The impact of low emission zones on PM_{10} levels in urban areas in Germany

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Abstract

High levels of particulate matter scaling less than 10 micrometers in diameter (PM_{10}) in many urban areas have led to the introduction of binding PM_{10} limit values by the European Commission in 2005. Road transport in inner city areas is believed to be one of the main contributors to accumulated PM_{10} levels and, thus, is the focus of regulation. One of the strongest regulatory mechanisms to meet the new PM_{10} air quality standard is the introduction of low emission zones (LEZs) in Germany. This policy allows local authorities to define geographical areas in urban agglomerations as LEZs, into which vehicles that do not meet predetermined emission standards are prohibited from entering. This paper evaluates the effectiveness of LEZs on reducing PM_{10} levels in German cities. We employ a fixed effects panel data model to analyze the effects of LEZs on daily PM_{10} levels using data from 2000 to 2009. We take into account daily data for meteorological conditions and traffic volume. The results of the analysis reveal that the introduction of LEZs has significantly reduced daily PM_{10} levels in urban areas. We can also show that PM_{10} levels are significantly driven down further when LEZ standards in cities become more stringent over time.

Key words: Particulate matter, low emission zones, panel data

JEL Classification: Q58, R49

1. Introduction

While road transport without doubt contributes significantly to growth and development of economies,[1] this positive impact, at the same time, comes at an environmental cost. Transportation has, inter alia, negative impacts in relation to global climate change, air quality, noise and land use. The annual environmental costs of transportation in Germany, for example, are estimated at 40 billion Euros of which 90 percent can be attributed to road transport.[2] These costs constitute a negative technological externality, which arises as a coproduct of the actual service of transporting goods and services. In the absence of environmental regulation, the negative environmental costs are not relevant for the polluter's decision making, because they are not borne by him but by the environment and general public. As a consequence, the resulting market equilibrium does not represent an optimal solution from a societal perspective. This suboptimality serves as a normative justification for environmental regulation of transportation markets.

Environmental regulation on emissions in relation to particulate matter, as one of the main current concerns regarding air quality, has increased over the last years. Particulate matter consists of solid matter and liquid droplets subdivided into fractions of different sizes that are suspended in the air as particles. So far, European regulations concentrate on particulate matter with an aerodynamic diameter less than 10 micrometers (PM_{10}), which will, therefore, be the focus of this paper.

Particulate matter emissions stem from both natural and anthropogenic sources.[3] Natural sources are, for example, soil erosion, sea salt, emission from volcanos or forest fires. The anthropogenic part of particulate matter emissions originates from motor vehicles, industrial activities, biomass burning and others. Ample literature shows that particulate matter has a significantly negative impact on human health.[4] Particulates contribute to premature mortality and morbidity because they cause cardiovascular and respiratory diseases by penetrating the lungs and, depending on their size, by entering the blood system.[5] Lahl/Steven (2005), for example, show that particulate matter emissions lead to a decrease in average life expectancy by more than 8 months in the EU 25. German life expectancy is decreased by 10 months on average.[6] Annualized costs of premature mortality and morbidity due to PM_{10} are estimated to amount to between 270 and 780 billion Euro across the EU-25.[7]

High PM_{10} levels are particularly found in urban areas along busy roads, where - at the same time - population density and the number of exposed people is among the highest, which exacerbates the health impact of PM_{10} emissions from road transport.[8] Road transport is actually the prime contrib-

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[1] See e.g. Fernald (1999) or Ozbay et al. (2007).

[2] See Schreyer et al. (2007).

[3] See for the following information Umweltbundesamt (2009).

[4] See e.g. Dockery et al. (1993), Ostro/Chestnust (1998), Chay/Greenstone (2003), Chay/Greenstone (2005). For a WHO-lead meta-analysis of more than 400 studies on the negative impact of air pollution on health see Anderson et al. (2004).

[5] See Dockery et al. (1993), Hoffmann et al. (2009) and Pope et al. (2002).

[6] See Lahl/Steven (2005).

[7] See Watkiss (2005).

[8] See Diegmann et al. (2006), Umweltbundesamt (2011).

utor to the inside city concentration of PM_{10}.[9] Motor vehicles add to the PM_{10} level by exhaust emissions, brake and tire abrasion, road wear and resuspension of road dust and soil. Consequently, policies to mitigate the impact of particulate matter often focus on road transportation in cities.

The European Union first addresses the issue of air pollution caused by PM_{10} in a 1996 framework directive, which, inter alia, lists PM_{10} as an air pollutant, for which limit values needed to be developed.[10] This is carried out by the Council Directive 1999/30/EC (also called the Clean Air Directive), which imposes binding PM_{10} limit values in ambient air within the EU.[11] Starting in 2005, member states have to implement provisions so that

(1) a 24 hour limit of 50 µg/m^3 PM_{10} is not exceeded on more than 35 times per calendar year and
(2) the annual average does not exceed 40 µg/m^3 PM_{10}.

Stricter limit values were originally planned to come into force in 2010 but have been abolished by the Council Directive 2008/50/EC.[12] Germany implemented the European Directive and its limit values into national law in 2002 with the 22nd Ordinance of the Federal Immission Control Act (Bundes-Immissionsschutzgesetz - BImSchG**)**. § 47 BImSchG postulates that local authorities have to implement an air quality plan (Luftreinhalteplan) for every city in which PM_{10} levels do not comply with EU regulation. The most stringent policy measure that has been implemented with air quality plans is the introduction of low emission zones (LEZs). Using the 35th Ordinance of marking vehicles with low emissions (Kennzeichnungsverordnung für Kraftfahrzeuge), which went into effect on March, 1st of 2007, Cities and Municipalities can define geographical areas in urban agglomerations as LEZs, into which vehicles that do not meet the standard of a predetermined emission category are prohibited from entering. Starting in January 2008 with just three initial cities, to date (Summer 2012) 43 LEZs have been established.

