How do low emission zones impact air quality and mobility? Evidence from Brussels.

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HOW DO LOW EMISSION ZONES IMPACT AIR QUALITY AND MOBILITY? EVIDENCE FROM BRUSSELS

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Glossary

AFV	Alternative fuel vehicles
BC	Black carbon
CELINE	Cellule Interrégionale de l'Environnement
DiD	Differences-in-differences
EU	European Union
HDV	Heavy duty vehicle
LEZ	Low emission zone
LDV	Light duty vehicle
NO_2	Nitrogen dioxide
NO_x	Nitrogen oxide
O_3	Ozone
OLS	Ordinary Least Squares
PM_{10}	Particulate matter with a diameter of less than 10 micrometers
$PM_{2.5}$	Particulate matter with a diameter of less than 2.5 micrometers
RMI	Royal Meteorological Institute
VOC	Volatile organic compounds
WHO	World Health Organization

1 Introduction

Over the last decades, concerns about air pollution and its adverse effects on human health and the environment have been growing. Ambient air pollution is recognized as one of the most significant environmental risks to health, contributing to diseases such as lung cancer, stroke, heart disease, and acute respiratory problems, as well as premature deaths. It is estimated to cause approximately 4.2 million premature deaths globally in 2019 (World Health Organization, 2022). Additionally, air pollution deteriorates the environment, leading to issues such as acidification, eutrophication, and damage to various ecosystems, negatively impacting the economy. The European Union estimates the annual economic cost of air pollution to be at least €330 billion (European Commission, n.d.-b). A major contributor to this pollution is the transport sector, which accounts for a quarter of Europe's greenhouse gas emissions, with road transport alone responsible for over 70% of these emissions in 2019 (European Environment Agency, 2022a).

To improve air quality and mitigate its negative consequences, the European Union has introduced air quality standards for various pollutants. In response to these standards, a new policy has become increasingly popular: low emission zones (LEZ). These zones, which restrict access to cities for certain types of vehicles, have been implemented in over 200 cities across Europe in recent decades. Belgium has recently followed this trend by implementing this policy in three cities: Antwerp (2017), Brussels (2018), and Gent (2020).

The primary objective of low emission zones is to mitigate air pollution. However, by restricting access to the city center for certain categories of vehicles, these zones may encourage road users to modify their transport habits. Consequently, this research examines the effectiveness of the LEZ policy in Brussels, which has the largest LEZ in Belgium, with respect to both air pollution and mobility within the city and its surrounding area. This comprehensive analysis aims to determine whether the policy is effective and sufficient on its own, or if it should be complemented by other measures.

To shed light on this issue, the differences-in-differences econometrics approach is employed, enabling a clear identification of the policy's impact on the outcomes of interest. More specifically, the policy's impact on mobility is studied through the composition of the vehicle fleet according to the fuel type. Particular attention is paid to the assumptions underlying this method to ensure the most accurate model possible. Furthermore, to study the impact on the municipalities affected by the policy and those located in its periphery, municipalities on the outskirts of Brussels are removed from the control group and studied separately.

The main findings of the study indicate that the implementation of a low emission zone in Brussels reduced the share of diesel vehicles by nearly 3% within the zone while having little to no impact on other fuel categories. In other words, residents within the area affected by the policy started using fewer diesel vehicles but did not transition to less polluting vehicles such as electric or gas ones following the policy's introduction. However, the policy does influence mobility on the periphery of the city, increasing the share of alternative fuel vehicles (0.04%) and hybrid vehicles (1.2%) and still a decrease in the share of diesel vehicles (-1.5%). This impact remains nevertheless quite small.

As for air pollution, levels of NO_2 inside the zone were reduced after the implementation of the LEZ by respectively 2.1%, while ozone levels experienced an increase of 5.5%. More specifically, the last phase of the LEZ, banning Euro 4 diesel vehicles, had the greatest impact on air pollution. The effect on NO_2 levels is even bigger on the outskirts of Brussels (approximately -5%). Results on the outskirts of Brussels mirroring those in the city, it indicates that road users are not bypassing the restricted area in response to its implementation.

Given the recent implementation of Brussels LEZ, its efficiency has not yet been thoroughly assessed in the literature. A few studies have looked at this case (Bruxelles environnement, 2023; Verbeek & Hincks, 2022), but none carry out an ex-post analysis. Although several studies have analyzed the impact of this policy in other cities using the same methodology (Gehrsitz, 2017; Margaryan, 2021; Sarmiento et al., 2023; Wolff, 2014; Zhai & Wolff, 2021), the lack of harmonization in LEZ designs and the unique characteristics of each city makes it challenging to generalize the results. Furthermore, limited research has focused on how LEZs affect the composition of vehicle fleets according to fuel types (Peters et al., 2021).

This research thus contributes to the literature by providing an ex-post assessment of the Brussels LEZ in terms of mobility and air pollution, using a reliable econometric method. Additionally, this study offers a novel contribution by assessing the impact of the policy on mobility in Brussels through changes in the composition of the car fleet, focusing on fuel types rather than merely the share of non-compliant cars.

This thesis is organized as follows: the context regarding air pollution and policies targeting it, as well as a focus on Brussels LEZ and its design are presented in section 2, followed by a review of the literature in section 3. Section 4 describes how air pollution can be viewed as an economic problem and how it can be dealt with in theory. The data and methodology used for this analysis are described in sections 5 and 6, while the results are presented in section 7. Finally, section 8 examines the influence of various factors on the results, with a focus on the municipalities of Brussels. This section also presents robustness checks and discusses potential limitations and extensions of the research.

2 Context

This section provides a detailed description of the concept of air pollution and its various impacts. It discusses the main policies on air pollution and mobility, with a particular focus on the design and implementation of low emission zones. Finally, it examines the specific case of Belgian LEZs, with an emphasis on Brussels.

2.1 Air pollution and its impact

The World Health Organization (WHO) defines air pollution as "the presence of one or more contaminants in the atmosphere, such as dust, fumes, gas, mist, odour, smoke or vapor, in quantities and duration that can be injurious to human health" (World Health Organization, n.d.). While greenhouse gases are commonly associated with air pollution, other pollutants such as particulate matter (PM), nitrogen dioxide (NO_2), ozone (O_3), and black carbon (BC) also pose significant health and environmental risks. Particulate matter, in particular, is a major source of health problems. The European Environment Agency estimated at 238,000 the number of premature deaths caused by high exposure to fine particulate matter ($PM_{2.5}$) in the European Union in 2020, in addition to 49,000 due to nitrogen dioxide and 24,000 to ozone (European Environment Agency, 2023). In Belgium, ambient air pollution is estimated to cause approximately 300 deaths per million inhabitants annually (Verbeek & Hincks, 2022; World Health Organization, 2016). As mentioned in the introduction, air pollution also degrades the environment with impacts such as acidification, eutrophication, and damage to other ecosystems, thus negatively affecting the economy. For instance, ground-level ozone (O_3) degrades crops, forests, and plants, resulting in over 1 billion Euro of yield lost for wheat in 2019 across the European countries (European Environment Agency, 2022b).

Sources of air pollution are various and change from one pollutant to another. However, transport contributes in general to an important part of air pollution. Road transport contributes significantly to emissions of different pollutants, ranging from 9% for particulate matter (PM_{10} and $PM_{2.5}$) to nearly a quarter of black carbon emissions (European Environment Agency, 2022c). Appendix 1 provides the detailed sources for each pollutant in the EU.

Nevertheless, according to the European Environment Agency (2022), a downward trend has been observed in the emissions of main pollutants over the period 2005-2020. The reasons behind this decrease can be numerous. Several policies have been implemented to reduce air pollution, which could have contributed to this decrease.

