PLASTIC TURKEY: INTERNATIONAL IMPACTS OF CHINA'S WASTE IMPORT BAN

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Abstract. In pollution haven theories, trade and environmental policies can have significant effects on the levels and incidence of pollution and economic activity across countries, leading potentially to a race to the bottom. This paper studies the international effects of China's ban on plastic waste imports. In recent decades, high-income countries had been reducing their plastic waste burden by exporting it to China. This changed in 2017 when China banned key plastic waste imports, raising concerns over creation of waste havens in other parts of the world. The paper shows that China's policy led to a diversion of trade that had repercussions for countries across the world. Turkey emerged as a major importer of plastic waste from high-income countries. We provide direct evidence that importers in Turkey gained economically from better access to plastic waste that could be recycled and re-used as inputs in production. But their gains did not outweigh the losses of domestic firms that generated plastic waste and were displaced by import competition after China's ban. These domestic waste generators became more likely to mismanage their plastic waste, including through open burning. Air pollution increased more in Turkish regions where these waste generators were located. We model the channels of waste and recycling in a gravity model of trade and the environment to explain the empirical findings and to quantify the environmental externalities from China's unilateral import ban.

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Preliminary & Incomplete. Please do not circulate, cite or quote. e-mails: da497@cam.ac.uk, banu.demirpakel@economics.ox.ac.uk, S.Dhingra@lse.ac.uk. We are grateful to Clare Balboni, Vasco Carvalho, Kamala Dawar, Helene Ollivier and various seminar participants for their comments. The usual disclaimer applies.

1. Introduction

Trade and environmental policies can have significant effects on the levels and incidence of pollution and economic activity across countries. Pollution haven theories suggest that a race to the bottom on environmental outcomes can arise when countries with weaker environmental regulations become the preferred choice of location of pollution-generating activities as a result of shifts in trade and environmental policies. This paper examines the international impacts of the ban on plastic waste imports imposed by China in 2017 that led to a dramatic reorganisation of the world plastic waste market.

Global plastic production has increased dramatically since the 1950s, and it has generated large volumes of plastic waste.[1] Most plastic waste poses pollution problems that persist for a long time due to the slow rate of natural removal of plastic (Brooks, Wang, and Jambeck (2018), Geyer, Jambeck, and Law (2017), MacLeod, Arp, Tekman, and Jahnke (2021), Worm, Lotze, Jubinville, Wilcox, and Jambeck (2017a)). The global 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.[2]

In recent decades, high-income countries had been reducing their plastic waste burden by exporting it to lower-income countries. The ability to export waste to countries with cheaper, but often poorer waste disposal practices, has been a source of controversy and research at least as far back as the 1980s when news hit of garbage ships from OECD ports attempting to dump their unapproved cargo in various low-income countries (Copeland (1991), Baggs (2009), Kellenberg (2012)). Since then, cheaper processing fees in emerging markets have led to a staggering rise in global waste trade.

Global exports of plastic waste increased by 817 percent between 1993 and 2016, and 87 percent of all exports have been shipped from high-income countries to developing countries

[1] From 2 million metric tons (MMT) in 1950 to 322 MMT in 2015, or close to the total weight of the human population produced in plastic every year (Worm, Lotze, Jubinville, Wilcox, and Jambeck (2017a)).

[2] See UNEP (2020). For example, UN Environment (2018) estimates that 99 percent of seabirds would have ingested plastic waste by 2050. New plastic material makes up 20 percent of virgin petroleum consumption and is expected to contribute 15 percent to global greenhouse gas emissions by 2050 (Ellen MacArthur Foundation (2016); Walker and McKay (2021)).

(Brooks, Wang, and Jambeck (2018)). This has raised concerns over creation of waste havens in parts of the world where environmental regulations are weaker. And, the concerns have been exacerbated by the reorganisation of the world market in plastic waste after China's import ban, that has culminated in the plastic waste amendment of the Basel Convention.

In the 1990s, emerging markets, particularly China, found that ships could efficiently deliver waste from developed economies and that material could be drawn from this waste to be used for further production (Lee, Wei, and Xu (2020), Brooks, Wang, and Jambeck (2018)). Between 2010 and 2016, China had amassed an additional 10 to 13 percent of plastic waste through imports. While recyclable material could be drawn from the imported waste, unused material added to China's already burgeoning problem of managing domestically-generated waste. Environmental and health concerns over processing and disposal became more salient in China over time, particularly after its winter haze in 2013. The Chinese government tightened a number of air pollution regulations including permits and targets for activities emitting volatile organic compounds (Li and Takeuchi (2023)). In 2017, it enacted a far-reaching policy to only allow waste imports that passed very stringent contamination criteria into the country. This policy, known as "Operation National Sword" (ONS), led to a collapse in waste imports into China, and immediately led to calls for global action to prevent the displaced waste from potentially finding its way to nations with weaker regulations (Brooks, Wang, and Jambeck (2018)).

The ONS policy presents a stark setting to understand the relocation of economic activity and pollution arising from unilateral policy action. The policy clearly had bite - it amounted to a complete collapse of waste imports into China which made up more than half of all world imports. The resulting shakeup of the global market could have improved global waste management had the displaced imports stayed back in their source countries or moved to third countries with more stringent environmental regulations. Or, it could have resulted in a deterioration, had the displaced waste got diverted to third countries that mismanaged it more. As it turned out, the displaced waste imports did find their way to other emerging market destinations (Martin, Oliveira, Oliveira, and Bezerra (2021), Wen, Zou,

Liu, Huang, Evrendilek, Yan, Li, and Liu (2021)). Yet we know little about the economic and environmental effects of the trade diversion that resulted from China's ban.

Research on these effects has been limited by the paucity of direct evidence. While trade data provides granular information on international flows, it is often hard to measure domestic responses in pollution-generating activities and the overall consequences for social welfare (Kellenberg (2015)). To make progress, we focus on Turkey which emerged as a key "dumping ground" for waste generated in advanced economies after the China ban (Interpol (2020), Human Rights Watch (2022)).[3] Despite the sharp bite of the policy, it appears at first glance that the pollution haven hypothesis finds little support during this episode because Turkey is not the worst at managing waste and many other lower-income countries in Asia, Africa or South America did not experience similar inflows of waste from advanced economies. This turns out to be a summary observation because with multiple countries in an integrated global economy, the pollution haven hypothesis is more subtle. Under the pollution haven hypothesis, tightening environmental policy in one country causes production of the polluting activity to relocate to other countries with weaker environmental policy.[4] Polluting activities leaving one country may not move to a country with the weakest environmental policy. And in the current context, Turkey was not the worst but it mismanaged more of its waste than China before the policy - 47 percent of plastic waste was mismanaged in Turkey compared to 25 percent in China in 2016 (Law, Starr, Siegler, Jambeck, Mallos, and Leonard (2020)).

Firm-level customs data show a notable increase in Turkish imports of plastic waste banned by China after 2017 (relative to similar products not subject to the ban). The vast majority of importers were manufacturing firms that recycled the plastic waste as inputs in their production process. These firms gained access to cheaper imported plastic waste material which enabled them to reduce their production costs and to increase their sales and domestic market share. As these importers gained access to better quality plastic waste from abroad,

[3]See Watch (2022). Also see Liu (2021), Department for Environment and (DEFRA), The Guardian (2022) and Greenpeace (2021).

[4]This definition is taken from Brian Copeland's 2013 lecture notes on the pollution haven hypothesis in a multi-country setting.

we find that they no longer wanted to buy as much of it from domestic firms that generated the plastic waste products banned by China. These domestic plastic waste suppliers experienced a decline in their sales (relative to other firms). But the negative impacts were not limited to the channel of sales.

After the China ban, these domestic firms were faced with greater competition from waste imports. Domestic firms that generated ONS-affected plastic by-products became less likely to recycle their waste. In fact, they became more likely to mismanage it in one of the most environmentally unsound ways - open or uncontrolled burning of plastic waste. Regions in Turkey that were more exposed to such domestic plastic waste generators experienced higher air pollution levels after the China policy, relative to less exposed regions.

Building on the empirical findings, we generalise the workhorse model of trade and the environment to externalities from waste generation and management (see Shapiro (2016) and Shapiro and Walker (2018)). The model conceptualises the distinction between waste and other pollution-generating activities through the ability to recycle waste. In line with a gravity model of trade, we obtain a mapping between trade outcomes and welfare, with the addition of recycling among the sufficient statistics to infer welfare. Along with changes in waste trade, global and national welfare impacts depend on three statistics: (i) the change in mismanagement of waste, (ii) the change in the usage of virgin resources that are replaced through recycling waste, and (iii) the degree of pollution generated from waste recycling, waste mismanagement and virgin resource extraction. The first channel is similar to the choice of installing abatement technology in the trade and environment literature. Our analysis enables its direct estimation for waste-generating firms in Turkey, and its inference from satellite data for other countries. The second channel makes waste different from pollution-generating activities, such as transport, that emit pollutants which cannot be recycled to conserve virgin resources. We infer this from substitutability between primary plastic production and plastic waste imports. The third channel is the relative degree of externalities generated from each waste activity and these are calibrated to emission factors for greenhouse gases and other pollutants from the natural sciences literature.

The model enables an examination of welfare effects through economic and environmental channels. Importantly, it provides market clearing conditions to pin down aggregate effects from estimates of relative effects in the causal difference-in-differences analysis of revenues, waste imports and virgin resource use. The main finding is that China's unilateral ban had a substantial trade destruction effect that reduced emissions from waste mismanagement in China, and resulted in global emission savings. But the economic and environmental burden was unequally distributed across countries. Turkey experienced negative economic and environmental consequences from the China ban: we find that total income fell because domestic plastic waste suppliers lost out to import competition, and waste mismanagement increased.

The rest of the paper is structured as follows. Section 2 places our paper in the context of the literature and then proceeds to a description of the China import ban and its trade diversion impacts. Section 3 presents empirical findings obtained from micro-level data on importers of plastic waste and waste-generating firms in Turkey. Section 4 introduces a theoretical framework to explain the key findings and to determine the aggregate welfare tradeoffs of the policy. Section 5 quantifies the economic and environmental impacts. Section 6 concludes.

2. Context and Policy

This section places the paper and the policy under study in the literature. It then provides details of the policy and its immediate consequences for Turkey's emergence as a key destination for plastic waste exports from advanced economies and for trade diversion across countries.

2.1. **Related Literature.**

Our paper makes contributions to four key areas of existing literature. We build on recent advances combining trade with environmental externalities to examine the welfare impacts of a large unilateral policy (e.g. Shapiro (2016), Shapiro and Walker (2018)). Within this literature, a number of studies empirically test the pollution

haven hypothesis of environmental policy differences determining the location of pollution-generating activities. However, pollution haven effects arising from differences in environmental policy stringency across countries have been empirically elusive because of various factors such as capital abundance that are correlated with environmental stringency and trade flows. In early work, Copeland and Taylor (2003, 2004) argue that pollution haven effects could be more precisely measured in a multi-country setting, and we exploit such variation induced by a more stringent environmental policy in one country to determine shifts in pollution-generating activities to third countries.

A multi-country setting provides new insights into the global effects of unilateral environmental policy. We find the channel of reduced domestic abatement in third countries to be empirically relevant, and the model suggests that it is the main driver of the increased environmental burden faced by Turkey after the ban. Previous work has mostly focused on the exporting channel of firm emissions while the more limited evidence on importing has typically suggested increased abatement through access to intermediate imports or escape-competition effects from technology upgrading with increased importing (see Cherniwchan, Copeland, and Taylor (2017) for a review, and Barrows and Ollivier (2018), Gutiérrez and Teshima (2018) for importing). We instead find evidence for the "distressed and dirty industry hypothesis" of reduced abatement expenditures among firms that are negatively affected by imports (Cherniwchan, Copeland, and Taylor (2017), also see Forslid, Okubo, and Ulltveit-Moe (2018)).

Waste trade has a long tradition of research and several papers consider its welfare consequences theoretically (e.g. Copeland (1991), Levinson (1999), Lee, Wei, and Xu (2020)). Prior empirical studies, such as Bunn and Blaney (1997), Baggs (2009), Kellenberg and Levinson (2014) and Thakur (2022), highlight patterns in waste trade, particularly the tendency for waste to flow from nations with more stringent environmental regulations to those with more lenient ones (Pollution Haven effects and hypothesis in waste). We confirm the finding of waste exports by advanced nations to more environmentally lax destinations in our

setting. Building on the comprehensive review by Kellenberg (2015), we advance the literature in several key dimensions identified as avenues for future research: the microeconomic upstream-downstream relationships (which we pin down using Turkish data), substitutability between virgin resources and recyclables (critical for our welfare calculations), and direct evidence of the social welfare effects of waste trade (which we explore through the lenses of pollution and income effects). The review notes the stark conclusion of Ray (2008) that the environmental costs associated with international trade in waste have outweighed any potential economic benefits for Asia and points to the need for evidence to examine this trade-off more thoroughly. We make progress on this front by evaluating the economic and environmental effects of a large policy change. This is closest in spirit to Tanaka, Teshima, and Verhoogen (2022) who also focus on the welfare consequences of an international environmental policy - birth outcomes in Mexico after the tightening of US air quality standards for lead.

The paper is also related to a large literature on the economics of waste that has studied a number of issues such as optimal recycling policy and impacts of cleanup of waste sites, primarily in developed countries (see Kinnaman (2006), Fullerton (2024) for reviews, and Viscusi, Huber, and Bell (2011), Currie, Greenstone, and Moretti (2011) for examples). Research on emerging and developing economies is a recent but growing field that has examined the rise in waste generation and domestic waste management solutions (see The World Bank (2018) for a review, Chong, Karlan, Shapiro, and Zinman (2015), Dhingra, Kondirolli, and Machin (2024) for examples).

Finally, our study contributes to a growing literature on environmental policies in emerging markets, including China and India. As emerging markets have grown and integrated with the world economy, they have seen substantial increases in consumption as well as pollution levels. Many of them have recently enacted more stringent environmental policies to address domestic concerns over pollution. With their growing importance in the world economy, emerging markets are increasingly likely to affect other countries through their environmental policies. This adds complexity to the international impacts of trade and environmental

policies, moving beyond the traditional North-South trade perspective. A number of papers have examined China's pollution policies (e.g. Greenstone, He, Li, and Zou (2021)) and specifically China's ONS policy, that we discuss in more detail in the next section. Previous research on China's waste import ban has provided evidence for improved environmental and health outcomes within China, demonstrating notable improvements in air quality in coastal areas (Li and Takeuchi (2023); Shi and Zhang (2023); Unfried and Wang (2022)). We account for these in our calculations of global effects, and provide direct evidence on waste mismanagement to show that the ban had unintended consequences in some other parts of the world.

2.2. **Policy.** This section provides the details of the policy context under study: China's waste import ban. In the 1990s, China's booming manufacturing sector led to a high demand for scrap materials as feedstock for its industries. As a result, many developed countries, especially in North America and Europe, began exporting large quantities of waste materials of paper, plastics and metals to China for recycling. These materials were often considered low-value or difficult to process domestically.

China's role as a global recycling hub led to the establishment of informal recycling and processing facilities, where imported waste materials were sorted, processed, and the residuals were disposed of. Over time, however, concerns grew about the environmental and health impacts of these practices. Many recycling operations lacked proper regulations and infrastructure, leading to pollution, groundwater contamination, and health hazards for workers.

The majority of these concerns were around the substantial influx of waste into China, often contaminated with food, garbage, and other pollutants. While paper, plastic, and metal were valuable to China, their recyclability was compromised if they arrived mixed with contaminants. To address this, China initiated "Operation Green Fence" in February 2013, an inspection effort aimed at reducing contaminated waste imports. While the value of plastic waste trade did not recover to levels seen before Green Fence in China, it remained significant in 2016. China imported 56% of the world's plastic waste in 2016 and the inspection policy

gave way to a more comprehensive step after 2016, which affected the entire world market in waste (Brooks, Wang, and Jambeck (2018), Tran, Goto, and Matsuda (2021).

In July 2017, China implemented "Operation National Sword". This policy banned 24 types of solid waste imports, including certain plastics and paper, while imposing strict quality standards on others. China ceased imports of banned products as illustrated in Figure 2.1. After China's waste import ban, Turkey became one of the main destinations for plastic scrap and waste products that were subject to the ONS policy and we discuss the trade diversion from the ban next.[5]

2.3. **Trade Diversion.** This sub-section examines the shift in world trade patterns of plastic waste that followed the ban imposed by China. We first show that Turkey emerged as an important new destination and then determine trade diversion through a gravity-type estimation.

Figure 2.1 starts with visual snapshots of the evolution of plastic waste imports annually from 2013 to 2020. It plots the volume of plastic waste for selected importers. China was the leading importer of waste until 2017, but its imports declined sharply thereafter. At the same time, plastic waste imports into Turkey jumped up, making it eventually the largest importer. Other countries such as Malaysia, India and Thailand also saw significant increases in imports as the dominance of China and Hong Kong plummeted after the ONS policy.

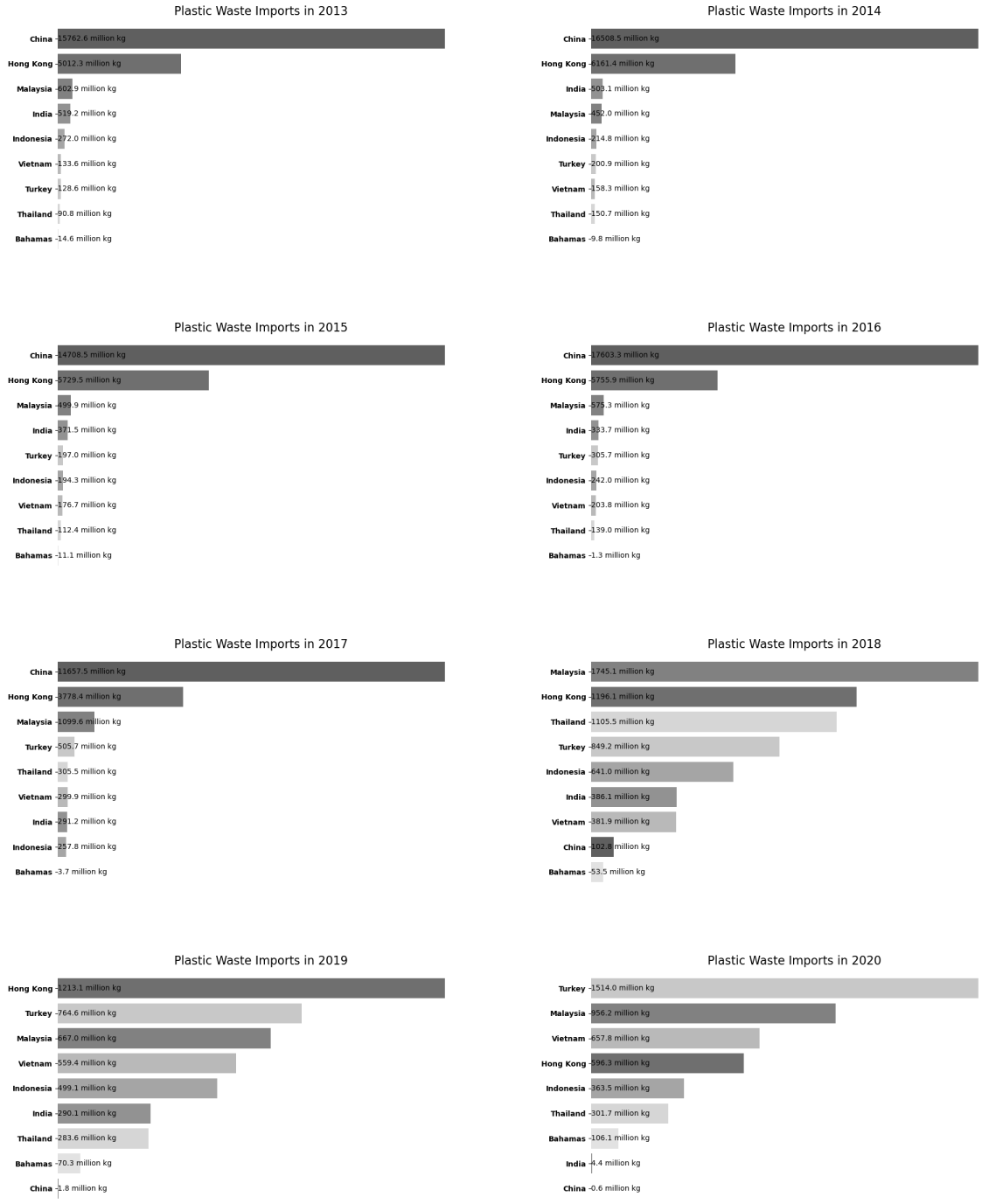
To examine this more systematically, we estimate a difference-in-differences specification:

$$\begin{aligned} \frac{Trade_{podt}}{\sum_{d'} Trade_{pod't}} &= \beta_1 Post_t * \text{Banned}_{po} * \text{CHN}_d + \beta_2 Post_t * \text{Banned}_{po} \\ &+ \alpha_{pod} + \alpha_{odt} + \alpha_{pdt} + e_{podt}, \end{aligned} \tag{2.1}$$

where the dependent variable is the share of exports of product p by origin country o to destination country d in year t. Banned_{po} indicates the set of China-banned plastic waste products and origin countries which had exported such products to China in 2015/2016,

[5]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.

FIGURE 2.1. Imports of Plastic Waste



Note: Each bar represents the percentage imports of plastic waste (in kg) by country in a given year.

i.e. before the implementation of ONS. The sample covers the years between 2013 to 2019. $Post_t$ takes on the value one for years after 2017, and zero otherwise. With the inclusion of product-origin-destination fixed effects (α_{pod}), we exploit variation within a triplet over time arising from the implementation of China's policy in 2017. Time-varying factors at the level of product-destination pairs origin-destination pairs with the inclusion of α_{pdt}, and α_{odt}.

TABLE 1. Change in Trade in Plastic Waste:

Dependent Variable: $\frac{Trade_{podt}}{\sum_{d'} Trade_{pod't}}$	(1)	(2)
$Post_t * \text{Banned}_{po} * \text{CHN}_d$	-0.173a	-0.173a
	(0.014)	(0.014)
$Post_t * \text{Banned}_{po}$	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 estimating equation 2.1, 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: $Post_t$*$Banned_{po}$*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 years 2013-2019. Letters indicate statistical significance: a indicates p<0.01, b indicates p<0.05, and c indicates p<0.10.

Results obtained from estimating equation (2.1) are presented in the first column of Table 1. The coefficient on the triple interaction term, which indicates whether the Chinese policy was binding, is estimated to be negative and statistically significant. Accordingly, existing exporters of plastic waste products subject to the ONS policy reduced their exports to China by 16 percent after 2017, relative to other products and origin countries. Evidence presented in the table also points to trade diversion: existing exporters of plastic waste products to China diverted their exports to other destinations after the introduction of the ONS policy, relative to other products. The second column presents results obtained from a

more stringent specification that also includes time-varying product-origin level fixed effects α_{pot}. The estimated coefficient on the triple interaction $Post_t * \text{Banned}_{po} * \text{CHN}_d$ remains robust to the inclusion of these additional fixed effects, providing further confidence in the effectiveness of the Chinese waste ban.[6]

China's decision to restrict waste imports had wide-ranging effects, including changes in waste management strategies in exporting countries, as discussed above. It highlighted the need for more sustainable waste management practices, improved recycling technologies, and international cooperation to address the challenges posed by waste trade. In the next section, we examine how this policy change affected firm-level decisions in a third country. We focus on Turkey as a case study for two reasons. First, Turkey became one of the main destinations for plastic waste. Second, comprehensive imports and waste data from Turkey allow us to directly examine the economic and environmental effects on firm-level production, input sourcing, and waste management practices.

3. Firm-Level Impacts of the Policy

We start our analysis by examining whether China's ban on the import of specific plastic waste products resulted in a redirection of these waste products to Turkey using detailed customs data. This is followed by an examination of the effects of the policy ban on domestic generators of plastic waste in Turkey. We also examine the performance of importers after the ban. We end with examining air pollution in ban-exposed provinces of Turkey.

