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The Chinese waste import ban and the emergence of waste havens within Europe

Konstantin Sommer[1]

1 University of Amsterdam, Vrije Universiteit Amsterdam and Tinbergen Institute

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Konstantin Sommer †

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Abstract

We study the implications of the Chinese waste import ban of 2018 on intra-European plastic waste trade. Specifically, we ask if it led to a "waste haven" effect, which would imply that countries with high disposal and recycling costs started to export more plastic waste to countries with lower costs. We study this question in a gravity difference-in-differences setting with detailed data on the costs of waste processing. We find strong evidence that countries with higher costs of disposal indeed started to export more waste towards lower cost countries as a result of this ban. We do not find consistent evidence that more waste was exported to countries with lower recycling costs. Our results raise distributional questions about the allocation of waste externalities in integrated markets and have implications for current debates on the legislation of international waste shipments.

Key words: Trade, Environment, Waste, Circular Economy, Europe
JEL codes: F18, Q53, Q27

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†University of Amsterdam, Vrije Universiteit Amsterdam.

1 Introduction

In this paper, we analyze the effects of an import ban on waste by the Chinese government in 2018 on cross country plastic waste trade within Europe. This import ban drastically influenced global waste trade, leading to large global waste trade diversions, and thus also severely affecting European waste processing and trading. We study if as a result of this shock countries with lower costs of waste processing became "waste havens" for countries with higher costs. This would imply that countries with higher costs, usually corresponding to higher income countries with stricter environmental regulation, outsourced some of their waste processing to countries with these respectively lower costs.

Waste processing can broadly be divided into two main activities: disposal and recycling. Disposal happens mostly in landfills and through incineration. Plastic waste disposal is linked to the release of toxic pollutants (and greenhouse gases) that can be harmful for both the environment and human health (see for a literature overview: Alabi et al., 2019). Even waste recycling, which is clearly preferable for most policy makers, can still entail important health costs for the workers in this labour intense industry (Huang et al., 2022). Within China, Unfried and Wang (2022) and Li and Takeuchi (2023) show that the import ban has reduced both air pollution and ozone emissions significantly, indicating the negative consequences of the waste processing before the ban.

Given these externalities, it is non-surprising that countries regulate these activities and dis-incentivize the disposal in landfills and partly in incineration, both through direct taxation and other regulations. Commonly, and also in the context of this study, countries do this with different stringency, usually implying that higher income countries have higher costs of waste processing. This, however, could induce a phenomena well known in the environmental and international economics literature: that of leakage and pollution havens. This would occur if firms and waste handlers outsource waste disposal and potentially recycling to countries where both of these activities are cheaper than at home. The EU has considered that possibility and passed a waste shipment legislation as early as 2006. Based on this, exports of waste to non-OECD countries is regulated and completely banned for waste destined for disposal. It is, however, not banned to trade waste within the OECD and waste flows relatively freely between EU and OECD countries. Regulation for waste processing, and especially the costs of it, however, vary widely between European countries, which leads to the question if curtailing exports to outside the block can lead to the emergence of waste havens here.

In this paper we study the potential emergence of such waste havens within Europe as a result of a shock to the global waste trade system that drastically and unexpectedly increased the amount of waste that had to be processed within Europe. This sudden increase comes from an import ban on waste goods by the Chinese government in 2018, which was preceded by a crackdown on such imports from March 2017 on. China was up to this point the by far biggest importer of plastic waste, importing about half of all globally traded plastic waste and about half of the plastic waste that left Europe. Global plastic waste exports to China reacted sharply, dropping throughout 2017 and then reaching almost zero in early 2018. This implied that waste handlers had to suddenly find new places for waste processing, and due to the relatively strict export regulations, most of the EU waste had to remain within the OECD, where waste handlers now had to reallocate the additional waste.

We study the effect of this ban on intra-European waste trade and analyze if the additional waste was traded based on the differences in waste processing costs between exporters and importers. The pollution haven hypothesis would naturally predict that more waste was now traded from countries with high disposal and recycling costs towards countries with respectively lower costs. We thus ask if we can observe a waste trade diversion from China towards new waste havens within Europe. We focus the study on plastic waste, as this was the by far most affected type of waste by the ban.

In addition to the effects of waste processing costs, both for disposal and recycling, we also study if waste disposal capacities influenced waste trade after the ban. We do this, as there are many reports that indicate an increase in waste disposal and an oversupply of waste destined for disposal in some countries in Europe after the ban (Tamma, 2018). We will therefore also study if countries with lower disposal capacities started to export more waste to countries with higher capacities. This is further motivated by large differences in disposal capacities and some anecdotal (see for example: Kazin & Bounds, 2023) but no academic evidence on their general importance for waste trade.

In contrast to most studies on pollution havens, our focus is not on how an increase in processing costs influences waste trade, but on how a sudden increase in the amount of waste that has to be processed within a set of countries is allocated among them. This allows us to study if pollution havens can emerge based on differences in the levels of processing costs instead of as a result of changes in processing costs. This helps us to avoid a likely endogeneity problem, as higher waste imports themselves could lead to changes in processing costs or capacities, therefore implying a reverse causality problem.

Our identification builds on estimating the effect of the waste ban on intra-European waste trade and then differentiating this effect by country-pair characteristics. For this, we also include data on trade in other goods than waste to proxy for waste trade in the absence of the import ban in the period thereafter. We thus apply a difference-in-differences strategy, with a treated group, i.e. waste trade, and an untreated group, i.e. comparable trade flows, where we estimate heterogenous treatment effects, based on country-pair characteristics linked to the outlined waste haven characteristics. We control for unobserved heterogeneity with theoretically motivated control variables and detailed fixed effects in order to make causal claims about our estimates. We are not aware of another study in the literature on pollution havens that is able to exploit an identification strategy that is based on a sudden and exogenous increase in the amount of goods associated with negative externalities.

In the literature on pollution havens it is traditionally very difficult to observe the direct costs of an environmental externality faced by firms in different countries, which we see as a great advantage of our study, as we have access to the costs of waste disposal in landfills for all countries in our sample. Observing disposal capacities is another novelty in the study on waste trade and waste havens.

We start the paper with a small institutional background on global and European trade in waste, show the distribution of our variables of interest and show the striking effects of the Chinese waste ban descriptively. Following this, we provide a small theoretical section in which we outline the important drivers of waste trade and streamline the thinking for our empirical estimation.

For the main focus of the paper, the empirical analysis, we compile a panel of monthly bilateral trade data on an HS6 level for all countries in the EU, Turkey, Norway and the UK between 2011 and 2019. We are able to select the affected waste goods based on the Chinese announcement of the ban to the WTO (China, 2017), which comprises a list of these HS codes. To measure the cost of waste disposal, we use landfill costs as a composite of taxes and fees and proxy for the costs of recycling by using wages in the waste processing sector. Disposal capacities are a combination of landfill and incineration capacities. Our panel also contains data on theoretically motivated confounders.

Based on this panel, we try to estimate the causal effect of the import ban on bilateral intra-European waste trade; differentiated by the differences in processing costs and disposal capacities between exporter and importer. For expositional ease, we first estimate the effect in a sample containing only waste trade, and study how our three variables of interest affected the effect of the ban. We do this in a gravity-style regression setting in which we interact the dummy-indicator for the import ban with our three continuous variables of interest. We then introduce a control group to the analysis that controls for general country pair dynamics and moves the analysis towards a difference-in-differences (DiD) estimation with heterogenous treatment effects. We include analyses with three different control groups that comprise bilateral trade in goods that were unaffected by the import ban. We include placebo test for parallel trends for all three of them and come to similar results, independent of the choice of the control group.

Our analysis leads to several conclusions. Most importantly, countries with higher landfill costs exported significantly more plastic waste to countries with lower landfill costs as a result of the waste ban. This relationship is significant throughout various specifications and robustness tests, as well as economically sizable. It indicates that a country pair where the exporter had a 2.5 times higher landfill cost than the importer, which is about the mean difference in landfill costs between countries, traded about 20 percent more waste in the direction of the low cost country as a result of the import ban. When studying this effect over time, it is clear that this difference only starts to matter throughout 2017 and becomes significant as soon as the waste ban fully takes effect in January 2018.

Related to this, we show that this increase in waste exports towards countries with lower disposal costs, went together with a decrease in the price of this waste. This indicates a reduction in the quality of that waste, further supporting the conclusion that countries with lower landfill costs started to take over some of the disposal activities for higher cost countries. It is important to note that countries with lower landfill costs are often lower income countries, raising distributional concerns on the outsourcing of waste externalities.

For the question if higher recycling cost countries started to export more waste to countries with lower recycling costs, the evidence is less robust and not always statistically significant, but the effect is also apparent here.

Our second main result is that countries with low disposal capacities started to export more waste to countries with higher disposal capacities. These countries are not necessarily the same as the countries with low landfill costs. This lends evidence to the hypothesis that more waste was disposed after the import ban, as less recycling opportunities were available.

We show that the most affected country by this ban was Turkey, which took over a lot of the waste processing for EU countries. Separating the effect for Turkey from the general effect, however, shows that the coefficient estimates remain statistically and economically significant.

Taken together, our results have strong policy implications. They imply that restricting waste exports can imply an increase in waste disposal as fewer recycling opportunities are then available. It is also likely that this leads to distributional concerns, as countries with lower costs for disposal can become waste havens for countries with higher costs. Importantly, this usually would also imply a waste transfer from higher to lower income countries. This should be an important consideration for the current reform of the EU waste shipment directive. While the directive might prevent the outsourcing of waste processing to non-OECD countries, it would not prevent the emergence of waste havens within Europe, and might in fact even make such effects more likely.

Our results also hold lessons for waste regulation in general. Higher taxes on waste disposal can be circumvented by exporting the waste and if these regulations are not coupled with further incentives for recycling it is not guaranteed that recycling will increase if waste leakage is strong enough.

Literature Our paper contributes to a small literature on waste trade, which has started with Copeland (2001), who builds a theoretical trade model that features legal and illegal waste disposal. He postulates that higher disposal taxes can lead to an increase in illegal disposal and exports of disposable waste to other countries.

Kellenberg (2012) is then the first to study the presence of " waste havens" by analyzing the effect of changes in environmental stringency on the waste trade between countries in a gravity panel estimation. He finds evidence for this. Higashida and Managi (2014) studies these waste haven effects in a more narrow setting by looking at recyclable wastes only with a similar estimation strategy as Kellenberg (2012). They find evidence that countries with lower wages can become waste havens for countries with higher wages. Our approach is related to both of these studies, as they are also using a gravity style framework with country differences in waste processing costs as the main explanatory variable. Our estimation strategy, however, is very different, as we use exogenous variation in the amount of waste that had to be treated and study the effect of such an increase on the distribution of waste between countries. We are also the first to actually observe disposal costs and capacities. Our results align with Kellenberg (2012) in that we also find evidence for waste haven effects, but when leaving out the exogenous variation of the import ban and focussing on the marginal effect of changes in environmental regulation on waste trade as he does, we cannot find evidence for it. In alignment with Higashida and Managi (2014), we also find some evidence for a waste haven effect for recycling, but the evidence is less strong than for disposal and is also absent when only studying the marginal effect of changes in recycling costs, as the authors do. Our results align with the theoretical predictions of the two papers, and show that when exploiting exogenous variation in the waste supply, we can establish causal evidence for the emergence of waste haven effects; especially driven by disposal and not necessarily by recycling motives.

Balkevicius et al. (2020) uses an earlier trade restriction on waste imports in China to study if this led to an increase of low quality waste exports to South East Asia, but finds no evidence for this. Our results indicate strong effects from the 2018 import ban, which is not at odds with their results, given the larger scale policy in our study compared to that in their study.

Our paper also contributes to a small literature that specifically looks at the Chinese import ban and the question about where waste would be redirected to has already been asked before the effects of the ban were visible (Brooks et al., 2018), as it was feared that one waste haven would be replaced by other ones. The ban has been shown to have reduced global waste trade in general (Wen et al., 2021), and it appears that a lot of waste was, at least initially, redirected towards South East Asian economies (descriptively shown in Wang et al., 2020). This is in line with our finding of waste havens within Europe.

In general our study falls into a larger literature on trade and the environment that tries to estimates leakage effects and pollution havens, by studying if stricter environmental regulations can lead to an outsourcing of the environmental externality. The seminal contribution in this field is by Antweiler et al. (2001) who provided the theoretical foundations for many of the upcoming studies. More recently economists have studied, for example, the effects of EU carbon trading (see for an overview Ekins et al., 2023), or of US environmental regulation (Shapiro & Walker, 2018). As exogenous policy variations for actually binding policies are rare, evidence for pollution havens so far is not as striking as it could theoretically be expected. More recently, studies of concrete environmental regulations have shown stronger effects, as for example in Tanaka et al. (2022) who show that stricter regulations on battery recycling in the US have moved much of the industry to Mexico, where infant health has worsened around these recycling facilities. In this literature we are not aware of another study with a similar identification strategy as ours, exploiting a sudden and exogenous increase in an externality that had to be distributed over countries.

In Section 2, we will give an institutional background and present some stylized facts about waste trade and waste treatment costs. In Section 3, we then give some theoretical background to motivate our empirical specification, and in Section 4, we describe the underlying data sources. Section 5 describes our estimation strategy, Section 6 our results and Section 7 and Section 8 present several more detailed analysis and robustness tests. Section 9 concludes.

2 Background and stylized facts

This section gives an overview over international waste trade, the important policies behind this study, and some first descriptive motivations for the main research question of this paper: Did the Chinese waste import ban lead to an increase in exports of waste to countries with low waste processing costs and high disposal capacities within Europe? The underlying data sources for the plots presented in this section are described in Section 4.

2.1 International shipments of waste

Most of global waste is handled domestically, but nevertheless millions of tonnes of waste are shipped between countries every year. The reasons for this vary, but as waste is either disposed, mostly in landfills and through incineration, or repurposed, through recycling or reusing, it is mostly determined by where these activities can be done most cheaply or where there is the highest demand for recycled materials. For certain types of waste, especially hazardous ones, it can also be based on the fact that disposal capacities are too small or none existing in some countries to properly dispose the waste domestically.

Waste is a non-homogenous good with different types of waste having different recycling properties and the contained materials can have very different economic values, thus determining how much effort is put into recycling it. For example, aluminum scraps and plastic waste have a very different economic value and different plastic wastes are themselves very differently recyclable. The quality of waste collection and sorting in a country additionally determines how easy it is to recycle municipal and industrial waste.

The trade of hazardous wastes has internationally been regulated since the Basel Convention, which entered into force in 1992. The convention set up a regulatory framework, under which the export of hazardous waste has to be notified to, and approved by, the importing country. By now most countries (but not the United States) have ratified the agreement. Kellenberg and Levinson (2014), however, find almost no evidence that it reduced the amount of international waste shipments.