The purpose of this paper is to investigate the impact of the introduction of low emission zones on urban PM_{10} levels in Germany. The contribution of the research is twofold. First, we add to the sparse literature on the evaluation of LEZs by using a more comprehensive dataset with respect to the time dimension and the urban areas considered. So far research on this topic includes only very few cities and very short time periods. Second, we use local information on traffic volume as explanatory variable for particulate matter emissions, which, to this point, has been omitted in previous analyses. Our results give valuable insight into the effectiveness of the introduction of LEZs in Germany and can function as a guide for policy makers.

The remainder of this paper is organized as follows. Section 2 gives a brief overview of current LEZs in Germany. Section 3 presents previous empirical work on the effect of policy measures for mitigating air quality related emissions. Section 4 deals with the econometric relationship between

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[9] See Krzyzanowski (2005), Diegmann et al. (2006), Jörß/Handke (2007).
[10] See European Commission (1996).
[11] See European Commission (1999).
[12] See European Commission (2008).

PM_{10} and LEZs. It starts by describing the data used, explains our model and presents and discusses estimation results. The final section concludes.

2. Classification and development of low emission zones in Germany

The Ordinance of marking vehicles with low emissions classifies vehicles according to their emission classes based on the European Emission standards.[13] The system applies for all vehicles and follows a simple color code (green, yellow, red). Vehicle owners can buy a colored sticker which shows that their car belongs to a certain emission class. The color a vehicle owner is allowed to purchase is determined by the emission classification number noted in the vehicle documentation. As shown in table 1 there are four different emission classifications, in which a vehicle will either not obtain a sticker because emission is too high, or it will, by decreasing emission level, receive a red, yellow or green sticker. A distinction is made between diesel driven vehicles and gasoline driven vehicles and according to whether the vehicle is equipped with particle filters, a catalytic converter or neither. Gasoline driven vehicles obtain either a green sticker – if they are equipped with a catalytic converter – or no sticker at all.

Table 1: Vehicle Emission classification system for low emission zones

	Requirement for each sticker category			
	no sticker	**red sticker**	**yellow sticker**	**green sticker**
Diesel driven vehicles	Euro 1 or older	Euro 2 or Euro 1 + particle filter	Euro 3 or Euro 2 + particle filter	Euro 4 or Euro 3 + particle filter
Gasoline driven vehicles	Without catalytic converter	--	--	Euro 1 with catalytic converter or better

Low emission zones are classified using corresponding colors. National as well as foreign vehicles which do not meet a certain emission limit according to their emission class are prohibited from entering a predefined geographical area. There are three types of LEZ: Stage 1 LEZs only ban very high emitting, non-sticker vehicles from entering the zone. Stage 2 LEZs ban non-sticker and red sticker vehicles and allow green and yellow sticker vehicles. Stage 3 LEZs only grant access to low emitting vehicles that receive a green sticker. In all three types of LEZs certain exceptions apply, for example for vehicles operated by disabled people, working machinery, vehicles on medical emergency calls, vehicles of the police and fire brigade and armed forces. Vehicle owners whose cars illegally enter any LEZs are fined EUR 40 and, if they reside in Germany, receive one penalty point in the Central Register of Traffic Offenders.

Most LEZs in Germany were initially introduced as stage 1 zones. In case PM_{10} levels in a city remain above the limit values, LEZs can be made more stringent by moving to stage 2 or 3. Currently, 31 of 43 LEZs are designated as stage 2. Seven cities (Stuttgart, Berlin, Bremen, Frankfurt a. M., Hannover, Osnabrück and Leipzig) have implemented stage 3 LEZs and most current stage 2 zones are sched-

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[13] For information regarding LEZs in Germany see Umweltbundesamt 2012(a).

uled to move to stage 3 in 2013. One additional stage 1 zone will be introduced in fall 2012 (Erfurt), which will increase the overall number of LEZs in Germany to 44.

The geographical scope of LEZs is designed as to capture inner city areas with the highest PM_{10} exposure. In some cases this area is large, as for the LEZ 'Ruhr', which was established in early 2012 by merging LEZs of 13 cities in the Ruhr area. It covers an area of 850 km^2 with 3.3 million citizens living in the zone. The second biggest LEZ in terms of area is found in Stuttgart with 207 km^2 (and 590,000 inhabitants). The geographical scope of Berlin's LEZ is small compared to Stuttgart (88 km^2) but it encompasses 1.1 million inhabitants. On the other side of the spectrum, the smallest LEZs in Germany cover only 1 to 5 km^2 and a few thousand citizens.

