times \bar{S}^u_f) - 1$, where $\bar{S}^u_f = 0.009$ denotes the mean exposure of *Using* firms and $\beta^P = -0.243$.

Combining with the accounting identity in Equation (C.4) yields:

$$\Delta E_k^u = \Delta E_k = \sum_{f \in \mathcal{F}_k^u} \Delta R_{fkk}^u + \Delta NX_k^r,$$

where ΔR_{fkk}^u denotes the change in revenues of domestic *Using* firms, estimated in Column (1) of Table 2, and ΔNX_k^r denotes the ONS-induced change in net exports of recycled plastic waste, recovered from the global trade analysis in Equation 2.1 and Table A.1. The estimates imply that revenues of *Using* firms increase by \$195.8 million following the ONS policy, while net exports of recycled plastic waste decline by \$92.2 million, yielding a net increase in expenditure in the *Using* sector of $\Delta R_k^u = \$103.6$ million.

Finally, we need information on sectoral expenditure in country k is given by

$$R_k^j = \sum_{f \in \mathcal{F}_k^j} R_{fkk}^j + \sum_{o \neq k} \sum_{f' \in \mathcal{F}_o^j} R_{f'ok}^j,$$

which includes both domestic expenditure and imports. However, since sectors are defined by production technology and waste-generation characteristics—information that is unobserved for foreign firms in trade statistics—imported varieties cannot be assigned to sectors. Substituting the empirical moments into compensating variation in equation D.2 yields:

$$CV_k = \frac{1}{8.6} \times 103.6 \text{ million USD} - E_k^u \left(-0.02 + 0.11\right),$$

where the term multiplying E_k^u is strictly positive, implying that compensating variation is decreasing in the size of the *Using* expenditure base.

Substitute the upper bound of CV into aggregate welfare,

$$\begin{aligned}
\Delta W_k =& \Delta Q_k - \Delta D_k \\
=& (1/P_k) \left(\Delta I_k - E_k^u \left(\Delta P_k^u / P_k^u \right) - D_k^{VSL} \right) \\
=& (1/P_k) \left(\Delta I_k - CV_k - D_k^{VSL} \right),
\end{aligned}$$

with $\Delta I_k = \$103.6$ million and $CV_k \leq \$12.05$ million, implies:

$$P_k \Delta W_k \leq 103.6 + 12.05 - D_k^{VSL}.$$

Substituting the maximum model-implied consumption gain into aggregate welfare yields:

$$P_k \Delta W_k \leq \begin{cases} 115.65 - 557.78 = -442.13 & \text{(a) Disposal only} \\ 115.65 - 570.63 = -454.98 & \text{(b) Disposal + recycling + Using firms} \\ 115.65 - 611.23 = -495.58 & \text{(c) Disposal + PM2.5 from recycling (low) + Using firms} \\ 115.65 - 654.86 = -539.21 & \text{(d) Disposal + PM2.5 from recycling (high) + Using firms.} \end{cases}$$

Thus, even under the most consumption-favorable assumptions of the model, welfare remains strictly negative across all empirically grounded mortality-cost specifications, with implied welfare losses ranging from \$442.13 million to \$539.21 million.

As an alternative exercise, we can also calculate the shadow price of PM10 that would be required for the ONS policy to be exactly welfare-neutral in Turkiye. Rather than relying on mortality-based value-of-statistical-life (VSL) estimates, this approach directly asks: what per-kilogram PM10 cost would equate total environmental damages to the maximum model-implied economic gains?

From the compensating-variation exercise above, the most consumption-favorable assumptions imply that the maximum possible consumption gain from the ONS policy is:

$$\Delta I_k + CV_k = \$115.65 \text{ million}.$$

Let $\Delta PM10_k$ denote the total increase in PM10 emissions induced by ONS. Then the break-even PM10 shadow price is:

$$p_k^{PM10,break-even} = \frac{115.65 \text{ million USD}}{\Delta PM10_k}.$$

Using the four PM10 estimates from Panel (A), this implies:

$$p_k^{PM10,break-even} = \begin{cases} \frac{115.65}{4{,}368{,}259} = \$26.48 & \text{(a) Disposal only} \\ \frac{115.65}{4{,}468{,}784} = \$25.88 & \text{(b) Disposal + recycling + Using firms} \\ \frac{115.65}{4{,}786{,}718} = \$24.16 & \text{(c) Disposal + PM2.5 from recycling (low) + Using firms} \\ \frac{115.65}{5{,}128{,}411} = \$22.55 & \text{(d) Disposal + PM2.5 from recycling (high) + Using firms,} \end{cases}$$

where PM10 quantities are expressed in kilograms.