To distinguish importing firms from domestic waste-generating firms, we index them with i and k respectively. Plastic waste is overwhelmingly imported by manufacturing firms that recycle them as inputs for further production. More than 70 percent of firms importing China-banned products belong to the manufacturing sector, and just 6% are waste management companies. Notably, less than 10 percent of the importers of China-banned products are themselves suppliers to waste management companies (see Figure A8 in the Appendix). This implies that the majority of imported waste is directly utilised as inputs by manufacturing firms rather than being processed by waste management companies, and we examine

[6]Figure A2 in Appendix shows no evidence of pre-trends in exports of treated products to China.

how access to imported waste affects the performance of importing firms in terms of sales and profitability.

The empirical analysis is based on three rich micro-level datasets from Turkey. Turkish Customs data provides information on annual exports and imports, disaggregated by firm, (destination/origin) country, and 8-digit Harmonized System (HS) product code. The customs data allow us to study the diversion of waste trade to Turkey after the ONS policy at a granular level. Firm registry and corporate financial statements contain firms' annual gross sales, material costs, and wage costs, as well as their employment, location (province), and industry of operation (4-digit NACE, the Statistical Classification of Economic Activities in the European Community, code). We augment this data with VAT declarations to identify domestic suppliers of the firms in our sample. We use this rich micro-level data to study the direct economic effects of the policy on waste importers. Finally, we use data on the production and management of domestically produced waste in the Turkish industry, as reported in the Manufacturing Industry Waste Statistics survey. This survey, conducted biennially, covers every formal firm with more than 50 employees, along with a representative sample of smaller firms. These firms are asked to report their annual waste production and the disposal methods employed (such as selling, reusing, dumping, burning, etc.), all categorized by waste type, classified according to the European Waste Classification for Statistics (EWC-Stat).[7]

3.1. **Importers of Banned Plastic Waste.** The trends presented in Figure 2.1 indicate a substantial increase in Turkish imports of plastic waste products after 2017. But they are based on raw data and thus do not account for time invariant factors at the product-origin level or changes in time varying country-specific factors that could correlate with the ban. To address this concern, we conduct a standard event study using the following specification:

$$\ln \text{Imports}_{pot} = \sum_{l=2013}^{2019} \beta_l D_t^l * \text{Banned}_p + \alpha_{po} + \alpha_{ot} + e_{pot}, \tag{3.1}$$

where Imports_{pot} denotes the value of imports of 8-digit HS product p from country o in year t. The sample covers the years from 2013 to 2019. We are interested in the estimates of β_l

[7]We then identify China-banned waste products using a crosswalk between EWC-Stat and HS6.

which measure the annual change in the imports of plastic waste products subject to the ONS policy in year l relative to the sample average at the level of product-origin country and controlling for time-varying factors affecting imports at the level of origin countries. Figure 3.1 presents point estimates for β_l, along with their 90% confidence intervals. The year preceding the Chinese ban on scrap and waste products, 2016, is excluded and serves as the reference year. In line with the pattern observed in Figure 2.1, Turkish imports of products banned under the ONS in 2017 significantly increased after 2016 and remained high until 2019.

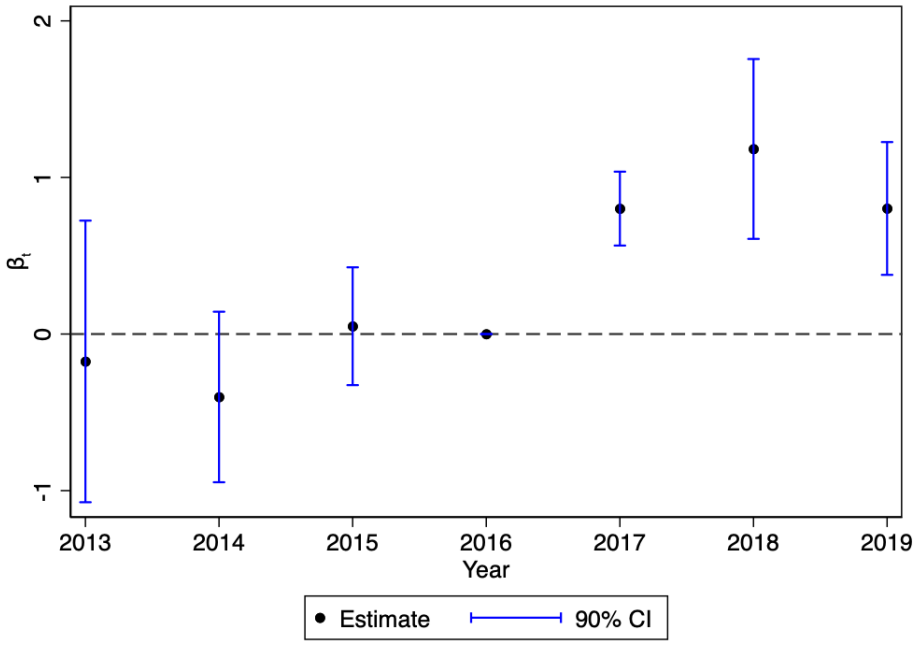
Figure A3 in the Appendix conducts the same analysis at a finer level of firm-product-origin country-time level to account for firm-specific demand shifts and shows that firms increased their imports of banned products after the ONS policy. Furthermore, Figure A4 in the Appendix decomposes the observed change in the value of imports into quantities and unit values. Accordingly, the estimates imply that the quantity of treated products increased (Panel A) after 2017 while their unit values declined (Panel B).[8] The ONS policy reduced global demand for the banned products. Prices for the banned products fell in international markets, making them more attractive to Turkish firms and driving up their import demand for the products banned by China.[9]

To mitigate concerns about potential bias in our findings stemming from the misclassification of products, we randomly allocate treatment status to 8-digit HS products within their corresponding 4-digit HS codes and then re-estimate Equation 3.1. This procedure is replicated 250 times. Figure A7 presents the the distribution of the estimates from the 250 simulations, which are clustered around zero, compared to our baseline estimate that is slightly higher than unity. This exercise lands credibility to our baseline findings, demonstrating that results are not driven by product misclassification following the ban.

[8]In Figure A5, we present evidence indicating a downward trend in quality-adjusted prices. This observation implies that the observed decrease in unit prices cannot be attributed to a reduction in quality.

[9]We also test whether Turkey only redirects these plastic waste products to other countries. Figure A6 shows no discernible increase in the exports of China-banned plastic waste products.

FIGURE 3.1. 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) value of imports of Turkish firm f of product p from origin o at year t. Coefficients on the interaction between year dummies D_t^l and an indicator for waste products banned by China $Banned_p$ are plotted in the figure. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

3.2. **Waste Management of Domestic Firms.** In the previous section we showed that following China's ban in 2017, firms in Turkey gained access to cheap plastic waste through imports. We now examine how domestic firms that generate plastic waste responded to the ban.

A representative survey of firms collects information on the annual quantity and type of waste that each firm generates using the EWC-Stat classification. Importantly, the survey asks about the waste management method of each firm, which includes selling waste, mismanaging waste, and re-using waste. Mismanaged waste encompasses waste that is either deposited into un-inspected storage facilities, discharged into rivers, streams, or lakes, dumped onto open land, or incinerated solely for disposal purposes rather than for energy

generation in controlled chambers. Re-used waste, on the other hand, consists of waste that is sold or recycled for the purpose of reuse.

We manually concord the EWC-Stat product codes to 6-digit HS codes to determine which domestic firms generate plastic by-products that were banned under China's ONS policy. We can therefore test whether domestic firms, facing more import competition due the Chinese policy, experienced a reduction in their waste sales by estimating the following specification:

$$\mathrm{X}_{kpt} = \sum_{l=2012}^{2020} \beta_l D_t^l * \mathrm{Banned}_p + \alpha_{kt} + \alpha_p + \epsilon_{kpt}. \tag{3.2}$$

The dependent variable is the share of firm k's volume of waste product p that is sold at time t. Banned_p is a dummy variable indicating whether waste product p is banned under China's ONS. We control for time varying firm-specific factors and time invariant product specific factors.

Estimates presented in Panel (A) of Figure 3.2 suggest a reduction in the share of plastic waste products banned by China in the total volume of waste sold by the firm. This is consistent with a fall in the demand for domestic waste as domestic firms gained access to cheaper imports. Panel (B) of Figure 3.2 shows heterogeneity in sales by firm type. The decline in domestic waste sales is more noticeable for small firms (with fewer than 250 employees) than for large ones. This suggests that larger firms were more able to compete with the heightened competition in recyclable waste compared to smaller firms.

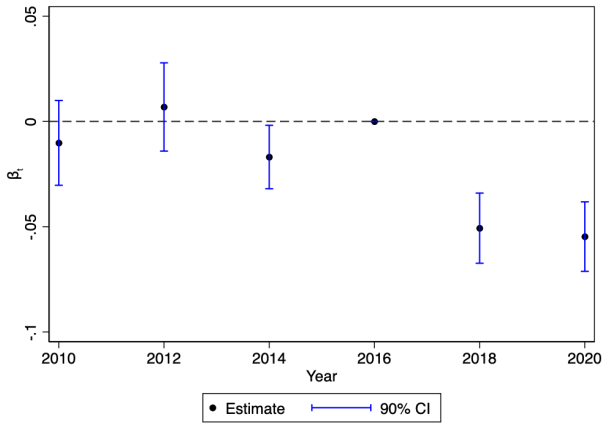
Domestic firms sold less of their waste products and this raises the question of what happened to the waste that they were unable to sell. To understand this, we change the dependent variable of Equation 3.2 to the share of firm k's volume of waste product p that is mismanaged or that is not re-used at time t.[10]

Figure 3.3 shows that following the China ban, waste products banned by China were more likely to be mismanaged and not re-used by Turkish waste generating firms. As before,

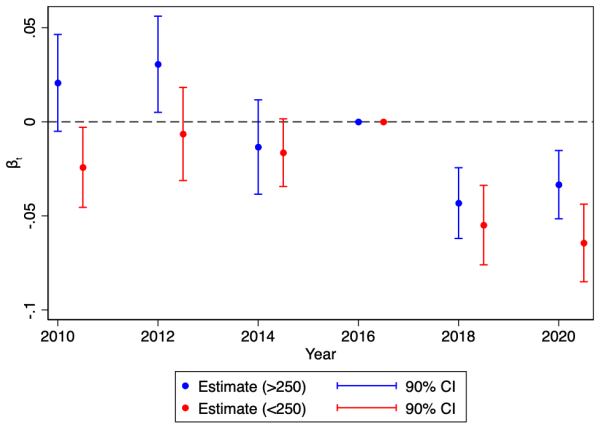
[10]The remainder category includes waste that is neither mismanaged nor reused. For example, plastic waste that is stored in an inspected facility is not categorised as mismanaged waste, and also not categorised as being re-used.

FIGURE 3.2. Event study: Waste Sales of Firms in Turkey

(A) Overall



(B) By Size



Note: These figures plot 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 firm k 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 $Banned_p$ are plotted in the figure. Panel (A) shows the estimates of the regression on all manufacturing firms, whereas Panel (B) presents the estimates of the same regression on firms greater than 250 employees (blue) and less than 250 employees (red) separately. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

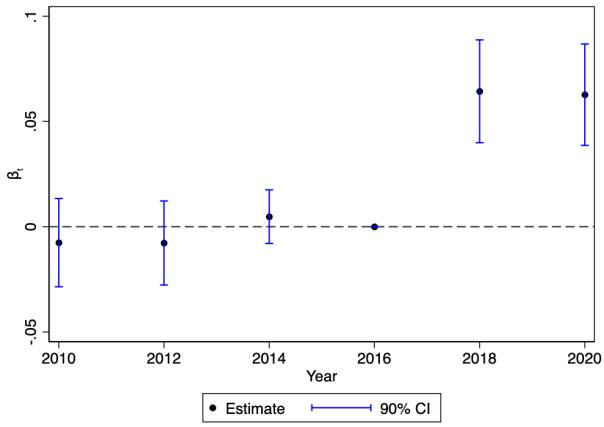
we also explore firm heterogeneity in the management of waste and find that the increase in mismanaged waste and not re-used waste is larger among smaller firms.

Overall, our results suggest that imported waste displaced domestic plastic waste. This can be explained by Turkish firms substituting towards higher quality and now cheaper waste sourced from abroad. Waste must undergo separation and cleaning for it to be recycled as an input. By international law, imported waste must already have undergone some amount of separation and cleaning before it is exported. It is therefore likely to be a better quality input that displaced lower quality domestic waste but did not have substantive displacement effects on higher quality domestic waste (Karasik (2022)).

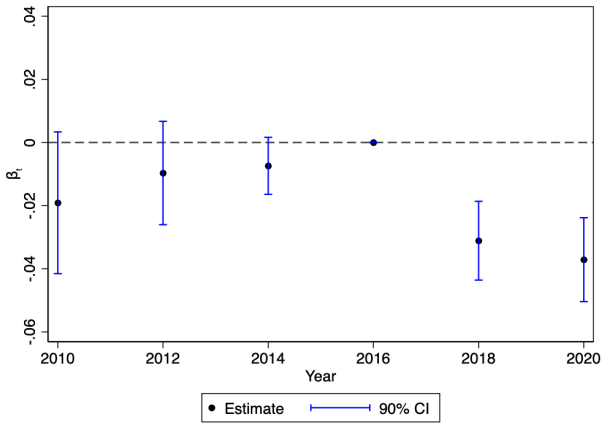
Our results so far suggest that following the China ban, domestic firms mismanaged a greater share of the waste they generated and re-used a smaller share of their waste. To ensure that these findings are not influenced by any changes in the total waste production (or the scale of waste generation) of these firms, we estimate the following specification:

FIGURE 3.3. Event study: Management of Waste by Firms in Turkey

(A) Mismanaged Waste



(B) Not re-used Waste



Note: These figures plot the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in 3.2. Each observation is at the firm-year level. The dependent variable changes across sub-figures as stated in the title. Coefficients on the interaction between year dummies D_t^l and an indicator for waste products banned by China $Banned_p$ are plotted in the figure. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

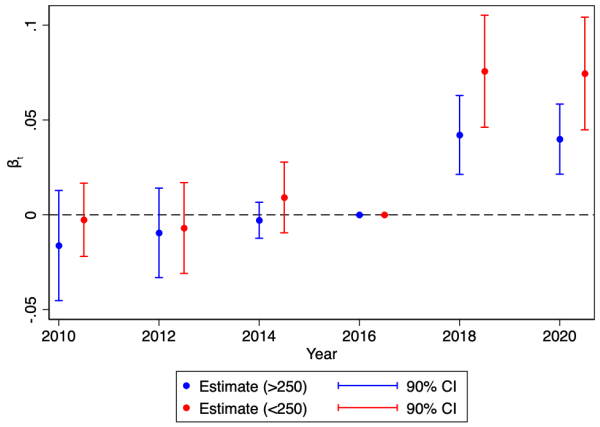
$$\text{Total waste}_{kt} = \sum_{l=2012}^{2020} \beta_l D_t^l * \text{Exposure}_k + \alpha_k + \alpha_{s(k)t} + \alpha_{r(k)t} + \epsilon_{kt} \tag{3.3}$$

where the dependent variable is the total amount of waste generated by firm k at time t. $Exposure_k$ is the share of China-banned waste products in the total waste generated by firm k before the ONS policy. The sector and region (province) of firm k are denoted by $s(k)$ and $r(k)$. Figure A9 plots the estimates, β_l, and there are no noticeable changes in the total waste generated by firms that were more intensive in ONS-banned waste products. Therefore, the main finding is a change in domestic waste mismanagement after the surge in import competition, and not a change in the scale of waste generation that could be driven by upstream plastic product demand.

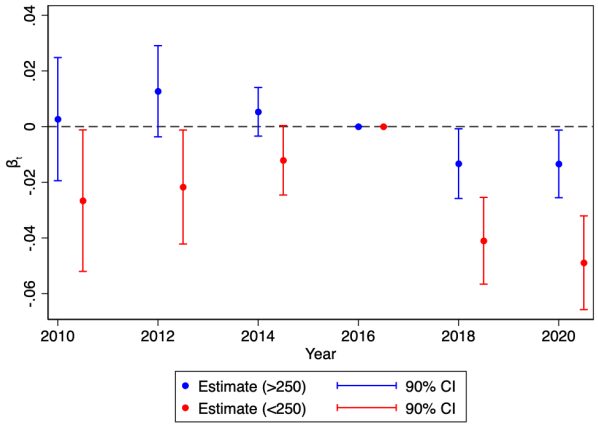
3.3. **Waste Sales of Domestic Firms.** Leveraging firm-product level waste management data, we observed a decline in the sales share of domestically-generated waste products that were banned by China, as illustrated in Figure 3.2. We now formally test for reduced

FIGURE 3.4. Event study: Management of Waste by Turkish Firms, By Firm Size

(A) Mismanaged Waste



(B) Not re-used Waste



Note: These figures plot the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in 3.2. Each observation is at the firm-year level. The dependent variable changes across sub-figures as stated in the title. Coefficients on the interaction between year dummies D_t^l and an indicator for waste products banned by China $Banned_p$ are plotted in the figure. The results are presented separately for two different samples of the data: (i) sample including firms with greater than 250 employees in blue, and (ii) sample including firms with less than 250 employees in red. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2013 to 2019.

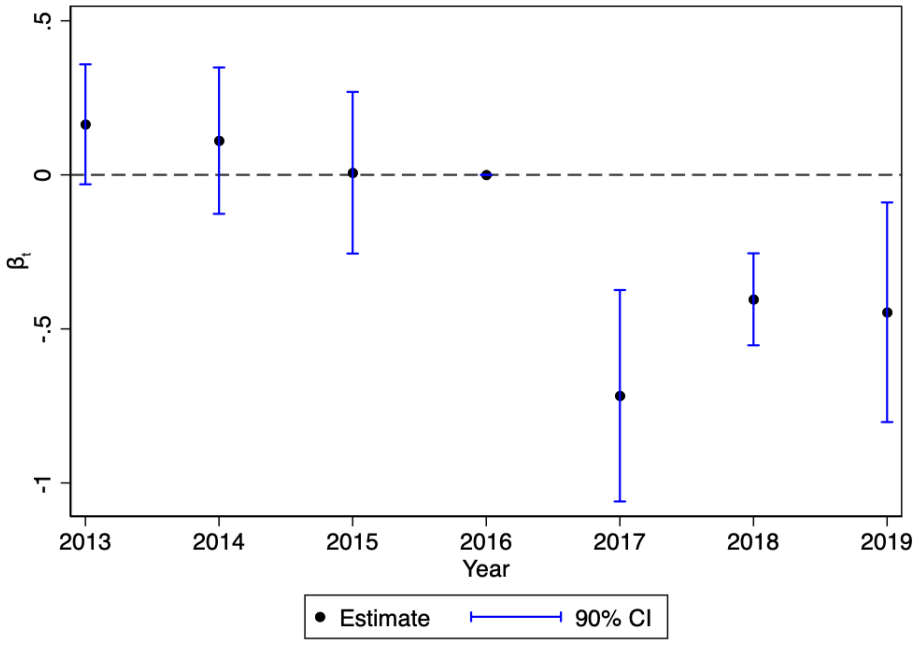
sales of domestic waste using firm-to-firm sales data, based on Turkish VAT declarations, by estimating the following specification:

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

The dependent variable is the logarithm of purchases by buyer firm i in the 4-digit NACE industry s from industry s' in year t. $Exposure_i$ is a continuous variable indicating the share of firm i's usage of China banned plastic products in its inputs in 2016, it is calculated as the share of ONS-banned plastic imports in total input costs of (wages + domestic purchases + imports). $Exposure_{s'}$ represents the share of plastic waste sold by industry s' in total plastic waste generated in Turkey in 2016. It serves as a continuous metric indicating the likelihood

of sector s' engaging in the sale of plastic waste. This measure is derived by aggregating data from the plastic waste management survey to determine the percentage of total plastic waste sold by each sector. We include firm-year fixed effects α_{it} which, among other factors, control for changes in firm scale and thus total input purchases over time. We also include firm-source industry and time-varying source-destination industry fixed effects. Therefore, identification comes from variation across supplying industries for a given buyer and year.

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 Turkey in 2016. The sample covers the years from 2013 to 2019.

The β_l coefficients, along with their 90% confidence intervals estimated from equation (3.4), are illustrated in Figure 3.5. These results suggest a significant shift of purchases away

from major domestic plastic waste producers following the ONS ban, indicating that Turkish buyers of plastic waste substituted domestic plastic waste inputs with imported varieties.

3.4. **Firm Performance of Importers.** Next, we investigate whether the importers of China-banned products benefited from the increased supply of imported plastic waste in terms of increased sales and profits. The majority of imported waste is directly utilised as inputs by manufacturing firms. We examine whether better access to material inputs from abroad impacted the economic performance of importing firms by estimating the following specification:

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

where X_{it} is the outcome of interest for firm i at time t such as sales, market share, and firm costs. $Exposure_i$ is a continuous variable indicating the share of firm i's usage of China banned plastic products in its inputs in 2016, and $Employment_i$ is the number of employees in firm i in 2016.[11] The specification controls for firm fixed effects and year fixed effects. The control group consists of importers of other products within the same 4-digit NACE industry as the importers of banned 8-digit HS plastic products.

First, we test whether reduced prices of imported plastic inputs led to a reduction in firms' costs, measured in terms of the ratio of material costs to sales or wages to sales. As illustrated in Figure 3.6, following China's ban, firms that relied more heavily on imports of China-banned products as inputs experienced a slight reduction in their expenditures on material inputs relative to their sales though the estimates are not very precisely estimated. Firms reduced their labour share in costs, likely because less sorting and processing was needed for imported plastic waste.[12]

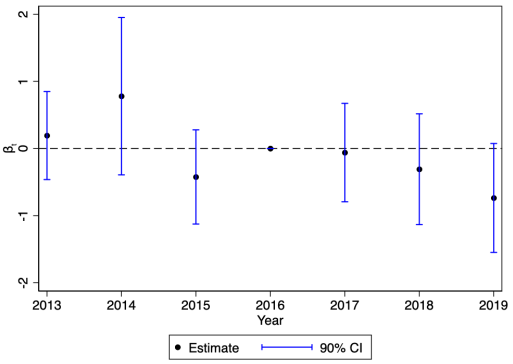
Our next test investigates whether firms that more intensively imported products that were banned by China experienced a differential change in their sales or profit margins. We

[11]Input costs are constructed as the sum of wage payments, purchases from domestic firms based on the VAT data, and imports.

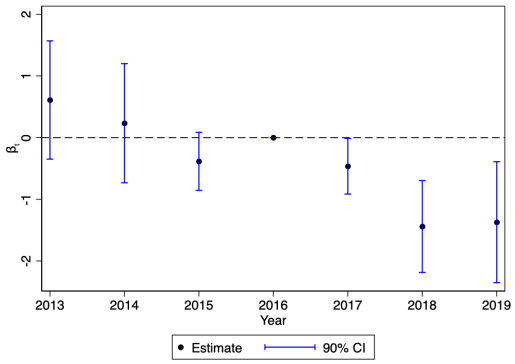
[12]This result is in line with Castro Vincenzi and Kleinman (2020), where they show causal evidence for the negative effect of materials prices on the labor share. Hummels, Jørgensen, Munch, and Xiang (2014) also shows that offshoring decreases low-skilled wages.

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 β_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.