2.2 European and OECD waste trade

The OECD and the European Union have based their waste shipment regulation originally on the Basel Convention, but especially the EU has adopted much stricter regulations since. Already in 1992, the OECD set up a control system to differentiate between low and high risk wastes, but they have not yet went beyond regulating shipments of hazardous wastes.

The EU adopted its waste shipment directive in 2006. Besides several additional notification and control mechanisms, this legislation also implemented several trade restrictions that go beyond hazardous wastes. Most importantly it bans the export of any waste destined for disposal to any non-OECD country, and any waste exports to non-OECD countries fall under stricter control and notification regulations than those within the OECD. Additionally, imports of waste meant for disposal are only allowed from OECD countries, and the export of hazardous and "other wastes", as defined in the Basel convention, is forbidden to non-OECD countries. As a consequence of this regulation most exports of EU wastes are already going towards other EU or OECD countries.

The waste shipment directive is currently in a reform process and will likely become stricter with respect to non-EU waste exports; this is also meant to protect developing countries from becoming havens for EU waste. What is, however, not considered is that exports within the block can also be driven by waste haven objectives, which is what this study is going to investigate. This idea is motivated by the sharp differences both in disposal and recycling regulations and costs as well as disposal capacities between member states and neighboring OECD countries, as can be seen in Figure 1.

As a measure of regulation cost, we plot the landfill tax charged by the government for the disposition on one tone of waste in a landfill in 2016 in panel (a). Non-surprisingly, richer economies dis-incentivize waste disposal more than poorer countries do, as can be seen by a concentration of high taxes in central and northern Europe, with some exceptions. These are notably Germany and Norway, which both do not have any tax in place any more.

As we will both study the effect of disposal and recycling costs (so in general: processing costs), we also present the same descriptive plots for our proxy of recycling costs, the wage in the recycling sector, in Figure 1 panel (b). We chose this measure as recycling remains a very labor intense process (it has the 7th highest labor share

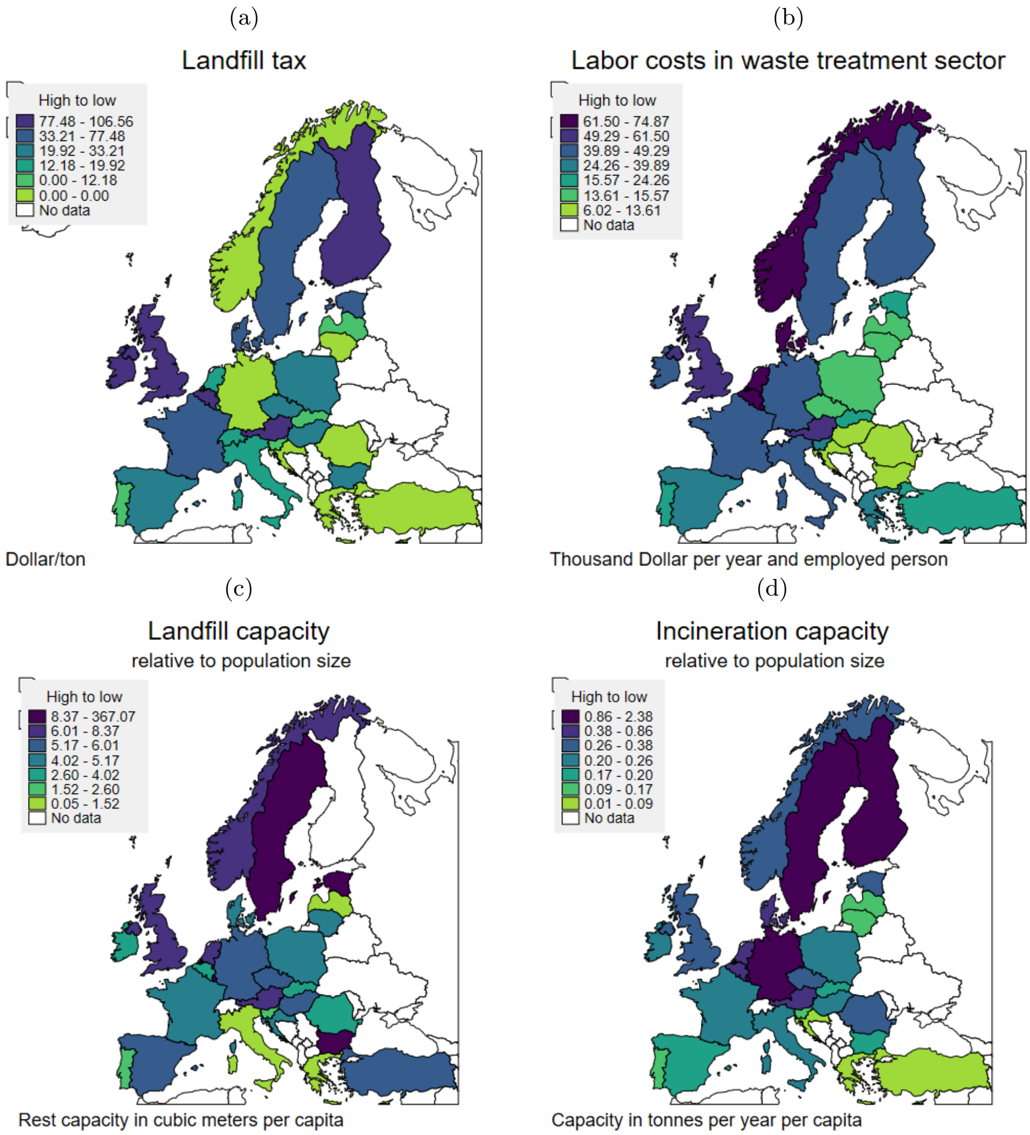

Figure 1: Distribution of main variables of interest in 2016

of all 30 manufacturing sectors based on EU averages, computed from Eurostat data), and because Higashida and Managi (2014) describe the wage rate as the determining factor for recycling cost differentials between countries. Not-surprisingly, wages are considerably higher in North-Western Europe than in the South and East, implying that recycling is much cheaper there than in high wage countries.

We additionally plot the distribution of landfill and incineration capacity by each country, relative to its population size in the same figure in panels (c) and (d). We can see that those again vary substantially by country and the pattern here is quite diverse. Small countries, for example, have relatively low landfill capacities, while Turkey and Sweden still have a lot of it. Incineration is especially prominent in Central and Western Europe, highly correlating with income levels.

2.3 Chinese importance for global waste trade and its import ban

Until 2017, China was, by far, the most important importer of global waste, especially made of plastic and paper, as can be seen in Figure 2. This had both to do with the exporting countries and China itself. Low wages and a large manufacturing base in China presented a cheap opportunity for Western countries to recycle waste there, and at the same time the growing Chinese economy was in need of materials to fuel their manufacturing sector. An additional reason that has often been promoted relates to cheap shipping costs. Many containers from western countries would have returned empty towards China, and so filling them with waste was almost free.

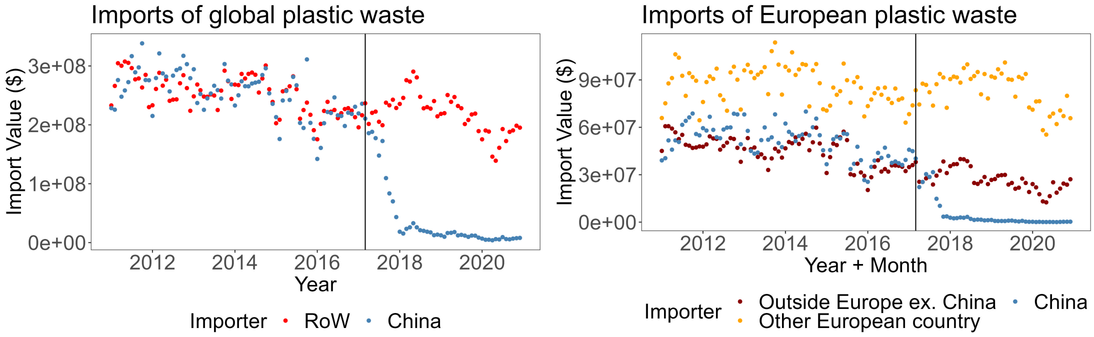

Figure 2: Plastic waste trade by importer

Note: Left panel plots all global plastic waste exports by importer and right pane all European waste exports by importer. Vertical lines indicate March 2017, i.e. the start of the waste import restrictions. Ban went into effect in January 2018.

However, a lot of the waste that was labelled as recyclable was in fact mixed with other unrecyclable wastes or was so contaminated that recycling proved impossible or far too expensive, leaving China with the waste that had to be disposed locally then. There are also widespread reports on detrimental health and environmental effects from this waste processing (for example Unfried & Wang, 2022).

In early 2017, China then started to surprisingly crack down on these imports under the newly announced operation "National Sword", citing environmental and health concerns as much of the waste labelled as recyclable was in fact mixed with other non-recyclable wastes. Under this operation the authorities originally started to go after illegal imports of wastes and permit fraud, but reports indicated that they also soon started to target imports of lower quality waste and in the summer exporters reported a sharp increase in importing fees. In July, China then informed the WTO of its intent to ban certain waste imports, which was then confirmed on July 18th, when China announced that it would ban almost all imports of plastic waste, several types of paper waste and some additional types of textile wastes and metal scraps starting from January 2018 (see China, 2017).

Waste exports to China plummeted throughout 2017 and then reached almost zero for plastic wastes, as can be seen in Figure 2. This also had tremendous effects for European exporters that had sent half of their non-Europe destined plastic waste to China up until this point, see Figure 2 panel (b). One can, however, also see that waste imports by other European countries did not pick up the whole decline in Chinese imports, indicating that more waste was absorbed domestically.

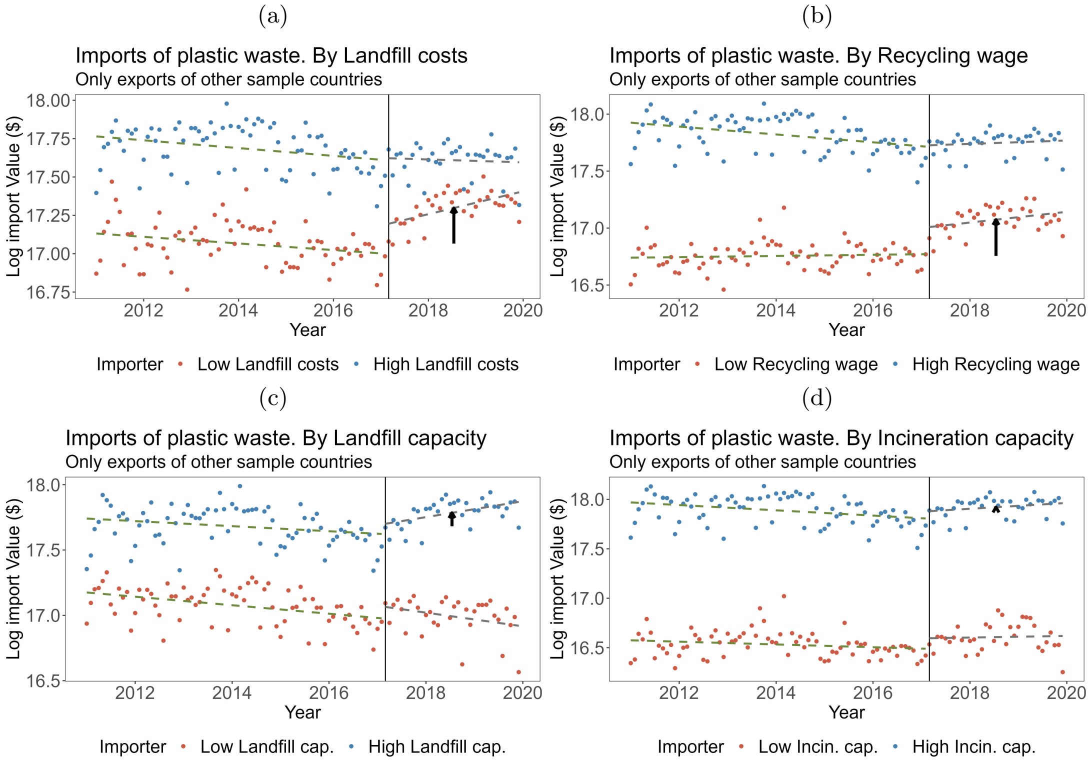

Figure 3: Development of waste imports for different country characteristics, split at the median for each variable.

Note: Vertical lines indicating March 2017, i.e. the start of the waste import restrictions. Ban went into effect in January 2018. Trend lines are included by country group and are based on pre and post restriction periods respectively. The black arrows indicate the average increase in waste imports after the ban for the group for which we expect an increase.

This does not imply that there were no changes to the patterns of trade within Europe, which is what we are showing in Figure 3, where we show the relation between our measures of interest and waste trade within Europe. For this, we split all countries in our sample at the median of our waste haven characteristics, respectively for one variable in each panel. In panel (a) for example, we do this for landfill costs and split countries into cheap and expensive landfill countries and plot the waste imports of both groups over time. We then indicate the average increase in waste imports after the ban for the country group for which we expect an increase, so in panel (a) for example for the countries with lower landfill costs. This depicts our research question visually, as it shows the uptake of waste imports by that respective group.

In panel (a), we can see that the uptake in waste imports within the sample is to the biggest extend based on the uptake of imports from low landfill costs countries, while high landfill cost countries do not see any increase in waste imports after the ban. The same can be observed for recycling costs, but to a less pronounced extend. It also looks as if countries with a higher than median landfill capacity also imported more waste after the import ban, this is less strong for countries with a higher incineration capacity, where in fact countries with a low capacity imported a bit more.

These are of course pure correlations and our empirical analysis will aim to disentangle these effects. The figures, however, already help to point to some potential caveats of an empirical estimation. These are among others: how much of the effect is driven by individual countries, how important is a shift in the trend compared to a level change and how much of the variation is left once we control for other factors. Our estimation and robustness tests aim to address all of these concerns.

3 Theoretical foundations

This section serves to outline the general ideas and mechanisms that are useful to understand and describe the underlying determinants of waste trade. These will then be used to motivate our empirical specification and to justify our choice of (control-) variables. We summarise all of the important determinants in Figure 4.

