# Import ban and clean air: estimating the effect of China's waste import ban on ozone pollution

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## Abstract

In this study, we investigate the effects of the ban on imports of plastic waste on local air quality in China. Using city-level daily ozone concentrations, we examine whether the pollution levels differ after the implementation of the import ban. The results demonstrate that the daily ozone concentration was reduced by 2.8% in treatment areas after the import ban. Additional analyses also suggest that the effect of the ban was larger during the later period and in coastal cities with ports while the effect of the ban might get weak in the long run. These findings highlight the effectiveness of policies that reduce the availability of inputs for pollutive activities.

## 1 Introduction

Air pollution is detrimental to human health and causes substantial loss, particularly in a fast-growing economy like China. In 2017, the Global Burden of Disease estimated that approximately 852,000 deaths in China were attributable to PM 2.5 exposures and an additional 178,000 chronic respiratory disease-related deaths to ozone (Health Effects Institute 2019). Economic losses from air pollution are huge and were estimated to be 12.9% of China's annual gross domestic product in 2019 (World Bank 2022). Recent studies suggest diverse effects of air pollution, including lower cognitive performance (Zhang et al. 2018), loss of labor productivity (He et al. 2019), and sleeplessness (Heyes and Zhu 2019).

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The Chinese government has initiated several actions to reduce the severity of air pollution. Consequently, the annual average concentrations of PM 2.5, PM 10, and $NO_2$ in China gradually decreased from 2014 to 2018 (Ministry of Ecology and Environment of China 2018). Despite witnessing a steady decline in PM 2.5 exposures, China still had the highest population-weighted seasonal concentration of ozone among the world's 11 most populous countries (Health Effects Institute 2019).

The melting process of material plastic waste recycling, however, could generate volatile organic compounds (VOCs) (Tsai et al. 2009; Yamashita et al. 2009), which react with NOx in the presence of sunlight to form ground-level ozone (Atkinson 2000). China has been the world's largest importer of plastic waste and processes huge amounts of hard-to-recycle plastics from other countries, accounting for approximately 56% of the global market (Brooks et al. 2018). Nonetheless, in July 2017, the Chinese government introduced a policy that imposed a ban, effective from January 1, 2018, on the import of several types of solid waste, including plastic waste.

In this study, we apply difference-in-differences (DID) methods to investigate the effects of the plastic waste ban on air quality in China by focusing on surface-level ozone. The import ban on plastic waste leads to a reduction in the supply of plastic waste which will slow down the plastic recycling process that releases VOCs, thereby decreasing VOC emissions. This reduction further reduces the ozone concentration, as VOCs are one of the main precursors of ozone formation. Therefore, we can expect an improvement in air quality in the areas around the plastic recycling facilities.

This study contributes to the literature on the economics of air pollution in developing countries in 2 ways. First, many studies on the effect of air pollution control focus on policies that directly affect emission sources, such as shutdown and installation of control equipment (Cao et al. 2009)), driving restrictions (Viard and Fu 2015), and subsidies to winter heating (Almond et al. 2009). However, in the case of plastic recycling, the number of pollution sources is large, and enforcing the regulation is often difficult. Although we do not have the exact number of recycling facilities dealing with plastic waste, Ministry of Commerce of the People's Republic of China (2017) reports that there are about one hundred thousand recycling facilities in China, which could be a piece of evidence. Import ban can be regarded as an indirect policy instrument that reduces the amount of plastic waste used by plastic waste recyclers, particularly some small-scale recyclers who do not have proper treatments for the VOC exhaust. In this study, we exploit this unique event and investigate the impact of a substantial drop in the availability of the input for a polluting activity. Second, although several studies have evaluated the global impact of the plastic waste ban very well (Brooks et al. 2018; Qu et al. 2019; Huang et al. 2020), they do not focus on the domestic environmental impact. One of the aims of the policy is to alleviate the severity of local pollution; hence, investigating whether and how the ban could improve the local environment is of relevance to policy makers.

The estimation results of our main analysis suggest that the daily ozone concentration was reduced by 2.8% in treatment areas after the import ban. We also extended the research period and change the outcome variables to other pollutants

to test the robustness. Additional analyses also suggest that the effect of the ban was larger during the last few months of 2018 and the first few months of 2019. Heavily ozone-polluted cities are little affected by the ban while coastal cities with ports are affected more.

The remainder of this paper is structured as follows. Section 2 briefly describes the policy background and explains air protection control in China related to ozone concentration. Section 3 describes our empirical strategy, the model, and the dataset. Section 4 presents the empirical results, robustness checks, and heterogeneous effects. Section 5 concludes our paper.

## 2 Background

### 2.1 The plastic recycling and VOC emission in China

A large volume of plastics is produced and recycled in China. In 2014, the production of all types of plastics in China amounted to 73.9 million tons, and the corresponding consumption was 93.3 million tons. In the same year, the amount of waste plastic recycled was 28.3 million tons, accounting for approximately 30% of the total consumption (Cao et al. 2019). Plastic recycling includes material recycling, chemical recycling, and energy recovery (Al-Salem et al. 2009; Lazarevic et al. 2010). Chemical recycling and energy recovery, however, are not encouraged in China because of fears about the environmental pollution produced by inappropriate treatment in the recycling processes, as a result of which, there are many material recycling plants in China (Huang et al. 2013).

A typical material plastic recycling process includes three main stages. The first stage is collection, where the plastic waste is collected and transported to the plastic recycling facilities. The second stage is sorting, where plastic waste is sorted from other materials and also sorted into different types. While the third stage, which is reprocessing, could cause air pollution if the melting process is treated inappropriately. During the reprocessing stage, plastic waste is first washed and shredded into pieces. Those plastic pieces will be heated to melt and then extruded into amounts of pellets, which are the final products of the plastic recycling activity. The heating temperatures in the melting process range from 200 to 300 $°C$, which may produce harmful components, such as VOCs. He et al. (2015) analyzed the emissions of different types of plastic solid waste and found that the melting extrusion of seven different thermoplastics could produce a substantial volume of VOCs. The total VOC concentrations emitted during recycling were highest in Acrylonitrile butadiene styrene (ABS) and lowest in polycarbonates, which also could bring definite cancer risks to the residents nearby.