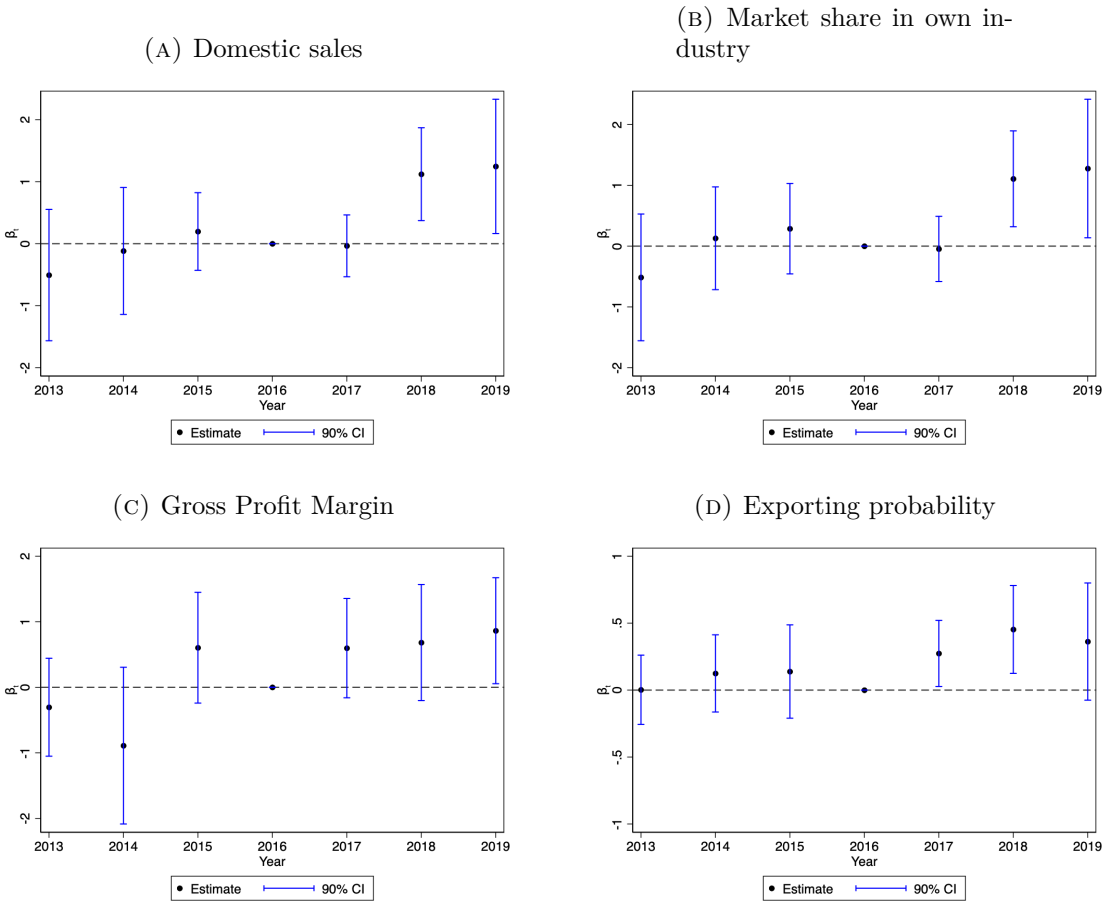
estimate the specification in equation (3.5) with domestic sales and industry (4-digit NACE category) market shares as dependent variables.[13] The top panels of Figure 3.7 plot the coefficients for domestic sales and industry market shares of more exposed firms, and show that they experienced relatively higher growth after the ban. The lower panels of Figure 3.7 examine their gross profit margins and probability of exporting (as a proxy for international competitiveness of firms). More exposed firms experienced an increase in their gross profit margin and to some degree, their export market participation, but these effects are not precisely estimated. These findings suggest that firms in Turkey benefited from the lower global price of plastic waste inputs from China's ONS policy.

4. Theoretical Framework and Quantification

To interpret the main empirical findings in light of the literature on trade and the environment, we generalise the canonical theoretical framework of Shapiro (2016) to waste as a pollutant. We focus on the minimal structure needed to explain the empirical findings and to

[13]Market share is defined as the ratio of firm's gross revenues to the sum of revenues in the firm's 4-digit NACE industry.

FIGURE 3.7. Effects of ONS on Importer-level Sales and Profits



Note: These figures plot the estimates of β_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 2013-2019.

provide a mapping from observable waste outcomes to an ex-post assessment of the welfare impacts of China's ONS policy in Turkey.[14] The section starts with describing production and waste choices of firms that use and supply waste and those that do neither. It then discusses welfare and environmental externalities from waste activities. Finally, it provides a mapping from waste imports to welfare to define the ONS policy. Details are relegated to the Appendix and the main findings are discussed here.

[14]We also report our calculations for global effects in Appendix D.

4.1. **Production and Trade.** There are three types of firms in the economy: plastic waste using firms u, plastic waste supplying firms s and firms that neither use nor supply plastic waste n. To fix ideas, a firm producing plastic traffic cones is a plastic waste-using firm and a firm producing medical syringes is a plastic waste supplying firm. In contrast, a carpentry firm that neither uses nor supplies plastic waste is not exposed through supply and use and denoted by n or referred to as a "neither" firm. They have zero exposure as measured by the exposure variables defined in the empirical section before. In other words, firm types are distinguished by their "netput" of plastic waste - using firms are net buyers of plastic waste, supplying firms are net sellers of plastic waste and non-exposed firms have zero net purchases/sales of plastic waste.

Waste Generation. Firms that use plastic in production of their main outputs generate plastic by-products, that can be treated to generate "managed" plastic waste. When the managed plastic waste is supplied to waste using firms, they can recycle it for use as an input into final production. For example, the plastic waste generated by the medical syringe producer is sold to the plastic cone producer to use as an input in cone production. Let a denote the waste treatment technology, that is familiar from the literature on pollution abatement. If a firm treats its by-products, it generates $\chi(x, a)$ units of managed waste per unit of virgin resource used in its main production process. We assume $\chi(x, a) = xa$ so that $0 \leq a \leq 1$ can be interpreted as the share of waste that is managed and has recyclable value. Treating waste requires workers to sort the by-products to separate out recyclable plastics and costs $w\gamma(a)$ where w is the wage rate and γ is an increasing and convex function. Better abatement technology costs more, and extracting more and more recyclables out of a given by-product requires more labour. Here we assume economies of scale in pollution abatement, but it is not necessary for the main results.

By-products that are mismanaged, such as through burning or open dumping, are more polluting than managed waste with emission rates $\xi^b > \xi^x$, that we will explain in more detail later. International law forbids sales of by-products, but allows trade in managed

waste material. Therefore, by-products stay in their own country while managed waste can be traded across countries.

Waste Recycling. Production in u needs labour and plastic which could be virgin material v or recycled plastic x. Let m denote the amount of material used in production in u. Then $m = m(v, x)$ and we assume that v and x are partly substitutable. This distinguishes waste from some other pollutants because recycling has an offsetting effect on pollution through conservation of virgin resources, that would otherwise need to be exploited and would contribute to environmental degradation.

Firm Decisions. Firms maximise profits, taking wages w_d and input prices as given. We begin with the decisions of supplying firms and then proceed to using firms and those neither supplying nor using plastic waste. The subscripts u, s, n denote the firm type and the subscripts d, d' denote countries.

Supplying firms s in country d use virgin plastic to earn revenues $R_{sd}(v_{sd})$. Virgin material costs z_{od} and this can be interpreted as the units of a freely traded world commodity, such as oil and natural gas from origin country o, needed to produce virgin plastic in country d. Then $z_{od} = z_d = \tau_d z$ where z is the world price of virgin resources and $\tau_d \geq 1$ is the iceberg transport cost in shipping from the world market to destination d.

Use of virgin material generates by-products. Management of these by-products results in recyclable waste that provides the supplying firms with revenues worth $\sum_{d'} r(x_{sdd'}) / \tau_{dd'}$ when sold to firms in country d' (where $\tau_{dd'} \geq 1$ is the usual iceberg transport cost that is set to 1 when $d' = d$). Firm chooses whether to pay $w_d \gamma(a_{sd})$ for the waste management technology. Having paid this, $x_{sd} a_{sd}$ units of managed waste are available to be recycled. λ_{sd} is the Lagrange multiplier on the constraint that recyclable waste demand cannot exceed the supply of managed waste.

Using firms u in country d choose labour and material (that could be virgin and/or recyclable). Plastic waste from origin country o used by u is denoted by x_{uod}, and it comes at a market price of p_{od}. Other firms n that neither buy nor sell plastic waste produce final products with labour, taking wages as given.

Let h index varieties, such as the HS code for plastic waste products. It will be convenient to refer to products, indexed by h, because the China ban applies to specific products within plastic that are more polluting or contaminated. Assume without loss of generality that ξ_h^x is increasing in h. China's ONS policy bans imports of $h > \bar{h}$ and therefore removes higher disutility imported waste destined for recycling in China.

The profit maximisation problem of each firm is summarised below:

$$\max_{l,m,v,x} \Pi_{ud} = R_u\left(l_{ud}, m_{ud}\right) - w_d l_{ud} - \sum_h \sum_o z_d v_{huod} - \sum_h \sum_o p_{hod} x_{huod} + \lambda_{ud}\left(m\left(v_{huod}, x_{huod}\right) - m_{ud}\right)$$

$$\begin{aligned}\max_{a,v,x} \Pi_{sd} =& R_s\left(v_{hsod}\right) - \sum_h \sum_o z_d v_{hsod} + \sum_h \sum_{d'} r_{sdd'}\left(x_{hsdd'}\right)/\tau_{hdd'} - \sum_h w_d \gamma\left(a_{hsd}\right) \sum_o v_{hsod} \\ &+ \sum_h \lambda_{hsd}\left(x_{hsd} a_{hsd} \sum_o v_{hsod} - \sum_{d'} x_{hsdd'}\right)\end{aligned}$$

$$\max_l \Pi_{nd} = R_n\left(l_{nd}\right) - w_d l_{nd}$$

Details are in the Appendix and the key observations are summarised below. Firms choose the share of waste to manage. A first observation is that optimal waste management rises with waste generation and falls with the sorting costs. This follows from the first-order condition (FOC) for optimal waste management choice: $r'_{sdd'} x_{hsd}/\tau_{hdd'} - w_d \gamma'\left(a_{hsd}\right) = 0$ and the second order conditions ensuring a concave profit maximisation problem (that marginal revenues are declining and the abatement cost function is convex). Given all else equal, the FOC implies a is a decreasing function of $w\tau/r'x$ because $\gamma'' > 0$, and we summarize it below.

Remark. *1.* *When $\gamma'' > 0$ and given all else equal, the share of waste that is managed and can be recycled, a_{sd}, rises with the scale of waste generated x_{sd} and falls with waste sorting costs w_d.*

A second observation is that waste trade follows gravity, as documented in the literature and in our empirical specification earlier. Under a power function for marginal revenues and a distance elasticity of -1, it can be shown that the imports of waste into destination d fall with the distance from the origin country The optimal choice of waste supply from o to d is

determined by the FOC: $r'(x_{hsod}) = \tau_{hod} w_o \gamma'(a_{hso}) / x_{hso}$. Dividing by equilibrium exports to China gives a form similar to gravity when τ is interpreted as a measure of distance:

$$\frac{r'(x_{hsod})}{r'(x_{hsoc})} = \frac{\tau_{hod}}{\tau_{hoc}}.$$

Summing across origins and products, waste imports of banned products into destination d are

$$x_d \equiv \sum_{h>\bar{h}} \sum_o x_{hsod} = \sum_{h>\bar{h}} \sum_o r'^{-1} \left(\frac{\tau_{hod}}{\tau_{hoc}} r'(x_{hsoc}) \right)$$

Let g_{od} denote the inverse of geographical distance between o and d and assume an elasticity of trade to distance of -1. Under a power function for marginal revenue $r'(x) \propto x^\eta$ (for $\eta < 1$), we get the gravity relationship:

$$x_d = \sum_o r'^{-1} \left(\frac{\tau_{od}}{\tau_{oc}} r'(x_{soc}) \right) = \sum_o \frac{g_{od}}{g_{oc}} x_{soc}.$$

Another insight from this relationship is that China's import ban would have larger impacts in third country destinations that are relatively closer to countries that initially exported larger volumes to China. We summarise this below and draw on it later to estimate a first stage equation to determine which countries received more of the displaced waste imports after the ban, and to infer the environmental burden of waste across different destinations for which we do not have direct evidence of waste management. While we focus on trade costs arising from geographical distance and bilateral tariffs in our specifications, a broader interpretation would include bilateral tariffs or destination-specific costs and taxes that have been the subject of research to disentangle pollution haven motivations from cost efficiency of locations.

Remark. 2. *Let g_{od} denote the inverse of geographical distance between o and d. Under a distance elasticity of -1 and a power function for marginal revenue $r'(x)$, equilibrium waste*

imports into destination d take a gravity form:

$$x_d = \sum_o r'^{-1}\left(\frac{\tau_{od}}{\tau_{oc}} r'\left(x_{soc}\right)\right) = \sum_o \frac{g_{od}}{g_{oc}} x_{soc}. \tag{4.1}$$

The profit maximisation problems, together with market clearing of products and factors, describes the economy as we discuss next.

4.1.1. *Market Equilibrium.* Give revenues and input prices $(z_d, w_d, p_{hod}, R_{ud}, R_{sd}, R_{nd})$, firms choose $(l_{ud}, v_{huod}, x_{huod}, a_{hsd}, v_{hsod}, x_{hsdd'}, \lambda_{hsd}, l_{nd})$. The market equilibrium is determined by the profit maximisation FOCs for these choices, given factor prices, revenue functions and the by-product function. In a market equilibrium, virgin resource prices are fixed by arbitrage and free trade. Let z denote the global energy price and τ_d denote the cost of shipping energy from a Walrasian global market to d. Because virgin resources are traded in a global market, the price of shipping it from o to d is $z_{od} = z_d = \tau_d z$. Wages w are determined by labour market clearing:

$$0 = l_{ud} + l_{nd} + l_{sd}. \tag{4.2}$$

Waste prices p_{iod} are determined by market clearing of demand and supply of using and supplying firms $x_{huod} = x_{hsod}$. Finally, factor incomes I equal expenditures Y on final consumption in the economy. And, these in turn, equal the profits and earnings of all agents (where domestic transfers such as wages get cancelled out). Let $\bar{V}_d$ denote the country's endowment of virgin resources (such as oil or natural gas) that can be directly converted to plastic. The national income identity is:

$$I_d = Y_{sd} + Y_{ud} + Y_{nd} \tag{4.3}$$

$$= R_{sd} + R_{ud} + R_{nd} + z\bar{V}_d \tag{4.4}$$

$$+ \sum_h \left(\sum_{d' \neq d} p_{hdd'} x_{hsdd'} - \sum_{o \neq d} z_d v_{huod} - \sum_{o \neq d} z_d v_{hsod} - \sum_{o \neq d} p_{hod} x_{huod} \right) \tag{4.5}$$

Together, the FOCs and the resource clearing conditions provide solutions to the unknowns in production, the factor prices and the income of the representative consumer, given demand for domestic and imported outputs summarised by R and Y.

Before proceeding to the welfare analysis, it is worth noting that the revenue functions above are defined over factors of production that appear in the empirical findings, but they can be generalised to make them symmetric across supplying, using and doing neither firms. This will add more factor choices without changing the mapping to moments that we need to calibrate the ex post assessment, as discussed in the Appendix. We show moreover that the economies of scale over by-products in abatement technologies is not necessary for the quantification exercise and can be altered to the context under consideration.

4.2. **Welfare and Policy.** The previous subsection takes the revenue functions as given and then determines firm decisions. We now generalise the usual formulation of welfare in the trade and environment literature to close the model with a specification of the demand side that determine the revenue functions. Then we specify the ONS policy and discuss the steps to quantify its economic and environmental impacts.

4.2.1. *Welfare.* The main departure from the standard welfare form is that we need to account for waste instead of other pollution-generating activities. Waste recycling reduces the use of virgin materials, which would otherwise generate pollution, such as emissions from resource extraction and use. Waste therefore differs from other pollution-generating activities that cannot be recycled for further production. For example, shipping emissions generated during transport cannot typically be recycled to offset virgin energy use. To account for this aspect, we consider three channels through which waste affects welfare:

(A) **Waste mismanagement** externality from waste generators' choices at home (e.g. Turkish firms dump or burn their own waste),

(B) **Waste recycling** externality from more processing of waste in the country (e.g. Turkey and China import waste that gets recycled within the country and produces emissions during processing or landfilling of the remainder)

(C) **Virgin resource** (offsetting) externality from less virgin material being used when waste is recycled (e.g. Turkey gets access to cheaper plastic waste imports and does not need as much virgin plastic for industrial production).

To formalise these features, a representative consumer in country d gets utility from consumption of goods produced by each firm. She faces externalities from pollution that lower her utility through extraction of virgin resources (such as fossil fuel pollution that generates disutility at a rate ξ^v) and from pollution generated by plastic waste (such as through marine pollution and greenhouse gas emissions that generates disutility at a rate ξ^x).

Under linear utility, welfare can be summarised in Welfare W1 below:

Welfare W1.

$$\begin{aligned} W_d \equiv & W\left(U_d, S_d, N_d, -V_d - Z_d\right) \\ V_d \equiv & \sum_h \xi_h^v \sum_{d'} \left(v_{hudd'} + v_{hsdd'}\right) \\ Z_d \equiv & \sum_h \left(\xi_h^b \left(1 - a_{hsd}\right) x_{hsd} + \sum_o \xi_h^x x_{huod}\right) \end{aligned}$$

where W is increasing in final consumption of goods U, S, N produced by firms u, s and n, and welfare maximisation generates demand relationships that firms take as given. W is decreasing in virgin resource extraction V and waste pollution Z.[15] Waste pollution arises from waste mismanagement at home and from waste recycling operations at home. By definition, mismanaged waste generates more disutility than managed waste $\xi_h^b > \xi_h^x$ as explained before.

We are interested in quantifying the welfare effects of China's Operation National Sword policy. Assume without loss of generality that ξ_h^x is increasing in h. China's ONS policy bans imports of $h > \bar{h}$ and therefore removes higher disutility imported waste destined for recycling in China. Let $\Delta x_{huoc} \equiv x_{huoc}\left(\tau_{hoc}^{ONS}\right) - x_{huoc}\left(\tau_{hoc}\right)$ denote the change in imports into China, indexed by c. Then τ_{hoc}^{ONS} is prohibitively high for $h > \bar{h}$ and for all $o \neq c$ after the ONS policy in 2017. Imports are zero under a prohibitive tariff, and the ban is

[15] Only virgin resources that are extracted within the country directly feature in the welfare function, though this can be extended to account for global virgin resource extraction in welfare.

defined as a shift to an equilibrium with a prohibitive price for the banned waste imports, $\Delta x_{huoc} = -x_{huoc}$.

We are interested in obtaining an ex post assessment of the welfare impacts of the ONS policy. To quantify the welfare effects of the China policy, we need to determine changes in each component of the welfare function: expenditures $P_u U + P_s S + P_n N$, managed and mismanaged waste $ax, (1-a)x$ and virgin resource use v. We also need the disutility rates associated with environmental pollution,$\xi_h^b, \xi_h^x, \xi_h^v$, and we discuss this in the next section when presenting the results. Finally, to evaluate consumer surplus impacts, we would also need marginal utilities of consumption. We consider each component in turn in this section and summarise the relationships - theoretical and empirical - underlying the determination of each component.

4.2.2. *Revenue Impacts.* From the national income identity in equation 4.3, aggregate changes in consumption equal aggregate changes in revenues. Aggregate expenditure impacts (through $P_u U + P_s S + P_n N$) can therefore be obtained from the revenue side, where we have firm level information on sales. The event studies have already demonstrated the sales impacts of the policy on firms that use or supply plastic waste in Turkey, relative to firms that neither use nor supply plastic waste in sections 3.3 and 3.4. We now formalise this in line with the theoretical framework to determine the relative changes in revenue for using and supplying firms. Utilising firm-level data, we estimate a difference-in-differences specification:

$$\ln(Revenue_{f(j)t}) = \alpha + \beta_u^R Post_t * Exposure_f^u + \beta_s^R Post_t * Exposure_f^s + \gamma_{jt} + \gamma_f + \epsilon_{ft}, \tag{4.6}$$

where $Revenue_{f(j)t}$ is the total revenue (or the export revenue) of firm f of type $j \in u, s, n$, $Exposure_f^u$ captures firm f's exposure to the China policy through its use of plastic waste u, measured as the share of plastic waste imports in total input costs in 2016. Similarly, $Exposure_f^s$ captures the firm's exposure through its supply of plastic waste s, measured as the share of plastic waste sales in total sales in 2016. The remainder category is firms that are exposed neither through the use nor the supply of plastic waste, and the coefficients

β_u^R, β_s^R estimate the sales effects of the China can for using and supplying firms, relative to non-exposed firms.

We can also estimate a similar equation for employment of firms:

$$\ln(Employment_{f(j)t}) = \alpha + \beta_u^E Post_t * Exposure_f^u + \beta_s^E Post_t * Exposure_f^s + \gamma_{jt} + \gamma_f + \epsilon_{ft}, \tag{4.7}$$

where the coefficients β_u^E, β_s^E again estimate the effects of the China ban on employment in using and supplying firms, relative to non-exposed firms. From labour market clearing in equation 4.2, the sum of changes in employment across all firms is zero, and we therefore have a relationship to determine the employment changes in absolute terms. Having inferred the employment changes and given a revenue function, we can also arrive at the absolute change in revenues and trade values among non-exposed firms. Once we have the absolute change for non-exposed firms, absolute changes in revenues and exports of supplying and using firms can be determined from the relative revenue effects estimated through equation 4.6 and the initial values for exposure of firms through supply and use of plastic waste.

4.2.3. *Trade Impacts.* Having obtained the revenue changes in absolute terms, we proceed to determining the trade impacts which in turn will enable us to quantify managed waste impacts.

From Remark 2 and the resulting gravity relationship for trade in equation 4.1, we estimate a difference-in-differences specification to isolate the effect of the ONS policy by comparing the changes over time between products that were affected by the policy and those that were not. The gravity estimating equation is:

$$\begin{aligned} \ln x_{pct} &= \sum_{l \in T} \sum_{c \in D} \beta_x^c \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{l = T\} \\ &+ \sum_{l \in T} \sum_{c \in D} \beta_x^{c,HS2} \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{l = T\} \\ &+ \sum_{l \in T} \mathbb{1}\{t \geq 2017\} * \text{Tariff Rate}_{pc,t=2016} * \mathbb{1}\{l = T\} + \alpha_{ct} + \sum_{l \in T} \alpha_{pcl} + \epsilon_{pct}, \end{aligned} \tag{4.8}$$

where the set of destination countries is $D = \{\text{China}, \text{Exporters}, \text{Turkey}, \text{Rest of the World}\}$, and trade is $T = \{\text{Imports}, \text{Exports}\}$. To account for potential spillover effects over similar products, we control for treatment status at the 2-digit HS product level. Banned products are defined at the 6-digit level and similar products are defined as the 2-digit sectors to which these banned products belong. For example, China banned plastic waste imports of polymers of vinyl chloride, with the HS code 391530. The banned product in this example is 391530 and similar products consist of all products in the 2-digit code 39 that covers plastics and articles thereof. The slope coefficients take the usual interpretation of the effect of the China policy on the relevant trade flow of banned and similar products for each country group, relative to that trade flow for other products. The ONS policy can also affect the non-plastic other products through general equilibrium effects, and we now discuss how these can be recovered from the gravity slope coefficients and the balance of trade conditions for each country.

The equation is the same as a standard gravity equation, except we have pooled imports and exports to be able to estimate country-time effects that are common to exports and imports. Had the country-time fixed effects been estimated separately for exports and imports, the specification would have been equivalent to estimating two separate regressions with exports and imports as dependent variables in the gravity equations. The reason for estimating common country-time fixed effects is that we are interested in determining the absolute effects on trade, but these are only identified together through balanced trade conditions that are defined over net imports (and not separately for gross exports and gross imports). Identifying separate country-time fixed effects for gross exports and gross imports would therefore require stronger assumptions on the economic structure or functional forms, and national income accounting would not suffice to identify the absolute net import effects.

For generality, we do not impose further economic structure but instead show in the Appendix that estimating separate country-time fixed effects makes very little difference to the estimated slope coefficients and hence the results are not sensitive to this empirical feature.[16]

To infer theoretically consistent absolute impacts, the national income identity of equation 4.3 can be written in terms of balanced trade in each country. We define net exports of waste of country d as $E_d \equiv R_{jd} - Y_{jd} + \sum_i \left(\sum_{o \neq d} p_{iod} x_{iuod} - \sum_{d' \neq d} p_{idd'} x_{isdd'} \right)$ and net exports of virgin resources as $V_d \equiv z\bar{V}_d - \sum_{o \neq d} z_d v_{iuod} - \sum_{o \neq d} z_d v_{isod}$. Let $E_{ud} \equiv R_{ud} - Y_{ud}$ denote net exports of using firms u in country d and similarly for E_{sd} for supplying firms and E_{nd} for non-exposed firms. Balanced trade in country d can be written in net exports and net imports notation and in changes, it is $\Delta X_d + \Delta E_{ud} + \Delta E_{sd} + \Delta E_{nd} + \Delta V_d = 0$.