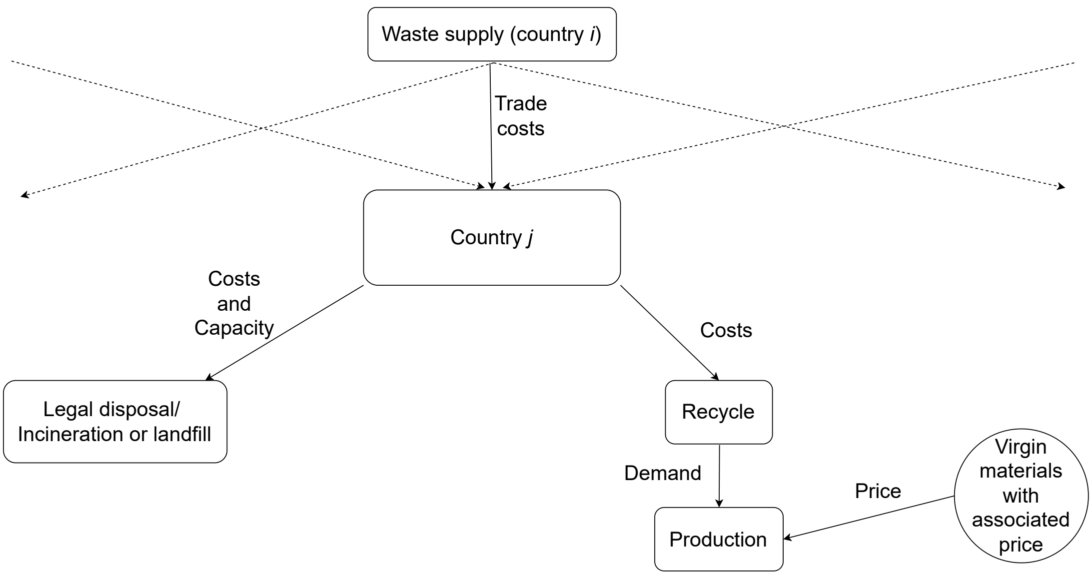

Figure 4: Stylized overview of determinants of waste trade.

3.1 Waste supply

Consumption and production in each country create waste. The share of waste that is recyclable is determined both by the quality of waste collecting and sorting as well as by production and design requirements or other regulation in the form of EPR schemes.[1]

Waste is collected and sorted by waste collectors, who can be municipal corporations or industrial organizations, and who (for simplification) supply this waste to domestic and foreign waste handlers, as can be seen at the top of Figure 4. The goal of the waste supplier is to minimize the costs of its waste processing. This implies that the supplier will choose the combination of recycling and disposal options that minimizes the costs of disposal and maximizes the revenues from recycling. These

[1]It is noteworthy that recyclable waste can of course be disposed if the price that one would get for selling waste to a recycler would become sufficiently negative or if there were not enough recycling opportunities available. In fact, our study will also shine some light on this possibility as a result of the Chinese import ban.

options could of course be spread over different countries leading to a reason for trade in waste, say from country i to country j.

This decision to export is thus determined both by the costs of shipping the waste abroad, including some potential tariffs and non-tariff barriers, as well as by the differences in disposal costs and recycling revenues in different countries. These factors will be the explanatory variables in our estimation, and so we will explain those in more detail in the following subsections.

The supply-side quality and collection policies are beyond the scope of this paper, in which we focus on the demand aspects of waste trade, but it shows that it will be important to control for country-specific characteristics in the supply of waste during the estimation.

3.2 Disposal

(Legal) disposing in practice implies either incineration or landfilling. Both options have some costs of operation, and incineration can sometimes be used for energy creation where the operator can gain some revenue. Country-specific regulation can increase the costs of disposing waste either through directed taxation or other regulatory requirements. In the EU, one of the main policy tools in this regard are unilateral taxes on landfilling, which are very prominent and aim at dis-incentivising the disposal of waste.

Disposal operators also charge a price for their disposal services, a so-called "gate fee" for landfills. Disposal is additionally affected by the remaining capacities for it. One of the prominent drivers of within Europe waste trades for disposal is based on the fact that some countries do not have sufficient capacities to dispose waste (Kazin & Bounds, 2023). This can either be because certain wastes can only be disposed in certified facilities, or because the country produces too much waste for the domestic waste processing system.

The trade of disposable waste between countries thus depends on the costs, including the regulatory costs, of disposal in different countries, and potentially on the capacities of disposal in the trading partners, as highlighted by the left side in Figure 4.

3.3 Final goods demand and recycling

Recycling is costly, and these costs can vary substantially between countries (da Cruz et al., 2014), with labor costs and thus local wages as a fundamental determinant of these recycling costs (Genc et al., 2019). Recyclers, however, can sell the recycled goods to a final producer who can use the recycled material as input in production.

This final producer can also substitute recycled materials with raw or virgin material, say iron ore instead of recycled iron. The price of this substitute together with the final demand for their product will thus determine the demand for recycled material and will therefore also determine the potential revenues for a recycling company.[2] All of these aspects we depict on the right side of Figure 4.

How much recycled material is demanded from each country thus depends on the final demand for materials, which itself depends on the availability or price of virgin materials and how easy it is to substitute from virgin materials to recycled materials; and on the costs of recycling in that country. The trade of recyclable wastes between

[2] These two companies could also be vertically integrated, with the same logic applying.

countries therefore depends on both the supply of waste in the exporting and the demand for recyclable waste in the demanding country.

3.4 The Chinese import shock, moving towards the estimation strategy

In this section we stretched how various factors can drive the direction of waste trade, but the challenge now becomes how to estimate the actual effect of these factors on waste trade. Using changes in the respective factors over time (as is commonly done in standard panel estimations) might lead biased results if policy makers and companies react to waste trade itself and adjust the costs and capacities of waste handling accordingly. In this paper we will thus rely on differences in the levels of these variables at one moment in time, namely when China stopped importing waste.

The breaking off of one trading partner implied that waste suppliers had to find new places for waste processing, and, as we argued, these considerations will among others have been based on the respective disposal and recycling costs as well as on the available disposal capacities in both exporting and importing countries.

We will therefore estimate the effect of the import ban on bilateral waste trade, and differentiate the effect by the differences in our factors of interest between exporter and importer. This will answer the main question of this paper, namely if waste was exported more to countries with low processing costs (i.e. waste havens) and countries with high disposal capacities within Europe as a result of the Chinese import ban.

As we described in this section, there are, additionally, several other factors that could determine waste trade and also the direction and amount of it after the import ban. Our empirical specification will thus have two main tasks, namely to (1) control for these other factors, such that they do not bias the estimation, and to (2) find good measures of processing costs, as to properly estimate this channel.

The fact that several of the described factors will be constant over either time or space will allow us to control for them in fixed effects, but for others, like the final demand for materials, or the commodity price of virgin materials, it might be crucial to control for with adequate measures. The next section will describe the measures that we use for this.

4 Data

Our main variable of interest is based on monthly bilateral trade data on an HS-6 level obtained from Comtrade. These data contain information on both trade values and weights. We restrict our sample to all trade between countries that are members of the EU-Turkey customs union, Norway and Great Britain.[3] Waste trade is then defined as the HS6 codes that were explicitly banned by the Chinese government, which comprises all plastic waste, but not all paper waste. For paper waste, we therefore exclude waste goods (based on their HS6 codes) that were not banned from the analysis, as these were still targeted by adjacent regulations. For plastic waste these banned HS codes comprise "Waste, parings and scrap, of" three different kinds of plastic polymers and an "other" category. See Appendix A for an overview of these HS codes.

[3] We are happy to share the code used for communicating with the Comtrade API to receive the desired data.

To measure the costs of disposal, we comprise a measure of landfill costs. Landfill tax data is uniquely available within Europe and is mostly based on data that we have obtained from the Confederation of European Waste-to-Energy Plants (CEWEP). CEWEP has gathered data on landfill taxes and average gate fees, i.e. the fees charged by owners of landfills, since 2011. We amend their data with information obtained from the European Environmental Agency (EEA) that has compiled multiple country assessments on the state of domestic waste handling both in 2013 and 2023 on all EU countries, and in 2013 also on several adjacent countries (European Environment Agency, 2013; European Environmental Agency, 2023). Our gate fees are based on these EEA reports, and are thus fixed in time, and the landfill tax rates are based on CEWEP and therefore updated every year. Compiling the data into a panel is based on certain adjustments that are documented in detail in Appendix A.

We measure recycling costs by the labor costs in the recycling industry. Wages in the waste treatment sector are taken from both Eurostat's national account data and the OECD STAN database. We use the wage rate in the sector "waste collection, treatment and disposal activities; materials recovery", as this is the closest that we can get to the materials recovery (recycling) itself. We divide total labour costs by the number of employees to get average wages. Where available, we rely on Eurostat data, but amend it with OECD data for countries that did not report to Eurostat. Eurostat and OECD data are almost identical for all countries for which both data sources are available.

Landfill and incineration capacities are taken from Eurostat and are given in cubic meters or tonnes per year respectively. We interpolate linearly between years, as data is only reported biannually. As it does not fluctuate much from year to year, this is an inconsequential adjustment. Incineration capacity contains the capacity that can be used for energy recovery through waste incineration. In the main analysis we combine landfill and incineration capacities into one measure that tries to capture how much could be disposed within one year in that country. We do this by converting the remaining landfill capacity into incineration capacity, based on a conversion by the US EPA that allows to convert both into a weight equivalent. Additional details can be found in Appendix A.

Industry output in the plastic and paper producing sectors is used to capture final demand for recycled material as well as the supply of waste and is taken from both Eurostat and the OECD again, and the choice between the two is made in the same way as for the recycling costs. We use the output in the sectors "Rubber and plastics products" and "Paper products" for paper waste and convert them into million dollars. Prices of substitutable virgin materials are taken from the OECD's "Merchandise Trade Price Index Database by CPA". We use the Laspeyres index and chose "rubber and plastic products" and "paper" respectively for the two waste types. Total waste generation by industry, which is used for the scaling of disposal capacities later on, is taken from the OECD waste generation database by sector, where we chose the total of all sectors.

Where necessary, we convert all monetary values into current year US Dollars using OECD exchange rate data, and capture remaining price trends in yearly fixed effects. We keep values in nominal terms, as this should be the determinant of cost-benefit calculations for waste handlers choosing between different countries. We later on control for purchasing power differences that we also obtain from the OECD. All additional data choices and transformations are described in Appendix A.

After all these choices we are left with a panel of 30 countries from 2011 to 2019 on a monthly basis. Focussing purely on the waste trade, this leaves us with around than 244,000 observations for plastic waste and 36,000 for paper waste. In Table A4, we summarize the relevant variables by year.

5 Estimation

This section translates our research question, namely if the Chinese import ban led to the emergence of waste havens within Europe, into an empirical model. We first describe a general relation between the waste ban, the waste haven variables and waste trade, and then gradually extend this relation to our baseline specification, which captures the effect of the waste ban through our three variables of interest. The first step in this approach nests a relation usually studied in the literature. We also discuss identification and estimation concerns and how we address those.

5.1 Motivating our baseline specification

We start with presenting a model that captures the general relation between waste trade, the import ban and the waste haven variables. Aligning with other waste trade papers, we focus on trade values in the main specification, but will study waste weights and prices in the discussion section. Assuming a constant elasticity model, and thus aligning with most of the theoretical and applied gravity literature (see for a current example and discussion Nagengast & Yotov, 2023), we estimate the relation between export EX, of waste good g, on an HS6 level, from country i to country j in month m[4], and the explanatory variables that vary by year y by:

$$EX_{ijmg} = exp[BAN_y \beta + C'_{ijy}\gamma + Z'_{ijyg}\alpha + D_{ijyg}] * \varepsilon_{ijmg} \ . \tag{1}$$

This specification also allows the inclusion of zero trade flows, that are frequent in our detailed panel. Section 5.2 will extend this relation to including the trade in non-waste goods as a control group.

The focus of this study is the effect of the waste ban, captured by the dummy variable BAN_y, which is zero until 2017 and turns one in 2018, on waste exports. We will differentiate its effect by the country characteristics linked to the three waste haven channels. We do this in three steps, by refining β in (1) in three increasingly detailed ways:

$$\beta = \beta_{0hom} \tag{1a}$$
$$\beta = \beta_{0imp} + C'_j \beta_{1imp} \tag{1b}$$
$$\beta = \beta_0 + C'_{ijy}\beta_1 \ , \tag{1c}$$

with

$$C_j = [LCost_j \quad RW_j \quad DCap_j] \ ,$$
$$C_{ijy} = [LCost_{ijy} \quad RW_{ijy} \quad DCap_{ijy}] \text{ , and}$$
$$\beta_{1imp} = [\beta^{LCo}_{1imp} \quad \beta^{RW}_{1imp} \quad \beta^{DCa}_{1imp}]$$

[4] We use bilateral monthly trade data, as we will use it to control for further factors later, but using aggregate annual data does not alter the results or changes the significance of any of the results.

and similarly for β_1.

In (1a) we purely estimate the effect of the import ban on intra-European waste trade. Even though we know that the import ban increased the amount of waste that had to be handled within Europe, we do not know if it increased the total amount of waste that was traded within this market, as more waste could have been absorbed domestically. The expected coefficient sign for β_{0hom} is thus a priori ambiguous.

We then move closer towards our research question, by studying if some countries started to import more waste after the import ban than others, based on their waste haven related characteristics. We study this divergence of trade flows towards countries with lower processing costs and higher disposal capacities in (1b). The variables contained in C_j are importer-specific variables that capture each of the waste haven characteristics in the year before the import ban, 2017. These variables are $LCost_j$, capturing the costs of landfilling, RW_j, capturing the recycling wage and thus proxying for recycling costs, and $DCap_j$, measuring the total waste disposal capacity. We include all three variables in logarithmic form.

We expect that the ban led to an increase of imports in countries with low waste processing costs and therefore expect negative estimates for β^{LC}_{1imp} and β^{RW}_{1imp}. Likewise, we expect an increase in imports of countries with high disposal capacities as this could indicate an increase in total disposal within the EU, this would be reflected in a positive estimate of β^{DC}_{1imp}.

To also capture if countries with higher costs and lower capacities with respect to their trade partners started to respectively export more waste, we make beta dependent on the variables contained in C, which capture the difference in waste haven characteristics between exporter and importer, in (1c). The three variables contained in C_{ijy} are all in the form of a ratio between exporter and importer, defined as $LCost_{ijy} = log(LCost_{iy}) - log(LCost_{jy})$ and similarly for RW_{ijy} and $DCap_{ijy}$. They capture the difference in landfill costs, $LCost_{ijy}$, recycling wages, RW_{ijy} and disposal capacities, $DCap_{ijy}$.

Our main estimate of interest is $\hat{\beta}_1 * LCost_{ijy}$, which gives the estimated effect of the waste ban on bilateral waste trade through differences in landfill costs. The choice of y is hereby naturally 2017, as the year before the ban. We expect that the ban led to an increase in exports from high to low landfill cost countries. $LCost_{ijy}$ is positive in the case of an exporter having higher landfill costs than the importer, and so we expect an increase in exports from i to j, implying a positive β^{LC}_1. Note that a positive β^{LC}_1 would be in line with a negative β^{LC}_{imp1} estimate.

For example $\hat{\beta}^{LC}_1 * LCost_{ij2017} = 0.1$ would imply an increase in waste exports from country i to country j of approximately 10 percent as a result of the ban through this channel. Taking a fixed country pair, say a pair where the exporter has a roughly 2.5 times higher landfill cost than the importer, which is about the mean difference in landfill costs between countries, a $\hat{\beta}^{LC}_1 = 0.2$, would imply a 20 percent increase in EX_{ij} ($2.5 \approx exp(1)$ and so $LCost_{ij2017} \approx 1$).