2.2 Policy background

Since the 1970s, several policies have been introduced to mitigate the adverse effects of air pollution, notably the convention on long-range transboundary air pollution by the United Nations Economic Commission for Europe (UNECE), displaying the general principles of international cooperation regarding air pollution (Convention on long-range transboundary air pollution, 1979). This convention led later to other policies, in particular the 1999 Gothenburg protocol, formally known as the protocol to abate acidification, eutrophication, and ground-level ozone, which established emission ceilings for the year 2010. This protocol underwent revision in 2012 and is enforced within the EU through the National Emissions Reduction Commitments (NEC) directive (Council of European Union, 2016). The latter sets emissions reduction commitments for 2020 and 2030 regarding five main pollutants.

At the EU level, it is noteworthy to reference the ambient air quality directives. Originating from the 1996 air quality framework directive, two primary directives deserve particular attention: the Cleaner Air for Europe directive (CAFE) of 2008 (Council of European Union, 2008) and the 2004 directive regarding arsenic, cadmium, mercury, nickel, and polycyclic aromatic hydrocarbons in ambient air (Council of European Union, 2004). The former establishes, among other things, quality standards for the emission of twelve different pollutants. These have undergone several amendments. In 2019 the European Commission published an analysis of these directives, concluding that while they have been partially effective in improving air quality, not all the objectives have been met. Consequently, within the framework of the European Green Deal, the European Commission revised these policies, notably by merging the two aforementioned directives into a single one and setting new standards more closely aligned with the recommendations of the WHO which are available in appendix 2 (European Commission, n.d.-c). The table hereunder presents these new standards for the major pollutants.

Table 1: Standards of the European Commission regarding air pollution. To be reached by 1 January 2030

Pollutant	Concentration	Averaging period	Legal nature
$PM_{2.5}$	25 µg/m3	1 day	Not to be exceeded more than 18 times per calender year
	10 µg/m3	1 year	
PM_{10}	45 µg/m3	1 day	Not to be exceeded more than 18 times per calender year
	20 µg/m3	1 year	
NO_2	50 µg/m3	1 day	Not to be exceeded more than 18 times per calender year
	20 µg/m3	1 year	
SO_2	50 µg/m3	1 day	Not to be exceeded more than 18 times per calender year
	20 µg/m3	1 year	

Source: European Commission (n.d.)

Compared to the previous standards, the new ones reduce by half the annual limit for $PM_{2.5}$ (from 20 to 10), NO_2, and PM_{10} (40 to 20). WHO recommendations available, in appendix 2, also cover O_3 and CO.

The previously mentioned European Green Deal, which aims to achieve climate neutrality by 2050, also includes policies to improve air quality, notably the Zero Pollution Action Plan. This plan sets several targets for 2030, including an improvement in air quality to reduce by 55% the number of premature deaths caused by air pollution and a 25% reduction of the EU ecosystems where air pollution threatens biodiversity (European Commission, n.d.-a). Specifically regarding transport, this European Green Deal aims to reduce transport-related greenhouse gas emissions.

Finally, policies not directly targeting air pollution can nevertheless influence it, such as the climate law introduced in 2021 (Council of European Union, 2021), policies on renewable energy, or policies targeting mobility. In terms of mobility and transport, numerous policies have been implemented over the years. Focusing on vehicles and their emissions, the EU has set CO2 emission performance standards for new passenger cars and vans since 2019 (Council of European Union, 2019) and introduced vehicle emissions standards, also known as Euro norms, which will be further described in the following section.

At a smaller scale, the city of Brussels has implemented several actions regarding mobility, many of which are included in the Good Move plan, implemented in August 2022. The actions range from enforcing a speed limit of 30km/h to increasing services related to bikes or electric scooters, as well as road infrastructure maintenance (Bruxelles Mobilité, n.d.). Another recent initiative is ”electrify brussels”, which aims to accelerate the transition to electric vehicles by increasing the number of charging stations for these vehicles.

2.3 Low emission zones

To comply with these European limits and mitigate urban air pollution, numerous cities have instituted low emission zones. These can be defined as restricted areas, that vehicles are allowed to enter only if they meet some criteria regarding their emissions (Holman et al., 2015; Wolff, 2014). Sweden led the implementation of this policy in 1998 with its environmental zones in Stockholm, Göteborg, and Malmö (Holman et al., 2015; Ku et al., 2020). It is more than 200 LEZs that emerged in Europe in the last decades, with Germany and Italy hosting a significant number (Sadler ldt, n.d.). However, the design of these LEZs is not harmonized across Europe, sometimes varying within individual countries. Germany, the Netherlands, and Denmark have established national frameworks, but most countries like Italy or Belgium did not (Holman et al., 2015).

The design of these zones can differ in several ways. Lurkin et al. (2021) distinguishes three main dimensions for setting up this policy: Vehicle, Area, and Time.

First, the type of vehicle targeted differs from one LEZ to another. Some zones restrict access solely for heavy-duty vehicles (HDVs) failing to meet emission criteria, such as London's initial phase of LEZ implementation or certain Dutch LEZs like Utrecht. In contrast, others encompass all vehicle types, including light-duty vehicles (LDVs) (Sadler ldt, n.d.). In addition, the stringency of entry restrictions varies. European emission standards, known as Euro norms, are frequently used to categorize authorized vehicles. These norms are based on the fuel type and registration date, with the highest norm representing the lowest emissions. These standards make a distinction between LDV and HDV. There are currently seven categories for LDV, represented with Arabic numbers, and six categories for HDV, represented with Roman numbers. The latest norm, Euro 7, was implemented in 2022 (European Parliament, 2023). Another important consideration is whether these non-compliant vehicles are entirely prohibited or can gain access by paying a fee.

Second, the area covered by the LEZ varies. London is the world's largest, covering over 1500 km² (Holman et al., 2015). As a comparison, the area concerned by the policy in Brussels is the whole region of Brussels, representing 161 km² (ibsa, n.d.).

Finally, the operational hours of LEZs also differ. While most operate 24/7 all year round, some only apply during specific times, such as in Lisbon and Athens, applying during the weekdays, from 7 am to 9 pm or 8 pm respectively (Sadler ldt, n.d.). Moreover, some Italian LEZs are only effective during winter for passenger cars (Holman et al., 2015).

These discrepancies can make it difficult for road users to comprehend the entry requirements. In addition, some cities are using stickers as a way to categorize allowed vehicles from others, such as German cities or Paris, while others use an automatic plate number recognition system such as Amsterdam, London or Brussels (Holman et al., 2015). These numerous differences represent a challenge for drawing general conclusions from studies on specific LEZs, emphasizing the importance of examining the specific case of Brussels.

As outlined in the policy background, other policies, that are sometimes combined with LEZ, can impact air pollution and mobility. One policy not yet discussed is the congestion charge, which imposes a fee on road users entering a city based on traffic conditions. For instance, London and Milan possess both an LEZ and this type of charge (Ku et al., 2020).

Although most LEZ examples cited are European LEZs due to their prevalence, similar policies exist in other continents, such as Mexico City's "Hoy No Circula" and Seoul's LEZ (Verbeek & Hincks, 2022). This thesis focuses on the case of Belgium, and more specifically Brussels, the following section is devoted to these particular instances.

2.3.1 Belgian's low emission zones

Compared to countries like Germany, which began implementing LEZs in 2008, Belgium was relatively late in establishing the policy. It was only in 2017 that the first Belgian LEZ was established in Antwerp, followed by Brussels in 2018 and Gent in 2020. All of them are operating 24/7 all year round (Sadler ldt, n.d.).

Brussels hosts the largest LEZ in Belgium, covering the entire region of Brussels with its 19 municipalities, representing 161 km² and 1.2 million inhabitants (ibsa, n.d.). As a result, the most extensive data are available for this city, making it the chosen case study for this research.

Figure 1 represents the Brussels region and its municipalities, all included in the LEZ. The main road encircling the region, the Brussels "Ring", is not included in the zone, nor are certain roads providing access to transit parking located on the outskirts of the zone and intended for people with non-compliant cars (LEZ Brussels, n.d.).