To operationalise it empirically, let D denote a country (or country group) - Turkey, China, Exporters and the Rest of the World (RoW). $E^D_{HS6} \equiv X_D$ denotes country D's net exports of the 6-digit plastic waste products banned by China, $E^D_{HS2} \equiv E_{sD} + E_{uD}$ is country D's net exports of plastic products, and E^D_n is country D's net exports of all other products. For each country group, the BoT is as follows:

$$0 = \Delta E^D_{HS6} + \Delta E^D_{HS2} + \Delta E^D_n + \Delta E^D_{virgin}$$

Substituting for equation D.1, the BoT for each country in terms of the difference-in-differences slope coefficients is:

[16] One way of inferring the absolute impacts on trade would be directly from the country-time fixed effects after the ban. But the country-time fixed effects would include other reasons for changes in trade and not just those from the China ban.

$$
\begin{aligned}
0 =& \Delta E^D_{HS6} + \Delta E^D_{HS2} + \Delta E^D_n \\
=& \underbrace{(e^{\beta^D_x+\alpha_D}-1)}_{\text{\% change banned exports}} Export^D_{HS6} - \underbrace{(e^{\beta^D_m+\alpha_D}-1)}_{\text{\% change banned imports}} Import^D_{HS6} \\
&+ \underbrace{(e^{\beta^{D,HS2}_x+\alpha_D}-1)}_{\text{\% change plastic exports}} Export^D_{HS2} - \underbrace{(e^{\beta^{D,HS2}_m+\alpha_D}-1)}_{\text{\% change plastic imports}} Import^D_{HS2} \\
&+ \underbrace{(e^{\alpha_D}-1)}_{\text{\% change neither exports}} Export^D_n - \underbrace{(e^{\alpha_D}-1)}_{\text{\% change neither imports}} Import^D_n \\
&\underbrace{\mu_D * (e^{\beta^D_x+\alpha_D}-1)}_{\text{\% change virgin exports}} Export^D_{virgin} - \underbrace{\mu_D(e^{\beta^D_m+\alpha_D}-1)}_{\text{\% change virgin imports}} Import^D_{HS6}
\end{aligned}
$$

where α_D is the log change of non-plastic other products due to the ONS (that is unknown) and μ_D summarises the elasticity between net imports of plastic waste and virgin resources in destination D, that we estimate for Turkey. In section 3, we estimated the impacts on inputs and outputs of waste using firms in equation 3.5 by exposure of firms using products banned by the ONS policy after the ban. When we focus only on plastic material inputs of using firms (other than plastic waste) as a dependent variable in equation 3.5, the effects are negligible and statistically insignificant.[17] We therefore determine $\mu_{Turkey} = 0$ and solve for balanced trade to obtain α_{Turkey}.

In the Appendix, we also show that the balanced trade conditions for other countries can also be solved to determine (α_{China}, $\alpha_{Exporter}$, α_{RoW}). It is worth noting that one BoT equation is linearly dependent, but because we have used domestic labour market clearing in Turkey, we in fact have a fourth equation to solve for the absolute changes in country-level imports and exports for non-plastic products due to the ONS policy (α_D). Having done so, we can compute the logarithmic change in trade of banned products for each country D as $\ln\left(e^{\beta^D_T+\alpha_D}\right)$ for $T = m, x$.[18]

[17]The elasticity between plastic waste imports and virgin plastic is theoretically $\mu \equiv xdv/vdx$ which can be obtained by totally differentiating FOC (2) in the Appendix with respect to x, taking z as the numeraire.
[18]For China, the value is -0.1 based on the literature and for Exporters and RoW, we also set the value to -0.1 (and confirm that virgin plastic use has an elasticity of -0.1 with respect to country exposure to the ban after the ONS policy in the Appendix).

4.2.4. *Waste and Virgin Resource Impacts.* The remaining outcomes that we need to determine welfare effects are changes in managed waste, mismanaged waste and virgin resources from the ONS policy, that we discuss next. Changes in output of s firms translate into changes in supply of plastic waste generated by them x_{hsD}. This can be in the form of managed waste that is sold to users a_{hsD} or it can be mismanaged waste $1 - a_{hsd}$, and we determine each separately.

Waste mismanagement impacts for Turkish firms are estimated directly in equation 3.2. Estimation of equation 3.3 shows moreover that total waste sales of domestic firms do not rise. Therefore, domestic firms mismanage the same waste volumes more after the ban. Putting these findings together, we infer that the ban increased the volume of managed waste from imported sources and increased the share of mismanaged waste in Turkey, but it did not increase the volume of waste generated at home nor reduce the use of virgin plastics.

With the environmentally relevant outcomes determined, we proceed to the economic and environmental effects arising from the ONS policy in the next section.

5. Economic and Environmental Impacts of the Policy

This section provides an ex-post evaluation of the economic and environmental effects of China's waste import ban in turn.

5.1. **Economic Impacts.** We begin with the impacts of the policy on revenues and employment in Turkey. Firm-level difference-in-differences estimates of the revenue and employment impacts on firms exposed through supply or use of plastic waste provides the economic effects, relative to non-exposed firms in the same industry. Results for these relative effects are presented in column 1 of Table 2.

As earlier, we find exposure through usage of waste increases sales of using firms while exposure through waste supply reduces sales of supplying firms in column 1. The findings need to be interpreted relative to firms classified as "neither using nor supplying" plastic, denoted by n. It is important to consider however that n firms could also be affected by the policy through indirect general equilibrium effects.

TABLE 2. Effects on Revenues

	(1)	(2)
	$\ln(Sales)$	$\ln(Employment)$
$Post_t * Exposure_f^u$	0.170a	0.107a
	(0.0238)	(0.0106)
$Post_t * Exposure_f^s$	-0.132b	0.00579
	(0.0551)	(0.0444)
$Post_t *$ Initial Employment$_f$	0.122a	0.0727a
	(0.00357)	(0.00171)
N	618907	662989
R^2	0.803	0.881
Fixed Effects:		
Firm	Yes	Yes
Industry × Time	Yes	Yes

Note: This table shows the results from estimating equation (4.6), where the dependent variable for the first column is the log sales in dollars of firm f, and the dependent variable for the second column is the log employment of firm f. $Exposure_f^u$ is the share of waste imports in the total input cost of firm f (wages + domestic purchases + imports), $Exposure_f^s$ is the share of waste sales in total sales of firm f, and $Post_t$ is a dummy variable indicating 1 if year is greater than 2017. The sample covers the years between 2013 to 2019. Letters indicate statistical significance: a indicates p<0.01, b indicates p<0.05, and c indicates p<0.10.

To assess these potential general equilibrium effects, we estimate the difference-in-differences (equation(4.6)) with employment as the dependent variable instead in Column 2 of Table 2. The effects are positive through exposure in the use of plastic waste and negligible through supply exposure, relative to the non-exposed firms that form the reference group. The labour market impact of these n firms, denoted as β_{ln}, can be inferred from the labour market clearing condition evaluated at the mean exposure and initial labor shares in the economy.

The labor market clearing condition in changes gives:

$$
\begin{aligned}
0 &= \Delta l_u^{TR} + \Delta l_s^{TR} + \Delta l_n^{TR} \\
&= (\beta_{lu} \cdot Exposure_u + \beta_{ln}) \left(\frac{l_{ud}}{L_d}\right) + (\beta_{ls} \cdot Exposure_s + \beta_{ln}) \left(\frac{l_{sd}}{L_d}\right) + \beta_{ln} \left(\frac{l_n}{L}\right) \\
&= \beta_{lu} \cdot Exposure_u \left(\frac{l_u}{L}\right) + \beta_{ls} \cdot Exposure_s \left(\frac{l_s}{L}\right) + \beta_{ln}
\end{aligned}
$$

$$= 0.107 \times 0.008 \times 0.0429 + 0 \times 0.064 \times 0.2194 + \beta_{ln}$$

$$= 0.0008\% + \beta_{ln}.$$

Under a standard assumption that revenues of n firms are increasing in their labour inputs, $R'_n(l_n) > 0$, we can infer that revenues of non-exposed firms are also not impacted by the China policy. Under an alternative assumption of Cobb-Douglas production (or a first order approximation to production changes) and constant markups, revenues change by $\Delta R_n = 0.21 \times \Delta l_n \approx 0$ because labour costs make up 21 percent of total costs for n firms in Turkey. We have therefore pinned down the revenue effect of n firms as negligible.[19]

We can now go back to the relative revenue effects from equation 4.6 to arrive at the absolute aggregate revenue impact as:

$$\begin{aligned}
\Delta R =& \Delta R_u + \Delta R_s + \Delta R_n \\
=& \beta_u Exposure_u \left(R_u/R\right) R + \beta_s Exposure_s \left(R_s/R\right) R + \beta_n R \\
=& \left(0.170 \times 0.0002 \times 0.0048 - 0.132 \times 0.0045 \times 0.4085 + 0\right) R_d \\
=& -0.02\% \times \$437.9 \text{ bn.} = -\$106.18 \text{ mn.}
\end{aligned}$$

where the second line follows from non-exposed firms by definition having zero exposure directly to the ban. We find that using firms gain from the ban. Their revenues rise as they gain access to cheaper imported inputs, but they account for a small share of the Turkish economy, and thus the displacement of domestic waste generated by supplying firms instead dominates. They lose their waste income to import competition, leading to revenue losses in the aggregate. While their revenue from waste falls, supplying firms do not cut back on overall employment. This is because by-products are not their main source of economic activity, and their main output is not substantially affected by reorganisation in the world

[19]Because the production impacts are negligible for n firms, it also implies that a generalised production function where n firms also pollute directly would not alter our main findings. More generally, the welfare function could be extended to include $v_{hndd'}$ to allow for emissions from n firms that get affected when they change their inputs and production. In our setting, this channel is not empirically operational because non-exposed firms do not alter their inputs and production in response to the China ban.

plastic waste market. In other words, medical syringe producers in Turkey lose some income from sales of plastic waste to plastic cone producers but their syringe business is not much affected by the ban.

Replacing total revenues with export revenues as the dependent variable, we find negligible relative effects across using and supplying firms.[20] Combined with negligible total revenue effects for non-exposed firms, we infer that export revenues did not change for any firms. Earlier we had seen some evidence for a rise in the exporting probability of using firms, but this does not result in a noticeable rise in the value of exports, perhaps because those who do start to export are small in size.

To sum up, using a difference-in-differences specification, and recovering the absolute effects from labor market clearing, we infer that the ONS policy decreased aggregate revenues by $106.18 million in Turkey. This number can be contextualised by comparing it with the influx of imports after China's ban. Turkey received an extra $205 to $295 million of plastic waste imports annually after 2017, implying less than full displacement of domestic sales from import competition.[21] It is worth reiterating that the drop in domestic waste sales does not translate directly into gains for consumer surplus. Waste is sold to using firms that now import more from abroad and their revenues rise. Inferring the consumption gains therefore requires additional information on prices or quantities sold by using firms to domestic consumers and demand side parameters to determine changes in consumer surplus at home.

5.2. **Trade Impacts.** An increase in waste imports affects environmental externalities both through the management and mismanagement of imported waste and by influencing the management and mismanagement of domestic waste. To quantify these effects, we first assess the relative trade impacts resulting from the ban through a gravity relationship and

[20]Results available upon request.

[21]Import values refer to the difference between the annual averages for 2012-2016 and 2018-2021. $205 million is the difference in actual trade values while $295 million refers to actual trade volumes evaluated at 2016 unit values to account for underestimation due to the drop in prices from the ban.

then determine the absolute impacts on trade using the relative effects and balanced trade identities.

For global trade, product-country-level difference-in-differences estimates of trade impacts are estimated for Exporters, China, Turkey, and the Rest of the World. The estimates are obtained for both banned products and products similar to the banned products, relative to other products.

The estimating equation is:

$$\begin{aligned}
\ln x_{pct} &= \sum_{l\in T}\sum_{c\in D} \beta_x^c \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{l = T\} \\
&+ \sum_{l\in T}\sum_{c\in D} \beta_x^{c,HS2} \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{l = T\} \\
&+ \sum_{l\in T} \mathbb{1}\{t \geq 2017\} * \text{Tariff Rate}_{pc,t=2016} * \mathbb{1}\{l = T\} + \alpha_{ct} + \alpha_{pc} + \epsilon_{pct}, \qquad (5.1)
\end{aligned}$$

where the set of destination countries is $D = \{\text{China}, \text{Exporters}, \text{Turkey}, \text{Rest of the World}\}$, and trade is $T = \{\text{Imports}, \text{Exports}\}$. To account for potential spillover effects over similar products, we control for treatment status at the 2-digit HS product level.

The results of this regression are presented in Table A3. The difference-in-difference coefficients should be interpreted relative to the log change in imports and exports of non-plastic products for each country group. We account for the possibility that the ONS policy may have indirectly affected non-plastic products through general equilibrium effects. We denote the log change in imports/exports of non-plastic products due to the ONS policy through GE to be denoted as α_c.

To estimate these general equilibrium effects, we rely on the balance of trade for each country. The balance of trade equation ensures that within a country, the combined change in net imports of 6-digit treated products, 2-digit treated products, and non-plastic products equals zero. Since we know the log changes in imports and exports of 6-digit (β_m^c, β_x^c) and 2-digit ($\beta_m^{HS2,c}$, $\beta_x^{HS2,c}$)treated products relative to non-plastic products, we can infer the log change in non-plastic products using the balance of trade equation. Detailed derivations of

this process can be found in Appendix D.1. The absolute log change of imports and exports of 6-digit treated products are then defined as $\beta_m^c + \alpha_c$ and $\beta_x^c + \alpha_c$. We show this for each country-group in Table 3.

TABLE 3. Log Change in ONS-Banned Products

Country Group	$\Delta \ln(Imports)$ $\beta_m^c + \alpha_c$	$\Delta \ln(Exports)$ $\beta_x^c + \alpha_c$
China	-4.252	-0.002
Turkey	1.090	-0.007
Exporters	-0.002	-0.723
RoW	0.176	-0.447

Note: This table indicates the total log changes in imports and exports for ONS-Banned products for each country-group (China, Turkey, Exporters, and RoW). The total log changes are calculated by the adding the estimates of Equation (D.1) shown in Table A2 with the missing intercept calculated using Balance of Trade conditions as detailed in Appendix D.1

We find that China's imports of banned plastic waste products declined by approximately 99 percent, while its exports of these products remained unchanged. Turkey's imports of banned plastic waste products increased by 199%, whereas its exports did not significantly change. Countries that heavily exported banned products to China before the ONS policy (such as the United States and the United Kingdom) reduced their exports by 51 percent and did not see a concomitant rise in their imports. Lastly, the rest of the world increased their imports of banned products by 20 percent and reduced their exports by 36 percent.[22]

5.3. **Environmental Impacts.** This sub-section quantifies the environmental impacts of the China policy, and a summary of the ways in which plastics affect the environment through their lifecycle from production to end-of-life is provided in the Appendix.

A large literature has documented the detrimental impacts of plastic pollution on public health, biodiversity and the environment (Marfella, Prattichizzo, Sardu, Fulgenzi, Graciotti, Spadoni, DâOnofrio, Scisciola, La Grotta, Frigé, et al. (2024), Trasande, Liu, and Bao (2022); Law (2017), Worm, Lotze, Jubinville, Wilcox, and Jambeck (2017b)). But there is a paucity

[22]Log changes are converted into percentage changes by taking the exponential of the coefficient and deducting 1: $e^{\beta^c + \alpha_c} - 1$

of broad-based measures of health and environmental outcomes that can be directly linked to plastic waste. For example, direct human health and marine impacts are limited to specific experiments or samples from certain coastal waters, while tracing of plastics through satellite data is a new area of inquiry with ongoing validation of techniques. Due to a lack of comprehensively available measures, the scientific literature has developed emission factors based on physical properties of the material and consequences of its leakages into the environment. We draw on the emission factors and develop them to determine the effects of the ONS policy in terms of air quality, greenhouse gas emissions, human toxicity and marine toxicity. For air quality, we can moreover validate the impacts across Turkish provinces by leveraging the fact that larger particulate matter is often a local occurrence and different provinces had differential exposure to waste mismanagement after the policy across Turkish provinces.

5.3.1. *Air Quality.* Plastic contributes to air quality through generation of particulate matter that poses health risks. PM10 is a commonly used measure of local air quality that consists of particulate matter of 10 micrometres or less. Plastic waste releases PM10 when it goes through wear and tear or degradation in the environment and particularly when it is incinerated. While its general degradation generates some fine particles, plastic waste combustion process releases a range of fine particulate matter, including soot, ash, and other combustion byproducts, directly into the atmosphere. These particles can easily become airborne, leading to elevated PM10 concentrations, which pose health risks when inhaled, such as reduced lung functioning and increased blood pressure. The immediate release of particulates during incineration makes burning the primary concern when addressing the impact of plastic waste on air quality. In contrast, general degradation has small effects and the production of virgin plastic does not have a direct effect on PM10 levels.

We start with direct evidence on the impacts of the China policy on air quality, measured as PM10 levels. Because the waste management survey in Turkey records the location of firms that mismanage waste, we can examine whether air quality deteriorated more in areas that were more exposed to mismanagement of plastic waste after the ban. The air quality

data is from Turkey's Ministry of Environment, Urbanization, and Climate Change based on real-time station level measurements of PM10. We aggregate the observations to the province-year level by constructing the maximum of all reported measurements for a given province and year. Exposure of a province is measured by the share of plastic waste produced by smaller firms in the province because we showed in Section 2 that these firms mismanaged their waste after the ban.

We estimate the following difference-in-differences specification for provinces:

$$\ln PM10_{rt} = \sum_{l=2015}^{2021} \beta_l D_t^l * \text{Exposure}_r + \alpha_r + \alpha_{NUTS2,t} + \epsilon_{ct} \tag{5.2}$$

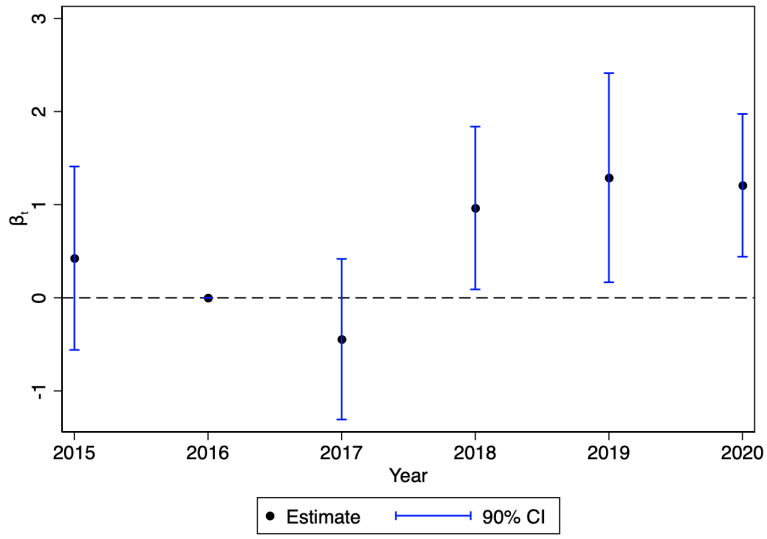
where $PM10_{rt}$ captures the extreme pollution readings within a province in year t.[23] Exposure_r is measured by the share of plastic waste produced by firms with less than 250 employees in province r. We construct provincial exposure based on the size distribution of plastic waste producers as, given the results presented above, these firms are more likely to mismanage unsold plastic waste products. Province fixed effects are included to focus on changes over time in a province and time-varying NUTS2-level fixed effects are included to account for weather fluctuations in broader areas.[24]

Figure 5.1 shows the estimated coefficients. There was an increase in PM10 levels after 2017 in provinces where the banned waste products were more intensively produced by smaller firms (with less than 250 employees). The estimated rise in PM10 of a province with mean exposure is 1.56 which corresponds to 32 percent of the mean exposure.

[23]We construct this variable using data on daily readings from multiple locations for each province. After adjusting these multiple readings from their month-year, day-month, and province level averages, we use the maximum value over a year for each province.

[24]Provinces in Turkey correspond to NUTS3-level units.

FIGURE 5.1. Air Pollution in Areas of Domestic Plastic Waste Generation



Note: This figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in 5.2. Each observation is at the province-year level. The dependent variable is the (log) pollution readings at province p and year t. The coefficient of interest is on an interaction term of year dummies D_t^l and $Exposure_p$, where $Exposure_p$ is the share of plastic waste generated by firms with less than 250 employees in province p. The interaction with year 2016 is excluded to serve as a reference year. The sample covers the years from 2015 to 2020.

We also follow the emission factor literature to quantify the national impacts on air quality using data from EXIOBASE on PM10 emissions resulting from plastic incineration. According to the World Bank, China generated 21.6 million tons of managed plastic waste in 2016, of which 25 percent was incinerated. The 6.91 million tons of controlled burning of plastic waste in China produced 47.48kg of PM10 from plastic incineration, according to EXIOBASE. Using this information, we obtain an emission factor of 0.007 tons of PM10 produced per million tons of plastic burned. We assume that burning plastics in open air (mismanaged burning) creates double the amount of PM10. So, for mismanaged waste, we use a conversion factor of 0.014 tons of PM10 produced per million.

Domestic plastic waste absorption at home is the sum of domestically-generated plastic waste and net imports of plastic waste. Based on our estimates of the impacts of the policy on imports, we find that domestic absorption of plastic waste rose from 1.04 to 1.23 million

tons. Domestically-generated waste and exports of waste in the economy do not change, as we have previously shown that these effects were negligible. And the main channel of firm responses to the policy was from direct imports of plastic waste and displacement of domestic plastic waste sales at home. We showed that domestic plastic waste generators started to mismanage their waste, primarily by burning it. The share of managed incineration was 12%, and decreased to 11% post-ONS, and the share of mismanaged (open-air) burning was 21% and increased to 23% post ONS. Multiplying the total amount of plastic burned by the emission factor (0.001 and 0.002 tons of PM10 produced per million tons of plastic burned for managed and mismanaged plastic respectively), an increase in plastic incineration after the China policy translates into an increase in PM10 levels from 3.90kg to 4.89kg annually. Annual PM10 levels in Turkey increased by 25%.

The calculated effect of the change in plastic waste mismanagement on PM10 levels in Turkey is consistent with the figures presented in Table 5.1. For the observed increase in the share of burned plastic waste, our estimates using EXIOBASE implies a 25% rise in PM10 levels, which remains within the same range as the 32% change reported by using a province-level diff-in-diff analysis.[25]

5.3.2. *Toxicity.* Plastic waste poses severe health hazards through the release of toxic chemical compounds. 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. Using emission factors from the Stockholm Convention on Persistent Organic Pollutants, we quantify the impacts of the China policy on toxicity resulting from plastic waste burning.[26]

The quantity of toxins released from the burning of municipal solid waste is dependent on the efficiency of the incineration plants. The Stockholm Convention on Persistent Organic

[25]We calculate the predicted change in PM10 levels from Table 5.1 as follows. The pre-ONS fraction of plastic waste produced by firms with less than 250 employees for our treated firms is 0.15. Using the implied difference-in-differences estimate of 1.154 for the post-ONS period, the predicted change in the PM10 levels for an average province is $(0.15 \times (exp(1.154) - 1))$.

[26]`https://toolkit.pops.int/Publish/Annexes/A_09_Annex09.html`

Pollutants categorizes incineration plants into four classes, each with a specific conversion rate from solid waste to toxic equivalents (TEQ/kg). We assume that managed waste is processed in the most efficient plants of class 4, while mismanaged waste that is burned amounts to incinerating in class 1 plants. The toxicity results should therefore be considered lower bounds because uncontrolled burning of plastic waste is more harmful. Additionally, there are few emission factors for burning of industrial plastic waste and we therefore follow the practice of using factors for burning of municipal waste. This is also likely to underestimate the release of toxic chemicals because industrial sites have been shown to have about six times higher concentrations of dioxins and related compounds than residential, commercial and rural areas (Cook and Velis (2021)).