We expect a similar effect on country-pairs where the exporter has a higher recycling cost than the importer, i.e. a positive β^{RW}_1. We expect the opposite for countries where the exporter has a higher disposal capacity than the importer, i.e. a negative β^{DC}_1.

In our estimation, we control for a general relation between the waste haven variables and waste trade through $C'_{ijyg}\gamma$, with $\gamma = [\gamma^{LCo} \;\; \gamma^{RW} \;\; \gamma^{DCa}]$. γ^{LCo} corresponds to the marginal effect of an increase in $LCost_{ijy}$ on EX_{ijyg}. It thus captures if an increase in the landfill cost difference between country pairs leads to an increase in waste trade between the two, where we would expect a positive estimate for γ^{LCo}. This marginal effect corresponds to those that are studied in comparable papers that use "standard" panel estimation techniques, to estimate this effect. Specification (1a) can thus be used to compare our study to those in the literature, keeping in mind the very different sample composition and measurement of the variables of interest.

This relation, however, is likely suffering from endogeneity concerns, as increases in waste exports from i to j could lead policy makers in the importing country j to react, for example by increasing landfill taxes to prevent further imports of waste. This would imply a decrease in $LCost_{ijy}$ as a result of an increase in EX_{ijmg}, implying a reverse causality problem putting a downwards bias on γ (see for a discussion on this: Millimet & Roy, 2016). To avoid the potential bias from including the waste haven variables directly in $C'\gamma$ we also estimate (1) excluding $C'\gamma$. The estimates of β_1 are almost identical.

The estimation of the ban, instead, should not suffer from the described type of endogeneity as no country in Europe had an influence on it's imposition, as it was unanticipated, not influenced by EX itself, and as countries could not respond with new regulations to it within our panel period[5]. The non-anticipation of the event can visually be checked in Figure 3, and will further be studied through an event study approach in Subsection 5.4. Given that our identification strategy is based on estimating the effect of an event in a treatment effect fashion, we need to control for pre-treatment trends and take care of various forms of unobserved heterogeneity, which we try to capture with our control variables and fixed effects outlined below.

The remaining threat to our identification remains that the period after 2018 might have been different to the period pre 2018 for some other reason than the import ban. This could be, among others, because of certain country-pair dynamics, or because of changes in individual country manufacturing structures. In order to control for this, we move the analysis to a difference-in-difference setting by including a control group in our estimation that captures these potentially biasing factors. We outline this strategy in detail in subsection 5.2 below and continue with describing the remaining model features of the baseline estimation here.

In all specifications, we include theoretically motivated control variables and detailed fixed effects to control for remaining omitted variable bias, which we will describe in the following. The vector Z_{ijyg} contains our control variables. We include variables that we have outlined to potentially also influence waste trade. VA_{iy} is defined as the value added in the plastic producing industry in the exporting country, which proxies for the supply of waste. VA_{jy} is the value added in the plastic producing industry in the importing country, which proxies the demand for recycled material. We also include a country-time specific commodity price index, $Pcom_{jy}$, for the commodity that can substitute recycled material. All variables are included in logarithmic form.

As we have explained, the waste ban was imposed from January 2018 on, but from March 2017 on, the Chinese government started to drastically restrict the imports of waste. Because of this, choosing 2018 as the starting date of the import ban, might be problematic as the estimation then compares the period after the ban to a baseline period containing these restrictions. Optimally the baseline period, however, should

[5] Governments did in fact react to the policy, but only starting later, for example through an import ban on waste in Turkey from 2021.

be the period without trade restrictions. We will thus add a control for this to Z:

$$RESTRICTION_m C'_{ijyg} \ , \tag{1d}$$

where the restriction dummy captures the period between March and December 2017 and the ban dummy the period thereafter.

In D we include detailed fixed effects that aim to capture remaining unobserved factors that could influence our estimation. As our sample purely consists of countries that are part of a customs union, we assume that trade frictions can be captured with a constant bilateral dummy, τ_{ij}. The trade frictions, however, could be depending on the traded good, which is why we will also add a specification with pair-good dummies as a robustness, τ_{ijg}. The waste quality in the exporting country should not vary by trading partner, but could vary over different goods, which is why we include exporter-good dummies, ξ_{ig}. To also control for a time-independent comparative advantage in waste handling of the importer, we include importer-good dummies, ρ_{ig}. To control for common shocks on the global commodity markets, we also include period-good fixed effects, θ_{mg}. When estimating (1a), our month fixed effects would be colinear with the BAN dummy, so we exclude them in this regression, as this serves mainly demonstrational purposes. This implies that in our main estimation, we will only estimate β_1, as β_0 will be collinear with our fixed effects. Our fixed effects in D are thus:

$$D_{ijyg} = \tau_{ij} + \xi_{ig} + \rho_{jg} \text{ for (1a), and} \tag{1e}$$

$$D_{ijmg} = \tau_{ij} + \xi_{ig} + \rho_{jg} + \theta_{mg} \text{ for (1b) and (1c).} \tag{1f}$$

We assume that the errors ε_{ijyg} are independent between country-pairs, and so we cluster the standard errors at the country-pair level.[6] We assume that $E[\varepsilon|X]$, where X captures all RHS variables, is constant, allowing us to estimate (1) with Poisson-Pseudo Maximum Likelihood (PPML), which is the current standard in applied gravity estimation. In our robustness section we test for dynamics in the relation through the inclusion of lagged dependent variables and linear trends, and show that our results are robust to this. We also estimate (1) on the non-zero observations with OLS by taking the logarithm on both sides of the equation there.

5.2 Including non-waste goods as a control group - Difference in difference estimation

So far, our sample has consisted solely of trade in waste. One could, however, argue that the shock might have correlated with other country (-pair) characteristics that could have influenced waste trade, and that could be picked up by our estimates. These could, among others, be changes in trade relations or barriers over time, but also trends in country-specific manufacturing characteristics. This is the classical problem if one wants to infer causal claims on the estimates, as we can not observe waste trade in the absence of the import ban after 2018. In order to control for this, i.e. to proxy for

[6] Clustering standard errors is standard in gravity settings (Egger & Tarlea, 2015) and usually done at the country-pair level (Nagengast & Yotov, 2023, see for example). But since our panel has an additional dimension to most gravity regressions, i.e. the variety level, we have also experimented with clustering at different levels. Clustering at the country-pair-good level and clustering at importer and exporter level does not change the estimated uncertainty so much that it would alter the main results of the study.

waste trade after 2018, we will add an analysis in which we include trade in non-waste goods, "control goods". Including these will allow us to capture such country(-pair) dynamics in additional fixed effects.

To find a suitable control group, i.e. a group of traded goods that were not affected by the import ban and that exhibit parallel trends to trade in wastes, we add three different specifications.

1. Total trade in all goods. As waste trade is a marginal share of this, total trade should hardly be affected by the ban.

2. Substitutes of waste trade as input in production. These are raw materials that could be used to substitute recycled plastic. The advantage of this choice is that trade in these goods is expected to behave similar as trade in plastic waste, but it also presents a risk for spillover effects. This is because an increase in recyclable waste might crowd out trade in raw materials. The complete list of the included HS codes for this can be found in Appendix A.

3. A random selection of HS codes. We currently draw 20 HS codes at random from all traded HS codes for this. We then remove one outlier from this sample as it exhibits a strong decline in overall trade between members at the end of the sample period, driven solely by one exporter (Germany). This HS code is 870333, Passenger Motor Vehicles With Compression-ignition Internal Combustion Piston Engine (diesel), Cylinder Capacity Over 2,500 Cc. We have found no causal link between the waste trade ban and the decline in exports of this variety and therefore decided to move on with the remaining 19 varieties.

We thus extend our sample and estimate:

$$EX_{ijmg} = exp[BAN_y WASTE_g \beta + C'_{ijy} WASTE_g \gamma + Z'_{ijyg} \alpha + D_{ijyg}] * \varepsilon_{ijmg} \ , \quad (2)$$

with β as in (1c) and the dummy variable $WASTE_g$ indicating if the traded good is a waste good. We increase the amount of fixed effects in D, in order to capture the remaining concerns about omitted variables:

$$D_{ijyg} = \tau_{ij} + \tau_{yg} + \tau_{ig} + \tau_{jg} + \tau_{im} + \tau_{jm} \quad (2a)$$

or

$$D_{ijyg} = \tau_{ijg} + \tau_{im} + \tau_{jm} \quad (2b)$$

In the robustness section, we also add specifications with country-pair and importer and exporter-specific trends.

Our estimation now is in fact a difference in difference estimation. In this interpretation, we have a treatment group, i.e. the waste goods, and a control group, i.e. the non-waste goods, and the respective treatment is the import ban. We do not estimate the effect of the treatment on our treated goods directly (as an ATT), but want to estimate how this treatment affects the treated goods based on the interaction with the continuous variables $Lcost_{ijy}$, W_{ijy} and $WDcap_{ijy}$. One could thus also interpret the coefficients as a heterogenous treatment effect on the treated waste goods.

In this interpretation, however, we need to ensure that our control goods, can indeed serve as an appropriate control group, for which, most importantly, the parallel trend assumption as well as the stable unit treatment value assumption (SUTVA) must hold. This implies that trade in these goods would have moved in parallel with the trade in waste-goods in absence of the ban, and that the ban did not affect these waste goods through spillovers. This interpretation also helps to sharpen how we can evaluate the choice of our control-goods.

To evaluate these two assumptions, we perform a placebo test. For this, we construct a sample containing all our three control goods, but no wastes. On this sample, we run three regressions, each in the style of (2), but replacing the waste dummy with a dummy for one of the three groups. In that, we test if we find significant coefficient estimates for our waste haven hypothesis, but for an actually unaffected group. If this were the case then this would imply that this group might be unsuitable as a control group.

5.3 Getting country aggregate results

To visualize the effects of the ban and to highlight the potential magnitude of it, we present the effect based on the estimates of (1) by country. We do this, by aggregating the effect on waste imports through our three variables of interest for each importing country j. This will show us by how much the ban increased waste imports from the other countries in our sample as predicted by our regression results. In detail, we calculate total predicted import changes TPI as:

$$TPI_j = \sum_i (\hat{\gamma}_1 LCost_{ij2017} + \hat{\omega}_1 W_{ij2017} + \hat{\pi}_1 WDCap_{ij2017}) * w_{ij} \tag{3}$$

with the trade-partner specific weights, w_{ij} defined as:

$$w_i = \frac{\sum_g EX_{ijg2017}}{\sum_i \sum_g EX_{ijg2017}},$$

such that we scale the change by the pre-ban import value with the respective trade partner. We calculate the respective standard errors as:

$$SE_j = \sqrt{\begin{pmatrix} \sum_i w_i * LCost_{ij2017} \\ \sum_i w_i * W_{ij2017} \\ \sum_i w_i * WDCap_{ij2017} \end{pmatrix} COV \begin{pmatrix} \sum_i w_i * LCost_{ij2017} \\ \sum_i w_i * W_{ij2017} \\ \sum_i w_i * WDCap_{ij2017} \end{pmatrix}'} \tag{4}$$

where COV is the estimated covariance matrix of the three variables from (1).

5.4 Event study design

To exploit the timing of the regulation in more detail, we check how the differences in treatment costs and capacities have influenced waste trade over time, and if the Chinese ban presented indeed a break in this relation. For expositional purposes we focus on changes in this relationship by quarter and estimate:

$$\begin{aligned} EX_{ijmg} = exp[\sum_{q=2014q1}^{2019q4} (LCost_{ijy}\gamma_q + w_{ijy}\omega_q + WDcap_{ijy}\pi_q) * \mathbb{1}\{m \in q\} + \\ Z_{gijy}\alpha + D_{gijm}] * \varepsilon_{ijmg} \ , \end{aligned} \tag{5}$$

where q describes a quarter and we include the interaction with our variables of interest with an indicator for each quarter since 2014. The coefficients now capture the correlation of waste trade between two countries in a specific quarter with the difference in our variable of interest. We expect that the waste import ban will significantly alter this relation and therefore that the coefficient estimates will vary with time. The choice of the fixed effects is as in (1f).

For each quarter, $LCost_{ijy}\hat{\gamma}_q$ will then give us the estimated difference in exports from i to j in waste g of this quarter compared to pre-2014, differentiated by $LCost_{ijy}$. Strong pre-ban effects could thus indicate the presence of other events that influenced waste trade and that could bias our estimation. This approach thus also allows us to check for potential anticipation effects, by analyzing when the coefficient estimates become significantly different from zero.

This approach bears similarities to an event study approach, where we interact the usual treatment indicator with three continuous variables. Including or excluding the control group leads almost identical results and we present the results without a control group here.

6 Results

In this section we start by discussing our placebo test on parallel trend for our three different controls groups and then present the estimated coefficients from (1) and (2) for plastic waste trade in values. We discuss the magnitude of the effects and what they imply for the different countries in our sample based on (3). We then present results from our event study approach (5).

6.1 Placebo tests for parallel trends

We present the results for the placebo test for parallel trends in Table B7. We hereby test the different control group choices against each other and see if we can find a treatment effect for a group that is actually not treated.

We can see that two out of three choices lead the expected zero results. This implies that neither the substitutes and the random control group experienced a treatment effect from the import ban compared to the respectively other groups. This does not proof the parallel trends assumption, which is by construction unprovable, but lends confidence to it.

For the total trade group, we find a significantly positive interaction with recycling wages of the ban indicator with one of the fixed effect specifications, leading us to present results without this group in the main results table. However, including this group as a control group leads the same conclusions, as can be seen in Table B6, where we show that the choice of the control group has almost no effects on our results.

6.2 Baseline approach

We present our main results in Table 1. The coefficient estimates of our main variables of interest, $\hat{\beta}$, which are presented in the top three boxes of the table, are the coefficients on the import ban and its interactions with our three waste haven variables. Below those, one finds the ones for the non-interacted variables, γ. We omit the control variables for better oversight in this table, but present a table including all controls in Appendix Table B1.

In the first column, we present the results of (1a), which estimates the effect of the ban on intra-European waste trade, but not differentiated by our waste haven variables. Columns 2 presents the results of (1b) and column 3 to 5 then present the results from our main research strategy, by estimating (1c). The last two columns then include a control group, thus moving the estimation to a DiD one. We include the substitutes and randomly drawn HS codes as a control group here, and show in Table B6 that this choice does not matter for our results.

The effect of the import ban on total waste trade within Europe is statistically insignificant. This is not at odds with our estimation strategy, and is implying that countries overall absorb more waste domestically. It, however, does not imply that waste trade patterns did not change as a result of the import ban.