Figure 1: Map of Brussels capital region and its municipalities

Source: https://www.cartograf.fr/ville/bruxelles.php

The implementation of this policy in Brussels was realized in several phases, getting progressively more restrictive. The LEZ was officially implemented on the 1st of January 2018, with a transitory period of 9 months during which non-compliant were not fined (Meurice, 2023). The amount of these fines is 350€, with however a maximum of four fines per year (LEZ Brussels, n.d.).

To be allowed in the LEZ, a vehicle can either comply with the restrictions or buy a day pass for 35€. Until 2022, each road user was allowed eight passes per year. This number was increased to 24 in 2022, notably to better support professionals and occasional visitors, according to Bruxelles Environnement (Bruxelles environnement, 2023). In 2022, a total of 47,610 day passes were purchased.

The restrictions have evolved over time. When the policy was introduced, the regulations applied to personal vehicles, vans, and minibusses. Two-wheeled vehicles and heavy goods vehicles were not included, and will only be from 2025. The first step was a ban on Euro 0 and 1 diesel vehicles, extended in 2019 to Euro 2 diesel vehicles and Euro 0 and 1 gasoline vehicles, then Euro 3 diesel in 2020. The more recent step was implemented on the 1st January 2022, with further restrictions on Euro 4 diesel vehicles, with a six-month transitory period, due to the Ukrainian war and its supply shock in the automobile sector (Meurice, 2023). This latest category, Euro 4, was the last generation of diesel vehicles not yet systematically fitted with a particulate filter, emitting up to 6 times more fine particles than those still allowed to circulate (Bruxelles environnement, 2023). Table 2 summarizes the different phases, past and future.

Table 2: Calendar for Brussels LEZ

Vehicle	Fuel	2018	2019	2020	2022	2025	2028	2030	2035
Personal vehicle (M1), vans (N1, class 1)	Diesel	Euro 2	Euro 3	Euro 4	Euro 5	Euro 6	Euro 6d	X	X
	Gasoline	All	Euro 2	Euro 2	Euro 2	Euro 3	Euro 4	Euro 6d	X
Minibuses (M2), vans (N1, class 2 and 3)	Diesel	Euro 2	Euro 3	Euro 4	Euro 5	Euro 6	Euro 6d	Euro 6d	X
	Gasoline	All	Euro 2	Euro 2	Euro 2	Euro 3	Euro 4	Euro 6d	X
Bus (M3)	Diesel	Euro II	Euro III	Euro IV	Euro V	Euro VI	Euro VI	Euro VI	Euro VId
	Gasoline	All	Euro II	Euro II	Euro II	Euro III	Euro IV	Euro VI	Euro VId
Moped (L1-L2)	Diesel					X	X	X	X
	Gasoline					All	Euro 5	X	X
Motorcycle (L3-L5)	Diesel					X	X	X	X
	Gasoline	All (not in scope)				Euro 3	Euro 4	Euro 5	X
Quadricycle (L6-L7)	Diesel					X	X	X	X
	Gasoline					All	Euro 4	Euro 5	X
Heavy vehicle (N2-N3)	Diesel					Euro VI	Euro VI	Euro VId	Euro VIe*
	Gasoline					Euro III	Euro IV	Euro VId	Euro VIe*

Sources: Bruxelles environnement (2023) ; RTBF (2018). Notes: *solely for N2 whose reference mass is above 2.610kg and for N3. The table shows from which category the vehicles are accepted, the red X means that all vehicles are banned.

In addition to the Brussels, Antwerp, and Gent LEZs, the plan to set up a low emission zone throughout the Walloon region was approved in 2019. Last April, however, this plan was repealed. The studies concluded that the implementation of this type of measure would be more relevant at the level of large cities such as Liège or Charleroi, and not for the region as a whole. It was observed that the exceedance of EU limits was very localized and that WHO recommendations were not respected solely in urban centers (wallonie.be, 2024).

EU limits in Brussels

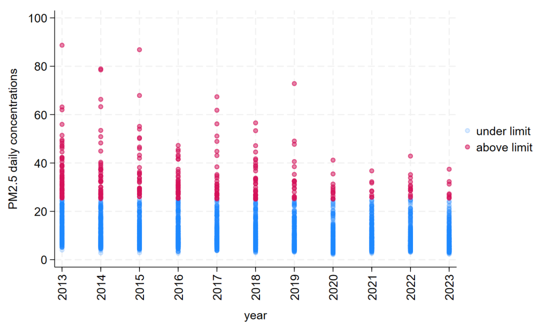
As outlined in the policy background, the EU has set limits for various pollutants, both on a daily and annual basis. Regarding daily limits, the 2022 amendment of these limits sets the number of excess days granted per year at 18. With daily data available for particulate matter and nitrogen dioxide, it is straightforward to verify whether this threshold is exceeded. This verification is particularly important given that Antwerp and

Brussels have both been ranked among the top ten cities with the highest NO_2 mortality burden (Khomenko et al., 2021). Figure 2 depicts the average daily concentrations in Brussels for each of the aforementioned pollutants by year, highlighting in red the ones above the limit.

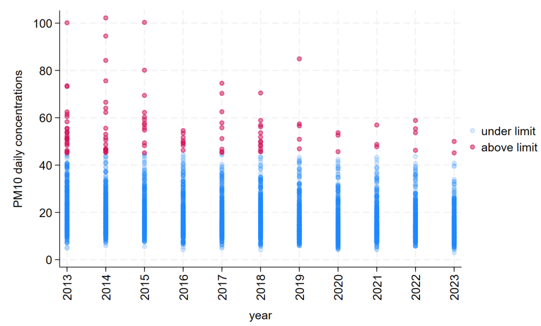
Analyses have indicated that the number of exceedance days for PM_{10} and NO_2 has been consistently above 18 until 2015. Compliance with the EU-set limits for $PM_{2.5}$ has only been achieved since 2020. Consequently, for two pollutants out of three, the goals set by the EU were already achieved before the implementation of the LEZ.

Figure 2: Exceedance days per pollutant

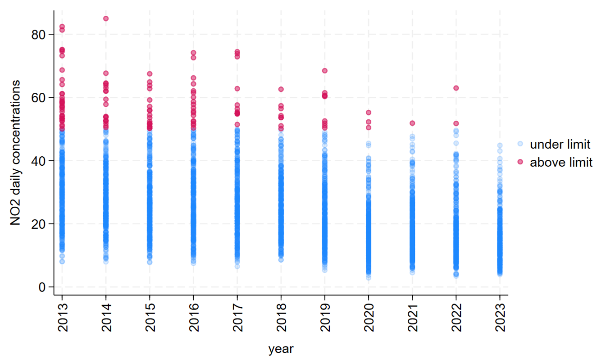
(a) $PM_{2.5}$ exceedance days



(b) PM_{10} exceedance days



(c) NO_2 exceedance days



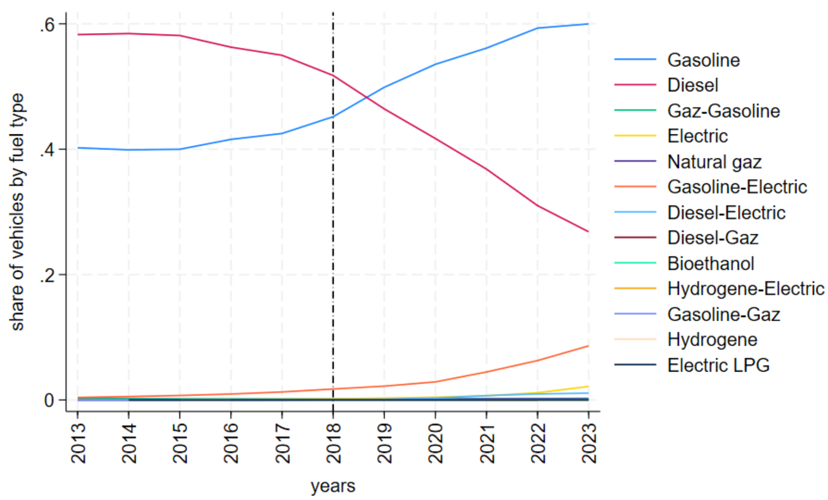
Source: CELINE, own computations and representations

The computations have been done by averaging the 19 municipalities of Brussels for the sake of clarity, but there are differences from one municipality to another. The municipalities located in the area covered by the policy are not all affected by the pollution in the same manner. Appendix 3 displays the pollution levels in the last year before the implementation of the policy, for five pollutants. Saint-Gilles, followed by Evere, appear to be the most polluted municipalities, whereas Auderghem, Watermael-Boitsfort, and Uccle are the least polluted for all pollutants except ozone (O_3). The reasons behind these divergences between pollutants and between municipalities will be further studied later on.