These pollutants are often emitted into the atmosphere without sufficient treatment in China. Many devices in small workshops recycling imported plastic wastes are not advanced or sophisticated, which could release harmful exhausts into the air (China Economic Daily 2018). For example, most of the plastic recycling activities in one of the largest plastic recycling centers in China, Xingtan (Guangdong

**Fig. 1** The percentage of Cities meeting the national air quality standard among 74 large Cities in 2016 and 2017 *Source: report on the State of the ecology and environment in China 2017*

The Percentage of Cities Meeting the National Air Quality Standard Among 74 Large Cities in 2016 and 2017
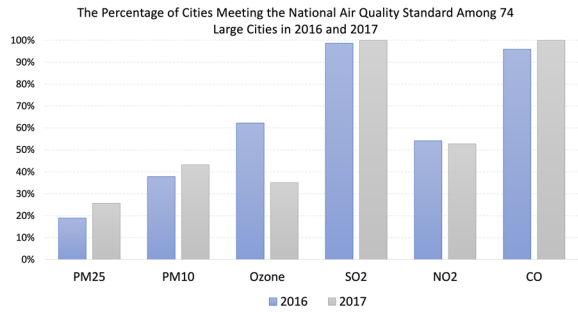


Province), where about 1000 plastic waste recycling plants are concentrated, are carried out in the open air with no measures to treat the emitted harmful gases. Those exhaust VOC gases emitted from plastic waste recycling granulation have an effect on the ambient environment in Xingtan (Huang et al. 2013).

Despite the small-scale recyclers, some large recycling companies also contributed to the pollution activities. The Ministry of Ecology and Environment of China initiated a campaign in July 2017 to manage and control the recycling industry. The ministry examined the acts of 1768 large recycling companies, many of which engaged in plastic waste recycling, and found that 60% of these companies were operating illegally. The illicit acts included using invalid filters, creating fake monitoring data, selling the permit for importing solid waste to other unqualified firms, and building new plants without permission (PRC Central Government 2017).

Notably, a company needs to obtain a permit from the authorities to import solid waste. The permits can be obtained only by companies with qualified facilities that meet the standards set by the government. Some of these companies, however, sell and transfer their permits or imported plastic waste to other firms and small workshops, as one of the government reports shows (Ministry of Ecology and Environment of China 2017a). These firms and workshops may not meet the standards or even not have a filter installed. The permits illegally sold by qualified companies help them and workshops with no permits to obtain materials and process them without enough care for the environment.

## 2.2 The control of the VOC emissions

In January and February 2013, severe haze covered many provinces in China. This led to a widespread environmental concern that pushed the Chinese government to tighten the air pollution regulations (Jin et al. 2016). During 2013 and 2014, China rolled out a new, nationwide air pollution program in three waves across cities to provide real-time monitoring of six air pollutants, PM2.5, PM10, $O_3$, CO, $NO_2$, and $SO_2$ (Greenstone et al. 2021). Since then, the Chinese government has implemented various policies to alleviate the severity of air pollution.

By contrast, the concentration of surface ozone has increased, emerging as one of the major air pollutants in 74 large cities (including cities in key regions such as Beijing, Shanghai, Guangzhou, and Shenzhen) based on the newly amended national

ambient air quality standard. As Fig. 1 shows, the share of cities meeting the standard for ozone among these 74 large cities was about 62.2% in 2016, while the share dropped to 35.1% in 2017. The average daily ozone concentration in the 74 large cities also increased from 154 $\mu g/m^3$ in 2016 to 167 $\mu g/m^3$ (Ministry of Ecology and Environment of China 2017b). As the acceptable concentration of ozone set by the WHO is about 100 $\mu g/m^3$ (World Health Organization 2006), the average daily ozone concentration in the 74 large cities is much larger than the WHO standard.

The control measures for ozone were largely untouched in China until the policy called *13th Five-Year Plan for the Prevention and Control of VOC Pollution* was introduced in 2017 (Health Effects Institute 2019). Through this policy, the government emphasized the importance of reducing VOCs as the main precursors of ozone. The government also set a target for the local governments of the heavily VOC polluted areas in 16 provinces to reduce the VOC emissions by 10% before 2020. This policy is very helpful for us in locating the provinces that also suffered from VOC pollution when we choose suitable control groups for the DID analysis afterward.

### 2.3 The control of the import of plastic waste

The Chinese government had been controlling the import of waste for a long time and eventually concluded it with a ban. The government has frequently revised the *Law of the People's Republic of China on Prevention and Control of Environmental Pollution by Solid Waste* in recent years. Since its introduction in 1996, the law has been revised four times in 2005, 2013, 2015, and 2016 (Sun 2019). Furthermore, the government has been strengthening border control. In 2013, the Green Fence campaign was implemented to enhance border controls. Sun (2019) employed difference-in-differences (DID) methods and found that the Green Fence campaign had a significant effect on reducing the waste imported into China. The government then revised the campaign to the so-called Blue-Sky Campaign and implemented stricter regulations every year from 2017 onward.

The ban on the import of plastic waste was announced in July 2017. The Chinese government set three targets in this policy called the *Implementing Plan for Banning Foreign Trash Imports and Promoting Reform in the Solid Waste Import Management System*. The first target was to reduce the import of solid waste to gradually lessen the environmental damage. The second was to regulate the recycling industry by inspecting illegal behaviors, revising related laws, monitoring smuggling, and promoting job changes. The last was to increase the recycling rate of domestic waste, improve the recycling technology, and build new and cleaner recycling plants.

As a result of this ban, the volume of plastic waste imported to China has greatly reduced. According to the *China Recycling Industry Report 2019* published by Ministry of Commerce of the People's Republic of China (2019), it has reduced from 5.83 million tons in 2017 to 0.05 million tons in 2018, by almost 99% (Fig. 2). We expect the great reduction would have a noticeable effect on the air quality.