To sum up, for the ONS policy to have been welfare-improving, the social cost of PM10 in Turkiye would need to be no greater than approximately \$22.55–\$26.48 per kilogram, depending on the pollution accounting framework.

Table D.5. Derived Consumption-Side Parameters

Term	Symbol	Value	Equation / Note
Panel A: Firm-Level Expenditure Changes			
Expenditure % change, domestic firm f in Neither	$\Delta R^n_{fkk}/R^n_{fkk}$	0.000	From labor market clearing, $\hat{\beta}^L_n = 0$, and Monotonicity Assumption 4
Expenditure % change, domestic firm f in Supplying	$\Delta R^s_{fkk}/R^s_{fkk}$	0.000	$\exp(\hat{\beta}^R_s \cdot \overline{S^s_{fk}}) - 1$
Expenditure % change, domestic firm f in Using	$\Delta R^u_{fkk}/R^u_{fkk}$	0.115	$\exp(\hat{\beta}^R_u \cdot \overline{S^u_{fk}}) - 1$
Panel B: Firm-Level Price Changes			
Price % change, domestic firm f in Neither	$\Delta p^n_{fkk}/p^n_{fkk}$	0.000	From $\hat{\beta}^l_n = 0$ and Monotonicity Assumption 4
Price % change, domestic firm f in Supplying	$\Delta p^s_{fkk}/p^s_{fkk}$	0.000	$\exp(\hat{\beta}^P_s \cdot \overline{S^s_{fk}}) - 1$
Price % change, domestic firm f in Using	$\Delta p^u_{fkk}/p^u_{fkk}$	−0.002	$\exp(\hat{\beta}^P_u \cdot \overline{S^u_{fk}}) - 1$
Panel C: Sectoral Expenditure Changes			
Sectoral expenditure change in Neither	ΔE^n_k	0.000	From CES assumption
Sectoral expenditure change in Supplying	ΔE^s_k	0.000	From CES assumption
Sectoral expenditure change in Using	ΔE^u_k	103.527 million USD	From National Income Identity in Assumption 3 $\Delta E^u_k = \Delta I_k = \Delta \text{Firm Sales} + \Delta NX^r$
Panel D: Elasticity			
Elasticity of substitution in Using	σ_u	9.6	$1 - \beta^D$

Notes: This table reports the parameters derived from the raw data and estimated coefficients in Table 2, used as inputs to the consumption-side welfare calculation. Firm-level expenditure and price changes are evaluated at the average sector exposure ($\overline{S^u_{fk}}$ and $\overline{S^s_{fk}}$). The sectoral expenditure change in Using combines the change in domestic firm sales and the change in net imports of ONS-banned recycled plastic waste (ΔNX^r).

Appendix E. Environmental Impacts of the ONS policy

E.1. **Data Sources for Environmental Impact Calculations.** This section details the data sources and methodology utilized to determine the environmental impact following the ONS. We will first outline the data sources for domestic plastic waste production, imports, exports, and their respective changes. Following that, we will explain the data sources for waste management levels.

The environmental impact calculations in Table 3 draw on three sets of inputs described below. PM10 emission intensities, used in Panel B scenarios (b)–(d), come from Li et al. (2024) and Kim et al. (2023). CO_2e emission factors for waste burning and virgin plastic production, used in Appendix E.3, come from Climate Trace and Li et al. (2024) respectively. The broader toxicity and ecotoxicity indicators reported in Appendix Table E.3 draw on the same LCA study.

Environmental Impact Intensities: Environmental impact intensities used in this analysis are from midpoint impacts of the life-cycle assessment (LCA) study by Li et al. (2024), which evaluates the environmental impacts of global plastic waste using the ReCiPe 2016 life-cycle impact assessment method. The study reports indicators for plastic waste treatment and substitution processes across multiple polymers (HDPE, LDPE, PET, PP, PS, PVC), waste treatment options and countries. Waste treatment pathways include recycling, incineration (with and without energy recovery), landfill, open dumping, and open burning. All impacts are calculated per functional unit of 1 kg of plastic waste treated or substituted, and reported using ReCiPe 2016 (H) v1.13 midpoint indicators (e.g. climate change in kg CO_2-eq, marine ecotoxicity in kg 1,4-DCB-eq). We extract the Turkiye-specific impact intensities from the publicly released LCA outputs through the data repository associated with the publication.