Vehicle fleet in Brussels

As outlined in the introduction, the policy is most likely to affect mobility by preventing access to some vehicles. Before realizing a more detailed analysis as presented in section 7, the mere evolution of the vehicle fleet in regions affected by the LEZ can be represented, according to the fuel types. As figure 3 illustrates, most private vehicles in Brussels are gasoline or diesel vehicles. To be more specific, during the 2013-2017 period, gasoline accounted on average for 40.8% in Brussels and 37% across Belgium, while diesel vehicles constituted 57% and 61% of the vehicle fleet, respectively. Over time, the share of diesel vehicles in the Brussels fleet has decreased, while the proportion of gasoline vehicles has increased. Regarding other categories, the most notable evolution is the rise in the share of hybrid gasoline-electric cars. The econometric analysis will allow to highlight which of these changes can be attributed to the low emission zone.

Figure 3: Evolution of private vehicle fleet in Brussels by fuel type



Source: SPF Economie, own representations

3 Literature review

With low emission zones becoming an increasingly popular policy in European countries, research into this policy has intensified over the last decade. While most of the studies focus on European cities, research on the recently established LEZs of Antwerp (2017), Brussels (2018), and Gent (2020) remains scarce. Apart from the report of the authority responsible for the Brussels' low emission zone (Bruxelles environnement, 2023), a single paper, to the best of my knowledge, is dedicated to the case of Brussels (Verbeek & Hincks, 2022). In contrast, research is way more extensive in the case of Germany, hosting more than 70 LEZs with a national framework (Gehrsitz, 2017; Malina & Scheffler, 2015; Margaryan, 2021; Wolff, 2014) or London being the largest LEZ (Ellison et al., 2013; Ma et al., 2021; Verbeek & Hincks, 2022; Zhai & Wolff, 2021).

Nevertheless, the specific features of each LEZ make it difficult to generalize the results. Moreover, this policy constitutes only one possibility among several policy instruments that can reduce air pollution and/or influence mobility habits. Consequently, it seems appropriate to assess whether the low emission zone is a good policy for Belgium, particularly given the possible extension of this policy to Wallonia.

Throughout the literature, studies on LEZs mainly revolve around three main dimensions: their impact on air pollution, health, and mobility. Some literature reviews have already been conducted (Holman et al., 2015; Ku et al., 2020; Moreno et al., 2022) reviewing the different designs of LEZs throughout Europe (Ku et al., 2020) or focusing on specific LEZs and their associated studies (Holman et al., 2015). However, none combine these three dimensions into a single review. Therefore, the objective of the following literature review is to combine these dimensions, as well as to review the existing literature concerning the inequality aspect of this measure and a specific focus on congestion charge, a policy sharing similarities with LEZ.

3.1 Air pollution

The primary purpose of creating low emission zones is to reduce air pollution and ensure compliance with the limits set by the EU, which explains why the majority of studies evaluating LEZs focus on this aspect.

Before delving into the results, it is important to acknowledge the variations in the way of measuring air pollution from one paper to another, making comparisons difficult when different outcome variables are used. While PM_{10} is the most commonly used measure (Ellison et al., 2013; Gehrsitz, 2017; Wolff, 2014), $PM_{2.5}$, NO_2, and NO_x are also frequently employed. The pollutant black carbon is often overlooked, even though some studies argue that it is a relevant measure of pollution, sometimes more than PM_{10} and $PM_{2.5}$ (Cyrys et al., 2014; Invernizzi et al., 2011). For instance, Invernizzi et al. (2011) observed a significant effect of Milan's eco pass zone on black carbon, while no significant effect on PM_{10} and $PM_{2.5}$ was identified. Lastly, ozone (O_3) is less frequently used as a way to determine the impact of LEZ (Ma et al., 2021; Sarmiento et al., 2023).

The prevailing conclusion of a majority of research is that LEZs have a statistically significant negative effect on at least one of the air pollution measures. Studies focusing on Germany all found a negative effect, going from 2% (Margaryan, 2021) up to 13% (Malina & Scheffler, 2015) on PM_{10}. The study realized by Wolff (2014) specifies that pollution has decreased in urban traffic centers on average by 9%, but not in non-traffic areas. Even though the sign of the effect in these studies is similar, the magnitude can differ by an important amount. The largest impact found is a 22% and 23% decrease in PM_{10} for the LEZ of Lisbon (Ferreira et al., 2015; Santos et al., 2019).

However, a negative impact on one type of pollutant does not imply that all pollutants are significantly reduced. The LEZ of London would have induced a significant reduction of 1-2% of PM$_{10}$ but no significant reduction for NO$_x$ (Ellison et al., 2013). Similarly, some research concluded that the Lisbon LEZ significantly reduced PM$_{10}$, but not PM$_{2.5}$ (Santos et al., 2019), and similarly for Germany with a significant reduction of monthly PM$_{10}$ but no statistically significant effect for NO$_2$ (Margaryan, 2021).

In some instances, LEZs were even found to increase pollution. During the initial phase of London's LEZ, the PM$_{10}$ levels measured at the roadside increased by approximately 14.8%. However, after a tightening of restriction in the second phase, a decrease in PM$_{10}$ levels was observed (Zhai & Wolff, 2021). This initial increase could be attributed to the first phase targeting only heavy good vehicles. In another case, Sarmiento et al. (2023) concluded in air quality improvements but still accompanied by a ground-level ozone increase.

Finally, little to no significant effects were found in a study on the Netherlands (Boogaard et al., 2012) and on the ultra low emission zone of London (Ma et al., 2021).

Several elements can influence the differences in those results. Results between pollutants can be explained by their different characteristics and sources. Given that low emission zones primarily target road usage, it can be inferred that pollutants predominantly emitted by road transport will experience the greatest reduction. As already discussed in the context, part of road transport's contribution to the emissions is different from one pollutant to another. Road transport is responsible for over a third of the emissions of NO$_x$ (37%) and almost a quarter of black carbon (23%) in the EU over the year 2020 (European Environment Agency, 2022c). As for particulate matter, smaller than 10 µm (PM$_{10}$) or than 2.5 µm (PM$_{2.5}$), its primary source is energy consumption in the residential, commercial, and institutional sectors, but road transport still accounts for almost 10% of its emissions (European Environment Agency, 2022c). Lastly, ozone differs from the aforementioned pollutants as it results from the interaction of two other pollutants — NO$_x$ and volatile organic compounds (VOCs) — both partially attributed to traffic. These pollutants react with sunlight to form ozone. As a consequence, emissions of ozone are more consequent in summer than in winter (Wallonair, n.d.). LEZs could therefore have different impacts according to the season (Sarmiento et al., 2023).

In addition to the pollutants themselves, other factors could influence the findings of LEZ-related studies. First of all, the design of the LEZ can play a major role. As discussed in section 2.3, three main factors can vary from one LEZ to another: the vehicles concerned, the area, and the time at which the LEZ operates (Lurkin et al., 2021). The Dutch LEZ, which showed no significant effect, only targeted heavy-duty vehicles, as well as the first phase of the London LEZ. Furthermore, Cruz and Montenon (2016) examine the differing impact of implementing a LEZ under a national versus a local scheme. A national framework has the advantage of being more consistent, thereby enhancing public acceptance, albeit at the expense of flexibility. Moreover, the effect may vary according to the adopted scheme. For instance, the national framework in Germany led to the implementation of numerous LEZs, preventing freight companies from redeploying their non-compliant fleet to other parts of the country, thus leading to a different impact than what might have occurred without the national plan (Cruz & Montenon, 2016).