**Fig. 2** Import of plastic waste by China *Source: China recycling industry report 2019*
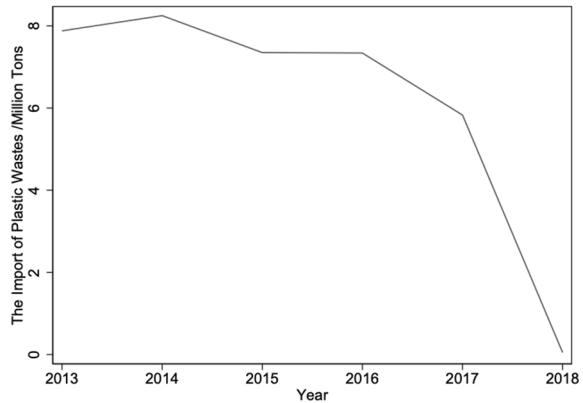


## 3 Methodology and data

### 3.1 Methodology

We employ a DID method to identify the treatment effects and mitigate possible endogeneity problems (Meyer 1995). City-level daily 8-hour average ozone concentration (*Ozone*) is used as a dependent variable to estimate the effect of the plastic waste ban. The baseline model is formulated as follows:

$$Ozone_{it} = \alpha_i + \gamma_t + \beta Treatment * Afterban_{it} + X_{it} + \epsilon_{it}, \tag{1}$$

where $Ozone_{it}$ denotes the ozone level in city $i$ in day $t$ and *Treatment\*Afterban* is the DID indicator that captures the treatment effect. The term is interacted by *Treatment*, which takes one for the cities in the treatment group and takes zero for cities in the control group, and *Afterban*, which takes one after the implementation of the ban and takes zero before the implementation of the ban. We expect the coefficient of the interaction term *Treatment\*Afterban* to be negative if the ban reduces the ozone concentration. We also include city fixed effect $\alpha_i$ to control the characteristics of cities and time fixed effect $\gamma_t$ to capture the seasonal effect of the ozone concentration. $X_{it}$ is a set of control variables including *Max_Temp*, which denotes the daily maximum temperatures, *Wind_Speed*, which denotes the highest wind speed during the day, and a dummy variable *Sunny* for sunny days. We choose these three control variables because we use the maximum 8-hour average ozone concentration covering the daytime, which is affected by the highest temperature, the weather, and the highest wind speed during the daytime. In general, sunshine and high temperature contribute to the formation of ozone while higher wind speed is associated with lower ozone concentration. $\epsilon_{it}$ is the error term and the standard errors are clustered at the city level.

We define the treatment group and the control group through the following steps. First, we follow the *The China Recycling Industry Report 2014* (Ministry of Commerce of the People's Republic of China 2014) and *The China Recycling Industry Report 2016* (Ministry of Commerce of the People's Republic of China 2016) to

locate the treatment group. According to these reports, most of the plastic waste recycling facilities are concentrated in four coastal provinces, Hebei Province, Shandong Province, Zhejiang Province, and Guangdong Province. We think this is reasonable because the plastic waste, as its name suggests, is unwanted garbage to the exporting countries. Compared to the cost of dealing with plastic waste in their own country, it is more beneficial to export the waste to some developing countries, like China. In this sense, the imported price of plastic waste is very low. When plastic waste arrives in the coastal provinces of China, transferring the light and bulky plastic waste from coastal provinces to inland provinces is not economically efficient, considering the value of the waste. Therefore, most of the imported plastic waste is supposed to be recycled in these coastal provinces.

Second, for the control group, choosing provinces having similarities to the treatment groups in terms of geographical and economic activities, particularly ozone pollution status, is important. To achieve this goal, we follow the *13th Five-Year Plan for the Prevention and Control of VOC Pollution*, which gave a list of provinces with heavy ozone pollution that needed to improve their air quality. As all the provinces in the treatment group are listed in this policy, we also choose control groups from the list to keep a similar VOC pollution status. To balance the dataset, we also choose four provinces for the control group, Henan Province, Anhui Province, Jiangxi Province, and Hunan Province, all of which are in proximity to the provinces in the treatment group. The geographical location of the provinces in the treatment in dark blue and control groups in light green is illustrated in Fig. 3.

Third, given the complex relationship between ozone concentrations and its precursor emissions, we further consider the ozone-NOx-VOC sensitivity regime to define the treatment and control groups. Cities in the NOx-limited regime may face a higher concentration of ozone if the concentration of VOCs decreases, which could cause a bias. Therefore we follow the ozone photochemical regime plotted by Jin and Holloway (2015) to find cities located in the VOC-limited and transitional regimes to confirm that the concentration of VOCs contributes to the formation of ozone. Consequently, we have 46 cities from treatment provinces and 31 cities from control provinces.

## 3.2 Data

We use city-level daily data. The data on ozone and other pollutants including PM2.5, PM10, SO2, CO, and NO2 in cities of selected provinces are collected from the China Air Quality Online Monitoring and Analysis Platform.[1]This website collects real-time data on pollutants from the Ministry of Ecology and Environment of China and automatically calculates daily average data. The data for temperature, weather, and wind speed were obtained from Tianqihoubao,[2] which collects historical weather data covering 34 provinces and provides data for 2290

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[1] http://www.aqistudy.cn

[2] http://www.tianqihoubao.com
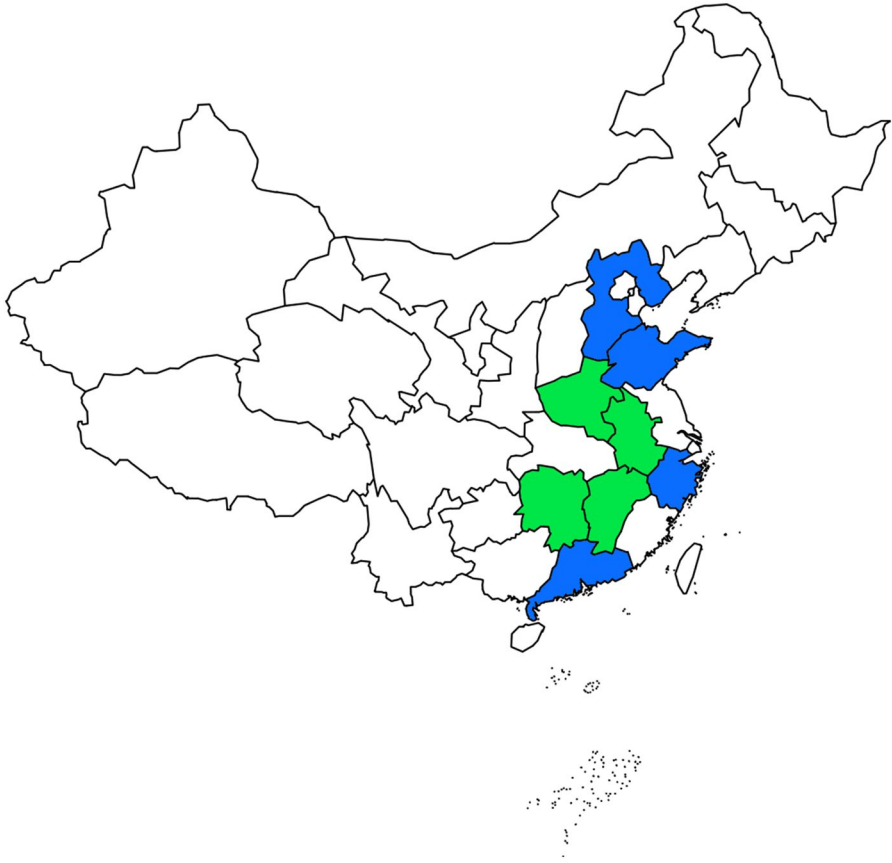
**Fig. 3** Control group (light green) and treatment group (dark blue) (color figure online)

counties. The baseline research period is from January 1, 2017, to December 31, 2018, spanning 1 year before and after the import ban. We could not build a balanced dataset because of some data loss and corruption.