Typically, 6.5 percent of burned waste gets converted into fly ash while 24.5 percent into bottom ash. For class 1 and 4 plants, on average, fly ash contains 30,000 ng and 1,000 ng of toxic equivalents (TEQ) per kg, and bottom ash contains 500ng and 5ng of TEQ per kg. Multiplying these factors with the rise in plastic waste burned gives the change in toxicity levels from the policy through the channels of more managed waste being incinerated in Turkey and through more plastic waste being mismanaged through inefficient burning. The production of virgin plastic, in contrast, does not have a direct effect on dioxin and furan levels in Turkey so there are no savings through this channel.

We estimate that the amount of fly ash and bottom ash released from managed burning rises 3% from its initial value of 0.01 and 0.03 million tons annually. Mismanaged burning, on the other hand, increases fly ash and bottom ash by 36% from its initial value of 0.01 and 0.05. Putting the managed and mismamanged waste channels together, the rise in plastic waste incineration through more managed waste being processed in Turkey and through more firms mismanaging their plastic waste results in toxicity rising from 1.539kg to 2.067kg annually.

The magnitude of the TEQ weights can be contextualised by comparing them to the tolerable daily intake of 1 to 4 pg/kilogram of body weight.[27] The change in TEQ from

[27] https://cdn.who.int/media/docs/default-source/chemical-safety/exe-sum-final_may1998.pdf?sfvrsn=bbb9e913_1&download=true

burning plastic due to ONS is calculated as 2.067 - 1.539 = 0.528 kg. Dividing this by 365 gives a daily measure of approximately 0.001 kg of TEQ. Given Turkey's population of around 80 million in 2016, this means that the daily per capita TEQ intake was about 12500 pg. Dividing by the average lifespan of about 80 years for an adult of about 70 kilograms, the increase in daily intake over the lifespan is about 2.23 pg/kg or the tolerable daily intake. The number can be further understood through the increase in cancer deaths associated with increased daily intakes. Using a standard body weight of about 70 to 75 kg, a daily intake of 1-2.3 pg/day per kg results in 70 to 173 pg per day per person. This level of exposure is associated with 14 to 33 cancer cases per million people per year or 242 to 586 DALYs (Disability-Adjusted Life Years) per million people per year in Western European countries.[28] Applying it to the average lifespan of 80 years and for Turkey's population of around 80 million, the estimated rise in the number of cancer cases would be 2167 to 2500 cancer patients per year or 1 percent of the 240,013 new cancer cases in 2020 in Turkey.[29]

5.3.3. *Marine Aquatic Ecotoxicity.* Plastics have penetrated into marine environments and this section quantifies the impacts of the policy on aquatic toxicity. The increasing presence of plastic waste poses a threat to marine biodiversity and contributes to the growing issue of Marine Aquatic Ecotoxicity. Marine Aquatic Ecotoxicity (MAE) refers to the toxic effects of pollutants on marine ecosystems, with plastic waste being a significant contributor. As plastics degrade in the ocean, they release harmful chemicals such as phthalates, bisphenol A (BPA), and persistent organic pollutants (POPs). 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 MAE 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.

[28]https://dioxin20xx.org/wp-content/uploads/pdfs/2012/1377.pdf
[29]`https://gco.iarc.who.int/media/globocan/factsheets/populations/792-turkiye-fact-sheet.pdf`

Furthermore, plastic debris can act as a vector for invasive species and pathogens, further destabilizing marine environments.

We use data from Exiobase on MAE resulting from plastic landfilling and burning in 2016. MAE is measured in kilograms of 1,4-dichlorobenzene equivalents (1,4-DCB eq), that provides a standardised way to quantify and compare the toxic impacts of different chemicals, including BPA, POPs, and others, on marine ecosystems. Turkey produced 0.011 tons of MAE from 1.04 million tons of plastic waste, resulting in an emission factor of 0.01 per million ton. Multiplying the amount of plastic waste generated before and after 2016 with this emission factor, we find a rise of about 0.002 million tons. Although the relative increase is 18 percent, the magnitudes are small. This is naturally influenced by the large landmass and relatively smaller exposure to marine environments in Turkey.

5.3.4. *Greenhouse Gas Impacts.* CO2e emissions from plastic waste primarily arise from the burning of plastics, with open-air burning being a particularly significant contributor. When plastic waste is burned, carbon dioxide (CO2), methane (CH4), and other greenhouse gases are released into the atmosphere, contributing to climate change. Open-air burning, which lacks the control and efficiency of managed incineration, leads to incomplete combustion, resulting in higher CO2e emissions due to the release of methane (a greenhouse gas with a much higher global warming potential than CO2) and other pollutants like carbon monoxide and black carbon. These additional compounds make open-air burning far more environmentally damaging than managed incineration. However, plastic waste does not always contribute negatively to CO2e emissions. If plastic waste is substituted for virgin plastic in production, it can reduce the demand for virgin plastic, which in turn lowers CO2e emissions associated with virgin plastic production. Because greenhouse gas emissions can be considered global in scope, it may be argued that the relevant point is whether overall greenhouse gas emissions were affected. This requires some assumptions on the domestic production of plastic waste in other countries that we do not observe directly, so this discussion is provided in the Appendix.

In this section, we analyze the impact of increased plastic waste imports in Turkey following China's ONS policy on CO2e emissions. We begin by quantifying CO2e emissions resulting from plastic waste management and mismanagement. Next, we examine whether virgin plastic and imported plastic waste are substitutable in Turkey.

To quantify the impact of China's ONS policy on CO2e emissions from plastic waste burning, we use greenhouse gas emission factors from the natural science literature, as reported by the United States Environmental Protection Agency (EPA). The EPA estimates that burning one ton of plastic waste emits approximately 2.40 tons of carbon dioxide equivalent (CO_2e) greenhouse gases.[30] In contrast, managed or landfilled plastic waste generates only 0.02 tons of CO_2e per ton. We calculate the total CO_2e emissions by multiplying the share of open-air burning by total plastic waste and 2.40, and the share of managed/dumped waste by total plastic waste and 0.02, separately for the pre- and post-ONS periods. Our results show that CO_2e emissions in Turkey from plastic waste burning and management increased from 1.30 million metric tons (mmt) to 1.70 mmt.

Whether there were any offsetting effects from changes in virgin plastic usage after the ban depends on whether recycled plastic waste imports substitute for virgin plastic in production. To formally test this substitution effect for Turkish firms, we analyze their post-ONS imports of virgin resin and polymers.[31] Our analysis shows no significant change in these imports. This finding is further supported by our country-level regressions, which also reveal no significant change in virgin plastic imports following the ONS ban.[32] Thus, for Turkey, we conclude there is no substitution between virgin plastic and recycled plastic waste.

As a result, we observe no offsetting effects from reduced virgin plastic production, as the elasticity between virgin plastic and imported plastic waste is effectively zero. The key finding, however, is the significant increase in environmental degradation due to waste mismanagement in Turkey. Had this mismanagement not occurred, emissions from Turkey would have been roughly half of what is reported in the table. The primary driver of

[30]Table A1 in the Appendix presents the figures broken down by different plastic types.
[31]It is important to note that Turkey imports coal, gas, and petroleum as primary energy sources.
[32]Results available upon request.

this environmental burden is reduced abatement in response to greater import competition. Evaluated at the current price of carbon credits in the European Union, Turkey's 0.4 mmt rise in emissions corresponds to an economic loss of approximately €42 million, adding an additional 40% to the economic burden from waste revenue losses.

Overall, Table 4 summarises the main results of a substantial increase in domestic waste in Turkey and consequent increases in air pollution and human toxicity arising from a greater burden of managing plastic waste and more mismanaged plastic waste.

TABLE 4. Environmental Impacts in Turkey

	$P^c + M^c - X^c$	Share of Open Air Burning	Share of Controlled Burning	PM10	Toxicity	MAE	GHG
Pre-ONS	1.04	0.21	0.12	3.90kg	1.54 kg	9.03kg	1.30mmt
Post-ONS	1.23	0.23	0.11	4.89kg	2.00kg	10.7kg	1.70mmt
% Change	18%	2pp	-1pp	25%	30%	18%	31%

Note: The first column presents the total amount of plastic waste in Turkey (in million tons), which includes domestic plastic waste production (P^c) sourced from the World Bank's What a Waste Database, along with imports (M^c) and exports (X^c) from UN Comtrade. The post-ONS values were calculated by applying the growth rates obtained from the diff-in-diff specifications to the pre-ONS values. The second column shows the share of plastic waste subject to open-air burning. This is derived by multiplying the mismanagement rate of plastic waste (from the World Bank) by the proportion of open-air burning within mismanaged waste (sourced from the OECD Global Plastics Outlook). The share of controlled burning is calculated by multiplying (1 - mismanagement rate) (from the World Bank) with the share of incineration in managed waste, also sourced from the OECD. Managed waste is defined as waste that is recycled, landfilled, or incinerated in a controlled manner. Columns 4, 5, and 6 display the calculated values for PM10, Toxicity, and Marine Aquatic Ecotoxicity, as explained in this section.

5.3.5. *Sensitivity Analysis.* To evaluate alternative scenarios in global trade and waste management, we conducted two hypothetical analyses concerning CO_2 emissions:

Scenario 1: No Trade Destruction and All Diversion to Turkey. We examine the environmental implications of no trade destruction. Under this scenario, all waste that would have been imported by China is redirected to Turkey. This reallocation would have resulted in Turkey's waste imports increasing by 736%, rather than the observed 199%.

Consequently, such a significant shift would have led to an increase in CO_2 emissions from 1.30 mmt to 11.49 mmt.

Scenario 2: No Country Composition Effect. Additionally, we assess the potential impact on CO_2 emissions if Turkey and the Rest of the World (RoW) adhered to the waste mismanagement levels observed in major exporting countries (primarily, the US and the EU). Under this scenario, if mismanagement practices were aligned, CO_2 emissions from waste in Turkey would have only seen a slight increase from 0.17 mmt to 0.20mmt.

6. 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 Turkey after China banned them, and by 2020 Turkey had become the top importer of plastic waste in the world. We find that plastic using firms in Turkey 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 Turkey dropped, resulting in higher levels of waste mismanagement by domestic firms that experienced reduced sales as a result of increased import competition. Air pollution increased in regions where these firms were concentrated, with negligible offsetting effects from reduced virgin plastic production in Turkey.

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, the model suggests that changes in global and national 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 model 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 Turkey lost economically through reduced aggregate revenue. Heightened import competition led to a fall in domestic waste sales and these firms became more likely to mismanage their plastic waste by-products. Turkey also experienced worse environmental outcomes in terms of increased air pollution, that we can measure directly and through emission factors. Our calculations suggest an increases in air pollution and human toxicity levels of about 25 and 30 percent, respectively.

The burden of the ban was borne unequally across the world. The displacement of exports from China to Turkey resulted in greater waste mismanagement and the economic gains from cheap imports were too small to offset the losses from displaced earnings of domestic waste suppliers.

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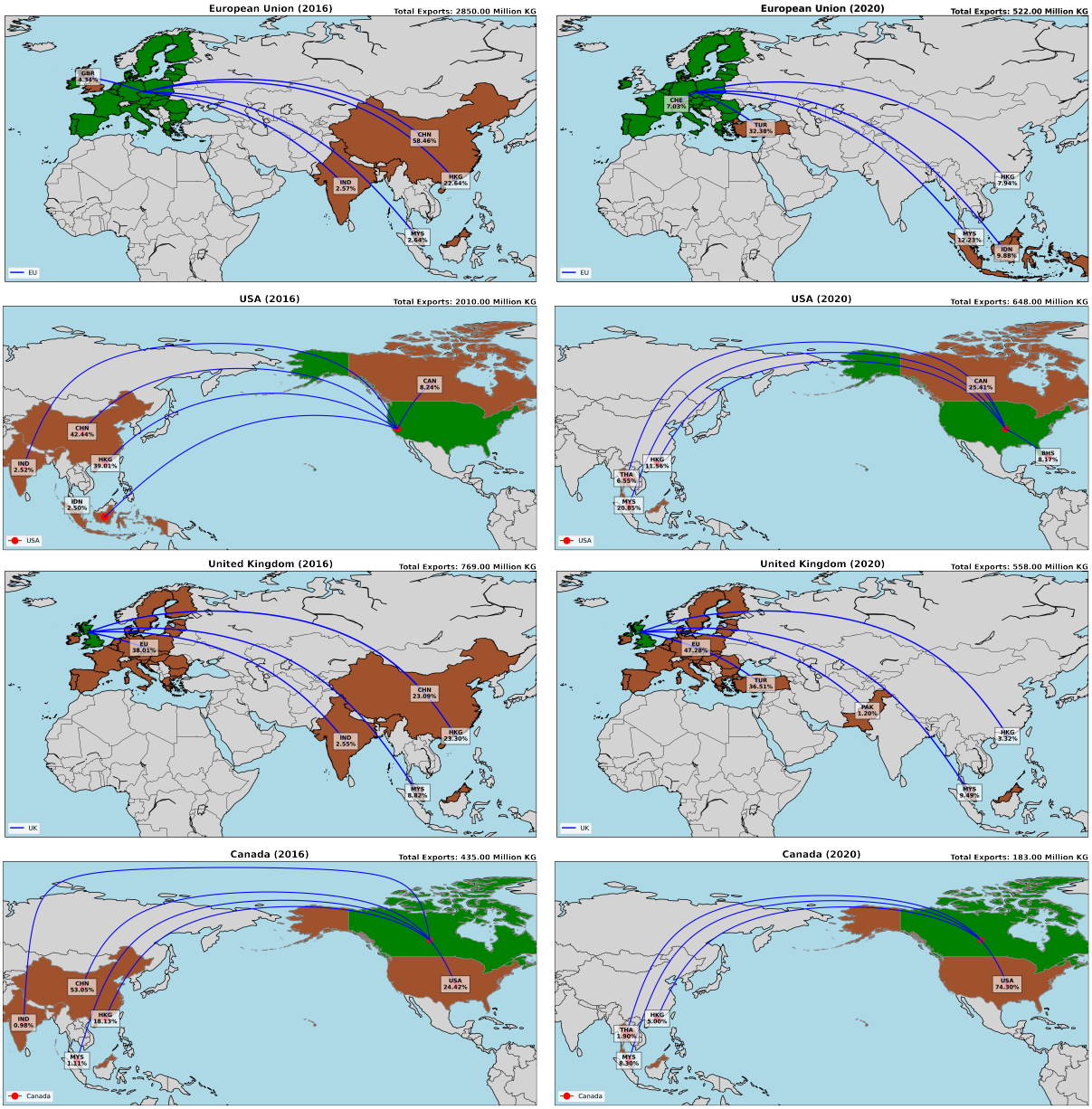
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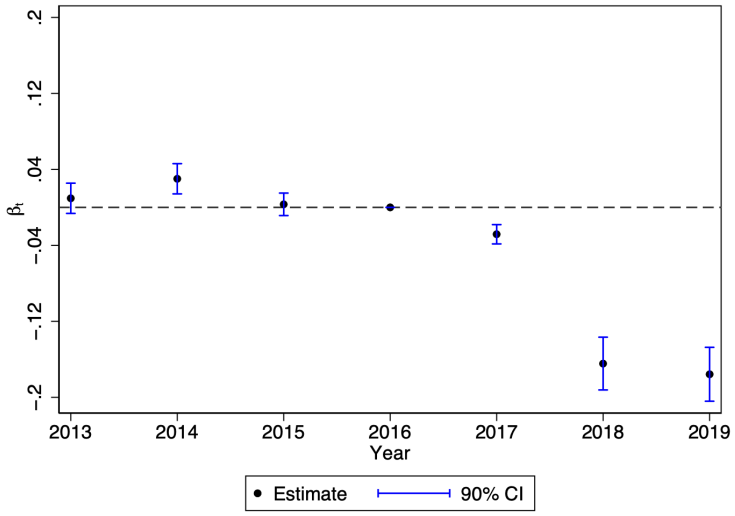
Appendix A. Additional Figures and Tables

Figure A1. Plastic Waste Exports by Major Exporting Country



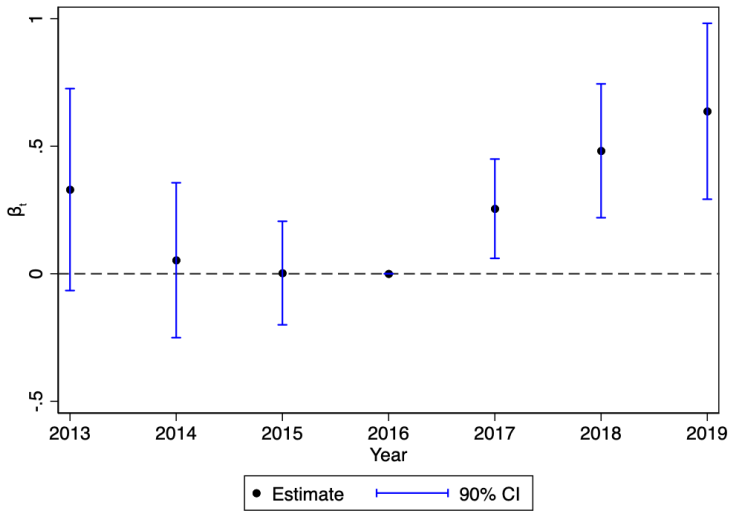
*Note:*Each map displays the top five plastic waste exporting destinations for different countries. The first row illustrates the European Union's major destinations in 2016 (right panel) and 2020 (left panel). The second row depicts the United States' top five exporting destinations for plastic waste for the same years, 2016 on the right and 2020 on the left. The third row presents data from the United Kingdom, while the final row focuses on Canada's main exporting destinations.

FIGURE A2. Event Study: Global Trade after the Operation National Sword Policy



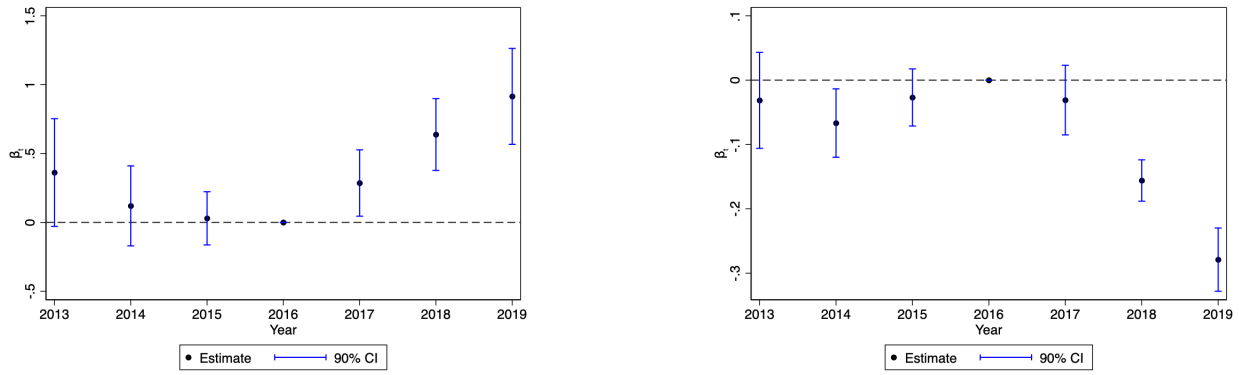
Note: This figure plots the estimates, together with 90% confidence intervals, obtained from an event study where the global trade of 6-digit HS products are regressed on an interaction of two dummy variables, $Treat_{HS6}$ and $Post_t$. $Treat_{HS6}$ is a dummy variable indicating whether that 6-digit HS product was banned by China through the ONS, and $Post_t$ is a dummy variable taking a value of 1 post 2017. The sample covers the years from 2013-2019.

FIGURE A3. Event Study: Firm-level Value of Imports



*Note:*The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from equation 3.1. 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. The estimating equation for the firm-product-origin-time-level imports is: $\ln \text{Imports}_{ipot} = \sum_{l=2013}^{2019} \beta_l D_t^l * \text{Banned}_p + \alpha_{ipo} + \alpha_{ot} + e_{ipot}$, where Imports_{ipot} 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 $Treat_p$ indicates the set of China-banned plastic waste products.

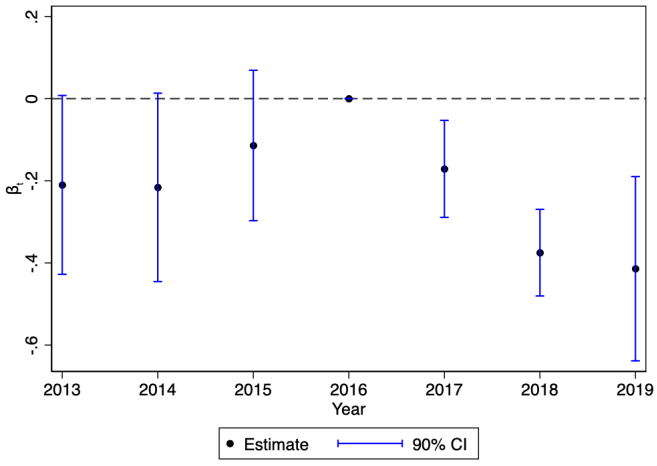
FIGURE A4. Event Study: Decomposition of Turkish Imports into Unit Values and Quantities



(A) Quantity (B) Unit prices

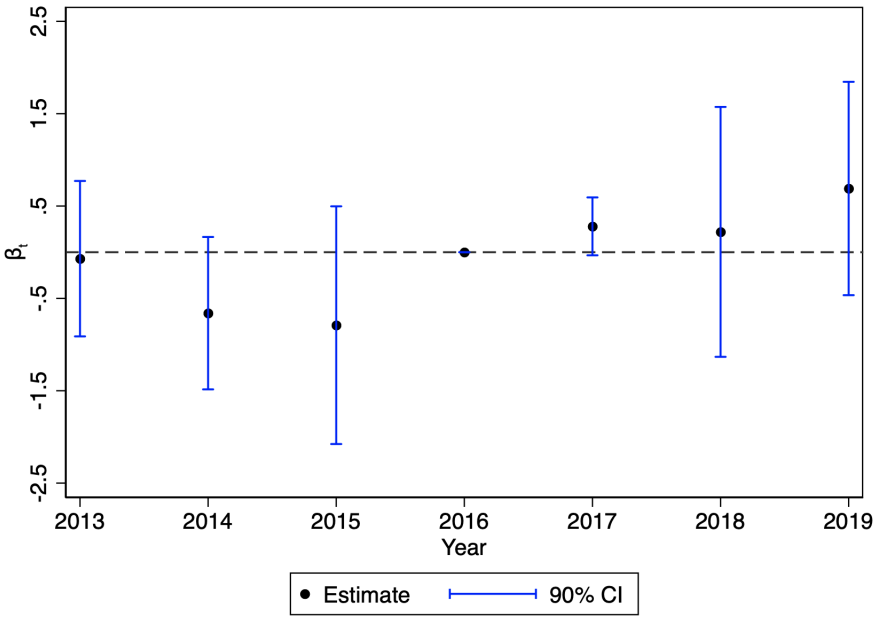
Note: The figure plots the estimates of β_l, together with 90% confidence intervals, obtained from equation 3.1, where the dependent variable in Panel (A) is quantity (measured in kg) of Turkish imports at the 8-digit HS level and in Panel (B) is the unit price (measured in dollars per kg) of Turkish imports at the 8-digit HS level. 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 A5. Event Study: Quality Adjusted Import Prices



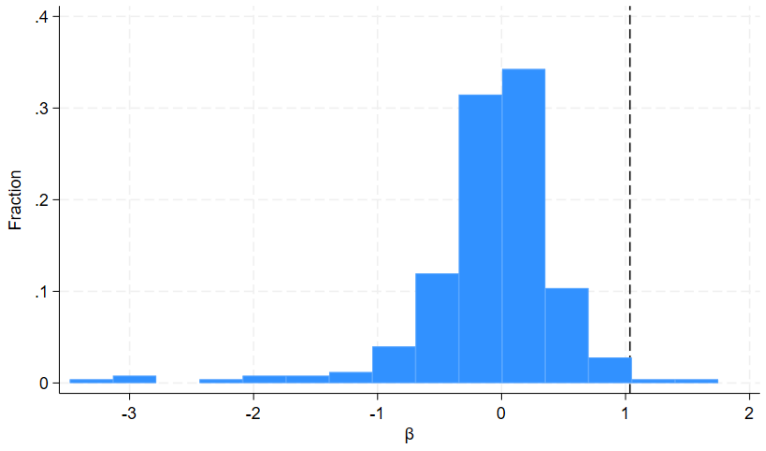
Note: This figure plots the estimates of β_l, together with the 90% confidence intervals, obtained from Equation 3.1, where the dependent variable is the quality adjusted price. Quality of a particular 8-digit HS product is the residual from regressing the log quantity + 5*log unit value on country-year,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

FIGURE A6. 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 at the 8-digit HS level. 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.