Figure 1: Spatial distribution of LEZs in Germany and their stages

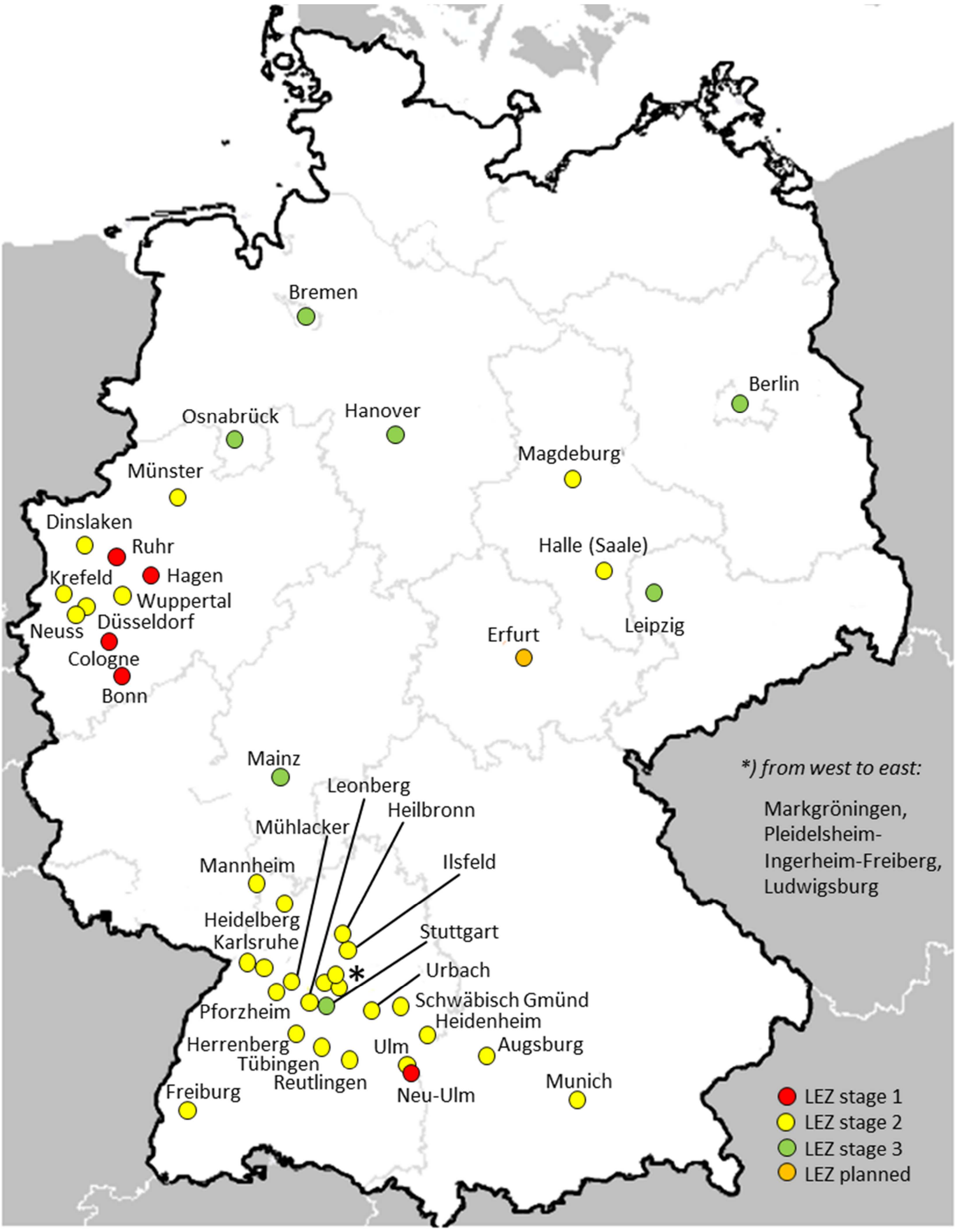

Figure 1 shows that there is a spatial concentration of LEZs in the wider metropolitan region of Stuttgart in the southwest of Germany, which accounts for approximately half of all LEZs. This area has high traffic volumes and unfavorable topographic conditions (mountain ridges), which exacerbates PM_{10} issues: Mountain ridges prevent the horizontal movement of pollutants out of the city, and therefore increase inner city PM_{10} levels.[14] A second cluster can be found in the west of Germany in the Rhine-Ruhr area (Ruhr, Cologne, Düsseldorf, and Bonn) which has high inner city traffic and is densely populated.

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[14] See Davis (2008).

3. Previous research

Air quality impacts of anthropogenic particulate matter emissions have received considerable scholarly attention in the last decades, and, consequently there is extensive literature available on emission measurement and sources of PM_{10}.[15] While these papers are not primarily concerned with researching as to how certain policy measures influence PM_{10} levels, they are still of interest to our work as they help in identifying the explanatory variables which we use for our regression model.

It is generally agreed that PM_{10} levels are influenced by meteorological conditions.[16] The less the atmosphere near the ground can mix, the more it can accumulate particles, because they are not removed through air movement. In a wide-ranging study of European high pollution episodes Baklanov (2010) finds that inversions near the surface, where a warmer air layer above a colder air layer near the ground hinders vertical mixing because it prevents the air from rising is one of the key meteorological factors influencing PM_{10} levels in Northern Europe.[17] Also, stable atmospheric stratification, in which stable layers are formed that hinder mixing and which is often associated with high atmospheric pressure and low wind speed contributes to an accumulation of PM_{10} emissions. In northern Europe, conditions that hinder mixing often occur in winter. Bauer et al. (2007) show that a lack of precipitation and sunny and dry weather conditions, ceteris paribus, also increase PM_{10} levels.[18] This is supported by Klinger and Sähn (2008) and Holst et al. (2008) who analyze the influence of meteorological conditions on PM_{10} levels in German cities. Consequently, we will include a wide range of meteorological data for our estimation, which will be described in more detail later in this paper.