Table 1 reports the descriptive statistics of the data on pollutants and weather for our baseline analysis. Our dataset comprises 53,234 observations, and the average concentration of ozone is 101.2 $\mu g/m^3$. As the acceptable concentration of ozone set by the WHO is 100 $\mu g/m^3$ (World Health Organization 2006), ozone pollution still poses a serious threat to Chinese citizens. As for the weather data, *Max_Temp* takes the real value of the highest temperature in each day at the city level. *Wind_Speed* takes the highest wind speed during the daytime classified by the Beaufort wind force scale at the city level. *Sunny* only takes one when the weather during daytime is sunny and takes zero for any other weather at the city level. The differences in variables between the two groups are insignificant, suggesting the validity of our group selection.

| Table 1 Descriptive statistics | Variables | Obs | Mean | Std. Dev | Min | Max |
|---|---|---|---|---|---|---|
| | *All* | | | | | |
| | Ozone ($\mu g/m^3$) | 53,234 | 101.2 | 50.08 | 1 | 586 |
| | PM2.5 ($\mu g/m^3$) | 53,234 | 48.19 | 37.56 | 2 | 539 |
| | PM10 ($\mu g/m^3$) | 53,234 | 81.55 | 53.88 | 4 | 646 |
| | SO2 ($\mu g/m^3$) | 53,234 | 18.69 | 15.48 | 1 | 217 |
| | NO2 ($\mu g/m^3$) | 53,234 | 32.90 | 18.2 | 2 | 183 |
| | CO (mg/$m^3$) | 53,234 | 0.9656 | 0.5280 | 0.1 | 11.8 |
| | Max_Temp (°$C$) | 53,233 | 21.99 | 10.10 | −20 | 41 |
| | Wind_Speed (*number*) | 53,076 | 3.277 | 0.8590 | 0 | 12 |
| | Sunny (*dummy*) | 53,233 | 0.2706 | 0.4442 | 0 | 1 |
| | *Treatment Group* | | | | | |
| | Ozone ($\mu g/m^3$) | 32,076 | 102.2 | 51.70 | 1 | 586 |
| | PM2.5 ($\mu g/m^3$) | 32,076 | 45.45 | 36.88 | 2 | 539 |
| | PM10 ($\mu g/m^3$) | 32,076 | 79.62 | 56.48 | 5 | 646 |
| | SO2 ($\mu g/m^3$) | 32,076 | 15.70 | 13.56 | 1 | 217 |
| | NO2 ($\mu g/m^3$) | 32,076 | 36.12 | 18.73 | 2 | 183 |
| | CO (mg/$m^3$) | 32,076 | 0.9505 | 0.5430 | 0.1 | 9 |
| | Max_Temp (°$C$) | 32,075 | 22.15 | 10.18 | −20 | 41 |
| | Wind_Speed (*number*) | 32,062 | 3.365 | 0.8919 | 0 | 12 |
| | Sunny (*dummy*) | 32,075 | 0.2938 | 0.4555 | 0 | 1 |
| | *Control Group* | | | | | |
| | Ozone ($\mu g/m^3$) | 21,158 | 99.76 | 47.49 | 4 | 316 |
| | PM2.5 ($\mu g/m^3$) | 21,158 | 52.36 | 38.21 | 3 | 450 |
| | PM10 ($\mu g/m^3$) | 21,158 | 84.47 | 49.56 | 4 | 583 |
| | SO2 ($\mu g/m^3$) | 21,158 | 22.23 | 17.04 | 1 | 126 |
| | NO2 ($\mu g/m^3$) | 21,158 | 28.02 | 16.28 | 2 | 139 |
| | CO (mg/$m^3$) | 21,158 | 0.9885 | 0.5037 | 0.2 | 11.8 |
| | Max_Temp (°$C$) | 21,158 | 21.74 | 9.970 | −7 | 40 |
| | Wind_Speed (*number*) | 21,014 | 3.142 | 0.7873 | 1 | 7 |
| | Sunny (*dummy*) | 21,158 | 0.2352 | 0.4241 | 0 | 1 |