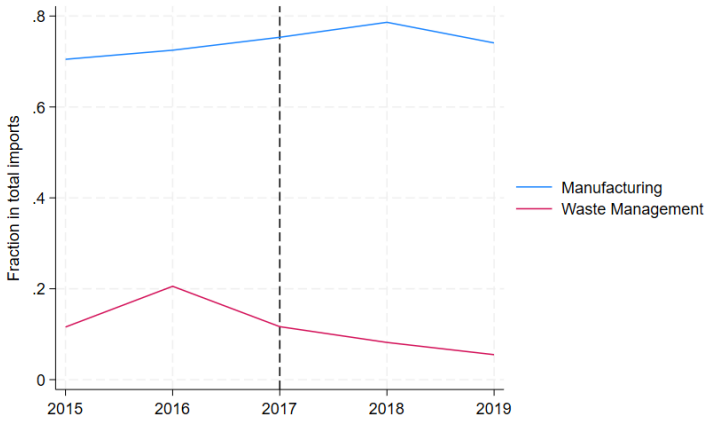
FIGURE A7. Imports of Plastic Products Randomly Assigned Treatment



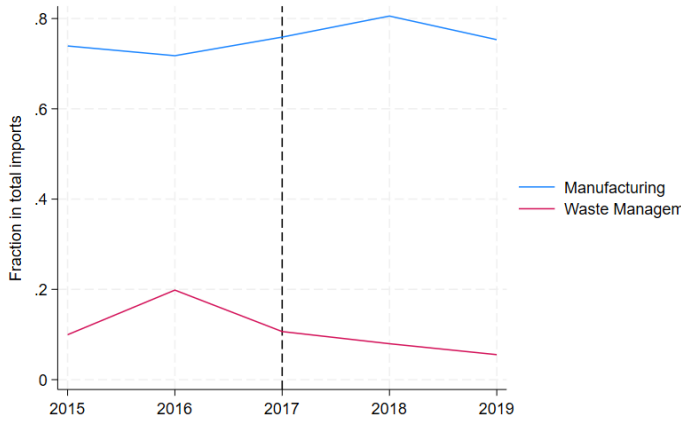
Note: This figure plots the β coefficients of Equation Equation 3.1. The treatment variable is randomly allocated treatment status to 8-digit HS products within their corresponding 4-digit HS codes and then re-estimated. This procedure is replicated 250 times. This is used as a robustness check.

FIGURE A8. Share of Manufacturing Firms in Imports

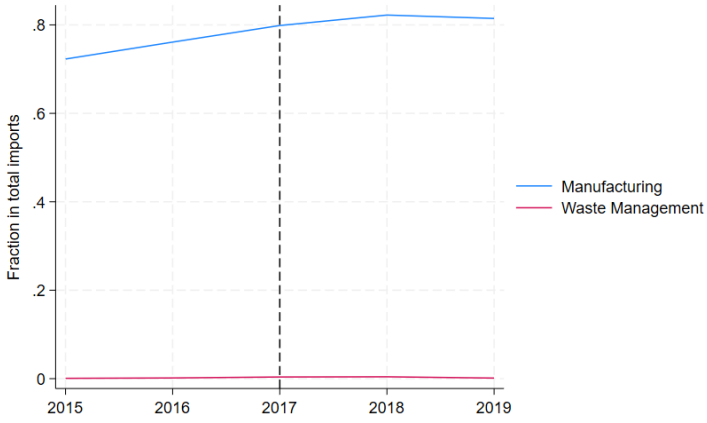
(A) Share of Imports by Quantity: Plastic Waste



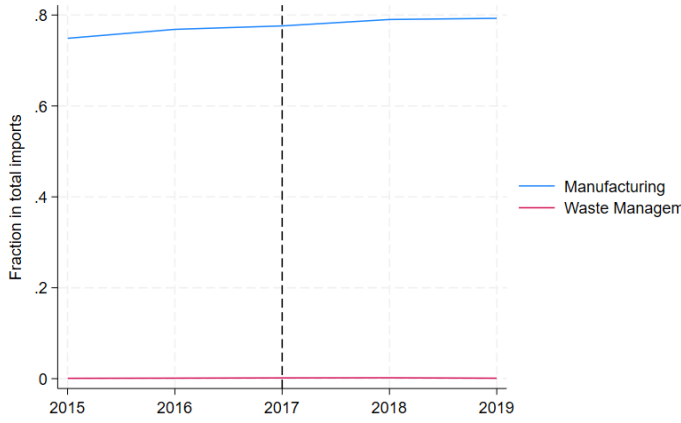
(B) Share of Imports by Value: Plastic Waste



(C) Share of Imports by Value: Other Goods

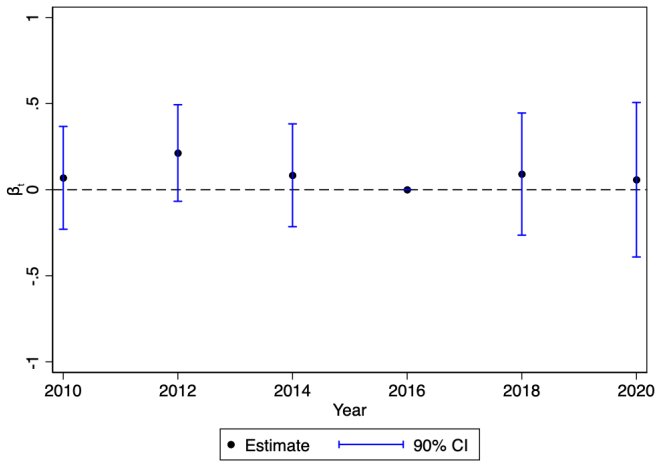


(D) Share of Imports by Value: Other Goods



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 A9. Total Waste Production of Domestic Waste Generators



Note: This figure plots the estimates of β_l, together with 90% confidence intervals, obtained from estimating the specification in 3.3. Each observation is at the firm-year level. The dependent variable is the amount of waste that firm k produces at year t. The coefficient of interest is on an interaction term of year dummies D_t^l and $Exposure_k$. Where $Exposure_k$ is the share of firm k's production of China banned waste 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 2013-2019.

TABLE A1. Emissions by Plastic Type

Plastic Type	**2016 Global Production (million t)**	**Share Global Production (%)**	**Mis- management/ Combustion ($MTCO_2E$ per Short Ton)**	**Managed ($MTCO_2E$ per Short Ton)**	**Raw Material Acquisition and Manufacturing Emission $MTCO_2E$ per Short Ton)**
HDPE	50.81	12.10	2.80	0.02	1.10
LLDPE/ LDPE	49.34	11.74	2.80	0.02	1.27
PET	22.99	0.05	2.05	0.02	1.75
PP	66.63	15.87	2.80	0.02	1.17
PS	19.23	0.05	3.02	0.02	1.87
PVC	46.09	10.97	1.26	0.02	1.69
Other	206.31	49.22	2.34	0.02	1.49
TOTAL	419.17	100	2.40	0.02	1.39

Notes: The global production of plastic in 2016 by plastic type is provided by Our World in Data: `https://ourworldindata.org/grapher/plastic-production-polymer?time=2013..latest`. Shares are calculated using the global production. The CO_2 emissions of mismanagement/combustion are taken from EPA's GHG emission hub. Similarly, the emissions from the raw material acquisition and manufacturing are obtained by US EPA: `https://archive.epa.gov/epawaste/conserve/tools/warm/pdfs/Plastics.pdf`

TABLE A2. Effect on Imports

	IMPORTS		EXPORTS	
	(1)	(2)	(3)	(4)
	$\ln(val_{pct})$	$\ln(val_{pct})$	$\ln(val_{pct})$	$\ln(val_{pct})$
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\}$	-4.246a	-4.244a	-0.0782	-0.0791
	(0.588)	(0.589)	(0.351)	(0.351)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\}$	0.137	0.137	-0.713a	-0.714a
	(0.108)	(0.108)	(0.107)	(0.107)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\}$	1.104a	1.104a	0.360	0.359
	(0.0898)	(0.0898)	(0.938)	(0.937)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\}$	0.192c	0.190c	-0.438a	-0.439a
	(0.0982)	(0.100)	(0.0761)	(0.0781)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\}$	-0.0254	-0.0255	0.129a	0.129a
	(0.0303)	(0.0303)	(0.0347)	(0.0347)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\}$	0.0367c	0.0396b	0.0543a	0.0582a
	(0.0196)	(0.0196)	(0.0207)	(0.0208)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\}$	0.121a	0.121a	-0.0186	-0.0184
	(0.0404)	(0.0404)	(0.0625)	(0.0625)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\}$	0.0481a	0.0466a	0.0750a	0.0778a
	(0.0108)	(0.0106)	(0.0156)	(0.0156)
$\mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\}$	0.0422a		0.00862	
	(0.0126)		(0.0119)	
$\mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\}$	-0.0736a		-0.102a	
	(0.00432)		(0.00530)	
$\mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\}$	-0.214a		0.0266c	
	(0.0120)		(0.0138)	
Tariff Percentage Points$_{pc2016} * \mathbb{1}\{t \geq 2017\}$	0.0798b	0.0984a	0.124a	0.0292
	(0.0327)	(0.0346)	(0.0472)	(0.0493)
# Observations	2,926,379	2,926,379	2,371,529	2,371,529
R^2	0.933	0.935	0.907	0.909
Fixed Effects:				
Year	Yes	No	Yes	No
Country x Year	No	Yes	No	Yes
Product x Country	Yes	Yes	Yes	Yes

Note: This table shows the β coefficient of the regression estimated by Eq. D.1. The sample period is between 2013-2019, and includes all 6-digit HS products. The data for this estimation is obtained from the UN Comtrade. Letters indicate statistical significance: a indicates p<0.01, b indicates p<0.05, and c indicates p<0.10.

Appendix B. Theory

B.1. **Firm Choices.** Firms maximise profits and let λ denote the Lagrange multipliers on the production constraints:

$$\begin{aligned}
\max_{l,m,v,x} \Pi_{ud} =& R_u\left(l_{ud}, m_{ud}\right) - w_d l_{ud} - \sum_h \sum_o z_d v_{huod} - \sum_h \sum_o p_{hod} x_{huod} + \lambda_{ud}\left(m\left(v_{huod}, x_{huod}\right) - m_{ud}\right) \\
\max_{a,v,x} \Pi_{sd} =& R_s\left(v_{hsod}\right) - \sum_h \sum_o z_d v_{hsod} + \sum_h \sum_{d'} r_{sdd'}\left(x_{hsdd'}\right)/\tau_{hdd'} - \sum_h w_d \gamma\left(a_{hsd}\right) \sum_o v_{hsod} \\
&+ \sum_h \lambda_{hsd}\left(x_{hsd} a_{hsd} \sum_o v_{hsod} - \sum_{d'} x_{hsdd'}\right) \\
\max_{l} \Pi_{nd} =& R_n\left(l_{nd}\right) - w_d l_{nd}
\end{aligned}$$

Optimal choices are given by the FOCs below:

$$\begin{aligned}
&(1) R_{lu}\left(l_{ud}, m\left(v_{huod}, x_{huod}\right)\right) - w_d = 0 \\
&(2) R_{mu}\left(l_{ud}, m\left(v_{huod}, x_{huod}\right)\right) m_{v_{huod}}\left(v_{huod}, x_{huod}\right) - z_d = 0 \\
&(3) R_{mu}\left(l_{ud}, m\left(v_{huod}, x_{huod}\right)\right) m_{x_{huod}}\left(v_{huod}, x_{huod}\right) - p_{hod} = 0 \\
&(4) r'_{sdd'}\left(\bar{x}_{hsd}\right) x_{hsd} - w_d \gamma'\left(a_{hsd}\right) = 0 \\
&(5) R_{v_h s}\left(v_{hsod}\right) - z_d + r'\left(\bar{x}_{hsd}\right) x_{hsd} a_{hsd} - w_d \gamma\left(a_{hsd}\right) = 0 \\
&(6) r'\left(x_{hsdd'}\right)/\tau_{hdd'} = \lambda_{hsd} \equiv r'_{sdd'}\left(\bar{x}_{hsd}\right) \\
&(7) R_{ln}\left(l_{nd}\right) - w_d = 0 \\
&(8) x_{hsd} a_{hsd} \sum_o v_{hsod} - \sum_{d'} x_{hsdd'} = 0
\end{aligned}$$

For a strictly concave maximisation problem, we must assume $R'', m'', r'', -\gamma'' < 0$ and $\gamma' > 0$. Given all else equal, equation (4) implies a is a decreasing function of $w/r'x$ because $\gamma'' > 0$. We have implicitly assumed r to be symmetric in all its arguments but this is not necessary.

B.2. **Market Equilibrium.** Give revenues and input prices$(z_d, w_d, p_{hod}, R_{ud}, R_{sd}, R_{nd})$, equations (1) to (8) have 8 unknowns $(l_{ud}, v_{huod}, x_{huod}, a_{hsd}, v_{hsod}, x_{hsdd'}, \bar{x}_{hsd}, l_{nd})$. The market equilibrium is determined by demand for labour and materials determined by the profit maximisation FOCs (1)-(3), given factor prices and revenue functions. Waste management is determined from the abatement FOC (4) and the virgin material choice of supplying firms in FOC (5), given factor prices, revenue functions and the by-product function. Waste demand and supply x is given by FOC (6) and the supply constraint (8). Labour choice of non-exposed firms is given by FOC (7).

In a market equilibrium, virgin resource prices are fixed by arbitrage and free trade. Let z denote the global energy price and τ_d denote the cost of shipping energy from a Walrasian global market to d. Because virgin resources are traded in a global market, the price of shipping it from o to d is $z_{od} = z_d = \tau_d z$, shown below as equation (9). Wages w are determined by labour market clearing in equation (10) below and waste prices p_{iod} are determined by market clearing of demand and supply in equation (11).

$$(9) z_d = \tau_d z$$

$$(10) L_d = l_{ud} + l_{nd} + l_{sd}, l_{sd} \equiv \sum_h \gamma\left(a_{hsd}\right) \sum_o v_{hsod}$$

$$(11) x_{hsod} = x_{huod}$$

Finally, factor incomes I equal expenditures Y on final consumption in the economy. And, these in turn, equal the profits and earnings of all agents (where domestic transfers such as wages get cancelled out). Let $\bar{V}_d$ denote the country's endowment of virgin resources (such as oil or natural gas) that can be directly converted to plastic. The national income identities are summarised in equation (12) below.

$$\begin{aligned}(12) I_d =& Y_{sd} + Y_{ud} + Y_{nd} \\ =& R_{sd} + R_{ud} + R_{nd} + z\bar{V}_d\end{aligned}$$

$$+\sum_h \left(\sum_{d' \neq d} p_{hdd'} x_{hsdd'} - \sum_{o \neq d} z_d v_{huod} - \sum_{o \neq d} z_d v_{hsod} - \sum_{o \neq d} p_{hod} x_{huod} \right)$$

Together, (1) to (12) provide solutions to the 8 unknowns in production, the 3 factor prices and the income of the representative consumer, given demand for domestic and imported outputs summarised by R and Y.

B.3. **Gravity Trade.** To show that the model is consistent with gravity in waste trade, equations (4) and (6) show waste supply from o to d is

$$r'\left(x_{hsod}\right) = \tau_{hod} w_o \gamma'\left(a_{hso}\right) / x_{hso}$$

Dividing by exports to China gives a form similar to gravity when τ is interpreted as a measure of distance:

$$\frac{r'\left(x_{hsod}\right)}{r'\left(x_{hsoc}\right)} = \frac{\tau_{hod}}{\tau_{hoc}}.$$

Summing across origins and products, waste imports of banned products into destination d are

$$x_d \equiv \sum_{h>\bar{h}} \sum_o x_{hsod} = \sum_{h>\bar{h}} \sum_o r'^{-1} \left(\frac{\tau_{hod}}{\tau_{hoc}} r'\left(x_{hsoc}\right) \right)$$

Let g_{od} denote the inverse of geographical distance between o and d and assume an elasticity of trade to distance of -1. Assuming a power function for marginal revenue $r'(x) \propto x^{\eta}$ (for $\eta < 1$), we get the gravity relationship:

$$x_d = \sum_o r'^{-1} \left(\frac{\tau_{od}}{\tau_{oc}} r'\left(x_{soc}\right) \right) = \sum_o \frac{g_{od}}{g_{oc}} x_{soc}.$$

In logs, this can be written accounting for zero waste imports into China after ONS as:

$$\Delta \ln x_d = -\ln \sum_{h>\bar{h}} \sum_o \frac{g_{od}}{g_{oc}} x_{hsoc}.$$

De-meaning by all exports to China, banned waste imports are

$$\Delta \ln x_d = -\ln \sum_{h>\bar{h}} \sum_o x_{hsoc} - \ln \sum_{h>\bar{h}} \sum_o \frac{g_{od}}{g_{oc}} \frac{x_{hsoc}}{\sum_{h>\bar{h}} \sum_o x_{hsoc}}$$

This is operationalized empirically as

$$\ln x_{dt} \equiv \alpha_d + \alpha_t + \beta_x \cdot \ln \sum_{h>\bar{h}} \sum_o \frac{g_{od}}{g_{oc}} \frac{x_{hsoc}}{\sum_{h>\bar{h}} \sum_o x_{hsoc}} \cdot Post_t$$

B.4. **Extension to Symmetric Revenue Functions.** To examine robustness to symmetry in revenue functions, we harmonize the arguments in the revenue function R_{jd}. We still maintain well-defined firm types and therefore the revenue functions will not be completely symmetric. In particular, the revenue functions are defined in terms of netput vectors - using firms u are net buyers of plastic waste and supplying firms s are net sellers of plastic waste while neither firms n have zero net sales or purchases of plastic waste. The profit functions are now re-written as:

$$\begin{aligned}
\max_{l,m,v,x} \Pi_{ud} =& R_u\left(l_{ud}, m_{ud}\right) - w_d l_{ud} - \sum_h \sum_o z_d v_{huod} - \sum_h \sum_o p_{hod} x_{huod} + \lambda_{ud}\left(m\left(v_{huod}, x_{huod}\right) - m_{ud}\right) \\
\max_{a,v,x} \Pi_{sd} =& R_s\left(l_{sd}, v_{sd}\right) - \sum_h \sum_o z_d v_{hsod} + \sum_h \sum_{d'} r_{sdd'}\left(x_{hsdd'}\right)/\tau_{hdd'} - \sum_h w_d \gamma\left(a_{hsd}\right) \sum_o v_{hsod} - w_d l_{sd} \\
&+ \sum_h \lambda_{hsd}\left(x_{hsd} a_{hsd} \sum_o v_{hsod} - \sum_{d'} x_{hsdd'}\right) \\
\max_l \Pi_{nd} =& R_n\left(l_{nd}, v_{nd}\right) - w_d l_{nd} - \sum_h \sum_o z_d v_{hnod}
\end{aligned}$$

We get 2 additional FOCs but the other FOCs (1) to (8) are not affected, except through additional arguments implicitly in the revnue functions:

$$(5)'' R_{ls}\left(l_{sd}, v_{sd}\right) - w_d = 0$$

$$(7)'' R_{v_{hnod}n}\left(l_{nd}, v_{nd}\right) - z_d = 0$$

The new FOCs will alter labour market clearing and virgin resource clearing to

$$(10)' L_d = l_{ud} + l_{nd} + l_{sd} + \sum_h \gamma\left(a_{hsd}\right) \sum_o v_{hsod}$$

$$\begin{aligned}(12)' I_d =& Y_{sd} + Y_{ud} + Y_{nd} \\ =& R_{sd} + R_{ud} + R_{nd} + z\bar{V}_d \\ &+ \sum_h \left(\sum_{d' \neq d} p_{hdd'} x_{hsdd'} - \sum_{o \neq d} z_d v_{huod} - \sum_{o \neq d} z_d v_{hsod} - \sum_{o \neq d} z_d v_{hnod} - \sum_{o \neq d} p_{hod} x_{huod} \right)\end{aligned}$$

where l_{sd} appears on the RHS of (10)' additionally and $\sum_{o \neq d} z_d v_{hnod}$ in (12)'. Because we are already including all labour and virgin resources used in the economy (as we cannot distinguish the purpose for which they are used within a firm), this does not alter the quantification.

B.5. **Extension to Per Unit Abatement Costs.** To examine robustness to constant return to scale in abatement, we re-define the abatement costs to $\sum_h w_d \gamma\left(a_{hsd}\right) x_{hsd} \sum_o v_{hsod}$ where the appearance of x_{hsd} implies that abatement costs are per unit of by-products, rather than amortized across them. Profit functions and optimal choices for u and n are the same as before. But the profit function and FOCs for s change and are shown below:

$$\begin{aligned}\max_{a,v,x} \Pi_{sd} =& R_s\left(v_{hsod}\right) - \sum_h \sum_o z_d v_{hsod} + \sum_h \sum_{d'} r_{sdd'}\left(x_{hsdd'}\right) / \tau_{hdd'} - \sum_i w_d \gamma\left(a_{hsd}\right) x_{hsd} \sum_o v_{hsod} \\ &+ \sum_h \lambda_{hsd} \left(x_{hsd} a_{hsd} \sum_o v_{hsod} - \sum_{d'} x_{hsdd'} \right)\end{aligned}$$

$$(4)' \lambda_{hsd} - w_d \gamma'\left(a_{hsd}\right) = 0$$

$$(5)' R_{v_{hsods}}\left(v_{hsod}\right) - z_d + \lambda_{hsd} x_{hsd} a_{hsd} - w_d \gamma\left(a_{hsd}\right) x_{hsd} = 0$$

$$(6) r'_{sdd'}\left(x_{hsdd'}\right) / \tau_{hdd'} = \lambda_{hsd} \equiv r'\left(\bar{x}_{hsd}\right)$$

$$(8)x_{hsd}a_{hsd}\sum_{o}v_{hsod}-\sum_{d'}x_{hsdd'}=0$$

Substituting for (6) in (4)' and (5)', we arrive at the following FOCs:

$$(4)'r'\left(\bar{x}_{hsd}\right)-w_d\gamma'\left(a_{hsd}\right)=0$$

$$(5)'R_{v_{hsod}s}\left(v_{hsod}\right)-z_d+r'\left(\bar{x}_{hsd}\right)x_{hsd}a_{hsd}-w_d\gamma\left(a_{hsd}\right)x_{hsd}=0$$

The forms of (4)' and (5)' are changed slightly from

$$(4)r'\left(\bar{x}_{hsd}\right)x_{hsd}-w_d\gamma'\left(a_{hsd}\right)=0$$

$$(5)R_{v_{hsod}s}\left(v_{hsod}\right)-z_d+r'\left(\bar{x}_{hsd}\right)x_{hsd}a_{hsd}-w_d\gamma\left(a_{hsd}\right)=0$$

but the main point of a being a decreasing function of w/r' is still true (under $\gamma''>0$), and (6) and (8) FOCs for s remain exactly as before. The key change is that the wage cost is no longer divided by x because it cannot be amortized across by-products. Accordingly, the labour market clearing condition is also altered to

$$(10)'L_d=l_{ud}+l_{nd}+l_{sd},l_{sd}\equiv\sum_{h}\gamma\left(a_{hsd}\right)x_{hsd}\sum_{o}v_{hsod}$$

to account for the per unit nature of abatement costs through x_{hsd}.