Manufacturing impacts of increased plastic product output are modeled using conversion-stage impacts (‘box production’) from Kim et al. (2023), normalized per kilogram of product and expressed in ReCiPe 2016 Midpoint (H) units. Material virgin production impacts are excluded.

The various pollutants and their descriptions are provided next.

Fine particulate matter with an aerodynamic diameter of 2.5 micrometres or less (PM2.5) is a major air pollutant because particles of this size can penetrate deep into the lungs and enter the bloodstream. Long-term exposure to PM2.5 is strongly associated with increased risks of cardiovascular disease (including ischaemic heart disease and stroke), chronic obstructive pulmonary disease, lower respiratory infections, and lung cancer. As a result, PM2.5 is considered one of the most health-relevant air pollutants globally, and is a leading environmental risk factor for premature mortality in assessments by the World Health Organization.[59] Unlike many other air pollutants, PM2.5 has no clear safe threshold, meaning that increases in emissions can contribute to health risks even in regions that already meet regulatory air-quality limits.

Human toxicity, carcinogenic, quantifies the potential contribution of emissions to human cancer risk through release of many carcinogenic substances (e.g. certain metals, organic compounds, and combustion by-products). **Human toxicity, non-carcinogenic**, instead captures neurological effects, developmental harm, respiratory and organ damage, and other health impacts that do not show up as cancer. As a midpoint indicator, these are intended for comparative and screening-level assessment, not for direct estimation of cancer cases or health outcomes. They exclude long-term emissions from landfills (beyond 100 years) and focuses on near- and medium-term emissions associated with waste management activities.

Terrestrial ecotoxicity measures the potential harm of chemical emissions to land-based ecosystems, such as soils, plants, invertebrates, and soil microorganisms. The indicator aggregates emissions of many toxic substances (e.g. heavy metals, persistent organic pollutants, combustion residues), allowing diverse toxic emissions to be expressed on a common scale, but again, it does not predict actual ecosystem damage or species loss without further site-specific exposure and ecological modeling, and is a screening assessment measure.

[59] https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health?

Baseline Plastic Waste Levels: The annual volume of plastic waste within Turkiye is computed by summing domestically produced waste with imported waste and subtracting exported waste. The pre-ONS levels of domestically produced waste is sourced from the World Bank, while data for imports and exports are obtained from UN Comtrade.

Waste Management Practices: We obtain the data on mismanaged waste shares for each country group from the World Bank's What a Waste Global Database.[60] Although the Turkish Statistical Institute provides data on waste management, for consistency across countries, we use the mismanagement share reported by the World Bank for Turkiye.

For Exporters and the Rest of the World (RoW), we calibrate waste mismanagement impacts with the elasticity of waste emissions with respect to waste imports. Emissions of greenhouse gases from waste disposal for countries are from Climate Trace. They draw on emissions from satellite data and have the advantage of capturing formal and informal waste mismanagement. But this measure attributes emission to all waste, and cannot be separately examined for those from just plastic waste. Waste imports of destinations are instrumented with the destination's proximity to countries that initially exported more of the banned plastic waste products to China. Following gravity in waste trade, the reasoning is that destinations that are closer to origin countries that exported more to China will be more likely to receive the displaced waste imports after China's ban. We find that a 1 percent higher gravity measure for a destination is associated with a 0.652 percent higher increase in waste imports after the ban. A 1 percent increase in ONS-banned plastic waste imports in turn translates into a 0.04 percent increase in emissions from waste in the destination. This emission elasticity is multiplied by the drop in waste exports for Exporters and the rise in waste imports for the RoW to arrive at the baseline values for the change in waste mismanagement.

E.2. **Government Objective Function and Optimal Fine.** We extend the model by allowing the government to set an expected per-unit fine on improperly disposed plastic

[60]World Bank's What a Waste Global database estimates that Turkiye generated 0.959 million tonnes of plastic waste annually before the ONS policy, which includes waste from all industrial sectors, households, and other non-industrial sources.

waste. Let the expected fine equal to the probability of detection times the statutory fee conditional on violation:

$$F_k \equiv p_k(\theta_k)\,\mathrm{Fee}_k$$

The fine enters Supplying firms' private disposal cost as

$$(w_k^r D_k + F_k) b_{fk}^s.$$

Let equilibrium outcomes under fine F_k be

$$Q_k(F_k), \quad B_k(F_k), \quad X_k(F_k), \quad V_k(F_k), \quad \Pi_k^j(F_k), \qquad j \in \{s, u, n\}.$$

Environmental damages are

$$\mathcal{D}_k(F_k) = \xi_{bk} B_k(F_k) + \xi_{xk} X_k(F_k) + \xi_{vk} V_k(F_k).$$

The government chooses F_k to maximize

$$\mathcal{G}_k(F_k) = Q_k(F_k) - \mathcal{D}_k(F_k) + \sum_{j \in \{s,u,n\}} (\lambda_k^j - 1)\Pi_k^j(F_k) - C_k(F_k), \tag{E.1}$$

where Π_k^j is aggregate sector-j profit, $\lambda_k^j \geq 1$ captures excess political weight on sectoral profits, and $C_k'(F_k) \geq 0$ is the marginal resource cost of enforcement.