In column 2, we thus differentiate the effect by importer characteristics. We can see that countries with higher landfill costs started to import significantly less waste after the ban, while countries with high disposal capacities increased their imports significantly. This aligns exactly with our waste haven hypothesis. Even if countries did not trade more waste in total, there appears to have been a significant rerouting of it towards places with lower costs and higher disposal capacities. The coefficient on recycling wages is as expected negative, but not statistically significant.

The magnitudes apply that countries with 1 percent higher landfill costs reduced their imports by 0.2% compared to the countries with respectively lower costs as a result of the ban; which is likely a combination of an increase in imports for the countries with lower costs and a decrease for the ones with higher ones, given the statistical zero effect of the ban overall. The significant estimate on the disposal capacity coefficient could indicate that countries with low capacities became processing-constraint and thus started to export more waste to countries with respectively higher capacities, also implying an overall increase in disposal. These estimates, however, say nothing about the difference in characteristics in a country pair, but only look at importer characteristics. We thus move to interacting the ban with the waste haven variables capturing the differential in characteristics between exporter and importer.

The estimation of those shows some clear and interesting results. Firstly, the import ban led to a significant increase in waste exports from countries with high landfill costs to countries with respectively lower costs. The coefficient estimate on this variable is significant at the one percent level throughout all specifications. This is a strong indicator for a waste haven effect in Europe as a result of the Chinese import ban. This seems to imply that after the ban countries had to deal with an extra amount of waste, of which a relatively high fraction was exported to countries with lower landfill costs. The coefficient implies that after the ban, a country pair with a 2.5 times higher landfill cost in the exporting than in the importing country, which is around the sample average, traded around 20 percent more waste (exporting it from high to low cost country) as a result of the ban. We show in Subsection 6.3 what these magnitudes imply per country.

Secondly, we also find the expected sign for disposal capacities. The sign of the coefficient implies that as a result of the import ban, countries with lower disposal capacities exported more to countries with higher capacities and the estimates are significant at the ten percent level. Given that this effect is absent, or if anything the opposite, before the import ban, this might also imply that more waste was disposed after the ban, compared to before.

We do not find a consistently significant effect for recycling wages as a result of the

Table 1: Main estimation of (1) and (2) on the value of plastic exports; all presented coefficients present the effect on waste exports

	Gradular build up			Adding control goods	
	(1a)	(1b)	(1c)	(2)	(2)
Effect of import ban					
General, β_0	0.02	0.45			
	(0.05)	(0.65)			
Ban effect dependent on :					
(a) importer characteristics, β_{1imp}					
Landfill costs		-0.24^{***}			
		(0.07)			
Recycling wage		-0.12			
		(0.10)			
Disposal capacity		0.06^{*}			
		(0.03)			
(b) exp-imp differential, β_1					
Landfill costs			0.23^{***}	0.28^{***}	0.33^{***}
			(0.07)	(0.07)	(0.07)
Recycling wage			0.10	0.14	0.09
			(0.09)	(0.10)	(0.11)
Disposal capacity			-0.05^{*}	-0.07^{***}	-0.07^{***}
			(0.02)	(0.02)	(0.03)
General exp-imp differential, γ					
Landfill costs	-0.08	-0.07	-0.09	-0.07	-0.06
	(0.07)	(0.06)	(0.07)	(0.06)	(0.06)
Recycling wage	-0.00	-0.12	-0.14	0.13	0.07
	(0.24)	(0.22)	(0.23)	(0.25)	(0.23)
Disposal capacity	0.07^{**}	0.03	0.02	0.02	0.02
	(0.03)	(0.03)	(0.03)	(0.03)	(0.04)
Controls included	✓	✓	✓	✓	✓
FEs as in	(1e)	(1f)	(1f)	(2a)	(2b)
Observations	244488	241416	244488	2589413	1558872

Clustered (country-pair) standard errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.
First three columns are estimated only on waste trade, columns 4 and 5 also contain trade in other goods to control for additional country(-pair) trends. All presented coefficients present the effects on waste trade only. The coefficients on the omitted control variables can be found in Table B1. Estimation of all models by PPML. β_0 is colinear with the fixed effects in the last three columns and therefore omitted.

import ban, even if the coefficient has the expected sign, pointing again towards higher exports from higher to lower cost countries. Together with the estimates in column 2 there thus seems to be some evidence for a waste haven effect for recycling, but the evidence is less consistent as for disposal.

When studying the marginal effect of an increase in the three waste haven variables, at the bottom of the table, we find no significant effects of changes in landfill and recycling costs on waste trade. This is different to the results of Kellenberg (2012) and Higashida and Managi (2014), but not necessarily surprising. As we explained these estimates might be biased, where the bias at least for disposal costs should be negative, which could be reflected in these estimates. We find a positive, albeit not consistently significant, effect of increases in disposal capacities on waste exports, which is the opposite of the effect that one would expect. This estimate, however, also could be biased for the same reason.

Among the control variables, which we show in Appendix Table B6, those that are significant also have the expected sign, but not all seem to matter for the direction of waste trade. A higher value added in the plastic manufacturing industry, implies much higher exports of plastic waste. The price of raw materials is found to have a positive effect on waste trade, which is also as expected, as this implies that the substitute of recycled waste becomes more expensive. The coefficients on the restriction period align with those on the ban period, lending further confidence to our results that show a significant rerouting of waste in line with the waste haven hypothesis as a result of an increase in waste that had to be processed within Europe.

6.3 Country aggregate results

To get a grip on what our results imply for the countries in our sample and on how large these estimated coefficients are when taken together, we plot the implied percentage effect for each country as based on (3) using the coefficient estimates from column 3 in Table 1 and plot those in Figure 5. The calculated effects are based on the 2017 values of the three variables of interest, and show the percentage change in imported waste through the Chinese import ban as predicted by our coefficient estimates. We split the countries into old EU member states that had joined the EU before 2004 and all other countries in our sample.

For most new member states and Turkey the waste import ban implied an increase in their waste imports, and for most of them this effect is also statistically significant. This result is not universal, and also for some old member states our results imply an increase in waste imports through our waste haven channels. For the Netherlands, for example, this is based on relatively high incineration capacities, and for Portugal on low disposal costs. This picture is mostly in line with the waste haven idea, of lower income countries becoming the waste haven for higher income countries, with the just noted exceptions.

The country with the by far highest effect is Turkey where the coefficients imply an almost 90 percent increase in waste imports. This comes as no surprise and is fully in line with the waste haven hypothesis as Turkey had the far lowest landfill costs. Turkey has been pointed out as the number one country that took over plastic waste imports from the EU after the Chinese ban in several news reports since, lending some credibility to our results. We will also see in the robustness section that Turkey is indeed responsible for a sizable part of the effect that we identified before, but the

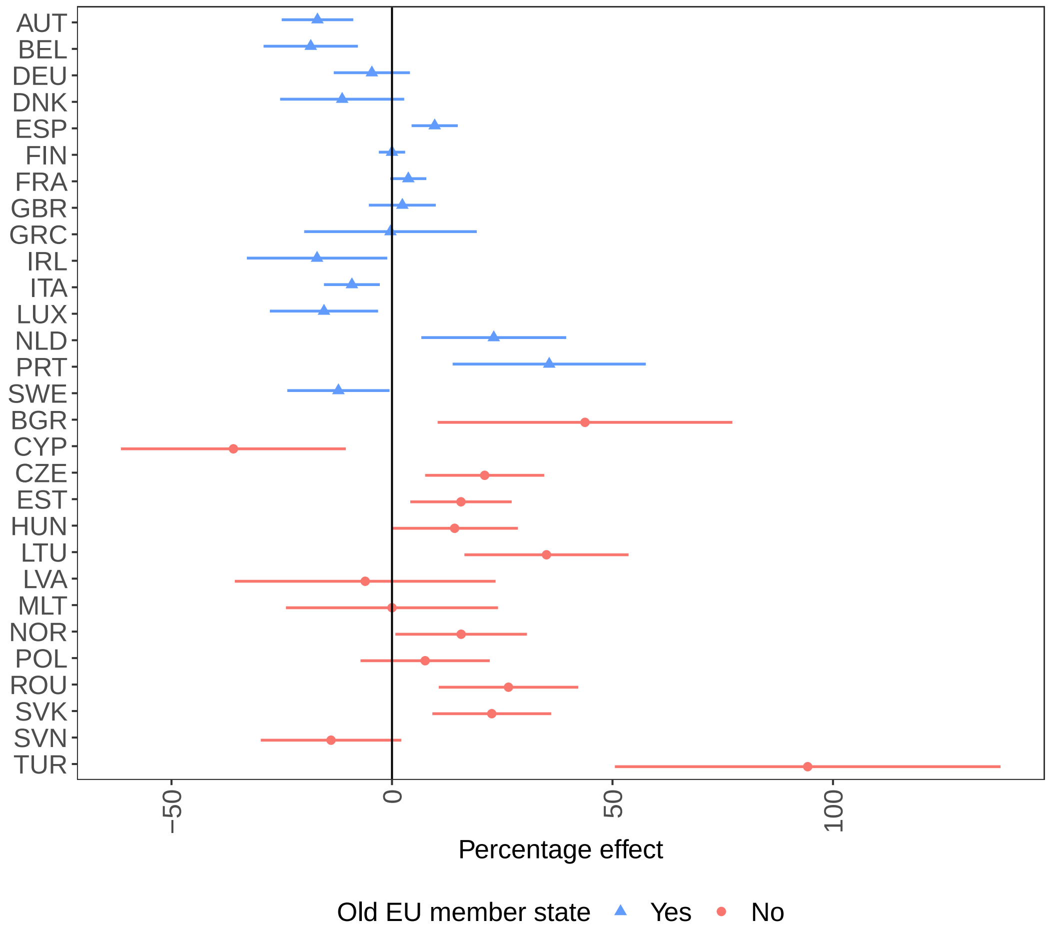

Figure 5: Effect of Chinese waste ban aggregated by importer characteristics

Note: Aggregating the waste haven coefficients after the ban from base results in column (3) of Table 1 based on trade values and explanatory variables in 2017. Point estimates calculated based on (3), standard errors based on (4). Bars present 95% confidence bars.

effect remains significant and sizable when separating the effect for Turkey from the main sample.

6.4 Effect size over time

Plotting the coefficient estimates from (5) for each of the three variables of interest in Figure 6, confirms our main results, and adds further insights. Most notably, the difference in landfill costs becomes sharply important in quarter 1 of 2018 and remains important until the end of the sample in 2019. Before that, the effect is never positive, insignificant, and shows no apparent trend. In quarter 2 in 2017 the effect becomes positive, but remains insignificant until quarter 4 in 2017. This is the period in which China started to reduce its waste imports, but did not fully forbid them.

For the disposal capacity, we can also see the timing of the regulation clearly in the estimated coefficients. Until quarter 2 in 2017 the disposal capacity seems to have an insignificant effect for the direction of waste trade, but when China started to reduce

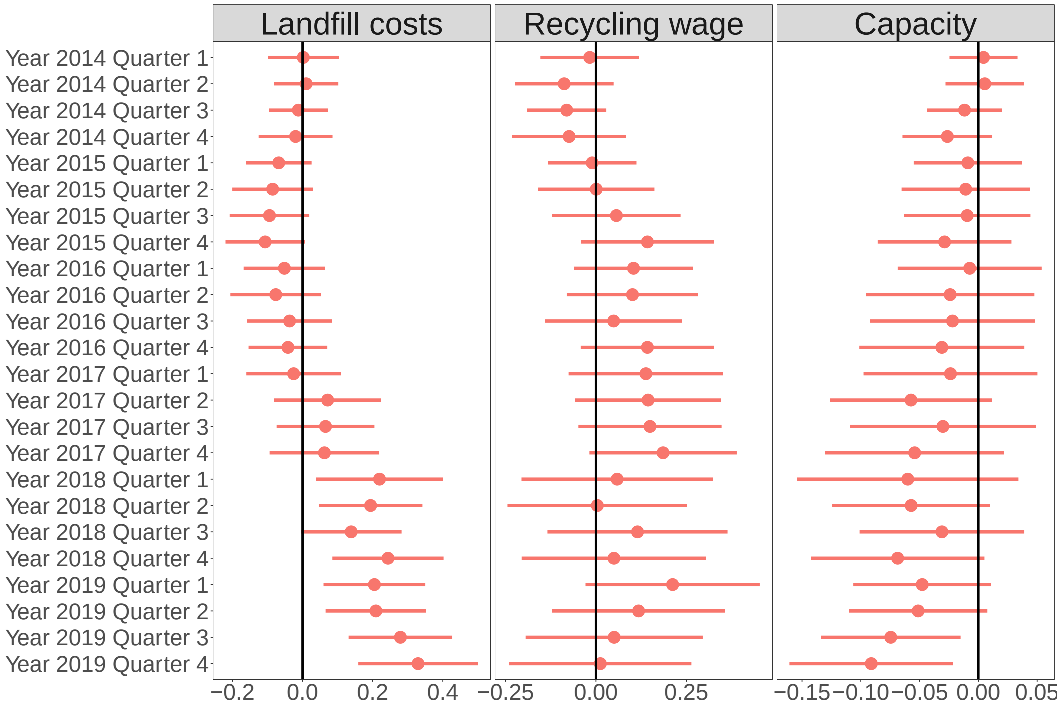

Figure 6: Event study plots.

Note: Plotting the coefficient estimates of (5). Results based on estimation without control group; including one does not change the results significantly.

its waste imports throughout 2017, the effect becomes negative, and larger over time until it becomes statistically significant in 2019.

As expected from the results in Table 1, there is no clear and consistent effect visible for recycling wages. During the restriction period before the ban, we can see back the positive effect that we found in Table 1, but it appears to have already started in the quarters before this, somewhat reducing the confidence in this result.

7 Discussion

In this section, we extend on the previous results, by studying if the effects that we found for trade values are driven by price or quantity effects, which will also give us some indication on changes in waste quality. We then study which kind of waste disposal capacity is more important for waste trade and if our results differ by plastic waste type. We also use this section to discuss policy implications of our research.

7.1 Effects on prices and quality

Our study has so far focussed on trade values, which aligns with other studies on waste trade, but is not the only possible choice. In fact, studying the two determinants of trade values, i.e. quantities and prices, can reveal interesting insights into a potentially changing nature of waste trade. One shortcoming of trade data is that it gives no indication of quality, which might be crucial for waste, as a lower quality of waste could imply that it is harder to recycle and might end up more in landfill or incineration. Luckily, however, we also have trade data in terms of weights, allowing us to calculate the average price of monthly bilateral waste trade, which has been used as a proxy for waste quality in Balkevicius et al. (2020) before.

Table 2: Estimation of (1) and (2) on plastic exports. Disentangling price and quantity effects.