Numerous studies have shown that PM_{10} levels in urban areas are also highly influenced by road transport.[19] Traffic volume can to a certain extend control for the different emission sources by road transport (tailpipe emissions, break and tire abrasion, road wear and resuspension of road dust), as they all vary with the volume of traffic. To explain different emission levels throughout the week, according to the literature it seems important to differentiate between weekdays and weekends, as there is reduced inner city traffic circulation during the weekend.[20] In addition, total PM_{10} levels are also influenced by the composition of the vehicle fleet in regard to the mix of vehicles, their ages and the emission standards. For example, Kelley et al. (2011) simulate traffic emissions in London. They include traffic volume and the composition of the vehicle fleet and also distinguish between different vehicle types like personal cars, motorcycles, taxis, light goods and heavy goods vehicles

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[15] See e.g. the meta-analysis of Viana et al. (2008), or Gillies et al. (2001), Vardoulakis/Kassomenos (2008), Pateraki et al. (2010), Lenschow et al. (2001).

[16] See e.g. the meta-analysis of Viana et al. (2008), or Vardoulakis/Kassomenos (2008), Juda-Rezler (2011), Beevers/Carslaw (2005).

[17] See Baklanov (2010), p. 25.

[18] See Bauer et al. (2006), p. 94.

[19] See e.g. the meta-analysis of Viana et al., or Fuller/Green (2006), Charron et al. (2007), Byrd et al. (2010), Juda-Retzler et al. (2011), See e.g. Lonati et al. (2006).

[20] See e.g. Davis (2008), Lonati et al. (2006), Qin et al. (2004), Morawska et al. (2003), Motabelli (2003), Blanchard/Tanenbaum (2002).

and buses. Consequently, we will also use variables that describe road transport within cities for our estimation.

In addition to analyzing sources of PM_{10} variations, some empirical work has been done on the effect of environmental regulation in relation to air quality in general, and in relation to PM_{10} emissions in particular. The studies briefly discussed below are of interest to us because of the specific econometric approach that was used and the variables that were chosen.

Auffhammer et al. (2009), for example, study the effect of the 1990 Clean Air Act Amendments on PM_{10} levels in the U.S. These amendments state that counties that are found to be in nonattainment with the limit values need to implement measures to reduce PM_{10} levels. If they fail to reduce emissions below the threshold, the U.S. Environmental Protection Agency may impose sanctions on the nonattainment county. The authors use annual average PM_{10} concentration data between 1990 and 2005 in a spatially disaggregated approach at the level of individual monitors. Monthly rainfall and temperature are used to control for climatic conditions. The influence of other socioeconomic factors (annual real personal income, population and employment) at county level is included in the sample as a proxy variable for economic activity - and in turn - transport activity. The estimation is based on a fixed effects model and is conducted in first differences, which eliminates monitor specific unobservable factors. The authors find that the regulation has the highest negative impact on PM_{10} levels at stations within counties that do not attain national limits, but does not lower PM_{10} levels much in the county average. Overall, the findings indicate that the regulation mitigates county specific hotspot problems in relation to PM_{10}.

In another air quality policy study Auffhammer and Kellogg (2011) analyze the impact of federal and state specific gasoline content regulation in the U.S. on ground level ozone pollution. Their unbalanced panel consists of data from measuring stations across different states in the U.S. from 1989 to 2003. They use two approaches, a regression discontinuity design and a difference-in-differences approach. The latter is employed to control for time varying unobservables between treated groups and control groups, the former is employed to control for changes in the ozone level directly before and after the implementation of the gasoline regulation. Meteorological data used consists of minimum and maximum temperatures as well as information on precipitation (rain and snow). They find that the federal gasoline regulation has not significantly reduced ozone pollution in the U.S. They attribute this to the fact that refineries had certain flexibility in relation to which compound to remove from the gasoline. They, therefore, chose to remove the compounds with the lowest abatement costs which were not those compounds with the highest impact on ozone formation. The state of California, contrary to the federal regulations, required certain harmful compounds to be removed from gasoline, which is shown in the study to significantly reduce ground level ozone pollution. The authors conclude that in this case direct measures appear to be more effective than allowing for too much flexibility.

Davis (2008) investigates whether a driving restriction that was introduced in Mexico City in 1989 improved air quality by reducing levels of carbon monoxide, nitrogen dioxide, ozone, nitrogen oxides, and sulfur dioxide. According to the last digit of a vehicle's license plate, a vehicle is prohibited from driving into the city center one weekday per week (Monday to Friday) from 5am to 10pm. Davis employs OLS estimations using a before and after comparison with a regression discontinuity specification to address the problem of endogeneity. He decides against a difference-in-difference approach due to Mexico City's unique geographic position that would make it difficult to find a comparable city as a control group. His unbalanced panel consists of data from 5 to 15 measuring stations on daily air pollution levels for 1986 through to 1993. Among others, he directly controls for temperature, humidity and wind. Davis finds no significant improvement in air quality on days and times the driving ban is in operation but finds a significant reduction in air quality on days and times the ban is not in force. He explains this result with a shift towards higher polluting vehicles, an increase of vehicles on the road and a postponing of trips to times and days when the restriction is not in place. In order to avoid the driving restriction, citizens bought additional and mostly old and high emitting cars or used taxis that are very old and high emitting in general.

For Europe, research on policy regulation in the transport market meant to improve air quality has been done on the impact of the introduction of the congestion charging scheme in London on engine emissions and ambient air quality, including PM_{10} levels, by Atkinson et al. (2009). The authors compare changes in geometric means before and after the introduction of the scheme from monitoring stations in the congestion zone and monitoring stations outside the zone which are unlikely affected by the scheme. They find evidence for a significant but small relative change in PM_{10} levels and attribute this to the introduction of the scheme. However, as the authors point out the study has to rely on only one traffic related monitoring station within the congestion charging zone, and, consequently, site specific factors might have distorted the outcome, so one should be cautious in generalizing the results.