The first-order condition for an interior optimum is

$$Q_k'(F_k^*) - \mathcal{D}_k'(F_k^*) + \sum_j (\lambda_k^j - 1)\Pi_k^{j\prime}(F_k^*) - C_k'(F_k^*) = 0. \tag{E.2}$$

where

$$\mathcal{D}_k'(F_k) = \xi_{bk} B_k'(F_k) + \xi_{xk} X_k'(F_k) + \xi_{vk} V_k'(F_k),$$

Defining

$$\delta_k(F_k) \equiv -B_k'(F_k) > 0,$$

gross marginal social damage per unit of mismanaged by-product is

$$MSD_k(F_k) \equiv -\frac{\mathcal{D}_k'(F_k)}{\delta_k(F_k)} = \xi_{bk} + \xi_{xk}\left(-\frac{X_k'(F_k)}{\delta_k(F_k)}\right) + \xi_{vk}\left(-\frac{V_k'(F_k)}{\delta_k(F_k)}\right). \tag{E.3}$$

When the fine only affects waste mismanagement, so that $X_k'(F_k) = V_k'(F_k) = 0$, then $MSD_k(F_k) = \xi_{bk}$. Substituting for $\mathcal{D}_k'(F_k)$ from FOC Equation (E.2), the marginal social damage can now be re-written as:

$$MSD_k(F_k^*) = \underbrace{-\frac{Q_k'(F_k^*)}{\delta_k(F_k^*)}}_{\text{consumption-cost wedge}} \underbrace{-\frac{\sum_j(\lambda_k^j - 1)\Pi_k^{j\prime}(F_k^*)}{\delta_k(F_k^*)}}_{\text{political-economy wedge}} + \underbrace{\frac{C_k'(F_k^*)}{\delta_k(F_k^*)}}_{\text{enforcement-cost wedge}}. \tag{E.4}$$

The consumption-cost wedge captures the real resource cost of reducing mismanagement. The political-economy wedge captures the government's excess weight on producer profits. The enforcement-cost wedge captures the resource cost of implementing a higher expected fine.

As a benchmark, when there is no lobbying by firms ($\lambda_k^j = 1$ for all j) and there are no enforcement costs $C_k'(F_k) = 0$, the government maximizes $Q_k(F_k) - \mathcal{D}_k(F_k)$, and Pigouvian implementation sets the private price of mismanagement equal to its gross marginal social damage:

$$F_k^{Pigouvian} = MSD_k(F_k^{Pigouvian}).$$

Equation (E.4) shows how the implemented fine differs from this Pigovian benchmark.

We use the following monotonicity conditions:

$$B_k'(F_k) < 0, \qquad Q_k'(F_k) \leq 0, \qquad \Pi_k^{j\prime}(F_k) \leq 0, \qquad C_k'(F_k) \geq 0.$$

The sign of $B_k'(F_k)$ follows from the Supplying firm's disposal choice: a higher expected fine raises the private marginal cost of improper disposal, $(w_k^r D_k + F_k)$, and weakly reduces optimal mismanagement. The sign of $Q_k'(F_k)$ is the equilibrium consumption-cost restriction: holding environmental damages fixed, stricter enforcement raises production or compliance costs and therefore weakly lowers the feasible CES consumption bundle.

The profit derivatives follow from envelope arguments. For Supplying firms, the fine enters profits only through $-(w_k^r D_k + F_k) b_{fk}^s$, so

$$\Pi_k^{s\prime}(F_k) = -\sum_{f \in \mathcal{F}_k^s} b_{fk}^{s*} = -B_k(F_k) \le 0.$$

For Using firms, the fine affects profits through the recycled-input price $\rho_k(F_k)$. Since $d\rho_k/dF_k \ge 0$, Hotelling's lemma gives

$$\Pi_k^{u\prime}(F_k) = -X_k(F_k) \frac{d\rho_k}{dF_k} \le 0,$$

where $X_k(F_k)$ is aggregate recycled-input demand. Firms in the Neither sector are affected only through general-equilibrium spillovers, which we assume are weakly non-positive or negligible. Enforcement costs are non-decreasing by construction.