## 3.3 Common trend assumption

One of the most important assumptions of the DID model is the common trend or parallel trend assumption. We provide the trend of the daily average and the monthly average of the ozone concentration in both the control and treatment groups, illustrated in Fig. 4, as the first evidence to validate this assumption. The difference in daily average ozone concentration between the two groups is also provided in Fig. 5 for a better understanding. The daily average ozone concentration indicates a similar trend in each group, while the monthly average shows that the ozone concentration trend in the two groups is parallel most of the time before being treated.
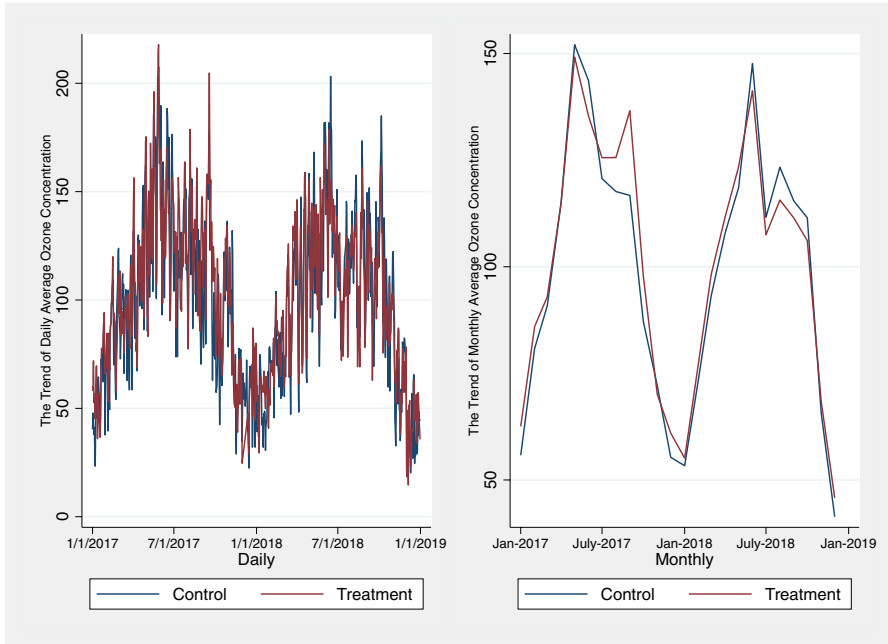
**Fig. 4** Daily and monthly average ozone concentration in control and treatment groups
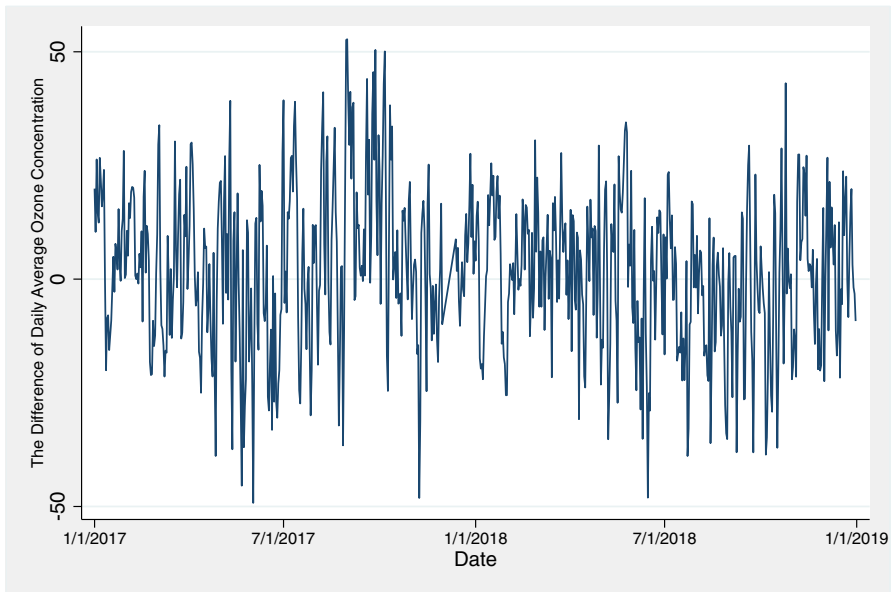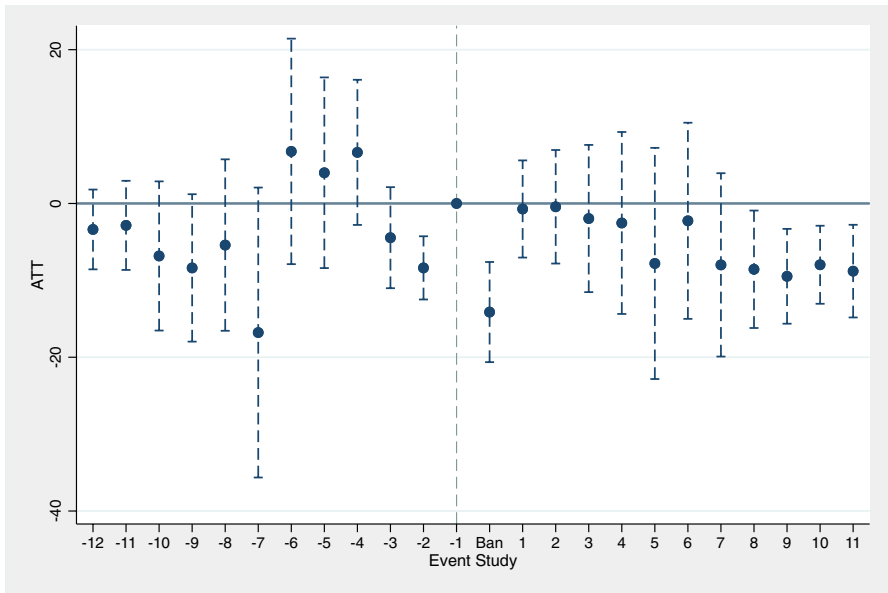


**Fig. 5** The difference of daily average ozone concentration between the two groups

**Fig. 6** Event study for the daily ozone concentration

To test the parallel trend assumption further, we adjust our main model to conduct an event study using the following model:

$$Ozone_{it} = \alpha_i + \gamma_t + \sum_{p=-2}^{-12} \beta_p treatment_i * M_p + \sum_{q=0}^{11} \beta_q treatment_i * M_q + X_{it} + \epsilon_{it},$$

(2)

where *treat* is a dummy variable that equals one when the observation is in the treatment area, $M_p$ is a dummy variable for $p$ months before the ban, and $M_q$ is a dummy variable for $q$ months after the ban. The event study covers the entire research period and the reference group is one month before the ban, that is, December 2017. The other settings are similar to our main model including the control variables and fixed effects. The result of the event study is plotted in Fig. 6 with 95% confidence intervals. The estimations of coefficients in the pre-treated period are not statistically significant, except for the one for November 2017. The figure shows a drop in ozone concentration immediately after the implementation of the ban, although the effect is not statistically significant until the last few months of the year 2018. Overall, the event study provides evidence that the dataset is suitable for performing DID analysis.

**Table 2** Effect of the import ban on the ozone concentration

| Variables | (1) | (2) | (3) | (4) |
|---|---|---|---|---|
| Treatment*Afterban | −2.915** | −3.614** | −4.516*** | −4.617*** |
| | (1.366) | (1.382) | (1.390) | (1.358) |
| Max_Temp | 5.011*** | 4.332*** | | |
| | (0.331) | (0.207) | | |
| Sunny | 4.550*** | 16.50*** | | |
| | (0.864) | (1.182) | | |
| Wind_Speed | −1.569*** | −1.797*** | | |
| | (0.356) | (0.370) | | |
| Constant | −8.395 | 26.68*** | 52.10*** | 60.05*** |
| | (6.217) | (2.927) | (4.595) | (2.224) |
| City FE | Y | Y | Y | Y |
| Date FE | Y | N | Y | N |
| Month FE | N | Y | N | Y |
| Year FE | N | Y | N | Y |
| Observations | 53,075 | 53,075 | 53,234 | 53,234 |
| R-squared | 0.626 | 0.510 | 0.512 | 0.328 |
| Number of City | 77 | 77 | 77 | 77 |

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

## 4 Results and discussion

### 4.1 Baseline results

Estimation results are summarized in Table 2.[3] We use the model in column (1) as the reference because it includes the control variables and the finer time fixed effects. In this case, the estimation of the coefficient of the DID indicator is statistically significant at the 5% level with a value of approximately −2.9. As the coefficient denotes the average treatment effect on the treated, this result suggests that the import ban reduced the average concentration of ozone by 2.9 $\mu g/m^3$, which is approximately 2.8% of the average ozone concentration in the treatment area. Coefficients for the control variables have expected signs and are statistically significant. We believe the variable *Max_Temp* and *Sunny* successfully captured the effect of temperature and sunshine on ozone formation, while strong wind speed, which is captured by the variable *Wind_Speed*, could reduce ozone concentration.