Focusing directly on LEZs in Germany and their impact on PM_{10} concentrations there is one series of revised but not yet peer reviewed working papers by Wolff and Perry.[21] They employ a difference-in-differences approach with control groups being formed based on either geographic proximity or similar PM_{10} levels in 2005. They use daily, disaggregated PM_{10} data from the beginning of 2005 to October 2008 and control, inter alia, for weather and population. Information on traffic volume was not taken into account. As LEZs in Germany were first introduced in early 2008 and in only very few cities at the beginning, their sample is rather small (it consists of only 4 cities with LEZs) and their time series is short. They find that LEZs decrease PM_{10} levels in urban traffic centers by around 9 percent while cities without LEZs do not experience any improvement in relation to PM_{10}.

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[21] See Wolff/Perry (2009), Wolf/Perry (2010a) and Wolf/Perry (2011). Some of their research on LEZs in Germany also appears in Wolff/Perry (2010b) which is otherwise a descriptive summary of PM_{10} regulation in Europe and the U.S.

4. Relationship between PM_{10} and LEZs

4.1 Data

Information regarding previous and current stages of LEZs is available from the German Federal Environment Agency, the Umweltbundesamt (UBA). Different start dates and stage evolvements are taken into account in the time dimension. Overall, we define three implementation groups: cities without LEZ, cities with stage 1 LEZ and cities with stage 2 LEZ.[22]

We obtain a panel of PM_{10} levels from January 1998 to December 2010 from the UBA. The data set includes aggregated daily means of PM_{10} at 613 air quality monitors (measuring stations) in 370 German cities. There are three categories of measuring stations: traffic stations, whose PM_{10} levels are mainly influenced by traffic, industrial stations, significantly affected by emissions from industry plants nearby and background stations, where the source of emission can neither be linked to traffic nor to industry plants. For our research, we use data from traffic stations as the policy measure under assessment in this paper is directed towards road transport. PM_{10} levels are measured by gravimetric and continuous measurement. We use gravimetric measurements whenever they are available, as this is the Europe-wide valid reference method.[23]

In order to take into account local meteorological conditions, we use a broad set of daily meteorological data from 74 measuring stations from 1998 to 2011 from the German National Meteorological Service, Deutscher Wetterdienst (DWD). As air quality and weather monitoring stations are not at the same location, we match air quality stations with the adjacent meteorological stations within a 50 km radius. PM_{10} monitors whose distance to the nearest meteorological station is larger than 50 km are removed from the analysis, as the actual meteorological conditions at the air quality monitor would not be well captured by far-off stations. For some air quality monitors we obtain up to four meteorological stations. In order to determine one indicator per air quality measuring station that captures the local meteorological conditions, the mean value across all assigned meteorological stations is calculated. While the physical connection between PM_{10} levels and meteorological conditions is well documented, there is no consensus among researchers about which variables to include into empirical analysis. In order to give the most detailed picture of the meteorological condition for each city we include all meteorological data that was available to us. Table 2 provides the summary statistics for the meteorological variables.

Additional meteorological factors influence ambient concentrations of PM_{10} and are modeled by using cross products or differences of meteorological variables. This lets us define the following meteorological conditions: days without precipitation in combination with mean air pressure (“dry & high air pressure”), days without precipitation in combination with cold temperatures (“dry & cold”), days with low mean wind force in combination with mean air pressure (“low wind & high pressure”), days with low temperature in combination with mean air pressure (“cold & high pres-

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[22] We limit the scope of the analysis to stage 1 and stage 2 LEZs as the first stage 3 zones were only introduced in 2010.

[23] See Umweltbundesamt (2012b).

sure”), daily mean temperature in combination with daily mean relative humidity (“hot & humid”), daily mean temperature in combination with daily total sunshine duration hours (“hot & sunny”) and the difference between daily maximum temperature and daily minimum temperature (extreme temperature variation during the day). Seasonal dummies for each month are included to control for seasonal climatic conditions and large scale weather phenomena which are not captured by the meteorological data, for example inversion in winter time.

Table 2: Summary statistics of meteorological variables

Meteorological Variables	Unit	Mean	St. Dev.	Min	Max
Daily mean temp. 2 m above ground (TMK)	°C	9.835913	7.522273	-19.3	30.6
Daily min temp. 2 m above ground (TNK)	°C	5.583967	6.720304	-25.1	23.5
Daily max temp. 2 m above ground (TXK)	°C	14.1427	8.8057577	-17.3	40.2
Daily mean relative humidity (UPM)	%	77.7269	12.50254	7	100
Daily mean wind force (FM)	Bft	3.680173	2.08871	0	25.2
Daily maximum wind speed(FX)	m/sec	10.48752	4.425605	0.7	54
Daily total sunshine duration hours (SDK)	hours	4.770284	4.403621	0	16.7
Daily mean degree of cloud cover (NM)	eighth	5.354201	2.136859	0	8
Daily total precipitation (RSK)	mm	1.977577	4.346241	0	158
Daily mean air pressure, station altitude (APM)	hpa	985.925	29.88925	819.6	1045.3
Daily mean vapor pressure (VPM)	hpa	9.910652	4.033439	0.5	24.5
Snow depth (SHK)	cm	1.453528	9.41217	0	250