Revealed-preference lower bound. Since $Q_k'(F_k^*) \le 0$, $\Pi_k^{j\prime}(F_k^*) \le 0$, $\lambda_k^j \ge 1$, and $C_k'(F_k^*) \ge 0$, all wedges in (E.4) are weakly non-negative. Therefore the observed expected fine provides a lower bound on gross marginal social damage:

$$F_k^* \le MSD_k(F_k^*). \tag{E.5}$$

If $X_k'(F_k) = V_k'(F_k) = 0$, this implies

$$F_k^* \le \xi_{bk}.$$

Thus, in the empirically relevant case where the fine primarily affects improper disposal, observed expected fines provide a revealed-preference lower bound on the per-unit damage parameter ξ_{bk}.

In 2021-2022, several high-profile media reports covered waste mismanagement in Turkiye. The government responded with increased inspections and fines to reduce waste pollution.[61]

[61] https://www.bbc.com/turkce/haberler-dunya-57142579, https://www.greenpeace.org/static/planet4-turkey-stateless/2022/02/be5d1ad3-game-of-waste-global-plastic-waste-trade-impact-on-turkey-greenpeace-report.pdf

These fines provide a revealed-preference lower bound on the planner's valuation of environmental damages. In 2024, the fines amounted to $F_k B_k = 212.92$ million USD (in 2016 prices).

E.3. **Global Pollution.** We reinterpret the environmental components of the model as contributing to a global externality. In particular, total emissions are given by the sum of emissions from waste disposal and virgin plastic production across all countries, $\sum_k B_k$ and $\sum_k V_k$, respectively. Thus, the impact of the policy operates through changes in these global aggregates rather than their country-level distribution.

Among the margins of the model, changes in global emissions are most directly linked to waste disposal, $\sum_k \Delta B_k$, as open-air burning of plastic waste releases carbon dioxide, methane, and other greenhouse gases into the atmosphere. Open-air burning is particularly emissions-intensive due to incomplete combustion, making it a key source of climate-related externalities in our setting. At the same time, increased availability of recycled plastic can, in principle, reduces emissions by displacing the production of virgin plastics, $\sum_k \Delta V_k$, which is highly carbon-intensive. While we do not observe substitution away from virgin plastic use within Turkiye (Column (3) of Table 2), this channel may still operate at the global level. The overall impact of the policy on CO_2e emissions therefore depends on the balance between increased emissions from waste disposal and potential reductions from lower demand for virgin plastic production.

We can directly examine these two channels using country-level data. In particular, we study how imports of plastic waste affect CO_2e emissions from waste sites—measured using satellite-based data and reported by Climate Trace—and the production of virgin plastics—reported by EXIOBASE. To do so, we estimate the following specifications to assess the effects of the ONS policy on per capita CO_2e emissions ($\mathrm{CO_2pc}_{dt}$), and virgin plastic production (m^v_{dt}), in destination country d at time t:

$$\begin{aligned} \ln \mathrm{CO_2pc}_{dt} &= \beta_x \ln x_{dt} + \Gamma_x X_{dt} + \alpha_d + \alpha_t + \varepsilon_{dt} \\ \ln m^v_{dt} &= \beta_v \ln x_{dt} + \Gamma_v X_{dt} + \alpha_d + \alpha_t + \nu_{dt}, \end{aligned} \tag{E.6}$$

where x_{dt} denotes imports of plastic waste into destination d at time t. The vector X_{dt} includes destination-specific controls—per capita income, the destination's pre-policy share in China's banned-waste imports, and initial virgin plastic production—all interacted with a post-ONS indicator. We include destination and year fixed effects.

To address the potential endogeneity of plastic waste imports, we instrument $\ln x_{dt}$ with two excluded instruments, both pre-determined and motivated by the gravity structure of waste trade. The first is an indicator for whether destination d runs a positive trade balance against origin countries that previously supplied China, weighted by each origin's share in China's pre-policy banned-waste imports. The second is the destination's average proximity to those origins, measured as the mean ratio of distance-to-China to distance-to-destination across origins that exported banned plastic waste to China in 2016. Both instruments capture the ease with which displaced waste flows can be redirected to destination d after the ban. The overidentification test reported in Table E.1 (J-statistic of 0.22, $p = 0.64$) does not reject instrument validity.