Appendix C. The Lifecycle of Plastics

This section explains the lifecycle of plastics and related environmental imapcts from it. The lifecycle of plastics begins with the extraction of raw materials, with 99 percent of plastics being derived from fossil fuels such as crude oil and natural gas. The extraction process involves pumping oil out of the ground or sea and extracting natural gas from formations deep under fields and waterways. This process is energy-intensive and poses significant environmental risks, such as oil spillages that can have immediate and long-term detrimental effects on sea life and wildlife. Additionally, the extraction and transportation

of raw materials contribute to emissions of pollutants such as carbon dioxide (CO2) and nitrogen dioxide (NO2).

Following extraction, the crude oil undergoes fractional distillation, a process where it is heated and separated into different components such as petroleum gas, gasoline, and naphtha. These hydrocarbons are then broken down into simpler alkenes or alkanes through cracking, which requires extremely high temperatures and pressures. This stage also leads to the emission of various pollutants, including CO2, NO2, sulphur dioxide (SO2) and potentially harmful substances such as carbon monoxide (CO) and Volatile Organic Compounds (VOCs).

The next stage, polymerization, involves linking hydrocarbon monomers through chemical processes to produce polymers, which are thick, viscous substances used to make plastic products. The polymerization process is also energy-intensive, requiring high heat and pressure, which further contributes to CO2, NO2, and SO2 emissions.

In the production phase, these polymers are converted into beads or pellets, which are then melted and molded into plastic products such as bottles or containers. Different types of additives, including toxic substances like chlorine in the production of Polyvinyl Chloride (PVC), as well as plasticizers and dyes, are added to enhance the properties of the plastic. This phase also contributes to the emission of CO2, NO2, and SO2, along with potential releases of hazardous gases like hydrogen chloride.

Plastics are then used across various sectors of the economy, including packaging, construction, textiles, consumer goods, transportation, electronics, and industrial machinery. While the use of plastics itself is not inherently polluting, certain additives can pose toxicity risks.

At the end of their life cycle, plastics are disposed of through various methods, each with its environmental consequences. Recycling is one option, where plastics are sorted, melted, and reused, though this process also requires energy and less than ten percent of plastic is

recycled across the world.[33] The recyclability of plastics is determined by the type of plastic, which is indicated by a recycling code typically found on plastic products:

(1) Polyethylene Terephthalate (PET or PETE): Commonly used in beverage bottles and food containers. It can be rinsed, softened at around 80 Celsius, and recycled into products like clothing fibers, carpet, and new containers.

(2) High-Density Polyethylene (HDPE): HDPE is used in products like milk jugs and detergent bottles. It can be melted and recycled into non-food containers, pipes, and plastic lumber.

(3) Polyvinyl Chloride (PVC): PVC is used in plumbing pipes and window frames. Due to its chlorine content and the use of plasticizers, PVC should not be melted or heated for recycling as it can release toxic substances. It is usually recycled for similar applications or downcycled.

(4) Low-Density Polyethylene (LDPE): LDPE is found in items like plastic bags and six-pack rings. It can be recycled for purposes other than food containers, often into products like bin liners and floor tiles.

(5) Polypropylene (PP): PP is used in yogurt containers and bottle caps. Although it can be reused, its recycling is limited because the process alters its structure, making it less durable. Recycled PP is often combined with virgin material for products like automotive parts.

(6) Polystyrene (PS): PS is used in foam products like disposable coffee cups and food trays. It can be compressed and reformed for reuse, but its recycling is less common due to the material's brittleness and low density.

(7) Other Plastics: This category includes a variety of plastics, such as polycarbonate and bioplastics, that are not easily recycled. These plastics often require specialized facilities and are not commonly accepted in standard recycling programs.

[33] https://data-explorer.oecd.org/vis?tenant=archive&df[ds]=DisseminateArchiveDMZ&df[id]=DF_PLASTIC_WASTE_5&df[ag]=OECD&dq=.&lom=LASTNPERIODS&lo=5&to[TIME_PERIOD]=false

Beyond recycling, other disposal methods include incineration, where burning plastic waste to create energy releases harmful pollutants such as CO2 and dioxins, furans, and polychlorinated biphenyls (PCBs). Dioxins, furans and PCBs, are highly toxic or carcinogenic. While incineration is undertaken by some municipal waste facilities, uncontrolled or open-air burning is also common. Uncontrolled burning disposes of plastics without energy recovery and is a key source of plastic pollution (Cottom et al. 2024). Landfilling, where plastics are buried, can affect wildlife and potentially contaminate groundwater with microplastics or toxic substances like chlorine from PVC. Finally, open-air dumping or water dumping of plastics is highly detrimental to the environment, as the low degradation rates of plastics can lead to long-lasting harm to wildlife and marine ecosystems, with plastics being carried into rivers and oceans by rain and wind, posing severe risks to sea life.

Appendix D. Quantification

D.1. **Derivations of Trade Impacts.** For global trade, product-country-level difference-in-differences estimates of trade impacts are estimated for Exporters, China, Turkey, and the Rest of the World. The estimates are obtained for both banned products and products similar to the banned products, relative to other products.

The estimating equation is:

$$\begin{aligned}
\ln x_{pct} &= \sum_{l\in T}\sum_{c\in D} \beta_x^c \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{l = T\} \\
&+ \sum_{l\in T}\sum_{c\in D} \beta_x^{c,HS2} \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{c = D\} * \mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{l = T\} \\
&+ \sum_{l\in T} \mathbb{1}\{t \geq 2017\} * \text{Tariff Rate}_{pc,t=2016} * \mathbb{1}\{l = T\} + \alpha_{ct} + \alpha_{pc} + \epsilon_{pct}, \qquad \text{(D.1)}
\end{aligned}$$

where the set of destination countries is $D = \{\text{China}, \text{Exporters}, \text{Turkey}, \text{Rest of the World}\}$, and trade is $T = \{\text{Imports}, \text{Exports}\}$. To account for potential spillover effects over similar products, we control for treatment status at the 2-digit HS product level.

The estimation results are shown in Table A3. The difference-in-difference coefficients should be interpreted in comparison to the growth rate of imports and exports of non-plastic products for each country-group. We allow for the ONS policy to have affected the non-plastic products through general equilibrium (GE). To estimate the GE effects, we rely on balance of trade for all countries.

For each country, the BoT is as follows:

$$0 = \Delta E_{HS6}^c + \Delta E_{HS2}^c + \Delta E_n^c$$

where E_{HS6}^c is country c's net exports of the 6-digit plastic waste products banned by China, E_{HS2}^c is country c's net exports of plastic products, and E_n^c is country c's net exports of all other products.

We can rewrite the BoT for each country in terms of the diff-in-diff coefficients we obtained from Table A3.

$$
\begin{aligned}
0 =& \Delta E^c_{HS6} + \Delta E^c_{HS2} + \Delta E^c_n \\
=& \underbrace{(e^{\beta^c_x+\alpha_c}-1)}_{\text{\% change banned exports}} Export^c_{HS6} - \underbrace{(e^{\beta^c_m+\alpha_c}-1)}_{\text{\% change banned imports}} Import^c_{HS6} \\
&+ \underbrace{(e^{\beta^{c,HS2}_x+\alpha_c}-1)}_{\text{\% change plastic exports}} Export^c_{HS2} - \underbrace{(e^{\beta^{c,HS2}_m+\alpha_c}-1)}_{\text{\% change plastic imports}} Import^c_{HS2} \\
&+ \underbrace{(e^{\alpha_c}-1)}_{\text{\% change neither exports}} Export^c_n - \underbrace{(e^{\alpha_c}-1)}_{\text{\% change neither imports}} Import^c_n \\
&\underbrace{(-0.1)*(e^{\beta^c_x+\alpha_c}-1)}_{\text{\% change virgin exports}} Export^c_{virgin} - \underbrace{(-0.1)(e^{\beta^c_m+\alpha_c}-1)}_{\text{\% change virgin imports}} Import^c_{HS6}
\end{aligned}
$$

where α_c is the log change of non-plastic products due to the ONS that is unknown to us.[34] Therefore, we have four unknowns (α_{TUR}, α_{CHN}, α_{EXP}, α_{RoW}) and four BoT equations. But one BoT equation is linearly dependent. We therefore draw on domestic labour market clearing in Turkey to obtain a fourth equation. We can then solve for the unknowns to arrive at the absolute change in country-level imports and exports for non-plastic products due to the ONS policy (α_D). Having done so, we can compute the logarithmic change in trade of banned products for each country D as $\ln\left(e^{\beta^D_T+\alpha_D}\right)$ for $T = m, x$.

The solution to this 4-unknown and 4-equation problem is:

$$\alpha_{TUR} = -0.0067$$

$$\alpha_{CHN} = -0.0016$$

$$\alpha_{EXP} = -0.0020$$

$$\alpha_{RoW} = -0.0030$$

Now that we have determined the change in country-level imports and exports for non-plastic products due to the ONS policy (α_c), we can compute the logarithmic change in

[34]The elasticity between imports and virgin plastic is taken as -0.1.

banned imports for each country c. This will allow us to estimate the impact of the ONS policy on emissions. The logarithmic change in imports for each country group c is:

$$\ln\left(e^{\beta_m^c+\alpha_c}\right) = \begin{cases} 1.090 & \text{if } c = \text{Turkey} \\ -4.252 & \text{if } c = \text{China} \\ 0.176 & \text{if } c = \text{RoW} \\ -0.002 & \text{if } c = \text{Exporters} \end{cases} \tag{D.2}$$

The logarithmic change in exports for each country group c is:

$$\ln\left(e^{\beta_x^c+\alpha_c}\right) = \begin{cases} -0.007 & \text{if } c = \text{Turkey} \\ -0.002 & \text{if } c = \text{China} \\ -0.447 & \text{if } c = \text{RoW} \\ -0.723 & \text{if } c = \text{Exporters} \end{cases} \tag{D.3}$$

TABLE A3. Effect on Imports and Exports

	(1) $\ln(val_{pct})$
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	-4.253a
	(0.590)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.125
	(0.107)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	1.096a
	(0.0899)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.179c
	(0.103)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	-0.0004
	(0.0263)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.0640a
	(0.0177)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.0096
	(0.0400)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.0577a
	(0.0102)
Tariff Percentage Points$_{pc2016} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Imports\}$	0.0903a
	(0.0332)
[1em] $\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	-0.0830
	(0.351)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	-0.721a
	(0.107)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.360
	(0.937)
$\mathbb{1}\{Banned_p^{HS6}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	-0.444a
	(0.0767)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{China_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.118a
	(0.0349)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{HeavyExporters_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.0529a
	(0.0203)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{Turkey_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.101
	(0.0635)
$\mathbb{1}\{Banned_p^{HS2}\} * \mathbb{1}\{ROW_c\} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.0711a
	(0.0158)
Tariff Percentage Points$_{pc2016} * \mathbb{1}\{t \geq 2017\} * \mathbb{1}\{Exports\}$	0.0303
	(0.0429)
# Observations	5,285,577
R^2	0.922
Fixed Effects:	
Country x Year	Yes
Product x Country x Trade	Yes

Note: This table shows the β coefficient of the regression estimated by Eq. D.1. The sample period is between 2013-2019, and includes all 6-digit HS products. The data for this estimation is obtained from the UN Comtrade. Letters indicate statistical significance: a indicates p<0.01, b indicates p<0.05, and c indicates p<0.10.

D.2. **Virgin Plastic Usage.** In addition to the baseline calibration of global estimates of emissions that relies on raw data and emission factors, we can directly examine emissions from waste using satellite data (from Climate Trace) and primary production of plastics from EXIOBASE. While this is more appealing in that the impacts can be directly estimated, the mapping to China's ONS policy is not exact because the data covers all waste emissions and all primary plastic production, and therefore we use the direct estimates for China and Turkey in our quantification, and consider scaling factors from emissions to waste mismanagement based on the direct evidence from Turkey.

We estimate the following specifications to measure the effects of the ONS on per capita CO_2 emissions, CO_2pc, and production of primary plastic for each destination d at year t:

$$\begin{aligned} \ln \mathrm{CO_2pc}_{dt} &= \beta_x \ln x_{dt} + \Gamma_x X_{dt} + \alpha_d + \alpha_t + e_{dt} \\ \ln m^v_{dt} &= \beta_v \ln x_{dt} + \Gamma_v X_{dt} + \alpha_d + \alpha_t + \varepsilon_{dt} \end{aligned} \tag{D.4}$$

where X_{dt} denotes the vector of destination specific controls, such as initial per capita income, CO_2 emissions, and virgin plastic production, all interacted with a post-ONS dummy.

To account for the potential endogeneity of plastic waste imports, we use a dummy variable that indicates whether destination d had, on average, a trade surplus against the main exporters of plastic waste to China before the implementation of the ONS. We calculate the trade balance as a weighted average of bilateral trade imbalances run by d against the main exporters of plastic waste, where weights reflect the importance of each exporter for Chinese plastic waste imports in 2016. The reasoning is that running a trade surplus against the main exporters of plastic waste reduces the cost of importing plastic waste due to a surplus of empty containers on the way back to the home country.

As presented in the first column of Table A4, the instrument is informative: running a trade surplus against the main exporters of plastic waste before China's ban was associated with more imports of plastic waste to destination d after the ONS. Both OLS and 2SLS results suggest that more imports of plastic waste after the ONS led to an increase in CO_2 emissions and a reduction in the production of virgin plastic in new destinations. This is

consistent with our baseline of increased imports of banned plastic waste increasing waste emissions but reducing virgin resource use.

TABLE A4. Effects on CO_2 Emissions and Virgin Plastic Production

	$\ln x_{dt}$ (1) FS	$\ln \text{CO}_2\text{pc}_{dt}^{waste}$ (2) OLS	$\ln \text{CO}_2\text{pc}_{dt}^{waste}$ (3) 2SLS	$\ln m_{dt}^{v}$ (4) OLS	$\ln m_{dt}^{v}$ (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)				
$\sum_o \ln\left(\frac{dist_{od}}{dist_{oChina}}\right) * \mathbb{1}\{t \geq 2017\}$	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)
$\ln m_{d16}^{v} * \mathbb{1}\{t \geq 2017\}$	-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
r2	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)

D.3. **Environmental Impact Following ONS.** This section details the data sources and methodology utilized to determine the environmental impact following the ONS.

D.3.1. *Data.* 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.

Baseline Plastic Waste Levels: The annual volume of plastic waste within Turkey 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.

World Bank's What a Waste Global database estimates that Turkey 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. The Turkish Statistical Institute's survey, targeting manufacturing firms with more than 20 employees, reported these firms produced approximately 0.168 million tonnes of plastic waste annually. Integrating this data with the World Bank's figures, we estimate that about 18% of Turkey's total plastic waste comes from larger manufacturing firms, consistent with the report that 23% of all waste in Turkey is industrial.

The post-ONS level of domestically produced waste is calculated by multiplying the pre-ONS level (as determined by the data) by the growth rate of Turkey's "supplying" industry, which we derived from the difference-in-differences estimates on employment (Column 2 of Table 2) and labor market clearing conditions. Column 2 indicates that supplying firms in Turkey did not experience any significant employment growth compared to "neither" firms following the ONS policy. Based on the labor market clearing condition, we also found that employment changes in neither firms were negligible.

Assuming Cobb-Douglas production (or using a first-order approximation of production changes) and constant markups, we estimate that revenue changes for supplying firms (ΔR_n) are approximately zero, i.e., $\Delta R_s = 0.21 \times \Delta l_s \approx 0$, given that labor costs constitute 21% of total costs for supplying firms. This suggests that domestic production for these firms was not affected by China's ban on plastic waste.[35]

As with Turkey, we use the World Bank's What a Waste Database to obtain the pre-ONS levels of domestic plastic waste production for China, the Rest of the World (RoW), and Exporters. For these country groups, we assume no growth in plastic waste production, applying a zero growth rate.

[35]Although Column 1 of Table 2 shows a decline in sales for supplying firms, we attribute this to a reduction in the sale of their byproduct, plastic waste, as illustrated in Figure 3.2, rather than a drop in their core production. This means the total amount of plastic waste produced by these firms remained unchanged, but the volume of plastic waste they were able to sell decreased. This is further supported by Figure A9, which shows no significant change in total waste production following the ONS.

We use 2015 data from UN Comtrade to determine the pre-ONS import and export levels of plastic waste for all country groups. Post-ONS values are calculated by applying the estimated growth rates to the pre-ONS levels. These growth rates are derived using a difference-in-differences approach and Balance of Trade conditions, as detailed in Section D.1.

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. 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 Turkey.

The raw data shows that mismanaged waste, as a share of disposed waste, was 71.6% in 2014. Using a difference-in-differences specification, we estimate that the mismanagement rate increased by 7 percentage points following the ONS ban (see 3.4). Therefore, we rely on our difference-in-differences estimate to quantify the growth in mismanagement due to the ONS. Since we are using 47% as the baseline mismanagement share from the World Bank's database, we adjust the 7 percentage point increase accordingly. If there is a 7 percentage point increase from 71.6%, this corresponds to a 4.6 percentage point increase from the 47% baseline.

Since comparable granular data on changes in waste management are not available for other countries, we rely on OECD plastic waste and satellite data to fill the gap. The OECD Global Plastic Outlook provides the share of plastic waste that is mismanaged for broad country groups, including China, Exporters (United States and OECD European Union) and the Rest of the World (RoW). To estimate the change in waste management following ONS in China, Shi and Zhang (2023) examine satellite data on open burning and estimate a 14 percent decline in such episodes in areas that were more exposed to the ONS policy. At a mean exposure of 38 percent, this amounts to a 5.3 percent reduction in mismanagement of waste in China.[36]

[36]One caveat to using satellite data is that any burning episode will be recorded in it but these are minimised through institutional knowledge of the seasons and regions where these are likely to be driven by other factors such as farming.

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 they apply to all waste and not 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.

Finally, we consider the shares of managed and mismanaged burning for each country group. For this, we rely on the OECD Global Plastics Outlook database, which provides data on open burning as a share of mismanaged waste and incineration as a share of managed waste. We assume these shares remain constant before and after the ONS policy. However, since the proportions of mismanaged and managed waste relative to total waste change post-ONS, the shares of open burning and managed incineration as a fraction of total plastic waste will also change accordingly.

One limitation is that the OECD does not provide separate data for Turkey. Therefore, we use the estimates for OECD non-EU countries as a proxy for Turkey. Exporters are defined using data from the US and OECD EU, while the Rest of the World (RoW) is represented by the average of the remaining country groups reported by the OECD.

The data explained above are summarized in Table A5.

TABLE A5. Country-Level Plastic Waste Management

	Turkey		China		Exporters		RoW	
Variable	pre	post	pre	post	pre	post	pre	post
P^c (million tons)	1.0	1.0	21.6	21.6	68.1	68.1	141.0	141.0
M^c (million tons)	0.10	0.29	7.4	0.1	4.6	5.2	3.1	3.6
X^c (million tons)	0.02	0.03	0.03	0.03	9.6	4.6	4.8	3.1
$P^c + M^c - X^c$ (million tons)	1.04	1.23	28.9	21.7	63.1	68.0	139.3	141.6
Mismanagement/Total Waste	47%	52%	25%	24%	5%	6%	52%	55%
Open Burning/Total Waste	21%	23%	11%	10%	2%	2%	17%	17%
Managed Burning/Total Waste	24%	24%	12%	11%	27%	27%	12%	11%

Notes: Data on domestic plastic waste production (P^c) and the share of plastic mismanaged were sourced from the World Bank's What a Waste Database. The share of inceneration is obtained from OECD Plastic Outlook. OECD reports Turkey under non-EU OECD countries. Information on plastic waste exports and imports (X^c and M^c) was obtained from UN Comtrade. Pre-ONS levels are taken from year 2015. The growth of plastic production, and hence post-ONS levels of P^c are estimated using the labor market clearing condition in Turkey. Post-ONS levels of imports and exports are calculated using the diff-in-diff specification on imports and exports. Post-ONS levels of mismanagement shares were estimated based on our calculations of the increase in mismanagement levels following ONS.

Given that burning is a primary method of waste disposal in Turkey, our environmental impact analysis focuses on PM10 emissions, toxicity levels, and CO2e emissions, as these are directly affected by incineration. The following sections outline the methods we use to estimate these impacts.[37]

D.3.2. *PM10.* Plastic burning is a significant contributor to PM10 levels, as the combustion process releases a range of fine particulate matter, including soot, ash, and other combustion byproducts, directly into the atmosphere. These particles can easily become airborne, leading to elevated PM10 concentrations, which pose health risks when inhaled. While the degradation of plastic in the environment can also generate some fine particles, its contribution to PM10 is generally considered negligible compared to the substantial increase caused by plastic burning. The immediate release of particulates during incineration makes burning the primary concern when addressing the impact of plastic waste on air quality.

[37]The raw data from the Turkish Statistical Institute shows that in 2014 waste mismanagement as a share of disposed waste was 0.716, and increased to 0.755 in 2018. Burned waste as a share of mismanaged waste increased from 0.072 in 2014 to 0.129 in 2018.

Although we previously conducted an empirical analysis on the change in PM10 levels in Turkey following the ONS policy, in this section, we will extend those results by calculating using some technical parameters combined with publicly available data broader impacts at the country level. To achieve this, we will use data from Exiobase, which provides information on PM10 emissions resulting from plastic incineration. With this data, we will develop a conversion factor to calculate global changes in PM10 levels. According to Exiobase, China produced 47.48 kg of PM10 from managed plastic incineration in 2016. Our calculations indicate that China generated 28.9 million tons of plastic waste that year, with 24% of it being incinerated, as per OECD data, resulting in 6.91 million tons of plastic waste that was burned in a managed manner. Using this information, we derive a conversion factor of 0.007 tons of PM10 produced per million tons of plastic burned. We assume that burning plastics in open air (mismanaged burning) creates double the amount of PM10 in comparison to managed burning. So, for mismanaged burning of plastic waste, we use a conversion factor of 0.014 tons of PM10 produced per million.

We calculate PM10 levels using the following formula: total plastic waste in a country x managed incineration rate x 0.007 + total plastic waste x mismanaged burning rate x 0.014. Itâs important to note that virgin plastic production does not directly affect PM10 levels, so any reduction in virgin plastic due to increased recycling is excluded from these calculations. The table below provides all the relevant data points and the resulting changes in PM10 levels.

Annual PM10 levels in Turkey rose from 3.90 kg to 4.89 kg, marking a 25% increase. Conversely, China experienced a 26% decline, with PM10 levels dropping from 90.64 kg to 66.85 kg. Exporting countries saw a modest rise from 130.21 kg to 140.38 kg, representing a 7.8% increase. Lastly, the Rest of the World (RoW) observed a 2% increase, with levels rising from 432.25 kg to 441.48 kg per year.

D.3.3. *Toxicity Levels from Burning Plastic.* Plastic waste presents significant environmental and health hazards, not only through the emission of greenhouse gases but also via the release of potentially carcinogenic substances when incinerated. The burning of plastics, particularly

TABLE A6. PM10 Calculations

	Turkey		China		Exporters		RoW	
Variable	pre	post	pre	post	pre	post	pre	post
$P^c + M^c - X^c$ (million tons)	1.04	1.20	28.92	21.68	63.11	68.02	139.29	141.60
Share of plastic burned managed	23.89%	24.32%	12.26%	11.10%	26.75%	26.74%	11.56%	11.46%
Share of open air burning	21.15%	23.40%	10.86%	10.28%	1.64%	1.65%	16.80%	16.96%
PM10 (kg)	3.90	4.89	90.64	66.85	130.21	140.38	432.25	441.48

Notes: Data on domestic plastic waste production (P^c) and the share of plastic burned were sourced from the OECD Plastic Outlook. OECD does not separately report the share of plastic burned for Turkey, so we take this information from the Waste Management Survey obtained from the Turkish Statistical Institute. Information on plastic waste exports and imports (X^c and M^c) was obtained from UN Comtrade. Post-ONS levels were estimated based on our calculations of the increase in mismanagement levels following ONS. PM10 emissions, measured in kilograms, were calculated by multiplying the total amount of plastic burned by the conversion factor (0.001 tons of PM10 produced per million tons of plastic burned), as determined from our earlier analysis.