The methodology employed can also play a significant role in the obtained results. The above-mentioned papers use a variety of methods. The most consistent methodology is the differences-in-differences approach, used by Gehrsitz (2017), Margaryan (2021), Wolff (2014), Zhai and Wolff (2021), and Sarmiento et al. (2023), with some different specifications influencing the statistical inference, such as the size of the clusters or the use of logarithms which will be further discussed in section 6. While other papers merely make a comparison of before and after measurement (Boogaard et al., 2012; Bruxelles environnement, 2023; Invernizzi et al., 2011), other methods such as a regression discontinuity design (Ma et al., 2021), a fixed effect model (Malina & Scheffler, 2015) or other specifications (Ferreira et al., 2015; Santos et al., 2019) are also used.

A large part of the success of a low emission zone follows from the way road users respond to the policy. Different behaviors can be observed after the implementation of the restricted area: drivers can either upgrade their vehicle if theirs is not compliant, can take a longer route to bypass the zone or in the case when the zone is not permanent, change their departure time (Lurkin et al., 2021). Lurkin et al. (2021) conclude that when individuals bypass the zone, the emissions are less reduced than in the other scenarios. Several studies focus on this phenomenon by looking at the spillover effects, such as Zhai and Wolff (2021) which found an increase of 13.1% in PM$_{10}$ levels along major roadways toward London during the first phase of the LEZ. Sarmiento et al. (2023) also observed spillover effects for O$_3$, arguing that this increase is an evidence that O$_3$ travels longer distances than some other pollutants. In other cases, some studies distinguish the effect of the low emission zone within the zone and outside, finding a more important effect inside LEZ versus outside (Ellison et al., 2013; Ferreira et al., 2015).

Finally, factors such as the situations before the introduction of the LEZ such as the fleet composition, and the supply of alternative transport can play an important role in explaining the heterogeneity of results (Lurkin et al., 2021).

Regarding Belgium, as previously mentioned, very few studies focus on this case. The only one evaluating the impact on air pollution is the one of Bruxelles environnement (2023), observing a 31% reduction of NO$_x$, 62% for black carbon, 19% for PM$_{10}$, and 30% for PM$_{2.5}$ between 2018 and 2022. However, it seems that the methodology followed consists only of a before/after comparison, using data on the characteristics of vehicles circulating in the zone. Therefore, analyzing the effect using a differences-in-differences approach should give more accurate information.

3.2 Health and cost-benefit analysis

As outlined in the context section, air pollution has a detrimental impact on health and the environment, which is why these thresholds have been established. Therefore, several papers translate the impact of the LEZ on health outcomes and monetize this impact. As it was the case for air pollution, the variables used to measure health outcomes vary. In the literature, the health impacts of LEZs are measured notably through cardiovascular health (Margaryan, 2021), premature mortalities (Malina & Scheffler, 2015), or birth outcome (Gehrsitz, 2017). Since the way of measuring health is different, it makes it difficult to compare the results. Nonetheless, except for birth outcomes for which no significant results were observed, LEZs have a positive influence on health. However, beyond the strictly physical impact, Sarmiento et al. (2023) focus on the effect on well-being. They conclude that the policy causes a “transitory yet long-lasting reduction in individuals’ life satisfaction despite health benefits”, implying that the influence on well-being caused by the restrictions on mobility potentially outweighs the positive effects on health (Sarmiento et al., 2023).

As previously addressed, lots of factors can influence the efficiency of the policy. Regarding health, for instance, a study by Moreno et al. (2022) concludes that, in the case of Paris, the most efficient scenario regarding health is a LEZ with a large perimeter and with strict standards. In other words, the design of the low emission zone could also influence the effect on health. Moreover, the population considered for the analysis is also a matter of debate. Some models measure exposure to air pollution based on activity, and not residence, arguing that people do not stay home all day (Dhondt et al., 2012; Poulhès & Proulhac, 2021). Dhondt et al. (2012) found significantly different estimates when using an activity-based approach, up to a 12% difference at a local level. Poulhès and Proulhac (2021), using a similar approach, argued that the beneficiaries of the LEZ are not necessarily the ones living inside its perimeter and that the category of people that is the most vulnerable to pollution problems, the youngest and oldest, receive the smallest benefit from the policy. This highlights the importance of considering who truly benefits from the LEZ, on top of merely the total health benefits.

To monetize the benefits of health, the most common methods are using a value of statistical life or cost-of-illness (Margaryan, 2021; Moreno et al., 2022; Wolff, 2014). Determining the cost of implementing an LEZ can be much more complicated. In the few cases where it has been realized (Börjesson et al., 2021; Margaryan, 2021; Savadogo et al., 2023; Wolff, 2014), the costs were based on the cost of upgrading the car fleet. Börjesson et al. (2021) compute the social cost based on the number of cars of each type and the price change of used cars. While there are other costs that could be considered, such as the cost to businesses located within the LEZ, monetizing these is much more difficult.

The cost-benefit analysis resulted in benefits larger than costs in some cases (Margaryan, 2021; Wolff, 2014), but not all the time. Börjesson et al. (2021) observe rather small benefits, notably because they focus on a LEZ targeting light vehicles, which are responsible for a small percentage of NO_2 emissions. Finally, Savadogo et al. (2023) also conclude in a negative net present value in various scenarios for a hypothetical LEZ in Lyon, thus implying that the costs outweigh the benefits. The way of measuring these costs and benefits as well as many factors such as the setup of the zone and the city itself can lead to a variety of results.

3.3 Mobility

By restricting access to city centers to certain types of vehicles, LEZs also have an impact on mobility. This aspect is usually studied through the car fleet evolution. Wolff (2014) and Margaryan (2021) studied this aspect in the case of Germany. Both papers concluded that LEZs lead to a reduction in the number of the most polluting vehicles and an increase in the number of more environmentally friendly vehicles. Moreover, by studying spatial substitution in vehicles' euro standard, Wolff (2014) observed that the closer car owners live to an LEZ, the higher the incentive to switch to less polluting cars is. Additionally, it seemed that the biggest impact was achieved with a strengthening of the rules rather than in the first phase after the implementation (Margaryan, 2021). The research of Ellison et al. (2013) who studied London was in line with the ones just mentioned, adding that the rate of fleet turnover for non-compliant vehicles increased after the implementation of the zone, but went back to its previous level after a few years. All in all, it seems that LEZs trigger an upgrade of the vehicle fleet to vehicles that complies with the rules of the LEZ, but that impact might depend on the stringency of these rules.

As outlined at the beginning of this literature review, the usual metric for assessing the impact on air pollution is the concentration of particulate matter or nitrogen dioxide, while the impact on mobility is typically evaluated through the share of compliant versus non-compliant vehicles, defined by Euro standards. However, by looking at CO2 level and the fuel category, Peters et al. (2021) observed that although the LEZ of Madrid has a significant impact on alternative fuel vehicle registrations, the shift is usually towards alternative fossil fuel-powered vehicles and plug-in hybrid electric vehicles, which have only limited potential to reduce CO2. The share of electric vehicles however does not seem to increase in consequence of the implementation of the policy. Consequently, it seems relevant to study the type of vehicle towards which individuals switch, and not only the compliance of the vehicles.

The vehicle fleet is not the only way through which mobility could be affected. In reaction to the restrictions, people could change their transport habits, for instance by turning to other forms of transportation, such as public transportation or biking. The latter, more specifically public bike sharing, was analysed for the ultra low emission zone of London (Ding et al., 2023). Using the propensity score matching approach, it was concluded that bicycle demand significantly increased by more than 25% following the introduction of this policy (Ding et al., 2023). Nonetheless, no papers focusing on the influence on public transportation were found, perhaps due to a lack of data. Another way to investigate how LEZs affect mobility is by looking at traffic and congestion (Moral-Carcedo, 2022). In Madrid, the LEZ resulted in a substitution effect with a reduction of traffic inside the restricted area but an increase in bordering regions (Moral-Carcedo, 2022).