We begin by establishing the relevance of the instrument. Column (1) of Table E.1 reports the first-stage results. The interaction between the pre-ONS weighted trade imbalance and the post period strongly predicts plastic waste imports, with a coefficient of 1.04. This indicates that countries with pre-existing trade surpluses vis-à-vis major exporters to China experience significantly larger increases in waste imports following the policy. The magnitude is economically meaningful, consistent with the idea that the availability of empty return containers lowers the marginal cost of importing plastic waste.

Table E.1 also reports the second-stage results. The IV estimates in Column (3) show that a 10 percent increase in plastic waste imports raises per capita CO_2e emissions from waste sites by approximately 0.35 percent, consistent with increased disposal and burning of waste. At the same time, Column (5) shows that higher imports reduce virgin plastic production: a 10 percent increase in imports lowers production by about 0.8 percent, indicating some substitutability between recycled plastic and virgin plastic.

Table E.1. Effects on CO_2 Emissions and Virgin Plastic Production

	$\ln x_{dt}$ (1) FS	$\ln \text{CO}_2\text{pc}^{waste}_{dt}$ (2) OLS	$\ln \text{CO}_2\text{pc}^{waste}_{dt}$ (3) 2SLS	$\ln m^v_{dt}$ (4) OLS	$\ln m^v_{dt}$ (5) 2SLS
$\ln x_{dt}$		0.00704a	0.0347a	-0.0262a	-0.0785c
		(0.00157)	(0.00519)	(0.00920)	(0.0409)
$\left(\sum_o \frac{x_{do}-x_{od}}{x_{do}+x_{od}} \cdot \frac{x_{oChina}}{\sum_o x_{oChina}} > 0\right) * \mathbb{1}\{t \geq 2017\}$	1.041a				
	(0.231)				
$\overline{\ln(dist_{oChina}/dist_{od})} * \mathbb{1}\{t \geq 2017\}$	0.813c				
	(0.469)				
GDP $\text{pc}_{dt} * \mathbb{1}\{t \geq 2017\}$	-0.0107	-0.000326	-0.000647b	-0.000637	-0.0000298
	(0.0103)	(0.000567)	(0.000313)	(0.00140)	(0.00154)
$\frac{x_{dChina}}{\sum_d x_{dChina}} * \mathbb{1}\{t \geq 2017\}$	14.64b	0.175	0.0526	1.541	1.774c
	(5.639)	(0.238)	(0.161)	(0.965)	(0.937)
$\ln m^v_{d16} * \mathbb{1}\{t \geq 2017\}$	-0.0660	0.00262	0.00327	-0.0615a	-0.0628a
	(0.0823)	(0.00193)	(0.00246)	(0.0182)	(0.0171)
N	273	273	273	273	273
R^2	0.793	1.000	-1.510	0.994	-0.0211
KP-Stat			13.30		13.30
J-Stat			0.220		0.219
			(0.6396)		(0.6390)
Fixed Effects:					
Country	Yes	Yes	Yes	Yes	Yes
Year	Yes	Yes	Yes	Yes	Yes

Notes: Notes: This table reports estimates of Equation (E.6). Column (1) presents the first stage, in which two excluded instruments—the share-weighted trade-balance indicator and the average log-distance ratio to former China-supplying origins—predict log plastic waste imports $\ln x_{dt}$. Columns (2) and (3) report OLS and 2SLS estimates of the effect of $\ln x_{dt}$ on per capita CO_2e emissions from waste sites, respectively. Columns (4) and (5) repeat the analysis with log virgin plastic production, $\ln m^v_{dt}$, as the dependent variable. Controls (interacted with the post-ONS indicator) include GDP per capita, the destination's pre-policy share in China's banned-waste imports, and initial virgin plastic production. The KP statistic reports the Kleibergen-Paap weak-identification F-statistic; the J-statistic reports Hansen's overidentification test (p-value in parentheses). The sample covers the years from 2015 to 2021 and includes country and year fixed effects. Standard errors are clustered at the country level. Statistical significance is denoted by letters: a for $p < 0.01$, b for $p < 0.05$, and c for $p < 0.10$.

While emissions from waste sites are directly observed in CO_2e terms, emissions from virgin plastic production are not. To quantify these, we convert changes in virgin plastic production into CO_2e using established emission factors. Following Li et al. (2024), we assume that the production of one ton of virgin plastic generates 2.22 tonnes of CO_2e.

We then combine baseline emissions with predicted changes in plastic waste imports to compute the implied change in global emissions. Specifically, we use initial CO_2e emissions from waste sites and inferred baseline emissions from virgin plastic production at the country level, together with the changes in imports estimated in Table 1 and the elasticities reported in Columns (3) and (5) of Table E.1. This yields the total change in global CO_2e emissions.