Contrary to other studies, which use proxy variables to account for the impact of road transport, we explicitly integrate traffic volume into our model. We obtain daily traffic volume data for selected counting stations from the Federal Highway Research Institute (Bundesanstalt für Straßenwesen - BASt) for January 1998 to December 2009 (most recent available month). To match counting stations with PM_{10} measuring stations we examine all counting stations close to PM_{10} monitors. The counting station is included into our sample if the destination of the counted vehicle is either close to the PM_{10} measuring station or will lead the vehicle past it. This approach leads to up to twelve counting stations per PM_{10} measuring station. We again calculate the average of this traffic data for each PM_{10} monitor. Using the data obtained from BASt we compute three daily traffic variables for each of the PM_{10} monitors: traffic volume of all motor vehicles, traffic volume of heavy goods vehicles and buses and traffic volume of personal motor vehicles (including motorbikes, delivery vans and cars with trailer). Table 3 provides the summary statistics for the traffic variables.

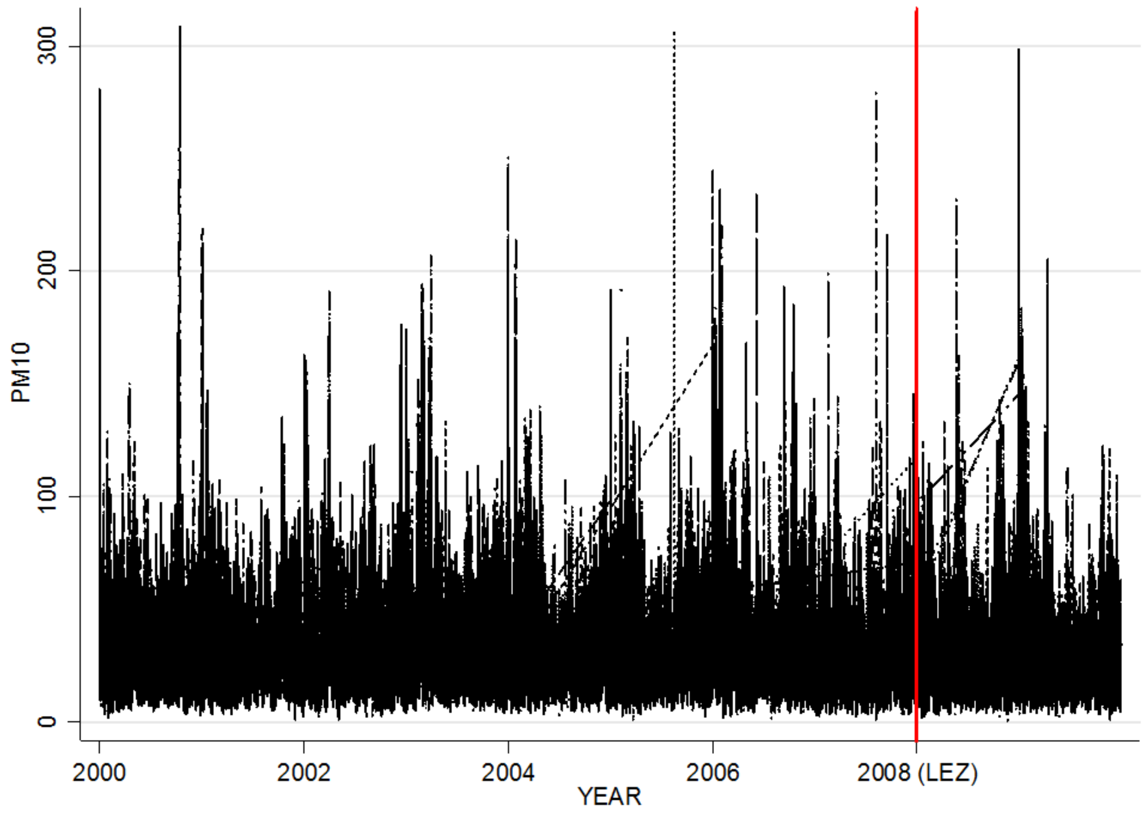
Table 3: Summary statistics of traffic variables

Traffic Variables (number of vehicles per day)	Mean	St. Dev.	Min	Max
traffic volume of all motor vehicles (MV)	41516.59	27803.57	949	198251
traffic volume of heavy goods vehicles/buses (HGV)	4593.969	4376.235	3	30126
traffic volume of personal motor vehicles (PMV)	35076.56	24112.23	833	185673

We exclude all PM_{10} observations from our data set, where no traffic data could be assigned. This is the case when no counting station is located nearby a PM_{10} measuring station or the requirements above could not be fulfilled, so the traffic volume could not be captured adequately. PM_{10} levels through road traffic generally differ on weekdays and weekends. Hence, we create variables for each observation distinguishing between weekdays and weekends. Finally, we drop stations for which less than 329 (90 per cent) of the 365 days of a year have been recorded.[24]

Overall, we obtain an unbalanced panel with data for PM_{10} levels, meteorology and traffic for 224 stations covering 136 cities from January 2000 through December 2009. Figure 2 plots daily average PM10 levels for every traffic station during the period under study.

Figure 2: Daily average PM_{10} levels at traffic measuring stations (2000 - 2009)



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[24] This assumption follows the criteria in Annex 11 of the BImSchV (Federal Emission Control Regulation) for the representativeness of monitoring stations.