	Export weight		Export price	
	(1c)	(2)	(1c)	(2)
Ban effect via exp-imp diff.				
Landfill costs	0.63***	0.69***	−0.29*	−0.29**
	(0.15)	(0.15)	(0.17)	(0.15)
Recycling wage	−0.33	−0.32	0.51*	0.39
	(0.23)	(0.22)	(0.29)	(0.25)
Disposal capacity	−0.16**	−0.16**	−0.07	−0.03
	(0.08)	(0.08)	(0.06)	(0.05)
General exp-imp differential, γ				
Landfill costs	0.04	−0.07	0.11	−0.00
	(0.15)	(0.16)	(0.21)	(0.23)
Recycling wage	−2.99***	−2.71***	0.27	−0.06
	(0.97)	(0.96)	(0.53)	(0.50)
Disposal capacity	−0.10	−0.14	0.05	0.13
	(0.13)	(0.13)	(0.09)	(0.08)
Controls included	✓	✓	✓	✓
FEs as in	(1f)	(2a)	(1f)	(2a)
Observations	243694	2563233	58533	606443

Clustered (country pair) stand.-errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. Column headers indicate the respective regression equation. First and third column are estimated only on waste trade, columns 2 and 4 contain additional trade in other goods; namely the substitute and random control group. All presented coefficients present the effects on waste trade only. The first block corresponds to β_1 in (1c), and the second to γ. The coefficients on the omitted control variables can be found in Table B1. Estimation of all models by PPML.

To study if the Chinese import ban indeed influenced trade quantities and prices, we run the same regression as before, but with trade weights and kilo prices (values divided by weights) and report the results in Table 2.

The results are even more striking than those for trade values. Trade weights seem to respond even stronger to differences in landfill costs and disposal capacities than trade values after the import ban. The coefficient size is more than twice as large as before, indicating an even larger effect of the import ban.

The same is true for the coefficient estimate for disposal capacities that increases two to threefold and stays significant at the ten percent level.

What is also interesting is the study of the waste prices. After the import ban, countries with higher landfill costs exported waste at significantly lower prices and presumably quality to countries with lower landfill costs, which is another indicator for an outsourcing of waste disposal to lower regulation countries.

The study on waste prices is also interesting for the recycling costs hypothesis. We find that waste prices increase for the exports from countries with high recycling costs to those with lower costs. This is in line with an outsourcing of recycling activities, as this implies an increase in waste quality towards countries with lower recycling costs.

Taken these results together, this adds further evidence for three interpretations of the results. Firstly, more waste, and especially more lower quality wast was exported from countries with high to countries with low disposal costs. Secondly, less waste at higher qualities is exported from countries high disposal capacities to those with lower capacities, indicating an increase in the outsourcing of waste disposal. And thirdly, countries with higher recycling costs exported higher quality waste to countries with lower recycling costs.

7.2 Disposal capacities- differentiating landfill and incineration and defining excess capacities

We have so far pooled all disposal capacities into one measure. In Table B4 we split these total capacities into capacities for landfill and for incineration to see if there are some differences between the two. For better oversight we now only focus on the import ban and leave out the restriction period before.

For plastic waste, the landfill capacity has the same, expected negative sign. For incineration capacity, however, when we include both capacities the coefficient becomes positive, and is also significant at the 10 percent level. Given that most landfilling capacity is located in the countries that also have lower landfilling costs relative to the countries with higher incineration capacity with respectively higher costs of landfilling, this results is not ad odds with our finding of waste haven effects. It might capture the correlation that more waste was exported towards Eastern and Southern Europe after the ban.

We also present a specification with a slightly adjusted measure of capacity. We have so far used the absolute capacity in terms of weight or volume, which might be misleading if trade is actually based on relative capacities, i.e. how much disposal capacities are there per unit of created waste. We thus rerun the analysis with an adjusted measure where we divide the capacities by the total waste creation in a country, before taking the ratio between exporter and importer. The results are presented in Table B3. Sign and economic uncertainty are similar between the absolute and the relative measure.

7.3 Results by plastic type

In this section we study how the different types of plastic waste reacted to the import ban; separately for all kinds of plastic waste that we can distinguish in our data. These are four kinds: Ethylene, styrene, vinyl chloride and a waste category consisting of all remaining plastic waste types. These types of waste are different to each other in terms of recyclability and economic value. Several Ethylene polymers belong to the most commonly used plastics and are often also more recyclable, while styrenes and vinyl chlorides can be recycled, but with less ease. The market for ethylene is therefore also the biggest, which is also reflected in our sample. The results of this exercise can be found in Table B5.

We can see that the two plastic types where we find statistically significant coefficient estimates for the landfill cost interaction with the waste ban are ethylene and vinyl chloride. These two thus also seem to drive the overall result, even though also for the other two waste types the coefficient has a positive sign. The disposal capacity coefficient is significant for ethylenes. One of the least recyclable waste types, styrene, has a significantly negative coefficient on the recycling wage variable, which is at odds with the idea of a recycling haven effect for this waste type.

7.4 Policy implications

Our results have important implications for policy designs that aim at restricting the export of waste, that target the pricing of disposal options and recycling and for circular economy policies in general. Most notably, waste export bans as currently planned by the EU are closely related to the setting in our study, where countries had to process more waste within one market in a short period of time. Our results indicate two important aspects. Firstly, such bans are likely to increase waste disposal, as we have found that waste trade responded strongly to both disposal capacities and prices after the ban. This is also an often raised concern about strict export bans like the EU's waste shipment directive.

Secondly, and probably even more important, even if these regulations usually aim to prevent waste haven behavior by forbidding the export to lower income countries outside the regulation, such regulation can still pose distributional concerns among the countries included in the regulation. Our results show that after the ban more waste was exported to countries with lower landfill costs and taxes. These countries are notably poorer and often have weaker environmental regulation in general, increasing the concerns about the hazardous effects of waste disposal in these places.

Our results also have implications for landfill and incineration taxation more broadly. Raising these could lead to an outsourcing of the problem to countries with lower costs, which is also a conclusion reached by Kellenberg (2012). This implies that such taxes should probably go together with an increase in recycling opportunities and could potentially be coupled with product requirements, making recycling less costly.

8 Robustness

In this section, we test several threats to our results. We start by adding trade in non-targeted goods to our sample, allowing us to control for additional omitted variable bias, turning our analysis into a difference in difference one, which does not alter our results. We then proceed with the concern that all of our results are driven by one country, namely Turkey. This proves not to be the case. We then test several other concerns and come to the conclusion that our results are robust to addressing these concerns.

8.1 Is everything driven by Turkey?

As we saw in Figure 5, Turkey was the country most affected by the import ban and the resulting waste haven effects. We show that this aligns with the observed pattern in waste trade over time in Figure B4, where we show that the share of waste imports on total imports increased drastically in Turkey from 2017 on. The same is true for Bulgaria, which our model predicts to be the second most affected country, while for example for Austria the share reduces clearly, again in line with our results in Figure 5.

The question arises if our results might be driven by this one outlier, i.e. Turkish waste imports. We therefore repeat our analysis but include an interaction of β_1 with a dummy that is one if the importer or exporter is Turkey. This allows us to study the effect in general as well as the potential difference for Turkey. We show the respective results for plastic waste trade in values, weights, and prices in Table B8.

The main conclusion that plastic waste exports from high to low landfill cost countries increased significantly after the import ban also holds when separating the effect for Turkey from the general effect. The coefficient magnitude becomes smaller, but remains economically and statistically significant. The effect is stronger for Turkey, especially for trade in values, and interestingly the coefficient on recycling costs after the ban is also significant for the deviation for Turkey. This could indicate that some of the waste sent to Turkey was indeed destined for recycling, but our results can only be indicative for this.

8.2 General robustness tests

In this section, we test some additional threats to our estimation and see how robust our results are to changes in the estimation approach. We report all results in Table B9 and summarize the conclusions here. We overall do not find any problems with our estimation approach.

It has been pointed out for some time that dynamics in gravity models could play an important role in the estimation (Bun & Klaassen, 2002), but there has yet been very little literature on how to adequately control for such dynamics in a trade regression. It is well known that including the lagged dependent variable as a regressor can lead to a Nickel bias, but there is no assessment of how big that could be in a gravity context. We nevertheless test if including the lag of the dependent variable in our estimation influences our results. We find that including the lag of trade values leads a similar estimate for the long run effect of landfill costs and disposal capacities (calculates as $\beta/(1-\delta)$ with δ the coefficient on the lagged dependent variable) as the β estimate before, and all previously significant estimates stay significant.

Estimating everything with OLS, to address the potential missspecification of the ML conditions, as described in 5, but thus excluding all the 0 values and accepting another potential bias, we find the same coefficient signs, but only find the coefficients on the recycling wages and disposal capacities to be significant.

One might also be concerned that countries with lower annual wages are usually also countries with higher working hours and that our measure of annual wages does thus not represent the acual costs of labor in recycling. We therefore adjust our wage measure by dividing the annual wage by 52 times the average hours in a working week by country, which we obtain from Eurostat. This also does not alter our results.

Including linear trends for each country-pair or for each importer and exporter in addition to the fixed effects as in (2a) or (2b) respectively does not alter our results.

One might also be concerned that our waste haven characteristics are picking up other omitted factors, like GDP and that their correlation with those is what is actually driving our coefficient estimates. We thus includes countries' real GDP differential interacted with BAN in the last column. The coefficient size and statistical significance of the landfill cost and disposal capacity variables stay unaltered. Additionally, it seems as if in addition to our explanatory variables, countries with higher GDP start to export more waste to countries with lower GDP as a result of the ban.

In a last check, we exclude countries that impose bans on different kinds of landfilling from the sample. We believe that our landfill costs are probably a good proxy for incineration costs as well, as regulations on one form of waste disposal likely correlate with those of other kinds of waste disposal. But to mitigate concerns that our results are driven by countries in which landfill is at least to a certain extent restricted, we also include an estimation of (1), while excluding countries that have strict forms of landfill bans. Landfilling is not restricted to 0 in any country, but countries impose varying levels of restrictions, and we exclude the countries with the strictest ones here (these are: Austria, Belgium, Denmark, Finland, Germany, Norway, Slovenia, Sweden). As one can see in the last column of Table B9, the results remain similar in terms of magnitude and stay statistically significant, despite the much smaller sample size.

9 Conclusion

In this paper, we study the effects of the Chinese waste import ban of 2018 on intra-European waste trade. We analyze if it led to a plastic waste haven effect within Europe, implying an increase in exports from high disposal and recycling cost countries to countries with lower costs and if countries with lower disposal capacity increased their exports to countries with higher disposal capacity.

The import ban implied that half of all plastic waste that was sent to outside of Europe before the ban now had to be reallocated. We therefore study where this waste was rerouted to and motivate an empirical gravity-style difference-in-differences estimating equation to study this. We compile a panel of bilateral, HS6-level goods trade, which includes the trade in waste types that were banned for import by the Chinese government. Our main variables of interest include the average costs of landfilling, which is the most detailed measure of waste disposal costs in any comparable study, the recycling wage as a proxy for recycling costs and actual disposal capacities.

We find that as a result of the ban countries with high costs of landfilling exported significantly more plastic waste to countries with low landfilling costs. We also find that the waste that countries with low costs imported increased in weight, but decreased in price, giving further confidence to the interpretation that these countries started to import more lower quality waste, destined for disposal towards countries with lowers costs. All of these are indications for a waste haven effect within Europe as a result of the Chinese waste import ban.

For recycling costs the effect is in most specifications not statistically significant and not consistently as expected after the import ban.

We also find that countries with low disposal capacities, especially for landfilling, exported more plastic waste to countries with high disposal capacities. This indicates that more waste was now being disposed within Europe, in countries that were "specialized" in this activity.

A lot of the effect is driven by one country that clearly became a waste haven for the rest of Europe. This country is Turkey. This has already been pointed out by popular press, but we show here that these waste imports were especially strong from countries with high disposal costs, where the incentive to outsource the waste processing was especially strong. Separating Turkey from the sample, does not remove the findings, implying that this waste haven emergence is not particular for this one country.

Our results have important implications for policy making. Policies that restrict the exports of waste to countries outside a jurisdiction, as is currently planned in the EU, can lead to both an increase in waste disposal and a distribution of waste treatment that favors countries with high regulation on disposal, but harms the ones with less stringent regulation. These countries are often lower income countries, which leads to questions about pollution outsourcing and fairness.

Our paper omits two important effects of the waste ban. It can, firstly, not control for effects on illegal waste trade. That is because illegal trade data is, naturally, not available. Given our strong results on legal waste trade, however, one could assume that including illegal waste trade might rather intensify our results. We, secondly, do not consider the emergence of waste havens outside of Europe. As we want to base our results on actually observable drivers of waste trade and abstain from very rough proxies (like income), we constrain our sample in this way. Again including other potential waste havens outside of Europe would probably rather strengthen our results.

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A Data details

A.1 Used HS codes

Our choice of what constitutes as an affected waste good is straight forward and based on the announcement by China to the WTO on the 18th of July 2017 (China, 2017). We compile the codes used for the analysis of plastic and paper waste in Table A1.

Waste type	Banned	Otherwise affected
Plastic	391510, 391520, 391530, 391590	
Paper	470790	470710, 470720, 470730

Table A1: HS codes banned and affected by 2017 and 2018 Chinese regulation

We complement those with other HS codes that were affected by adjacent regulations and are thus excluded from the analysis. In Table A2 we list the HS codes that we used in the substitutes control group in this study. These are all goods that can be seen as substitutes for recycled waste. We have thus decided to currently only include goods where there exist waste types that can also be recyclable, which include polyethylene, propylene, polystyrene, and poly vinyl for plastic; and wood pulp for paper.

Waste type	Control goods in substitutes category; HS codes
Plastic	390110, 390120, 390130, 390140, 390190, 390290, 390311, 390319, 390320, 390330, 390410, 390421, 390422, 390430, 390440, 390450, 390461, 390469, 390490
Paper	470100, 470200, 470311, 470319, 470321, 470329, 470411, 470419, 470421, 470429, 470500, 470610, 470620, 470630

Table A2: HS codes used as control goods

In the first group of total trade, we just use the aggregate bilateral trade between countries. The HS codes for the random category can be shared upon request and can also be found when running our replication package.

A.2 Landfill tax rates and gate fees

We describe here the specifics on the landfill tax rates in the cases where the choice was not fully unambiguous. These choices refer to the CEWEP data and their definition of the landfill tax rate. Their data is saved in a pdf file per yer, and we have compiled them into a panel. For consistency we had to make some choices in this, which are documented in Table A3. For Turkey, for which no data from CEWEP was available, we complemented our data with Bakas and Milios (2013) and an online search for the period after 2013; revealing a landfill tax rate of 0 throughout the whole panel period. To get a value for the gate fee in Turkey, we take the gate fee at Istanbul's (the country's biggest city) landfill sites, which we accessed on 16.08.2023 and converted into 2013 Euros for consistency with the other data. The url for the fee schedule can be found here.

Table A3: Notes on the determination of the landfill tax rate.