The results are reported in Table E.2. Row (a) shows that CO_2e emissions from waste sites increase in the rest of the world (excluding China), reflecting the diversion of plastic waste toward countries with higher mismanagement rates. Depending on whether we use IV or OLS estimates, this implies an increase of between 6.68 and 32.92 million kg of CO_2e, corresponding to a monetary cost of approximately 0.96 to 4.76 million USD using conversion factors from the *Environmental Prices Handbook*. In contrast, row (b) shows that China's reduction in plastic waste imports lowers emissions from waste sites by between 2.55 and 12.57 million kg of CO_2e, generating environmental gains of roughly 0.37 to 1.82 million USD.

TABLE E.2. Global Pollution (CO_2e)

Pollution Sources	Δ CO_2e (million kg)		Value (million USD)	
	IV	OLS	IV	OLS
(a) Waste-site emissions (excl. China)	32.92	6.68	4.76	0.96
(b) Waste-site emissions (China)	-12.57	-2.55	-1.82	-0.37
(c) Virgin plastic emissions (excl. China)	-6,386.58	-2,131.57	-922.51	-307.89
(d) Virgin plastic emissions (China)	10,141.94	3,384.95	1,464.95	488.94
(e) World emissions	3,772.53	1,256.86	544.92	181.55
(f) Emissions related to Turkiye	36.83	12.27	5.32	1.77

Notes: This table reports global CO_2e consequences through two channels: waste-site emissions and virgin plastic production. Waste-site emissions combine the estimated change in plastic waste imports with baseline country-level CO_2e emissions from waste disposal. Virgin-plastic emissions combine the estimated change in plastic waste imports with the implied change in virgin plastic production from EXIOBASE, converted to CO_2e at 2.22 tonnes per tonne of virgin plastic (Li et al., 2024). Rows (a)–(d) report estimates separately for China and the rest of the world. Row (e) aggregates channels (a)–(d) across all countries. Row (f) allocates world emissions to Turkiye by population share; other allocation keys yield similar orders of magnitude. The carbon price applied is 0.144 USD per kg CO_2e. All values are in 2016 USD.

The quantitatively dominant channel turns out to be virgin plastic substitution rather than waste disposal, consistent with the carbon intensity of primary plastic production. Since the production of one tonne of virgin plastic generates approximately 2.22 tonnes of CO_2e (Li et al., 2024), even modest changes in virgin output translate into large greenhouse gas effects, substantially exceeding the direct emissions from open burning. Row (c) shows that the diversion of plastic waste to the rest of the world reduces virgin plastic production, lowering emissions by between 2,131.57 and 6,386.58 million kg of CO_2e, equivalent to environmental gains of approximately 307.89 to 922.51 million USD. Conversely, row (d) shows that China substitutes toward virgin plastic production following the import ban, increasing emissions by between 3,384.95 and 10,141.94 million kg of CO_2e, corresponding to costs of roughly 488.94 to 1,464.95 million USD.

Aggregating across these channels, row (e) shows that the ONS policy increases global CO_2e emissions by between 1,256.86 and 3,772.53 million kg, corresponding to a total global cost of approximately 181.55 to 544.92 million USD. To contextualise the magnitude, this is roughly equivalent to the annual CO_2e emissions of approximately 0.3 to 1 million passenger vehicles. To assess the implications for Turkiye, row (f) allocates this global burden based on its population share, yielding an estimated cost of between 1.77 and 5.32 million USD.

E.4. **Other Pollutants.** Using impact intensities from Li et al. (2024), we quantify the impacts of the China ban on different environmental impact categories. Table E.3 summarizes the consequent increases in air pollution and human toxicity arising from the rise in domestic waste mismanagement following the ONS policy. Each of the 18 impacts shown in Table E.3 follows the lifecycle assessments recommended by the European Union. Consequently, they can be converted to a money metric with the corresponding EU environmental prices.