The change in transport habits, i.e. the modal shift defined as "the decision-making process employed in choosing between different transport alternatives" (Tarriño-Ortiz et al., 2022), results from a multitude of factors. Tarriño-Ortiz et al. (2022) observed that factors such as private car availability, education, household composition, income, or political ideology influenced the modal shift followed by the introduction of a low emission zone. For instance, individuals with university degrees were found to be more likely to keep using their private vehicle to reach the LEZ, as are families with children or elderly, and people with high-income levels.

Bruxelles Environnement provided in their study an analysis of the vehicle fleet in the Brussels region (Bruxelles environnement, 2023). The main conclusions were that the amount of diesel-fuelled passenger cars dropped almost by half since the implementation of the LEZ, amounting to 34% in 2022. The percentage of gasoline passenger cars on the other hand increased by approximately 25% and the share of electric vehicles remains a small percentage of the vehicle fleet, experiencing nevertheless a slight increase. As it was the case for air pollution, these numbers are solely based on an observation of the data from the camera at the border of the LEZ, identifying the amount and the type of vehicle circulating in the zone. Nothing however guarantees that this evolution is in fact due to the LEZ and no other circumstances. Using a differences-in-differences approach, as Margaryan (2021) and Peters et al. (2021), should enable to identify the variation in the vehicle fleet truly caused by the policy.

3.4 LEZ and inequalities

After mentioning the various ways in which the LEZ has an impact, the fairness of the policy should also be considered.

The concept of environmental justice has been increasingly popular in the recent years. It has been defined as *"the fair treatment and meaningful involvement of all people regardless of race, color, national origin, Tribal affiliation, or disability, in agency decision-making and other Federal activities that affect human health and the environment so that people are fully protected from disproportionate and adverse human health and environmental effects (including risks) and hazards, including those related to climate change, the cumulative impacts of environmental and other burdens, and the legacy of racism or other structural or systemic barriers"* (US Environmental Protection Agency, 2024).

In the context of air pollution, part of the literature indicates that low socio-economic groups are exposed to higher levels of pollution (Hajat et al., 2015; Moreno et al., 2022). More specifically, a literature review found that American studies generally support this claim, whereas European studies present mixed results, influenced by the choice of metrics and the particular characteristics of the cities examined (Hajat et al., 2015). Focusing on Brussels, Verbeek and Hincks (2022) identify a negative relationship between air pollution and income.

In line with the concept of environmental justice, low emission zones should not impose a disproportionate burden on any segment of the population in terms of air pollution. As previously cited, Poulhès and Proulhac (2021) highlight a disparity in the benefits derived from LEZs. Their study on Paris concludes that the wealthiest, who are most exposed to air pollution, benefit the most from the policy, whereas the most vulnerable, who contribute little to emissions, gain the least advantage. Furthermore, Moreno et al. (2022) suggest that expanding the coverage area of the policy and tightening restrictions would lead to a more equitable distribution of health benefits.

Beyond air pollution, the ban on older vehicles can also be inequitable, as these vehicles are predominantly owned by lower-income groups in Brussels, according to Verbeek and Hincks (2022).

To conclude, when designing a policy such as a low emission zone, considerations of environmental justice and policy fairness are crucial. This will be the subject of a brief study in the discussion.

3.5 Congestion charge

Low emission zones are not the only type of policy that can influence air pollution in a given area. As it will be further developed in the theoretical framework, policy instruments are often classified into price-based and quantity-based instruments. LEZ falls more into the category of price-based instruments, with its use of emission standards. In the context of this research, a similar policy, falling in the price-based category, is worth discussing: congestion charges, or urban tolls.

This policy consists of charging road users with a fee to enter, exit, or room in the city center during peak hours. Therefore, as LEZ, it targets a specific region, not by restricting its access but by imposing a fee on it. The primary goal of this policy is to mitigate congestion rather than directly reducing pollution. Nevertheless, by charging drivers to enter a certain area, this instrument can influence their transportation choices, thus influencing pollution. Conversely, the main objective of low emission zones is to reduce pollution, but can also influence congestion by restricting access to specific categories of vehicles (Bernardo et al., 2021).

Considering the clear relationship between congestion and pollution (Bernardo et al., 2021), along with the similarities between the two policies, both applying to urban areas and having potential effects on congestion and pollution, one can wonder which is more appropriate. An in-depth comparison seems therefore relevant, as congestion charge could be seen as an alternative to LEZs.

Congestion charges have been implemented in several European cities such as London, Stockholm, Gothenburg, Milan, or Palermo. In some cities, both policies, congestion charges and LEZs, are implemented. There of course have been studies on the efficiency of congestion fees. Börjesson and Kristoffersson (2015) demonstrate the efficiency of the Gothenburg charge in reducing traffic volumes. The authors emphasize that important differences can arise between cities, using the example of Stockholm, due to the development of public transport and the local characteristics (Börjesson & Kristoffersson, 2015), which has also been observed for LEZs.

A theoretical model comparing the effectiveness of LEZs and urban tolls was developed by Bernardo et al. (2021). The conclusion was that a LEZ is more efficient when pollution is more important than congestion, and inversely when congestion is more severe than pollution, urban tolls are more effective. In the second scenario, LEZ remains effective in curbing pollution but not in mitigating congestion (Bernardo et al., 2021). Furthermore, Anas and Lindsey (2011) highlight that road pricing allows for variable pricing according to the extent of the congestion, the time of the day, the place, or the type of vehicle, a flexibility that LEZs do not possess.

Beyond these considerations, a prevalence of quantity over price tools has been observed (Fageda et al., 2022). Indeed, in Europe, it is more than 200 LEZs that have been implemented versus only a few congestion charge systems. This could be explained by public acceptability. First of all, pollution is generally perceived as an externality more severe than congestion. In addition, urban tolls affect all road users while LEZs only affect the ones that do not meet the criteria (Bernardo et al., 2021). Moreover, it seems that LEZs are most applied in high-income cities (Bernardo et al., 2021), and more accepted by high-income users (Fageda et al., 2022), most likely seeing its potential inequality aspect. Additionally, the public acceptability of LEZs are influenced by factors such as individuals' political ideology, environmental awareness, and main transport mode (Fageda et al., 2022).

For what concerns Belgium, there are currently no such schemes implemented (Sadler ldt, n.d.). However, a project is currently under development to implement such a policy in Brussels. The "Smart Move" project aims, among other things, to introduce a vehicle tax based on the use and not on the ownership. To be more specific, each car, motorcycle, or van traveling in Brussels would have to pay a tax based on the time of the day, with a higher fee during rush hours, the number of kilometers, and the vehicle's power (De Ceuster et al., 2020). An analysis of the potential impact of this policy has been conducted, concluded that a reduction in traffic from 6.4% up to 9% could be observed. The impact on air pollution has also been estimated, resulting in a decrease of 4 to 6% per year for PM_{10} and NO_x, up to 11% for NO_x in rush hours. This however takes into account the impact of the LEZ as well (De Ceuster et al., 2020).

3.6 Conclusion and contributions

As a conclusion, although the literature on the impact of LEZs on air pollution is already quite complete, variety in the results due to different methodologies, a lack of common design across the many LEZs in Europe, and possible different reactions of road users to the policy, makes it question the external validity of the studies and the generalization of the results to other LEZs. Regarding mobility, although the LEZ appears to encourage the modernization of the vehicle fleet, the question remains as to which types of vehicles are being adopted. Additionally, there are other aspects related to mobility that would deserve consideration, such as the impact on alternative modes of transportation. Finally, the question of the fairness of the policy has been raised in several papers.

In light of the difficult generalization of the results, the existence of alternative policies such as the congestion charge, the scarcity of ex-post studies on Belgian LEZs, and the prospect of new zones in Wallonia, it seems pertinent to investigate the applicability and effectiveness of LEZs in the specific context of Brussels.