Country	Note
Belgium	Average of Flanders and Wallonia for combustible and general waste chosen
Chzechia	For early years we are relying on EEA data
Germany	Germany has a landfill ban in place and so we exclude it in some robustness analysis.
Denmark	Pre VAT rate used, consistent with other countries
Estonia	For early years we are relying on EEA data
Greece	According to the EEA an original fee was never implemented. We chose a tax of zero for that time frame.
Spain	Spain has no national landfill tax but waste authorities in different regions of Spain enact their own tariffs. We use the tax in Catalonia as this is the biggest economic region.
France	Rate for “Other authorized landfills” chosen.
Italy	We took the average of the possible range, consistent with the EEA
Netherlands	The tax is levied on all Dutch waste, from 2015 on also on waste that is exported from the Netherlands. We thus exclue Dutch exports from the analysis
Slovakia	Average of different separation levels chosen

A.3 Conversion of landfill capacity into tonnes

To create a measure that captures the total disposal capacity in a country, we have to convert both landfill and incineration capacity into a common unit. We chose to convert both into weights, where the interpretation would then be how many kilos of waste could be disposed in a given year. We use data from EPA (2016) to convert the middle between small and large landfill MSW waste from cubic meters into kilos. This middle is equivalent to 1700 kilos per cubic yard, which we convert into cubic meters.

A.4 Additional data choices and manipulations

As capacity data for the UK is only available from Eurostat until the Brexit, we complement data for afterwards with data from the UK Office for National Statistics. Since Dutch exports destined are subject to the same fee as domestically treated waste, we exclude Dutch exports from the analysis.

	Tax		Cap In		Cap Disp		Wage		VA Plastic	
Year	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
2011	35.47	41.51	6.45	11.69	130.74	179.17	41.59	24.52	4.22	7.18
2012	29.23	34.35	7.27	12.90	139.25	184.77	39.08	24.24	3.99	6.70
2013	32.36	37.45	6.95	12.74	137.56	180.40	40.01	25.07	4.29	7.14
2014	35.25	37.93	6.62	12.83	135.86	177.05	41.14	25.40	4.43	7.32
2015	29.35	32.73	6.87	13.04	136.37	175.59	34.12	20.97	3.97	6.52
2016	29.85	33.61	7.12	13.27	136.05	171.70	34.93	20.63	4.06	6.75
2017	33.20	35.40	7.52	13.63	133.99	167.34	36.65	21.39	4.14	6.82
2018	36.86	37.43	7.92	14.05	131.94	164.24	38.88	22.11	4.56	7.42
2019	36.98	36.26	8.49	14.97	167.09	215.07	37.68	21.48	4.44	7.12

Table A4: Summary statistics by Year. Tax is in Dollar per ton, capacities are in million kilo per year or cubic meters respectively, wages are in thousand dollar per person and year, VA is in billion dollars

B Additional results

B.1 Main results table, containing estimates on control variables

Table B1: Estimation of (1) and (2) on plastic exports. Disentangling price and quantity effects.

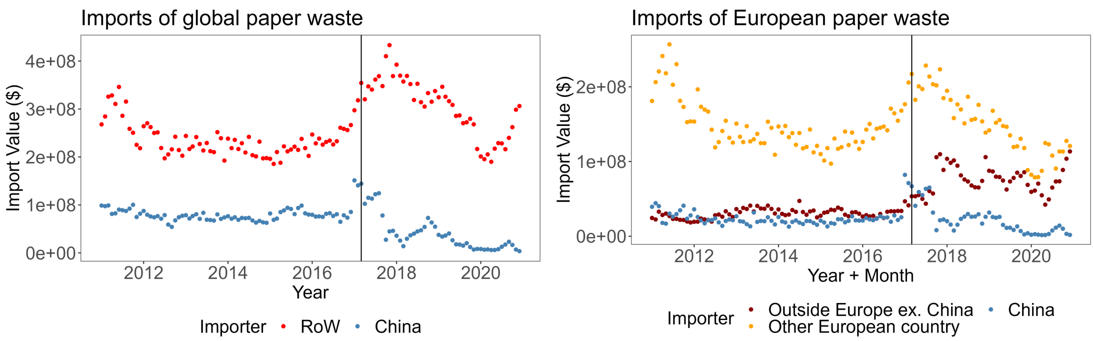
	Export value		Export weight		Export price	
	(1c)	(2)	(1c)	(2)	(1c)	(2)
Ban effect via exp-imp diff.						
Landfill costs	0.23***	0.25***	0.63***	0.69***	−0.29*	−0.29**
	(0.07)	(0.07)	(0.15)	(0.15)	(0.17)	(0.15)
Recycling wage	0.10	0.04	−0.33	−0.32	0.51*	0.39
	(0.09)	(0.10)	(0.23)	(0.22)	(0.29)	(0.25)
Disposal capacity	−0.05*	−0.04*	−0.16**	−0.16**	−0.07	−0.03
	(0.02)	(0.02)	(0.08)	(0.08)	(0.06)	(0.05)
General exp-imp differential						
Landfill costs	−0.09	−0.06	0.04	−0.07	0.11	−0.00
	(0.07)	(0.06)	(0.15)	(0.16)	(0.21)	(0.23)
Recycling wage	−0.14	−0.02	−2.99***	−2.71***	0.27	−0.06
	(0.23)	(0.24)	(0.97)	(0.96)	(0.53)	(0.50)
Disposal capacity	0.02	0.03	−0.10	−0.14	0.05	0.13
	(0.03)	(0.04)	(0.13)	(0.13)	(0.09)	(0.08)
Restriction eff. via exp-imp diff.:						
Landfill costs	0.07	0.08	0.25**	0.26**	−0.14	−0.25**
	(0.05)	(0.05)	(0.11)	(0.10)	(0.12)	(0.12)
Recycling wage	0.17**	0.13*	−0.22	−0.19	0.36	0.40**
	(0.07)	(0.07)	(0.20)	(0.19)	(0.23)	(0.20)
Disposal capacity	−0.02	−0.03	−0.04	−0.04	0.07	0.13**
	(0.02)	(0.02)	(0.06)	(0.06)	(0.05)	(0.05)
Other controls						
Industry supply	0.98***	0.22*	1.20***	0.52***	−0.81*	0.26
	(0.26)	(0.11)	(0.40)	(0.20)	(0.48)	(0.29)
Industry demand	0.05	0.18***	0.80	0.26	−1.62***	−0.31
	(0.27)	(0.07)	(0.71)	(0.25)	(0.47)	(0.22)
Substitute prices	0.52	0.01	5.22***	0.69	−1.10	0.94
	(0.57)	(0.18)	(1.94)	(0.47)	(1.76)	(1.18)
Controls included	✓	✓	✓	✓	✓	✓
FEs as in	(1f)	(2a)	(1f)	(2a)	(1f)	(2a)
Observations	244488	2574816	243694	2563233	58533	606443

Clustered (Country pair) standard-errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.
Column headers indicate the respective regression equation. First, third and fifth column are estimated only on waste trade, columns 2, 4 and 5 contain additional trade in other goods; namely the substitute and random control group. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

B.2 Paper waste

Our analysis has so far purely focused on plastic waste, as this was the most affected one from the Chinese import ban. The Chinese government simultaneously also banned the import of some paper wastes, for which it was also a crucial import destination (see Figure B1). The import ban was fuzzier and it took longer until Chinese imports actually reached zero, but certainly also affected European waste trade. Additionally important was the choice of the paper wastes, which were restricted to unsorted waste paper that is generally harder to recycle. We should therefore expect even less effects for the recycling cost interaction.

Figure B1: Paper waste imports



Note: Left panel plots all global paper waste exports, of waste types banned by the import ban, by importer and right pane all European waste exports by importer. Vertical lines indicate March 2017, i.e. the start of the waste import restrictions. Ban went into effect in January 2018.

We run the same regression as for plastic imports, (1) and (2), and report the estimated coefficients in Table B2. We include results with trade values, weights, and prices as dependent variables.

The results are a little bit more uncertain than for plastic waste, but contain some similar conclusions. The coefficient on the interaction of the waste ban with landfill costs is positive and significant at least for trade in weights. The magnitude of the effect for paper is smaller than for plastic, but still sizable. As for plastic, the magnitude is higher for weights, and we also find a significantly negative effect on the price of waste that is exported from high to low disposal cost countries.

The interaction of the disposal capacity with the waste import ban remains significant for values and weights, again suggesting that waste exports increased towards places with higher disposal capacities. This is again consistent with reports of higher waste disposal after the import ban.

The coefficients for recycling costs turn negative and are statistically significant at the 1 percent level for both values and weights. Given that the waste is likely not recyclable, we attach little value to this result, as we did not expect to find a positive relation between the two.

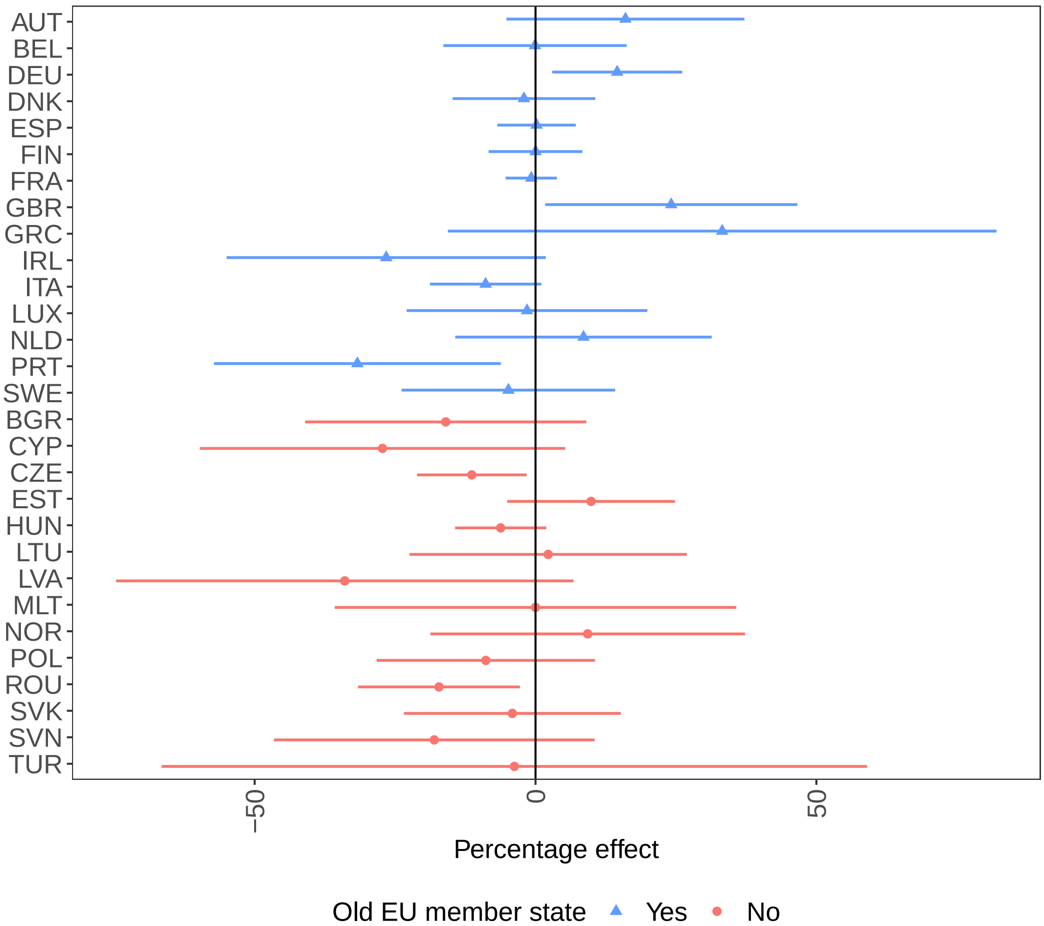
We again plot the implications of our results for values for individual countries (see Figure B2), but our results imply statistically insignificant consequences for most countries.

Table B2: Estimation of (1) and (2) on paper exports. Disentangling price and quantity effects.

	Export value		Export weight		Export price	
	(1c)	(2)	(1c)	(2)	(1c)	(2)
Ban effect via exp-imp diff.						
Landfill costs	0.03	-0.00	0.40**	0.46**	-0.03	-0.04
	(0.10)	(0.10)	(0.17)	(0.19)	(0.22)	(0.24)
Recycling wage	-0.35**	-0.23	-1.04***	-0.97***	-0.51	-0.59
	(0.15)	(0.16)	(0.37)	(0.37)	(0.44)	(0.40)
Disposal capacity	-0.04	-0.03	-0.19**	-0.22***	0.16	0.16
	(0.04)	(0.04)	(0.08)	(0.08)	(0.14)	(0.12)
General exp-imp differential, γ						
Landfill costs	0.04	0.05	0.21	0.16	0.30	0.34
	(0.06)	(0.06)	(0.29)	(0.29)	(0.41)	(0.35)
Recycling wage	0.04	0.09	-4.80***	-4.41***	1.72	1.43
	(0.28)	(0.29)	(1.63)	(1.48)	(1.82)	(2.05)
Disposal capacity	0.03	0.03	-0.29	-0.29	-0.07	0.11
	(0.05)	(0.05)	(0.18)	(0.20)	(0.21)	(0.24)
Controls included	✓	✓	✓	✓	✓	✓
FEs as in	(1f)	(2a)	(1f)	(2a)	(1f)	(2a)
Observations	36300	569548	36078	564339	13885	64211

Clustered (Country pair) standard-errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$.
Column headers indicate the respective regression equation. First, third and fifth column are estimated only on waste trade, columns 2, 4 and 5 contain additional trade in other goods. All presented coefficients present the effects on waste trade only. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

Figure B2: Aggregating the ban effect on paper waste trade by importer



Note: Aggregating the waste haven coefficients after the ban from base results in column (3) of Table B2 based on trade values and explanatory variables in 2017. Point estimates calculated based on (3), standard errors based on (4). Bars present 95% confidence bars.

B.3 Disposal capacities

Table B3: Using excess capacity instead of absolute capacity; plastic waste

	Base	Landfill	Incineration	Both
Ban effect via exp-imp diff.				
Landfill costs	0.20***	0.20***	0.14**	0.12*
	(0.06)	(0.06)	(0.07)	(0.07)
Recycling wage	0.02	0.01	0.03	0.00
	(0.11)	(0.11)	(0.11)	(0.11)
Rel. Disposal cap	−0.11**			
	(0.05)			
Rel. Landfill cap		−0.11**		−0.14***
		(0.05)		(0.05)
Rel. Incineration cap			0.11*	0.15**
			(0.07)	(0.07)
General exp-imp differential, γ				
Landfill costs	−0.07	−0.08	−0.05	−0.07
	(0.07)	(0.08)	(0.07)	(0.08)
Recycling wage	−0.03	−0.04	0.03	0.02
	(0.24)	(0.24)	(0.23)	(0.23)
Rel .Disposal cap	0.03			
	(0.03)			
Rel. Landfill cap		0.00		0.00
		(0.00)		(0.00)
Rel. Incineration cap			0.10	0.09
			(0.07)	(0.07)
Controls included	✓	✓	✓	✓
FEs as in	(1f)	(1f)	(1f)	(1f)
Observations	183504	179424	191712	179424

Clustered (Country pair) standard-errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. All columns are based on regressions containing both waste and control goods; namely the substitute and random control group. All presented coefficients present the effects on waste trade only. Estimation of all models by except for column 2 are estimated by PPML.