These results support our hypothesis that the import ban reduced the recycling activities of plastic wastes that potentially release VOCs into the atmosphere and contribute to ozone pollution. As one of the main targets of the import ban is to

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[3] We do not include the model using all fixed effects because the month fixed effect and the year fixed effect will cause collinearity with the date fixed effect.

improve the local environmental status, the ban achieves this objective by reducing the inputs. However, the estimated impact is only about 2.8%, suggesting that the recycling of plastic wastes might not be a major source of VOCs in the treatment area.

Notably, Unfried and Wang (2022) also investigate the effect of the import ban while leading to a result different from ours. They find a reduction in the concentration of PM2.5 rather than ozone. We believe the reason for it could be the difference in the research settings. For example, we only include cities in the treatment and nearby provinces under VOC-limited or transitional regimes, which are very likely to suffer from VOC pollution in our study to better capture the effect of the ban on the ozone concentration, whereas they cover all Chinese prefecture-level cities in their study.

## 4.2 Robustness analysis

In our baseline analysis, we focus on ozone concentration to estimate the effect of the import ban on plastic waste. In this subsection, we carry out some analyses to prove the robustness of our baseline result. We first extend our research period from 2017–2018 to 2016–2019 to analyze the effect of the ban in the long run. Next, we change the outcome variables to other pollutants to check the casualty of the baseline analysis.[4]

First, in our baseline analysis, the research period is set from 2017 to 2018, which is one year before the ban and one year after the ban. In this way, we can focus on the effect brought by the ban more clearly and directly considering the uncertainty brought about by longer research periods. The longer research period, however, is effective in evaluating the validity of the policy in the long run and providing further evidence for the common trend assumption. Therefore, we extend the research period to two years before and after the ban, which is from 2016 to 2019. The descriptive statistics for the 4-year dataset are provided in Table 3. The dataset includes 109,376 observations, and the average concentration of ozone is 99.28 $\mu$ g/$m^3$, which is slightly lower than the baseline dataset.

Similar to the baseline estimation, we first plot the extended 4-year event study with 95% confidence intervals in Fig. 7 to examine the trend of the two groups. The estimations of coefficients in the pre-treated period are not statistically significant, except for the one for November 2017, which is the same as the event study of the baseline estimation. The extended one, however, shows that the ban is effective during the last few months of 2018 and the first few months of 2019 while getting weaker from then on. The DID regression results for the expended dataset are presented in Table 4. The estimation of the coefficient of the DID indicator in column (1) is approximately −3.913 and statistically significant at the 1%

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[4] We also intended to extend our dataset to include the other two coastal provinces in China, which are Fujian and Jiangsu Province, and use these two provinces as the treatment group for a robustness check. However, as the common trend assumption is apparently not satisfied according to the event study analysis, we did not include it in this paper.

**Table 3** Descriptive statistics of the extended dataset

| Variables | Obs | Mean | Std. Dev | Min | Max |
|---|---|---|---|---|---|
| *All* | | | | | |
| Ozone ($\mu g/m^3$) | 109,376 | 99.28 | 49.83 | 1 | 586 |
| PM2.5 ($\mu g/m^3$) | 109,376 | 49.37 | 39.81 | 1 | 703 |
| PM10 ($\mu g/m^3$) | 109,376 | 83.41 | 57.64 | 4 | 886 |
| SO2 ($\mu g/m^3$) | 109,376 | 18.33 | 15.99 | 1 | 217 |
| NO2 ($\mu g/m^3$) | 109,376 | 34.50 | 18.42 | 2 | 183 |
| CO (mg/$m^3$) | 109,376 | 0.9750 | 0.5628 | 0.1 | 18.4 |
| Max_Temp (°C) | 109,361 | 21.78 | 10.05 | −30 | 41 |
| Wind_Speed (*number*) | 108,992 | 3.232 | 0.9483 | 0 | 12 |
| Sunny (*dummy*) | 109,361 | 0.2607 | 0.4390 | 0 | 1 |
| *Treatment Group* | | | | | |
| Ozone ($\mu g/m^3$) | 64,933 | 100.1 | 51.42 | 1 | 586 |
| PM2.5 ($\mu g/m^3$) | 64,933 | 46.44 | 39.12 | 1 | 703 |
| PM10 ($\mu g/m^3$) | 64,933 | 81.13 | 59.65 | 4 | 886 |
| SO2 ($\mu g/m^3$) | 64,933 | 16.47 | 15.52 | 1 | 217 |
| NO2 ($\mu g/m^3$) | 64,933 | 36.38 | 19.02 | 2 | 183 |
| CO (mg/$m^3$) | 64,933 | 0.9569 | 0.5808 | 0.1 | 18.4 |
| Max_Temp (°C) | 64,918 | 21.96 | 10.13 | −30 | 41 |
| Wind_Speed (*number*) | 64,866 | 3.317 | 0.9892 | 0 | 12 |
| Sunny (*dummy*) | 64,918 | 0.2861 | 0.4519 | 0 | 1 |
| *Control Group* | | | | | |
| Ozone ($\mu g/m^3$) | 44,443 | 98.11 | 47.38 | 2 | 316 |
| PM2.5 ($\mu g/m^3$) | 44,443 | 53.66 | 40.41 | 3 | 665 |
| PM10 ($\mu g/m^3$) | 44,443 | 86.75 | 54.41 | 4 | 823 |
| SO2 ($\mu g/m^3$) | 44,443 | 21.04 | 16.29 | 1 | 176 |
| NO2 ($\mu g/m^3$) | 44,443 | 31.76 | 17.15 | 2 | 172 |
| CO (mg/$m^3$) | 44,443 | 1.001 | 0.5344 | 0.1 | 11.8 |
| Max_Temp (°C) | 44,443 | 21.51 | 9.912 | −7 | 40 |
| Wind_Speed (*number*) | 44,126 | 3.107 | 0.8698 | 1 | 7 |
| Sunny (*dummy*) | 44,443 | 0.2237 | 0.4168 | 0 | 1 |

level. The impact of the ban during the extended research period is about 3.9%, which is a bit larger than the baseline model. This is mainly because the ban is still effective during the first few months of 2019 as the above event study shows. We think this analysis using an extended research period supports our main analysis and gives evidence of the effect of the ban in the long run.