Since the health effects of LEZs are more long-term effects, this thesis primarily focuses on the impact of the Brussels LEZ on mobility, through the evolution of the vehicle fleet by fuel types, and air pollution. The analysis will encompass the effects within the zone itself and the spillover effects on the periphery of the city. To provide more insight into the inequality aspect of the policy, the influence of income and inequality on the results will be highlighted, along with the impact of other relevant factors.

4 Theoretical framework

This section aims to define to what extent air pollution can be viewed as an economic problem and what economic theory states regarding adequate policy instruments.

Pollution arises from market inefficiency, which occurs when there is a divergence between private and social costs or benefits. This difference in this case is caused by externalities. By definition, *"an externality is present whenever some economic agent's welfare is directly affected by the action of another agent"* (Hindriks & Myles, 2013, chap. 8, p.224). The "directly" emphasizes that any effects mediated by prices are excluded. More specifically, the concept declines into production externality, affecting profit, and consumption externality, affecting utility (Hindriks & Myles, 2013).

In the presence of externalities, the market outcome equalizes the private marginal benefit (PMB) to the marginal cost (MC) of the good. In contrast, the Pareto efficient outcome equalizes the social marginal benefit (SMB), the sum of the PMB and the marginal external effect, to the marginal cost. In the presence of externalities, the PMB thus diverges from the SMB, leading to a market outcome that is not Pareto-optimal and to the wrong quantities of consumption or production. As a reminder, an equilibrium is Pareto-optimal when the well-being of one individual cannot be increased without reducing the one of another individual (Hindriks & Myles, 2013).

In the context of this research, the focus is on externalities resulting from transportation, as low emission zones specifically target this aspect. One can distinguish mainly two types of negative consumption externalities: congestion and environmental externalities. When choosing their transport mode, road users do not take into account the negative effect that their car will have on other users, by increasing congestion thus resulting in traffic jams on one hand, and by increasing air pollution on the other hand (Proost & Van Dender, 2012). As a consequence, the number of cars on the road resulting from the market outcome, taking into account only private valuations, is too high compared to what is Pareto optimal. It is important to note a further distinction between environmental and congestion externalities. Environmental damages caused by road transportation can be reduced by filter technology, meaning switching to less polluting vehicles, therefore without changing car use. However, this does not apply to congestion (Proost & Van Dender, 2012).

Another economic concept arising from a divergence of individual and social incentives is the tragedy of the commons. This concept, developed the first time by Garret Hardin in 1968, refers to a situation where individuals having access to a public resource end up depleting it by acting in their own interest (Spiliakos, 2019). The type of good that may be subject to this tragedy of the commons can be described as a common good. A common good is both non-excludable, meaning that no one can be excluded from consuming it, and rivalrous, that its consumption by one individual reduces the supply available for others (Hindriks & Myles, 2013). Natural resources are often described as such (Penn State, n.d.)

Public space can be seen as being impacted by the tragedy of the commons. Everyone has access to it, but individuals by only acting in their own interest can end up over-consuming it, which will result in its degradation and congestion. Since low emission zones target vehicles, it can be assumed that it will affect the transport habits of people, therefore affecting public space, especially roads inside and in the surroundings of the area.

As externalities are a market failure, they must be internalized.

4.1 Internalizing the externality

The Coase theorem states that *"in a competitive economy with complete information and zero transaction costs, the allocation of resources will be efficient and invariant with respect to legal rules of entitlement"* (Hindriks & Myles, 2013, chap. 8, p.214). In other words, it implies that, when property rights are well defined, policy intervention is not needed to resolve the externality problem because the market will achieve an efficient allocation of resources. It assumes that economic agents can reach private agreements when property rights are known, compensating for the externality and thus leading to the right price to emerge (Hindriks & Myles, 2013).

The assumptions upon which the theorem relies are however not always respected. Especially, property rights can be unclear. For instance, in the case of clean air, it is difficult to establish who owns the clean air. Other solutions must therefore be sought.

Two types of outcomes can be reached, the first-best and second-best. On the one hand, the first-best can be achieved when the only restrictions relate to the production technology and the limitation on endowments. It corresponds to what would be chosen by an omniscient planner. On the other hand, when there are other constraints, the second-best arises (Hindriks & Myles, 2013).

According to De Borger and Proost (2013), in the first best scenario, the optimal approach to reduce urban traffic emissions entails both making cars cleaner by reducing emissions per kilometer and decreasing the overall number of kilometers driven. This objective could be accomplished through the implementation of a Pigouvian tax.

As previously discussed, in the case of a negative consumption externality such as the one arising with air pollution, the market outcome results in a higher level of consumption than what is socially and Pareto optimal. The Pigouvian tax enables to achieve this optimum and, as a result, efficiency. The idea is that the consumer or firm responsible for the externality should pay a tax equal to the marginal damage caused by this externality (Hindriks & Myles, 2013).

Applied to air pollution, this can correspond to an emission tax, that would therefore be equal to the marginal damage of pollution (De Borger & Proost, 2013; Proost & Van Dender, 2012). This would ensure that, at the optimum, the number of kilometers driven is such that the unit cost of driving is equal to the marginal social cost (De Borger & Proost, 2013).

Nonetheless, in reality, multiple agents are generating multiple externalities. Therefore, for this type of tax to actually reach efficiency, a distinct tax must be set for each consumer so that every one of them fully internalizes the externalities they generate (Hindriks & Myles, 2013). Moreover, regarding the emission tax, emissions are not always directly proportional to the type and the quantity of fuel, adding another difficulty. It is actually only the case for carbon emissions and sulfur content (Proost & Van Dender, 2012). Consequently, second-best instruments are going to be considered.

A distinction can be made between price and quantity tools. An example of a price tool is a product tax, such as a fuel tax, which should not be mistaken for the previously described emission tax. The issue with a product tax is that it is a tax based on the kilometers driven, but it does not target specific abatement technologies, thus failing to incentivize a switch to "greener" vehicles (Proost & Van Dender, 2012).

In terms of quantity instruments, emission standards can be highlighted. These involve setting emission reduction efforts for vehicles, like the Euro norms previously mentioned. Unlike a product tax, the pollution that occurs with vehicle use is not taxed. It incentivizes the “greening” of the cars but not the reduction of kilometers driven (De Borger & Proost, 2013; Proost & Van Dender, 2012).

Low emission zones combine these emission standards with entrance fees in case of non-compliance. Therefore, with this system, the mileage is not affected. One particularity of low emission zones is that, unlike the previously mentioned instruments, it only applies to a specific region. The externalities in the surrounding region might be mitigated with spillover effects, but they are not the target of this policy.

As addressed in the literature review, a similar policy also targeting a specific region, is the congestion charge, which falls into the price-based instrument category. Although its primary goal is to mitigate congestion rather than directly reducing pollution, by charging drivers to enter a certain area, this instrument can influence their transportation choices, thus influencing pollution. Contrarily, LEZs’ main goal is to reduce pollution, but by restricting access to some vehicles it can also influence congestion.

Nevertheless, the outcomes achieved with quantity-based and price-based instruments are likely to differ. With a quantity-based instrument, the level of emission reduction is certain, as it is fixed by the government setting the policy. On the contrary, with a price-based instrument, consumers determine the optimal level of emission reduction according to the price they face with the tax. Due to the uncertainty regarding the cost function of consumers, setting a tax will most likely lead to different outcomes compared to the quantity-based approach.

In reality, as previously discussed, a prevalence of quantity-based over price-based instruments to deal with air pollution is observed. From an economic perspective, it is difficult to determine which one is more efficient as the theory has proved. This prevalence could be due to other factors than these theoretical considerations, such as public acceptability. Bernardo et al. (2021) provides some element to choose between LEZ and urban toll, arguing that a LEZ is more efficient when pollution is more important than congestion, and inversely when congestion is more severe than pollution, urban tolls are more effective. Moreover, it is also possible to combine these instruments, as it is the case in London, Milan, or Stockholm, which have both a congestion charge and a low emission zone.