Table B4: Estimation of (1) with differing measures of disposal capacity

	Base	Landfill	Incineration	Both
Ban effect via exp-imp diff.				
Landfill costs	0.23***	0.24***	0.20***	0.16***
	(0.06)	(0.06)	(0.06)	(0.06)
Recycling wage	0.06	0.03	0.06	0.01
	(0.08)	(0.09)	(0.09)	(0.09)
Disposal capacity	−0.05**			
	(0.02)			
Landfill capacity		−0.05**		−0.13***
		(0.02)		(0.04)
Incineration capacity			−0.02	0.10**
			(0.03)	(0.05)
General exp-imp differential, γ				
Landfill costs	−0.10	−0.12*	−0.10	−0.13*
	(0.07)	(0.07)	(0.07)	(0.07)
Recycling wage	−0.18	−0.25	−0.06	−0.14
	(0.23)	(0.24)	(0.23)	(0.23)
Disposal capacity	0.02			
	(0.03)			
Landfill capacity		0.00		−0.00
		(0.00)		(0.00)
Incineration capacity			0.10	0.09
			(0.07)	(0.07)
Controls included	✓	✓	✓	✓
FEs as in	(1f)	(1f)	(1f)	(1f)
Observations	244488	241680	252888	239856

Clustered (Country pair) standard-errors in parentheses, $^{*}p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. All columns are based on regressions containing both waste and control goods; namely the substitute and random control group. All presented coefficients present the effects on plastic waste trade only. Estimation of all models by except for column 2 are estimated by PPML.

B.4 Results by plastic waste type

Table B5: Estimation of (1c) on plastic exports. Results by waste type.

	Ethylene	Styrene	Vinyl Chloride	Others
Ban effect via exp-imp diff.				
Landfill costs	0.37***	0.13	0.37*	0.11
	(0.10)	(0.23)	(0.21)	(0.08)
Recycling wage	0.14	−0.75**	−0.37	0.12
	(0.14)	(0.32)	(0.32)	(0.14)
Disposal capacity	−0.06*	−0.09	0.14	−0.04
	(0.03)	(0.07)	(0.09)	(0.03)
General exp-imp differential, γ				
Landfill costs	−0.06	0.03	0.24	−0.14
	(0.08)	(0.25)	(0.30)	(0.10)
Recycling wage	0.21	−2.47***	0.90	−0.35
	(0.36)	(0.63)	(0.59)	(0.32)
Disposal capacity	−0.03	0.18**	−0.06	0.05
	(0.06)	(0.08)	(0.13)	(0.04)
Controls included	✓	✓	✓	✓
FEs as in	(1f)	(1f)	(1f)	(1f)
Observations	51072	31380	29520	58056

Clustered (Country pair) standard-errors in parentheses, $^*p < 0.10$, $^{**}p < 0.05$, $^{***}p < 0.01$. All columns are estimated only on waste trade, each with a different subsample, containing only one specific HS code as indicated in the column header. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

B.5 Difference difference in difference specifications and placebo tests

Table B6: Main estimation of (2) including non-waste control goods on plastic exports

	Total	Substitutes	Random
Ban effect via exp-imp diff.			
Landfill costs	0.26***	0.28***	0.25***
	(0.07)	(0.07)	(0.08)
Recycling wage	0.09	0.12	0.14
	(0.09)	(0.10)	(0.12)
Disposal capacity	−0.07***	−0.07***	−0.06*
	(0.02)	(0.02)	(0.03)
General exp-imp differential, γ			
Landfill costs	−0.06	−0.10	−0.08
	(0.06)	(0.06)	(0.08)
Recycling wage	0.02	0.06	0.35
	(0.25)	(0.26)	(0.28)
Disposal capacity	0.03	0.02	0.05
	(0.04)	(0.03)	(0.05)
Controls included	✓	✓	✓
FEs as in	(2a)	(2a)	(2a)
Observations	361072	1492292	1360813

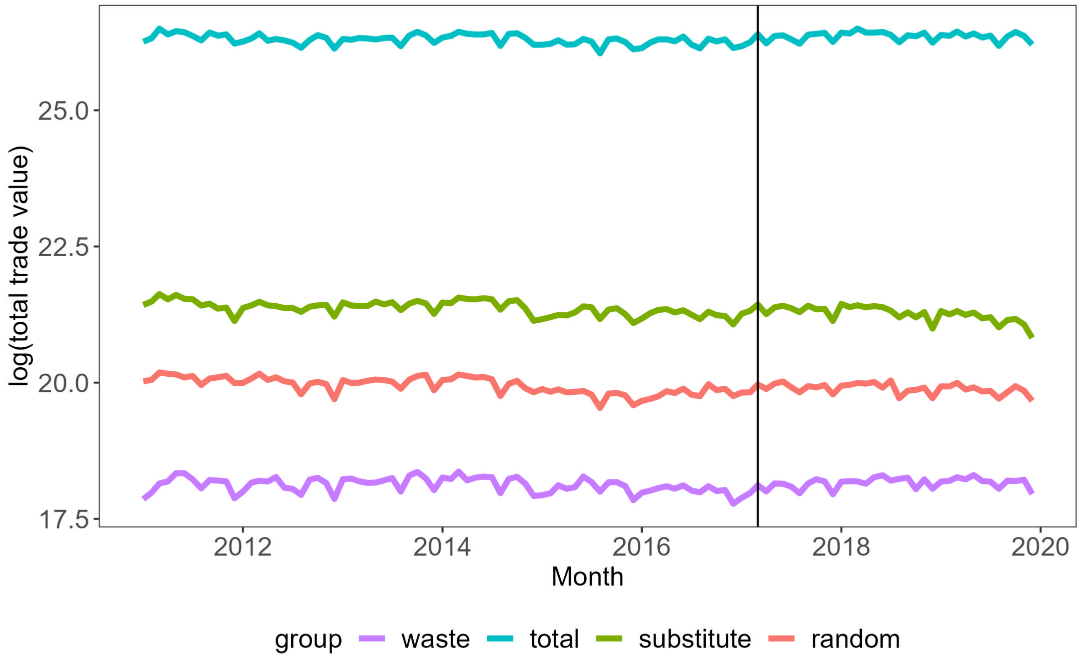
All columns are estimated on different subsample, containing trade in waste as well as a different control group in each column, as indicated in the column header. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

Table B7: Placebo estimation with varying placebo treatment groups

	Total	Total	Substitutes	Substitutes	Random	Random
Ban effect total						
Landfill costs	−0.02	0.06*				
	(0.03)	(0.03)				
Recycling wage	0.07**	−0.01				
	(0.03)	(0.04)				
Disposal capacity	−0.01	−0.01				
	(0.01)	(0.01)				
Ban effect substitutes						
Landfill costs			0.01	−0.05		
			(0.03)	(0.03)		
Disposal capacity			0.01	0.01		
			(0.01)	(0.01)		
Recycling wage			−0.05	−0.02		
			(0.03)	(0.04)		
Ban effect random						
Landfill costs					0.02	−0.06
					(0.05)	(0.06)
Disposal capacity					−0.00	−0.02
					(0.02)	(0.02)
Recycling wage					−0.13*	0.05
					(0.07)	(0.07)
Controls included	✓	✓	✓	✓	✓	✓
FEs as in	(2a)	(2b)	(2a)	(2b)	(2a)	(2b)
Observations	2400818	1461455	2400818	1461455	2400818	1461455

All columns are estimated on a sample containing trade in all control groups, but excluding waste trade. The waste dummy is replaced in each column by a dummy capturing a different control group. The differential variables always capture the difference in the variable between exporter and importer, with a positive value indicating a higher value for the exporter than the importer. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

Figure B3: Total trade in plastic wastes and the three control groups between sample countries



Note: The four lines depict total trade within all sample countries in each of the four groups respectively. We take the logarithm of total trade to make the lines visually comparable. The vertical bar indicates the beginning of the import restrictions in China from March 2017.

B.6 Leaving Turkey out of the sample

Figure B4: Plastic waste imports as a share of total imports from other sample members by country over time.

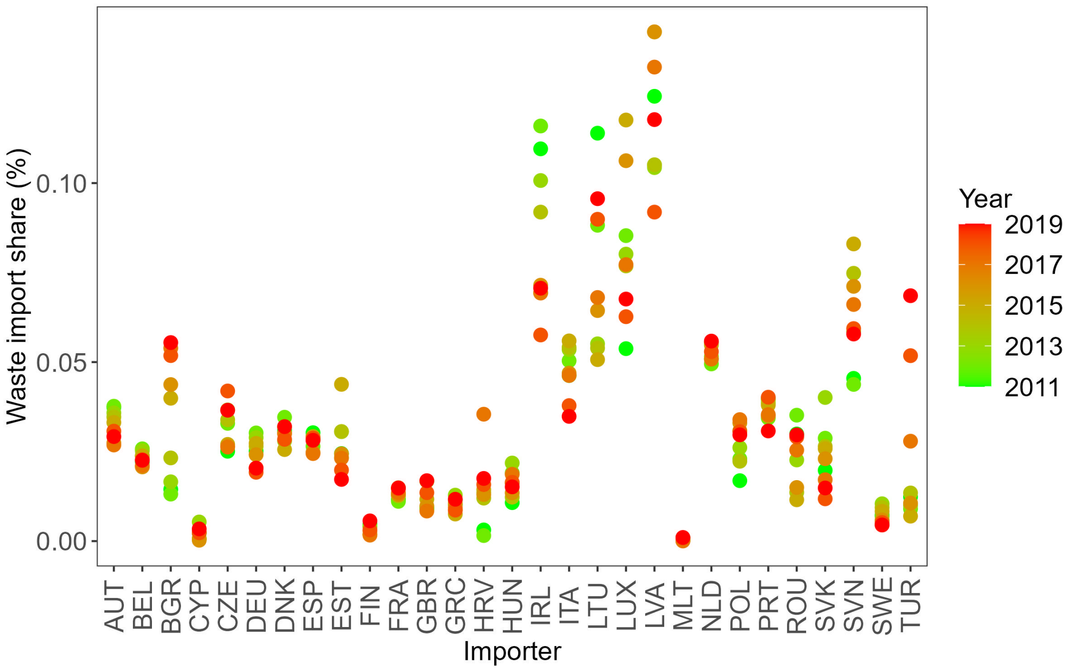


Table B8: Estimation of (1) and (2) on plastic exports. Including an interaction with Turkey as importer and exporter with ban effect.

	Export value		Export weight		Export price	
	(1c)	(2)	(1c)	(2)	(1c)	(2)
Ban effect via exp-imp diff.						
Landfill costs	0.13*	0.18**	0.51**	0.23***	−0.21	−0.56**
	(0.07)	(0.08)	(0.20)	(0.09)	(0.22)	(0.23)
Recycling wage	0.09	0.16	−0.35	0.09	0.50*	0.72**
	(0.10)	(0.10)	(0.24)	(0.14)	(0.30)	(0.29)
Disposal capacity	−0.03	−0.06**	−0.15*	−0.04	−0.09	−0.00
	(0.03)	(0.03)	(0.09)	(0.03)	(0.06)	(0.05)
Ban effect via exp-imp diff. for Turkey only						
Landfill costs	0.02	0.26**	0.01	0.26	−0.01	0.09
	(0.11)	(0.12)	(0.21)	(0.18)	(0.24)	(0.24)
Recycling wage	0.54**	−0.15	0.59*	0.40	−0.56*	−0.30
	(0.21)	(0.40)	(0.31)	(0.57)	(0.30)	(0.40)
Disposal capacity	−0.10	0.09	−0.08	0.09	0.11	−0.22*
	(0.12)	(0.28)	(0.12)	(0.22)	(0.13)	(0.12)
General exp-imp differential, γ						
Landfill costs	−0.11*	−0.09	0.01	−0.30**	0.12	0.03
	(0.07)	(0.06)	(0.15)	(0.13)	(0.21)	(0.26)
Recycling wage	−0.17	0.10	−3.02***	−0.36	0.29	−0.06
	(0.23)	(0.26)	(0.97)	(0.31)	(0.53)	(0.60)
Disposal capacity	0.03	0.04	−0.09	0.03	0.03	0.09
	(0.03)	(0.03)	(0.14)	(0.10)	(0.09)	(0.09)
Controls included	✓	✓	✓	✓	✓	✓
FEs as in	(1f)	(2a)	(1f)	(2a)	(1f)	(2a)
Observations	244488	2551901	243694	2539888	58533	606443

Clustered (Country pair) standard-errors in parentheses, $*p < 0.10$, $**p < 0.05$, $***p < 0.01$.
Column headers indicate the respective regression equation. First and third column are estimated only on waste trade, columns 2, 4 and 5 contain additional trade in other goods; namely the substitute and random control group. The first block corresponds to β_1 in (1c), and the second to γ. Estimation of all models by PPML.

B.7 General robustness

Table B9: Estimation of (2) on plastic exports with differing robustness specifications.

	Dynamics	OLS	Hourly	C-pair trend	Ex. Im. trend	GDP	Ban
Ban effect via exp-imp diff.							
Landfill costs	0.10^{***}	0.03	0.27^{***}	0.35^{***}	0.35^{***}	0.28^{***}	0.27^{*}
	(0.02)	(0.05)	(0.07)	(0.07)	(0.07)	(0.07)	(0.14)
Recycling wage	-0.00	0.24^{**}		0.05	-0.02	0.10	0.06
	(0.03)	(0.10)		(0.11)	(0.11)	(0.10)	(0.17)
Disposal capacity	-0.02^{**}	-0.09^{***}	-0.03	-0.05^{**}	-0.05^{**}	-0.14^{***}	-0.16^{***}
	(0.01)	(0.02)	(0.03)	(0.02)	(0.02)	(0.03)	(0.04)
Hourly rec. wage			-0.04				
			(0.10)				
GDP						0.11^{**}	
						(0.05)	
Lagged Exports	0.66^{***}						
	(0.01)						
Controls included	✓	✓	✓	✓	✓	✓	✓
FEs as in	(1f)	(1f)	(1f)			(2a)	(2a)
Observations	633280	640217	2117808	2686930	1572192	2589413	1115807

All columns are based on regressions containing both waste and control goods; namely the substitute and random control group. All presented coefficients present the effects on waste trade only. Estimation of all models by except for column 2 are estimated by PPML. Column 4 contains importer- and exporter-time and importer- and exporter-good FEs in addition to the country-pair trends, and column 5 contains country-pair-good FEs in addition to the importer and exporter trends.