Second, we run the same model as column 1 in Table 2 but using the concentration of PM2.5, PM10, SO2, NO2, and CO as outcome variables. The results are reported in Table 5 and we do not find statistically significant results for any of these pollutants. We think these results also support our results as our
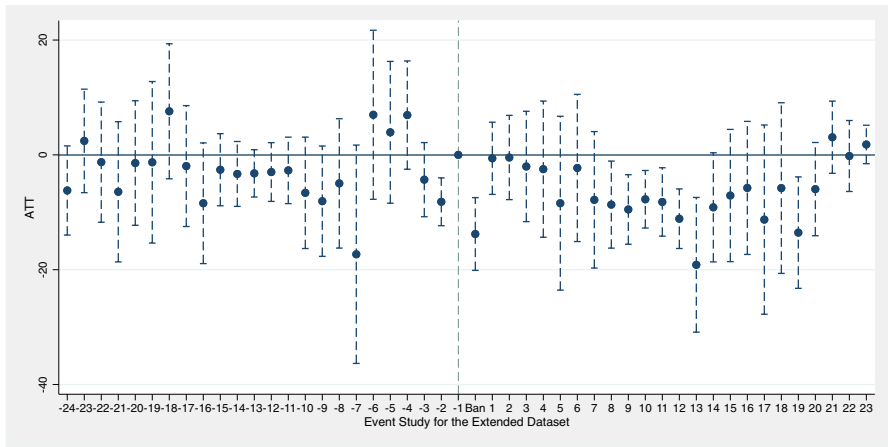
**Fig. 7** Event study for the extended dataset

**Table 4** DID estimation result of the extended dataset

| Variables | (1) | (2) | (3) | (4) |
|---|---|---|---|---|
| Treatment*Afterban | −3.913*** | −4.748*** | −4.438*** | −4.504*** |
| | (1.355) | (1.393) | (1.437) | (1.415) |
| Max_Temp | 4.822*** | 4.094*** | | |
| | (0.313) | (0.193) | | |
| Sunny | 5.682*** | 16.71*** | | |
| | (0.825) | (1.305) | | |
| Wind_Speed | −1.148*** | −1.161*** | | |
| | (0.354) | (0.322) | | |
| Constant | 0.757 | 16.19*** | 57.22*** | 47.12*** |
| | (5.392) | (2.698) | (3.267) | (2.120) |
| City FE | Y | Y | Y | Y |
| Date FE | Y | N | Y | N |
| Month FE | N | Y | N | Y |
| Year FE | N | Y | N | Y |
| Observations | 108,988 | 108,977 | 109,376 | 109,365 |
| R-squared | 0.623 | 0.510 | 0.513 | 0.340 |
| Number of City | 77 | 77 | 77 | 77 |

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

hypothesis suggests that other pollutants should not or be affected or only margin-ally affected by the import ban.

Furthermore, according to United States Environmental Protection Agency (2018), in the U.S., while there are 7.56 million tons of VOCs emitted from other

**Table 5** Effect of the import ban on other pollutants

| Variables | (1) PM2.5 | (2) PM10 | (3) SO2 | (4) NO2 | (5) CO |
|---|---|---|---|---|---|
| Treatment*Afterban | 0.696 | 1.930 | 0.485 | 0.722 | −0.0172 |
|  | (1.030) | (1.689) | (0.720) | (0.617) | (0.0245) |
| Max_Temp | −0.0959 | 0.725*** | −0.0837 | 0.146* | −0.00989*** |
|  | (0.151) | (0.226) | (0.0556) | (0.0786) | (0.00194) |
| Sunny | −3.506*** | −1.272 | 1.043*** | 0.631* | −0.0420*** |
|  | (0.770) | (1.076) | (0.294) | (0.321) | (0.0127) |
| Wind_Speed | −2.807*** | −2.109*** | −0.999*** | −2.846*** | −0.0467*** |
|  | (0.354) | (0.490) | (0.177) | (0.246) | (0.00674) |
| Constant | 165.3*** | 219.9*** | 46.47*** | 77.76*** | 2.790*** |
|  | (8.246) | (11.76) | (2.724) | (2.621) | (0.191) |
| City FE | Y | Y | Y | Y | Y |
| Date FE | Y | Y | Y | Y | Y |
| Observations | 53,075 | 53,075 | 53,075 | 53,075 | 53,075 |
| R-squared | 0.517 | 0.501 | 0.366 | 0.562 | 0.441 |
| Number of City | 77 | 77 | 77 | 77 | 77 |

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

industrial processes in 2014, there are also 2.38 million tons of VOCs emitted by on-road vehicles and 1.76 million tons of VOCs emitted by nonroad vehicles and engines, which takes a large share of the VOC emission. Therefore, there is a concern that changes in transportation might have an impact on our results. However, as European Environment Agency (2021) reports, about 55.38% of NOx is related to the transportation sector while only 9.28% of non-methane VOCs are related to the transportation sector. If our results are affected by some variables related to the transportation sector, we are supposed to observe a greater impact on NO2 concentration. As we do not find statistically significant results for the NO2 concentration, this robustness analysis also indicates our results are not affected by changes in the transportation sector.

### 4.3 Heterogeneous effects

In this subsection, we explore the heterogeneity in the treatment effect to better understand the impact of the import ban. We first investigate if the effect of the ban varies as time evolves. Next, we explore the difference in effects among cities with higher or lower baseline ozone concentrations. Further, we examine the difference between harbor cities and inland cities.

    First, the effect of the ban may not appear immediately because many companies are supposed to keep stock of recyclable materials to maintain a stable supply. Even if the import of plastic waste is banned, many recycling firms can keep

| Variables | Ozone |
|---|---|
| **Table 6** Time effect of the import ban | |
| Q1 | −2.107 |
| | (1.907) |
| Q2 | −1.001 |
| | (2.726) |
| Q3 | −3.053 |
| | (2.996) |
| Q4 | −5.553** |
| | (2.551) |
| Max_Temp | 5.015*** |
| | (0.336) |
| Sunny | 4.524*** |
| | (0.872) |
| Win_Speed | −1.581*** |
| | (0.351) |
| Constant | −8.401 |
| | (6.207) |
| City FE | Y |
| Date FE | Y |
| Observations | 53,075 |
| Number of City | 77 |
| R-squared | 0.626 |

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

processing plastic waste using stocks for some periods. Thus, the effect of the ban may change with time.