Table 4 summarizes PM_{10} levels between 2000 and 2009, as well as annual summary information for the monitors included in our analysis. PM_{10} levels vary widely across days and monitoring stations over a range from below 1 µg/m³ to over 200 µg/m³. On this aggregated level we cannot identify any clear long-term pattern nor a visible decrease in PM_{10} emissions that coincides with the implementation of LEZs.

Table 4: Summary statistics of PM_{10} variables (2000-2009)

Year	Number of monitors	Number of cities	Number of monitored LEZ cities	Mean annual PM_{10} level	Standard deviation	Min	Max
2000	75	54	N/A	29.95509	14.79878	1.8	308.51
2001	79	56	N/A	29.68446	15.65726	1	218.92
2002	83	58	N/A	31.30207	17.47218	1	190.81
2003	100	75	N/A	33.90038	19.12275	3.6	207
2004	102	75	N/A	28.78458	15.38273	3	250.8
2005	126	86	N/A	30.21547	15.53637	0	307.16
2006	153	89	N/A	31.48013	18.55836	1	245.42
2007	166	112	N/A	26.98251	14.81859	1	280.94
2008	158	115	14	25.8918	13.73162	0	232
2009	150	109	27	27.19948	16.86834	0	298.88

4.2 Econometric model

We assume a linear relationship between PM_{10} emissions and our explanatory variables and need to make a choice between a fixed effects and a random effects panel data model. A Hausman test is employed to find the preferred method. The test leads us to reject the null hypothesis of no correlation between individual effects and other regressors in the model, which indicates that a fixed effects model is appropriate. According to the Fisher-ADF and Fisher-PP unit root tests for unbalanced panels, the null hypothesis of nonstationarity can be rejected. A White Test is executed to test for heteroscedasticity. The null hypothesis of homoscedasticity is rejected. Furthermore, we employ the Wooldridge test for serial correlation in panel data with the null hypothesis of no first-order correlation. We reject the null hypothesis and conclude serial correlation. As a consequence, we estimate a robust fixed effects model to control for heteroscedasticity and autocorrelation.[25]

Let $PM10_{i,t}$ denote the PM_{10} level of monitor i on day t. Our econometric model is given by

$$PM10_{i,t} = \beta_0 + \beta_1 LEZ_{j(i),t} + \beta_2 LEZ2_{j(i),t} + \boldsymbol{\gamma W_{i,t}} + \boldsymbol{\delta T_{i,t}} + \beta_3 dweek_t + \boldsymbol{\varepsilon M_t} + \boldsymbol{\theta Y_t} + \tau_i + v_{it}.$$

Let $LEZ_{j,t}$ be a dummy variable that equals 1 when city *j* has implemented a LEZ, no matter the stage, on day *t* and 0 if this is not the case. The index *j(i)* indicates the city *j* to which monitor *i* is

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[25] See Baltagi (2008) for further details.

linked. $LEZ2_{j,t}$ is a dummy variable that takes the value of 1 when city *j* has a stage 2 LEZ on day *t* and the value of 0 otherwise. β_1 and β_2 are the parameters of interest. β_1 measures the difference in PM_{10} levels between LEZ and non-LEZ cities. β_2 explains the difference in impact between LEZs in general and stage 2 LEZs.

$\boldsymbol{W}_{i,t}$ is a vector of the explanatory meteorological variables, which vary over time at the monitor level. $\boldsymbol{\gamma}$ is a vector of unknown meteorological parameters to be estimated. $\boldsymbol{T}_{i,t}$ is a vector of the traffic volume variables presented, which also vary over time at the monitor level. $\boldsymbol{\delta}$ is a vector of traffic volume parameters to be estimated. Additionally, $dweek_t$ is a binary dummy variable that equals 1 when the time variable describes a weekday (Monday to Friday). To control for seasonal climatic conditions, $\boldsymbol{M}_t$ is a vector of binary dummy variables for each month (January to December) that equals 1 when the time variable defines any day of the respective month. The year fixed effects $\boldsymbol{Y}_t$ capture macroeconomic factors that vary over time (for example regulatory measures such as the introduction of the German car scrapping scheme and the European Emission Standards Euro 5/Euro 6, fuel price cycles or the financial crisis). τ_i is the unobserved time invariant monitor specific fixed effect and v_{it} is the idiosyncratic unobserved error component.

4.3 Results and Interpretation

We estimate our model using STATA 12.1. Table 5 displays the main estimation results. The entries are the parameter estimates and their estimated robust standard errors in parentheses.

The key finding is that, after controlling for meteorological conditions, traffic volume and time and seasonal variations, as well as the time invariant unobservables at the monitor level, the implementation of LEZs explains a statistically significant share of the variation in PM_{10} levels in urban traffic areas. The estimated coefficient β_1 is negative and statistically different from zero at the 1 per cent level. The parameter describes a decrease in daily average PM_{10} concentration due to the introduction of LEZs in general. β_2 is also significantly different from zero at the 1 per cent level. Therefore the estimate presents evidence for a significant difference of the impact of LEZs in general and stage 2 LEZs on PM_{10} levels. As the negative parameter indicates that stage 2 LEZs reduce PM_{10} levels slightly more than stage 1 LEZs, this result shows that a more restrictive LEZ does explain a change in PM_{10} levels. This implies that after banning vehicles with the highest PM_{10} emissions by introducing stage 1 LEZs, more restrictive zones reduce PM_{10} levels further.

The coefficients of the meteorological control variables are generally statistically significant at the 1 per cent or 5 per cent level. This supports previous findings that local meteorological conditions have a significant influence on daily average PM_{10} levels, as shown in section 3.