The results indicate moderate climate and air-quality impacts, but large terrestrial ecotoxicity and human toxicity pressures, consistent with waste-handling pathways that release persistent toxic substances to land rather than just short-lived air pollutants. Although plastics are often perceived as inert materials, plastic waste systems can be significant drivers of human and ecological toxicity due to the release of toxic additives, contaminants, and combustion by-products during burning, recycling, and secondary manufacturing. The burning of plastics, particularly those with additives and chlorine, can result in the formation of highly toxic compounds such as dioxins and furans. These compounds are produced during the combustion process as chlorinated plastics break down into smaller, more harmful chemicals that can bind to particles and disseminate through the air. The large human

Table E.3. Percentage Increases in Environmental Impact Categories

Human toxicity		**Air pollution & atmosphere**		**Ecotoxicity / resources**	
Cancer risk	23	Air pollution (health)	13	*Ecotoxicity*	
Non-cancer risks	19	Smog (human health)	21	Land ecosystems	35
		Smog (ecosystems)	21	Freshwater ecosystems	23
		Ozone layer damage	27	Marine ecosystems	24
		Climate change impact	24	*Resources & other*	
		Acid rain potential	14	Nutrient pollution (freshwater)	15
				Nutrient pollution (oceans)	16
				Freshwater use	15
				Land use	16
				Mineral depletion	10
				Fossil fuel scarcity	12
				Radiation exposure	16

Notes: Intensities are from Li et al. (2024).

toxicity and ecotoxicity results observed here indicate negative effects on long-term health and ecosystem risks.

To put the scale of the change in perspective, the actual amounts of the key pollutants that we have estimated can be contextualized by comparing them with other sources of pollution. The annual PM2.5 burden is comparable to emissions from a large coal plant or a medium-sized city of 500,000 to 1 million residents. The cancer toxicity impacts are of a magnitude similar to those associated with major lead-emitting activities, of about 5,000 to 10,000 tonnes of lead per year (or about 5 to 10 percent of annual production of large primary lead smelter). The non-cancer toxicity impacts are much larger, equivalent to over 50,000 to 100,000 tonnes of lead per year, that would amount to population-scale chronic exposure to contaminants. Other atmospheric impacts - acidification or smog burdens - are assessed to be small. However, there are very large modeled negative impacts on land-based ecosystems, such as soils, plants and microorganisms. For scale, the impacts are on the order of those from 25 to 50% of annual global lead production.

Aquatic impacts are assessed to be modest but these are likely to be underestimated due to inadequate modeling of microplastics in the environment. Plastics have penetrated deep into marine environments. As plastics degrade in the ocean, they release harmful chemicals such as phthalates, bisphenol A (BPA), and persistent organic pollutants. These substances can cause severe toxic effects on marine organisms, including fish, invertebrates, and plants. Microplastics, which result from the breakdown of larger plastic items, exacerbate the problem when ingested by marine life, leading to physical harm, chemical exposure, and disruption of the food chain. The accumulation of these toxic substances in marine organisms can damage reproductive, developmental, and immune systems, ultimately affecting the health of entire marine ecosystems. Furthermore, plastic debris can act as a vector for invasive species and pathogens, further destabilizing marine environments.

E.5. **Local Air Pollution: Other Regions.** In the absence of comparable data for the rest of the world, we draw on the substitution mechanism estimated for Turkiye to other country groups. Specifically, we aggregate countries into China, Top Exporters, and the Rest of the World (RoW), and compute land-area-weighted average PM10 concentrations for each group using country-level data. We then map observed changes in plastic waste imports (reported in Table 1) into implied changes in PM10 using the elasticity estimated for Turkiye. The results indicate substantial heterogeneity across regions. In China, the sharp decline in plastic waste imports (-86.47%) leads to a 3.57% reduction in PM10, corresponding to a decrease of approximately 13,254 tonnes. In contrast, the increase in plastic waste imports among Top Exporters (8.83%) and the RoW (11.66%) raises PM10 by 0.36% and 0.48%, respectively, implying increases of 1,235 and 10,223 tonnes.[62]

[62]These results are obtained in four steps. First, we compute country-level PM10 concentrations from the *WHO database* and aggregate them to the group level using land-area weights based on *World Bank data*. Second, we map changes in plastic waste imports into PM10 changes using the elasticity estimated for Turkiye, where a 517% increase in waste is associated with a 21.4% increase in PM10; we assume this relationship scales proportionally across countries. Third, we convert percentage changes in PM10 into changes in total PM10 mass by multiplying group-level concentrations by land area and a mixing height of 500 meters, converting units from $\mu g/m^3$ to tonnes. Finally, we monetize these changes using the benchmark value above. While this valuation approach aligns with commonly used environmental pricing handbooks, it remains conservative relative to VSL-based approaches that incorporate fuller mortality and morbidity risks. This pattern is consistent with previous work showing that China's ban improved domestic air quality and health outcomes within China (Li and Takeuchi, 2023; Shi and Zhang, 2023; Unfried and Wang, 2022) while also shifting trade patterns internationally (Martin et al., 2021a; Wen et al., 2021; Chunsuttiwat and Coxhead, 2024).