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. In this section, we will calculate the changes in levels of dioxins and furans in each country following China's plastic waste import ban, using scientific data from the Stockholm Convention on Persistent Organic Pollutants.[38] The incineration of plastics, especially those containing chlorine or other additives, leads to the creation of toxic compounds such as dioxins and furans. During combustion, the heat causes the plastic's molecular structure to break down, releasing chlorine and other elements that combine with organic molecules to form these toxic compounds. These toxins are emitted into the air and accumulate in the ash, particularly in finer particles like fly ash. The unburned carbon in the ash enhances the adsorption of dioxins and furans, concentrating these toxins in the residual ash. This ash can become a significant source of environmental contamination, releasing dioxins and furans over time as it spreads via wind or water.[39]

The quantity of toxins released from the burning of municipal solid waste is dependent on the efficiency of the incineration plants. The Stockholm Convention on Persistent Organic Pollutants categorizes incineration plants into four classes, each with a specific conversion

[38] `https://toolkit.pops.int/Publish/Annexes/A_09_Annex09.html`

[39] Note that the production of virgin plastic does not have a direct effect on dioxin and furan levels. Therefore, any reduction in virgin plastic production due to increased recycling is not included in these calculations.

rate from solid waste to toxic equivalents (TEQ/kg). Lacking data on the class of plants burning plastic waste in each country group, we will conservatively assume that all plants burning plastic are class 4, the most efficient class, whereas we assume open air burning is equivalent to a class 1 plant in terms of efficiency.

According to the Stockholm Convention on Persistent Organic Pollutants, 6.5% of burned plastic converts into fly ash and 24.5% into bottom ash. For class 1 and 4 plants, on average, fly ash contains 30,000 ng and 1,000 ng of toxic equivalents (TEQ) per kg, and bottom ash contains 500ng and 5ng of TEQ per kg. Multiplying these factors with the rise in plastic waste burned gives the change in toxicity levels from the policy through the channels of more managed waste being incinerated in Turkey and through more plastic waste being mismanaged through inefficient burning. The production of virgin plastic, in contrast, does not have a direct effect on dioxin and furan levels in Turkey so there are no savings through this channel.

We estimate that the amount of fly ash and bottom ash released from managed burning rises 3% from its initial value of 0.01 and 0.03 million tons annually. Mismanaged burning, on the other hand, increases fly ash and bottom ash by 36% from its initial value of 0.01 and 0.05. Putting the managed and mismamanged waste channels together, the rise in plastic waste incineration through more managed waste being processed in Turkey and through more firms mismanaging their plastic waste results in toxicity rising from 1.539kg to 2.067kg annually.

The magnitude of the TEQ weights can be contextualised by comparing them to the tolerable daily intake of 1 to 4 pg/kilogram of body weight. [40] The change in TEQ from burning plastic due to ONS is calculated as 2.067 - 1.539 = 0.528 kg. Dividing this by 365 gives a daily measure of approximately 0.001 kg of TEQ. Given Turkey's population of around 80 million in 2016, this means that the daily per capita TEQ intake was about 12500 pg. Dividing by the average lifespan of about 80 years for an adult of about 70 kilograms, the increase in daily intake over the lifespan is about 2.23 pg/kg or the tolerable daily intake.

[40] https://cdn.who.int/media/docs/default-source/chemical-safety/exe-sum-final_may1998.pdf?sfvrsn=bbb9e913_1&download=true

The number can be further understood through the increase in cancer deaths associated with increased daily intakes. Using a standard body weight of about 70 to 75 kg, a daily intake of 1-2.3 pg/day per kg results in 70 to 173 pg per day per person. This level of exposure is associated with 14 to 33 cancer cases per million people per year or 242 to 586 DALYs (Disability-Adjusted Life Years) per million people per year in Western European countries. [41] Applying it to the average lifespan of 80 years and for Turkey's population of around 80 million, the estimated rise in the number of cancer cases would be 2167 to 2500 cancer patients per year or 1 percent of the 240,013 new cancer cases in 2020 in Turkey. [42]

Below is a summary of the calculations for each country-group:

TABLE A7. TEQ Calculations

	Turkey		**China**		**Exporters**		**RoW**	
Variable	pre	post	pre	post	pre	post	pre	post
$P^c + M^c - X^c$ (million tons)	1.04	1.20	28.92	21.68	63.11	68.02	139.29	141.60
Share of open air burning	21.15%	23.40%	10.86%	10.28%	1.64%	1.65%	16.80%	16.96%
Amount of mismanaged fly ash (mt)	0.01	0.02	0.38	0.14	0.07	0.07	1.52	1.56
Amount of mismanaged bottom ash (mt)	0.05	0.07	1.43	0.55	0.25	0.27	5.73	5.88
Share of plastic burned managed	23.89%	24.32%	12.26%	11.10%	26.75%	26.74%	11.56%	11.46%
Amount of managed fly ash (mt)	0.01	0.01	0.32	0.34	1.10	1.18	1.05	1.06
Amount of managed bottom ash (mt)	0.03	0.03	1.20	1.29	4.14	4.46	3.95	3.98
TEQ (in kg)	1.54	2.00	41.68	17.65	15.65	16.91	165.05	169.20
% change in TEQ	29.67		-57.65		8.06		2.51	

Notes: Data on domestic plastic waste production (P^c) and the share of plastic burned were sourced from the OECD Plastic Outlook. OECD does not separately report the share of plastic burned for Turkey, so we take this information from the Waste Management Survey obtained from the Turkish Statistical Institute. Information on plastic waste exports and imports (X^c and M^c) was retrieved from UN Comtrade. Post-ONS levels were estimated based on our calculations of the increase in mismanagement levels after ONS. The quantity of fly ash was determined by multiplying the total plastic waste by the share burned and then by the average fly ash yield from plastic combustion (6.50%), as estimated by UNEP. Similarly, the amount of bottom ash was calculated by multiplying the total plastic waste by the share burned and then by the average bottom ash yield (24.50%), also from UNEP estimations. Finally, toxic equivalence (TEQ) was calculated by converting the total amount of fly and bottom ash (in millions of tons) to TEQ, using conversion factors: 1 ton of fly ash corresponds to 4,535 ng TEQ, and 1 ton of bottom ash corresponds to 907,185 ng TEQ.

For Turkey, we observe that TEQ increased from 0.421 kg to 0.565 kg, representing a 34% increase. In contrast, China's TEQ decreased from 8.322 kg to 6.67 kg, a decline of 20%. Exporters saw a modest rise in TEQ from 10.879 kg to 11.066 kg, an increase of 1.72%.

[41] https://dioxin20xx.org/wp-content/uploads/pdfs/2012/1377.pdf

[42] `https://gco.iarc.who.int/media/globocan/factsheets/populations/792-turkiye-fact-sheet.pdf`

Lastly, the Rest of the World (RoW) experienced an 8.86% increase in TEQ, from 11.960 kg to 13.020 kg.

It is important to note that both Exporters and RoW encompass multiple countries. When we adjusted these figures on a per capita basis, Turkey's per capita increase in TEQ was significantly higher compared to Exporters and RoW.

To convert TEQ into cancer cases, we reference a study that provides guidelines based on picograms of TEQ per day per body weight.[43] The standard body weight used is typically 70-75 kg. With a daily intake of 1-2.3 pg/day per kg, this results in 70 to 173 pg per day per person. This level of exposure has been observed to translate into 14 to 33 cancer cases per million people per year or 242 to 586 DALYs (Disability-Adjusted Life Years) per million people per year in Western European countries.

For Turkey, the change in TEQ from burning plastic due to ONS is calculated as 0.565 - 0.421 = 0.144 kg. Dividing this by 365 gives a daily measure of approximately 0.00155 kg of TEQ. Given Turkey's population of around 80 million in 2016, this means that the daily per capita TEQ intake was approximately 4,931.515 pg. This intake translates into approximately (4,931.515/75)*14 to (4,931.515/70)*33 cancer cases per million people per year. This is then divided by the average lifespan, taken as 80 years. Since Turkey's population is around 80 million, the estimated number of cancer cases would be multiplied by 80 to obtain the national impact. Therefore, there was an increase of 920 to 2324 cancer patients per year in Turkey due to China's ONS policy. Cancer patients in Turkey in 2018 was 210,537.

D.3.4. *Marine Aquatic Ecotoxicity.* Marine Aquatic Ecotoxicity (MAE) refers to the toxic effects of pollutants on marine ecosystems, with plastic waste being a significant contributor. As plastics degrade in the ocean, they release harmful chemicals such as phthalates, bisphenol A (BPA), and persistent organic pollutants (POPs). 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 MAE by being ingested

[43]https://dioxin20xx.org/wp-content/uploads/pdfs/2012/1377.pdf

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. The increasing presence of plastic waste in the oceans thus poses a significant threat to marine biodiversity and contributes to the growing issue of Marine Aquatic Ecotoxicity.

In this section, we estimate the impacts of plastic waste on Marine Aquatic Ecotoxicity (MAE) at the country level. We use data from Exiobase, which provides MAE measurements in kg 1,4-dichlorobenzene equivalents (1,4-DCB eq) resulting from plastic landfilling and burning for each country in 2016. The use of 1,4-DCB equivalents as a unit of measurement allows for a standardized way to quantify and compare the toxic impacts of different chemicals, including BPA, POPs, and others, on marine ecosystems. Based on this data, we develop a conversion factor to estimate global changes in MAE levels. It's important to note that these conversion factors may vary by country due to differences in exposure to marine environments. For example, countries with less exposure to water bodies are less likely to contribute to MAE from plastic waste, even if they generate significant amounts of plastic waste. According to Exiobase, China produced 4.48 tons of MAE, measured in 1,4-dichlorobenzene equivalents, from plastic waste in 2016. Our calculations show that China generated 28.92 million tons of plastic waste that year, leading to a conversion factor of 0.154 tons of MAE per million tons of plastic waste. For Turkey, Exiobase reports 0.009 tons of MAE from 1.04 million tons of plastic waste, resulting in a conversion rate of 0.008. We calculate the conversion rate for Exporters using United States data from Exiobase, which shows that the U.S. produced 0.33 tons of MAE in 2016, giving a conversion rate of 0.005. Since Exiobase does not provide specific data for the Rest of the World (RoW), we assume that the conversion rate for RoW is equal to Turkey's.

By applying these conversion rates to the amount of plastic waste generated before and after 2016, we can estimate the total MAE in tons. For China, MAE declined from 4.48

tons to 3.56 tons, representing a 25% decrease. In Turkey, there was virtually no absolute change in MAE, but the relative change was 18%. For the RoW, MAE increased from 0.33 tons to 0.35 tons, a 7.7% increase. Finally, for Exporters, MAE rose from 1.21 tons to 1.23 tons, corresponding to a 1.7% increase.

TABLE A8. Marine Aquatic Ecotoxicity Calculations

	Turkey		**China**		**Exporters**		**RoW**	
Variable	pre	post	pre	post	pre	post	pre	post
$P^c + M^c - X^c$ (million tons)	1.04	1.20	28.92	21.68	63.11	68.02	139.29	141.60
MAE conversion (tons 1,4-dichlorobenzene eq. per million tons of plastic waste)	0.009		0.155		0.005		0.009	
MAE (tons)	0.009	0.010	4.48	3.56	0.33	0.35	1.21	1.23

Notes: Data on domestic plastic waste production (P^c) and the share of plastic burned were sourced from the OECD Plastic Outlook. Information on plastic waste exports and imports (X^c and M^c) was obtained from UN Comtrade. Post-ONS levels were estimated based on our calculations of the increase in mismanagement levels following ONS. MAE emissions, measured in tons of 1,4-dichlorobenzene eq., were calculated by multiplying the total amount of plastic waste by the conversion factor for each country, as determined from our earlier analysis.

D.3.5. *Derivations of CO_2 Emission Estimations.* For each country-group we estimate the environmental burden from the mismanagement and management of waste and virgin plastic consumption, and then convert them to comparable environmental impacts. To do so, we need to calibrate the relative environmental burdens or disutility rates ξ for all three types of waste activities - waste mismanagement, waste recycling and virgin resource use - to arrive at their net effects. For this, we take greenhouse gas emission factors from the natural science literature that are available from the United States Environmental Protection Agency. EPA reports that the combustion of one ton of plastic waste results in the emission of approximately 2.40 tons of carbon dioxide equivalent greenhouse gases CO_2e. In contrast, waste management/dumping produces 0.02 tons while the production of one ton of virgin plastic emits an average of 1.39 tons of CO_2e.

The formula to compute the net change in emission levels from waste management and mismanagement for each country c is given by:

$$\Delta CO_2^{w,c} = \underbrace{(P^c_{post} - X^c_{post} + M^c_{post}) * (j^c_{post} * b^c) * 2.40}_{\text{Emissions from Open Air Burning of Plastic Waste post ONS}}$$
$$+ \underbrace{(P^c_{post} - X^c_{post} + M^c_{post}) * (1 - j^c_{post} * b^c) * 0.02}_{\text{Emissions from Managed and Dumped Plastic Waste post ONS}}$$
$$- \{ \underbrace{(P^c_{pre} - X^c_{pre} + M^c_{pre}) * (j^c_{pre} * b^c) * 2.40}_{\text{Emissions from Open Air Burning of Plastic Waste pre ONS}}$$
$$+ \underbrace{(P^c_{pre} - X^c_{pre} + M^c_{pre}) * (1 - j^c_{pre} * b^c) * 0.02}_{\text{Emissions from Managed and Dumped Plastic Waste post ONS}}$$

where:

- P^c_{post} and P^c_{pre} are the amounts of domestic plastic waste produced in country c post- and pre-ONS policy, respectively.
- X^c_{post} and X^c_{pre} are the exports of plastic waste from country c post- and pre-ONS policy, respectively.
- M^c_{post} and M^c_{pre} are the imports of plastic waste into country c post- and pre-ONS policy, respectively.
- j^c_{post}, and j^c_{pre} represent the proportion of mismanaged plastic waste post- and pre-ONS policy, respectively.
- b^c represents the proportion of open air burning of plastic waste as a share of mismanaged waste pre-ONS policy.

The values for each of these are referenced in Table A9.

Solving the change in CO_2 emissions from mismanaged/managed waste for each country group, we obtain:

$$\Delta CO_2^{w,c} = \begin{cases} 0.4 & \text{if } c = \text{Turkey} \\ -5.7 & \text{if } c = \text{China} \\ 1.0 & \text{if } c = \text{RoW} \\ 4.6 & \text{if } c = \text{Exporters} \end{cases} \tag{D.5}$$

TABLE A9. Variables

Variable	Turkey	China	Exporters	RoW
P^c_{pre} (million tons)	0.96	21.60	68.10	141.00
P^c_{post} (million tons)	0.96	21.60	68.10	141.00
X^c_{pre} (million tons)	0.02	0.03	9.55	4.76
X^c_{post} (million tons)	$e^{0.36} * 0.02 = 0.03$	$e^{-0.08} * 0.03 = 0.03$	$e^{-0.72} * 9.55 = 4.64$	$e^{-0.44} * 4.76 = 3.05$
M^c_{pre} (million tons)	0.10	7.35	4.57	3.05
M^c_{post} (million tons)	$e^{1.10} * 0.34 = 0.29$	$e^{-4.25} * 19.4 = 0.10$	$e^{0.13} * 9.97 = 5.17$	$e^{0.18} * 6.41 = 3.65$
$j^c_{d,pre}$	47.0%	25.3%	5.5%	48.0%
$j^c_{d,post}$	52.0%	23.9%	5.5%	48.4%
b^c_d	43%	45%	30%	35%

Notes: Data on domestic plastic waste production, P^c, and plastic mismanagement shares, j^c are obtained from the OECD Plastic Outlook. Data on exports and imports of plastic waste, X^c and M^c, is obtained from UN Comtrade.

We have calculated the impact of waste trade on CO_2 from waste management and waste mismanagemet and now we turn to the offsetting impacts from virgin plastic substitution. The impact on virgin plastic usage after the ban would depend on whether recycled plastic waste imports substitute for virgin plastic in production. To formally test this substitution effect for Turkish firms, we analyze their imports of virgin resin and polymers post-ONS.[44] Our analysis reveals no significant change in these imports. These firm-level findings are further supported by our country-level regressions, which also show no significant change in virgin plastic imports following the ONS ban (see Table **??**). Therefore, for Turkey, we conclude that there is no substitution between virgin plastic and recycled plastic waste.

For China, Sun and Tabata (2021) found that the 99.8% reduction in plastic waste imports was accompanied by a 10% increase in virgin resin production. This suggests that a 1% decrease in plastic waste imports leads to a 0.1% rise in demand for virgin resin. We consider this as an upper bound, as the increase in virgin resin documented by Sun and Tabata (2021) could also be driven by factors such as price fluctuations or changes in aggregate demand, independent of the ONS policy. To account for this uncertainty, we conduct a sensitivity analysis on the elasticity. Specifically, we examine the GHG impact of the ONS policy under

[44]It is important to note that Turkey imports coal, gas, and petroleum as primary energy sources.

different elasticity scenarios: 0 (as observed for Turkey), -0.02, -0.04, and -0.1 for China, Exporters, and RoW.[45]

To determine the pre-ONS consumption levels of basic plastics, we turn to data from EXIOBASE. This data provides basic plastic consumption expressed in million euros. Using the 2016 commodity price of basic plastic, valued at 1410 Euros per ton we convert the EXIOBASE basic plastic consumption from euros to tons.

The change in CO2 emissions from virgin plastic consumption, assuming an elasticity of -0.02 is calculated as:

$$(\mathrm{e}^{0.13}-1)*(-0.02)*\frac{(231{,}960.8\text{million euros})}{1410\text{Euros per ton}}*(1.39\text{ton CO}_2)=-0.7\text{million tons CO}_2 \quad \text{if } c=\text{Exporters}$$

$$(e^{0.18}-1)*(-0.02)*\frac{(239{,}147.9\text{million euros})}{1410\text{Euros per ton}}*(1.39\text{ton CO}_2)=-1.0\text{million tons CO}_2 \quad \text{if } c=\text{RoW}$$

$$\Delta CO_2^{w,c}=\begin{cases}0.4 & \text{if } c=\text{Turkey}\\ -5.7 & \text{if } c=\text{China}\\ 1.0 & \text{if } c=\text{RoW}\\ 4.6 & \text{if } c=\text{Exporters}\end{cases}$$

Table A10 shows results for CO_2e emissions from each activity across country groups. The findings show a mixed picture of environmental burden by region and activity. Specifically, the ban directly reduced China's environmental burden from waste management. However, whether this reduction outweighs the emissions from increased virgin plastic production depends on the assumed elasticity. If the elasticity is below 0.05, the reduction in emissions from waste management leads to an overall environmental benefit. On the other hand, if the elasticity between virgin and imported plastic is larger than 0.05, the resulting increase

[45]When we follow the gravity instrumental variable approach that we used for calibrating waste mismanagement and estimate a virgin plastic production elasticity with EXIOBASE, we find a highly similar substitution elasticity estimate. A 1 percent increase in waste imports, instrumented with a gravity measure after the ban, translates into a 0.7 percent decrease in primary production of plastic. Results are detailed in Appendix D.2.

in emissions from virgin plastic production dominates, resulting in a net environmental loss for China. Turkey's burden of waste management increased with imports, and there are no offsetting effects from reduced virgin plastic production as the elasticity between virgin plastic and imported plastic is zero for Turkey. The main finding though is of a large increase in environmental degradation from increased waste mismanagement in Turkey. If this had not occurred, estimated emissions from Turkey would have been half of those reported in the Table. Therefore, the bulk of the environmental burden came from reduced abatement in the face of greater import competition in the country. Exporters experienced greater emissions, net of savings from reduced virgin resource use assuming a elasticity less than 0.03.

TABLE A10. Welfare Estimations

Region	Waste Imports ΔX_d (%)	Waste Emm. ΔZ_d (CO2e mmt)	**Virgin Emmisions Δy_d^v (CO2e mmt)** elas = 0	elas = 0.02	elas = 0.04	elas = 0.1
China	-98.6	-5.7	0	2.6	5.2	12.9
Turkey	199.2	0.4	0	0	0	0
Exporters	13.3	1.0	0	-0.7	-1.3	-3.4
Rest of the World	19.6	4.6	0	-1.0	-2.0	-5.1
Global	-38.8	0.3	0	0.9	1.8	4.5

Note: This table displays the percentage decline in banned plastic waste imports for China, Turkey, Exporters and the Rest of the World in the first column. The second column presents the CO_2e emissions resulting from both waste management and mismanagement for each country group. The final four columns provides the estimated CO_2e emissions from virgin plastic consumption assuming different elasticity rates between virgin plastic and imported plastic waste. Detailed calculations for these emissions can be found in Appendix. D.3.5.

Overall, the analysis suggests that global emissions associated with plastic waste increased after China's ONS policy. The trade diversion effect of the ban was greater than its trade destruction effect in aggregate. But the economic and environmental impacts differed across countries. Strikingly, Turkey experienced both economic and environmental losses from the influx of imports that resulted after China banned the most polluting waste imports. Evaluated at the price of carbon credits in the European Union, Turkey's 0.4mmt rise in emissions amount to a loss of about €42 million, which adds an additional 40% to the economic burden from waste revenue losses.

D.3.6. *Sensitivity Analysis.* To evaluate alternative scenarios in global trade and waste management, we conducted two hypothetical analyses concerning CO_2e emissions:

Scenario 1: No Trade Destruction and All Diversion to Turkey. We examine the environmental implications of no trade destruction. Under this scenario, all waste that would have been imported by China is redirected to Turkey. This reallocation would have resulted in Turkey's waste imports increasing by 736%, rather than the observed 199%. Consequently, such a significant shift would have led to an increase in global CO_2e emissions by approximately 7.1 million tons.

TABLE A11. Welfare Estimations with No Trade Destruction and All Diversion to Turkey

Region	Waste Imports ΔX_d (%)	Waste Emissions ΔZ_d (CO2e mmt)	Virgin Consumption Δy_d^v (CO2e mmt)
China	-98.8%	-5.7	2.6
Turkey	736%	10.2	0
Exporters	0%	0	0
Rest of the World	0%	0	0
Global	0%	4.5	2.6

Note: The first column of this table displays the percentage change in banned plastic waste imports for each country group in the case that all plastic waste was diverted from China to Turkey. The second column presents the CO_2e emissions resulting from both waste management and mismanagement for each country group. The final column provides the estimated CO_2e emissions from virgin plastic consumption.

Scenario 2: No Country Composition Effect. Additionally, we assess the potential impact on global emissions if Turkey and the Rest of the World (RoW) adhered to the waste mismanagement levels observed in major exporting countries (primarily, the US and the EU). Under this scenario, if mismanagement practices were aligned, global CO_2e emissions from waste would have decreased by 3.29 million tons, which is 4.49 million tons less than what occurred.

Finally, we conduct two more robustness checks. First, we examine sensitivity of the main findings to alternative parameter values of mismanagement impacts in Exporters and the Rest of the World. We follow the gravity instrumental variables procedure for inferring mismanagement impacts in Turkey from satellite data. We can compare this with the actual mismanagement change in Turkey to obtain a scaling factor for how much emission elasticities translate into actual mismanagement elasticities. Scaling the mismanagement changes for

TABLE A12. Welfare Estimations with Turkey and the Rest of the World at the Mismanagement Levels of Exporters Assuming Elasticity of 0.02

Region	Waste Imports ΔX_d (%)	Waste Emissions ΔZ_d (CO2e mmt)	Virgin Consumption Δy_d^v (CO2e mmt)
China	-98.8%	-5.72	2.60
Turkey	199.2%	0.03	0.00
Exporters	13.3	0.97	-0.67
Rest of the World	19.6	0.50	-1.02
Global	-38.8%	-4.20	0.91

Note: The first column of this table displays the percentage change in banned plastic waste imports for each country group. The second column presents the CO_2e emissions resulting from both waste management and mismanagement for each country group provided that waste mismanagement levels in Turkey and the Rest of the World (RoW) are the same as the mismanagement levels of Exporters. The final column provides the estimated CO_2e emissions from virgin plastic consumption.

Exporters and the RoW with this factor of 1.6 does not alter the main finding, but the global emission savings are reduced to -4 mmt CO_2e. Exporters and the RoW are estimated to have an increased environmental burden, with the latter of about 1MMT of CO_2e. A second point of robustness is that the increased environmental burden of some Exporters within the group may be larger. 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. But the estimates are relative and also apply to all waste (and not just plastic waste).