The coefficients δ_2 for traffic volume of heavy goods vehicles and buses and δ_2 for traffic volume of personal motor vehicles are significantly different from zero at the 1 per cent and 5 per cent level. The daily number of heavy goods vehicles and buses as well as personal vehicles in urban areas has a positive influence on PM_{10} emissions, which means that PM_{10} levels increase with the throughput of these vehicles. The variable for motor vehicle traffic is statistical insignificant, which indicates that traffic volume in general has no impact on daily PM_{10} concentrations. This might be due to the fact that the content of the variable is very heterogeneous as it includes all types of vehicles regardless of the emissions. It might also already be explained by the other two variables. Another possible explanation is that the amount of motor vehicles in general does not cause high emission levels, but that traffic volume of high emitting vehicles like trucks and a part of the personal motor vehicles are the decisive factors. Furthermore, the result might suggest that other key metrics for the transport sector such as vehicle kilometers and the composition of the vehicle fleet, which has changed a lot over the last years, may have a higher influence than the traffic volume on PM_{10} levels.[26]

The estimated coefficient for 'the monitoring day is a weekday' (β_3) is positive and significantly different from zero at the 1 per cent level. PM_{10} levels at weekdays are significantly higher than at weekends. This cannot be explained with less traffic activity at weekends due to lower levels of commuter and commercial traffic, because it is already captured by the traffic volume variables. The significance of this parameter shows that even when controlled for traffic volume, PM_{10} levels on weekdays are significantly higher than on weekends. One possible explanation might be that leisure traffic, which dominates traffic on weekends, is driving slower and with lower levels of acceleration than weekday traffic, which, in turn, ceteris paribus reduces PM_{10} levels on weekends.

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[26] See IFEU (2010).

Table 5: Fixed effects regression results (389,689 observations; 223 groups)

Parameter	Description	Estimates
β_1	LEZ	-1.967996*** (0.5644889)
β_2	LEZ2	-5.53039*** (0.4931328)
γ_1	TMK	0.4639963*** (0.15123)
γ_2	TXK-TNK	0.7966155*** (0.0324453)
γ_3	RSK	-0.0301607** (0.0120167)
γ_4	UPM	-0.1855625*** (0.0238352)
γ_5	SDK	-0.1672004*** (0.0515972)
γ_6	FM	-1.135524*** (0.179676)
γ_7	FX	-0.6762339*** (0.0309879)
γ_8	NM	-0.6613102*** (0.0461564)
γ_9	APM	0.1123637*** (0.0079322)
γ_{10}	SHK	-0.0444987*** (0.0093866)
γ_{11}	VPM	3.941866*** (0.1184217)
γ_{12}	COLDDAYAPM	0.000935*** (0.0002687)
γ_{13}	LOWWINDAPM	0.001907*** (0.0003202)
γ_{14}	NORAINAPM	0.001678*** (0.0000907)
γ_{15}	COLDDAYNORAIN	4.058733*** (0.3568162)
γ_{16}	TMKUPM	-0.0308413*** (0.0026322)
γ_{17}	TMKSDK	-0.037043*** (0.0036908)
β_3	Dweek	3.754757*** (0.1965514)
δ_1	MV	-0.0001679 (0.0001031)
δ_2	HGV	0.0003783*** (0.0000747)
δ_3	PMV	0.0001724* (0.0001001)
β_0	Constant	-71.4685*** (7.806219)
	R^2 (within)	0.3755

***Significant at 1 per cent, **significant at 5 per cent, *significant at 10 per cent.

5. Conclusions

This paper contributes to the literature on the effects of environmental regulation by evaluating whether the implementation of LEZs in Germany has decreased levels of PM_{10} in urban areas. We use a novel data set containing information about daily traffic volume, which allows us to directly capture the influence of road transportation on PM_{10} levels, which has been omitted in previous research.

Our analysis is conducted at the monitor level, using data from traffic related air quality monitors in urban areas. Our key finding is a decrease in urban traffic related PM_{10} values that can be attributed to the introduction of LEZs. Additionally, we find a higher incremental impact on PM_{10} levels for stage 2 LEZs than for stage 1 LEZs, which shows that more restrictive LEZs can reduce PM_{10} levels further.

Subsequent analyses will refine this work in several areas. So far we averaged the values for all meteorological stations within a 50 km radius of the air quality monitor in order to attribute meteorological conditions to the monitor. Instead one could weight stations according to the distance to the air quality monitor or just use the nearest station. Applying these different matching schemes we will assess the robustness of the estimation results. The same applies for the matching scheme of traffic stations and air quality monitors that can be adjusted accordingly. We also aim at augmenting the estimations by explicitly controlling for the impact of other regulatory measures on PM_{10} levels, such as the introduction of a car scrapping scheme and the implementation of European Emission Standards Euro 5/Euro 6. Furthermore, we will try to incorporate additional explanatory variables, if we are able to gain access to this data. For the next version of our study we need to find supplementary PM_{10} data from further stations. There are cities that implemented LEZs but needed to be dropped from our estimation because important data points were missing. We also intend to include post 2009 data for traffic volume so we can evaluate the influence of Stage 3 LEZs. Further, we aim at including data on the vehicle fleet and its composition on a city level. This is an important explanatory variable as changes in vehicle stock, which to a certain extend might have been caused by the introduction of LEZs, impact total vehicle emissions and PM_{10} levels.

Our econometric approach, so far, relies on a fixed effects model. While this is a feasible approach in the context of our research question and dataset we will also employ additional models, compare the results and thereby will gain insight into the robustness of our findings.

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