Therefore, we divide the DID indicator *Treatment\*Afterban* into 4 variables ($Q_1$, $Q_2$, $Q_3$, $Q_4$) to capture the change during the four quarters in one year. The variables $Q_1$, $Q_2$, $Q_3$, and $Q_4$ take a value of 1 when they are in the first, second, third, and fourth quarters of the year 2018, respectively. Hence, the model is represented as follows:

$$Ozone_{it} = \alpha_i + \gamma_t + \beta_1 Q_{1it} + \beta_2 Q_{2it} + \beta_3 Q_{3it} + \beta_4 Q_{4it} + X_{it} + \epsilon_{it}. \tag{3}$$

The result is presented in Table 6. Although the estimation of the coefficients of Q1, Q2, and Q3 are negative, the coefficients are not statistically significant. However, the coefficient of Q4 is negative, statistically significant at the 5% level, and larger in value than those in the other quarters. This result indicates that the effect of the ban was not so strong during the first three quarters of 2018, and the effect of the ban appeared in the last quarter of 2018. Both of the event studies plotted for the baseline analysis and the robustness analysis also provide a similar trend. We believe this is because the stock of plastic waste could maintain the recycling process for a while until the material was consumed. Alternative plastic

**Table 7** Effect of the import ban in heavy polluted areas

| Variables | Ozone |
|---|---|
| Treatment*Afterban | −2.886 |
| | (1.943) |
| Max_Temp | 4.985*** |
| | (0.411) |
| Sunny | 2.040** |
| | (0.927) |
| Wind_Speed | −1.189** |
| | (0.507) |
| Constant | −5.936 |
| | (4.890) |
| City FE | Y |
| Date FE | Y |
| Observations | 29,358 |
| Number of City | 43 |
| R-squared | 0.776 |

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

waste sources such as domestic plastic waste, however, will offset the effect of the ban afterward.

Second, we examine if the ozone pollution status before the implementation of the ban influences the effects of the ban. For example, areas with higher average ozone concentrations in the treatment group might suffer more from the VOC exhaust released by plastic waste recycling, and therefore, the effects might be larger in those areas. To verify this hypothesis, we conduct a sub-sample analysis with the cities where the ozone concentration before the ban is higher than the average value. After the resampling, we have 26 cities in the treatment group and 17 cities in the control group.

The result is reported in Table 7. In this analysis, we obtain a negative but not statistically significant result: the import ban does not have a significant impact on the ozone concentration in heavily polluted areas. We believe this is reasonable as the recycling of plastic wastes might not be a large source of VOC pollution in these cities.

Last, we compare the impact among cities with and without ports. As most of the plastic wastes are imported by ship and the transportation costs compared to the value of these plastic wastes are high, we believe considerable plastic wastes are recycled in coastal cities with ports. Thus, the harbor cities in the treatment areas are very likely to experience a larger effect from the ban. Therefore, we use the cities with harbors in the treatment group and keep the control group the same as before. After this resampling, we have 21 cities in the treatment group and 32 cities in the control group. Furthermore, we also use cities with big harbors (Harbors with over 10,000,000 tons total trading amount per year) in the treatment group to confirm our expectations.

| Variables | (1) | (2) |
|---|---|---|
| | Cities with Ports | Cities with Big Ports |
| Treatment*Afterban | −3.312* | −5.466** |
| | (1.810) | (2.038) |
| Max_Temp | 5.070*** | 5.127*** |
| | (0.207) | (0.256) |
| Sunny | 5.595*** | 3.624*** |
| | (1.064) | (1.119) |
| Wind_Speed | −1.984*** | −1.404*** |
| | (0.402) | (0.438) |
| Constant | −11.22* | −12.24* |
| | (5.946) | (6.326) |
| City FE | Y | Y |
| Date FE | Y | Y |
| Observations | 35,693 | 27,975 |
| R-squared | 0.610 | 0.668 |
| Number of City | 52 | 41 |

**Table 8** Effect of the import ban on harbor cities

Robust standard errors clustered at city-level in parentheses

***$p<0.01$, **$p<0.05$, *$p<0.1$

The result is reported in Table 8. The estimation of the coefficients for the DID indicator *Treatment*Afterban* is still negative and statistically significant. Compared to our baseline result, the coefficient of the DID indicator *Treatment*Afterban* has a larger absolute value in both columns. This result supports our hypothesis that the import ban indeed has a larger effect on the harbor cities where a substantial amount of plastic wastes are recycled.

## 5 Conclusion and implications

In this study, we investigated the effects of the ban on the import of plastic waste on air quality using China's city-level daily ozone concentration data. First, we tested the common trend assumption by event study. The results of our baseline DID model suggested that the ban might help reduce the ozone concentrations in the treatment area where many plastic recycling firms are located. The estimated coefficients indicated that the ban contributed to a decrease in the ozone concentration in the treatment group by approximately 2.9 $\mu g/m^3$. Second, we extended our research period to two years before and after the ban and examined the effect of other pollutants to provide further evidence supporting our baseline results. Furthermore, we tested the coefficients among different quarters of the treated year of 2018. Combined this result with the event study for the extended research period, we found that the ban was only effective during the last few months of 2018 and the first few months of 2019. Our heterogeneous analysis also demonstrated that the effect of the ban on areas heavily polluted by ozone was weaker

while the effect on harbor cities was larger. In summary, we captured the environmental effects caused by the ban in our research settings.

The data availability prevents us from implementing a more detailed analysis. As we could not capture the exact location and the situation of the plastic recycling firms and workshops, defining the treatment and control groups at the exact city level or even the monitoring-point level is challenging. We also do not know the exact share of VOCs emitted by the plastic waste recycling firms which could have led to more precise results. Despite these limitations, our approach allowed us to examine the impact of the import ban in a simple framework.

We believe that in the long run, the import ban will lead to an increase in the recycling of domestic plastic waste. Therefore, the effect that we found might be temporal because of a possible increase in the supply of recyclable materials from domestic waste. For instance, several large cities in China, such as Shanghai, have already started waste sorting in 2019. This will increase the domestic supply of recyclables, reviving the domestic plastic waste recycling industry. To maintain the local air quality, however, the ban might not be enough. We think the plastic waste recycling industry needs to transform its process by adopting new technologies, which could be supported by policymakers.

## Declarations

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