5 Data

Data used in the framework of this research are described in this section, along with their sources and their pre-reform statistics for the two studied groups.

In analyzing air pollution, primary data were sourced from the Belgian Interregional Environment Agency (IRCEL in Dutch or CELINE[1] in French) and include daily observations across municipalities for five types of pollutants: particulate matter with aerodynamic diameters smaller than 10 µm (PM_{10}) and 2.5 µm ($PM_{2.5}$), nitrogen dioxide (NO_2), ozone (O_3), and black carbon (BC). To obtain municipality-level data, the original data were interpolated by CELINE due to the absence of measuring stations in each municipality. The data used in this research covers the years 2013 to 2023. However, the COVID period and its multiple lockdowns are susceptible to exert an important influence on air pollution. As a result, the years 2020 and 2021 have been excluded from the analysis. A robustness check will be carried out by including these years. Altogether, this gives five years of data pre-reform and four years post-implementation of the LEZ.

A series of control variables are necessary for the analysis. First of all, weather covariates are included since weather influences pollution. These covariates include average daily temperature, minimum temperature, maximum temperature, precipitation quantity in mm, average wind speed, maximum wind speed, and average pressure. These were obtained from the Belgian Royal Meteorological Institute (RMI) in hourly increments, subsequently aggregated into daily ones. However, the weather data are not available by municipality. By first retrieving the GPS coordinates of each observation, and then computing the minimum distance, each municipality has been matched with the nearest weather station. Population and employment rate were also integrated as control variables. Although these variables are recorded annually and not daily, since they are used only as control, this does not constitute a problem for the analysis. Regarding the employment rate, only province-level data are available. All of these have been retrieved from Statbel, the Belgian statistical office. Additionally, a variable distinguishing weekdays from weekends is included, as well as dummy variables for seasons.

Regarding the impact of the LEZ on mobility, the primary data is the car fleet composition, available from 2013 to 2023[2]. It details for private vehicles[3], the number of cars by type of fuel and by municipality. Data on other vehicle categories are not available, but since several categories such as HDV or two-wheel vehicles have not yet been targeted by the policy, this does not represent an issue. These have been obtained through a request made to the SPF Economie. To be more specific, 10 categories of fuel are available in the dataset, which can be grouped into larger categories as follows:

- Gasoline vehicles
- Diesel vehicles
- Alternative fuel vehicles (AFV): electric, natural gas, bioethanol, and hydrogen
- Hybrid vehicles: gas and gasoline, gas and diesel, electric and gasoline, electric and diesel

Control variables for this analysis consist of population and employment rate. Unfortunately, no other historical data relating to mobility were available. There are nevertheless several databases containing information on the current situation regarding mobility. More precisely, data on the number of public bike stands (”Villo!”), public car sharing (”Cambio”) and electric vehicle charging stations were retrieved for the Brussels region, through the website Datastore.brussels. Although these cannot be used as control due to

[1] Cellule Interregionale de l'environnement.
[2] Data realized on each 1st of August.
[3] Company cars are not included.

the lack of historical data, they will nevertheless be studied in the discussion section.

Moreover, to further investigate the factors influencing the results, data on income, inequality, and political affiliation were collected. Income and inequality data were obtained through Statbel. These variables were not included as controls in the regression due to their unavailability in recent years and their lack of use in the literature. The inequality measure consists of the interquartile difference by municipality. Political affiliation was computed based on the results of the 2019 federal elections, obtained on IBZ, generating the percentage for each main political ideology by municipalities. These will be used to perform heterogeneity tests in the discussion, to examine how results are distributed according to each of these variables.

Table 3 hereunder presents the pre-reform means and standard deviations of the aforementioned variables for municipalities inside and outside the low emission zone of Brussels, as well as the t-test for the difference in means between the two groups. It appears that pollution levels are higher for municipalities within the LEZ, except for ozone, as highlighted in appendix 3. The distribution of the car fleet is quite similar between the two groups, as indicated by the t-tests that do not reject the null hypothesis of similar means. However, there are differences in the characteristics of the municipalities. The t-tests for weather variables indicate differences in the means for all variables except precipitation and maximum temperature. The means are however still quite close.

Table 3: Pre reform means of variables, inside and outside LEZ

	Inside LEZ		Outside LEZ		T-tests
	Mean	Sd	Mean	Sd	
Pollution					
Emissions of PM10	19.15	(10.14)	18.72	(10.95)	0.00
Emissions of PM2.5	12.81	(8.97)	11.72	(9.83)	0.00
Emissions of NO2	27.61	(12.94)	17.09	(8.89)	0.00
Emissions of O3	37.39	(18.55)	45.02	(19.42)	0.00
Emissions of black carbon	1.26	(0.72)	0.92	(0.64)	0.00
Vehicle fleet					
Share of alternative fuel vehicles	0.00	(0.00)	0.00	(0.00)	0.48
Share of hybrid vehicles	0.02	(0.00)	0.01	(0.01)	0.16
Share of gasoline vehicles	0.43	(0.11)	0.42	(0.05)	0.39
Share of diesel vehicles	0.55	(0.11)	0.56	(0.05)	0.71
Municipality characteristics					
Population*	62,716	(41546)	40,802	(30315)	0.00
Employment rate*	0.61	(0.00)	0.70	(0.06)	0.00
Income per capita	15,140	(3218)	18,763	(2463)	0.00
Interquartile difference	21,545	(5008)	24,965	(4431)	0.00
% vote for left	0.33	(0.14)	0.27	(0.17)	0.00
% vote for right	0.27	(0.08)	0.47	(0.12)	0.00
Weather					
Precipitation*	0.74	(1.37)	0.77	(1.59)	0.14
Temperature*	11.26	(6.47)	10.98	(6.39)	0.00
Minimum temperature*	7.98	(5.81)	7.43	(5.82)	0.00
Maximum temperature*	15.22	(7.65)	15.04	(7.63)	0.07
Wind speed*	3.23	(1.31)	3.53	(1.76)	0.00
Maximum wind speed*	7.13	(2.87)	6.78	(2.88)	0.00
Pressure*	1016.80	(9.08)	1017.27	(9.16)	0.00

Sources: CELINE, Statbel, RMI, Datastore.brussels, SPF mobilité, own computations. Notes: Means computed on 2017 data except for votes that are from 2019 federal elections; outside LEZ = control group only ; * = variables used as control in the regressions; pollution data in ug/m^3; mobility data in percentage; daily data for pollution and weather, annual data for vehicle fleet and municipality characteristics.

6 Methodology

This section outlines the methodology employed to analyze the impact of Brussels LEZ on air pollution and mobility. Furthermore, the assumptions underlying the method, specifically the assumptions of Ordinary Least Squares (OLS) and the common trends assumption, are scrutinized to develop the most accurate model possible.

The main challenge in assessing the impact of a policy such as the low emission zone on an outcome variable, such as air pollution or the vehicle fleet composition, is to isolate changes in the outcome that are due to the implementation of the LEZ. The differences-in-differences (DiD) method allows for the identification of the effect by comparing a treatment group and a control group before and after the implementation of the policy, assuming that the parallel trends assumption holds, which will be discussed in the subsequent section (Bertrand et al., 2004). More specifically, it identifies the average treatment effect on the treated (ATT), thus focusing on the impact on air pollution for the treated units. The DiD estimator represents the difference between a control and a treatment group (the first difference) before and after the treatment (the second difference) (Goodman-Bacon, 2021).

Since there are two outcomes of interest, two DiD will be realized: one estimating the impact on air pollution, and the other the impact on the vehicle fleet composition. To do so, the DiD estimator can be computed with the following OLS regression:

$$\text{Y}_{it} = \alpha + \beta\text{LEZ}_i + \lambda\text{PostLEZ}_t + \tau\text{LEZ}_i \times \text{PostLEZ}_t + \text{A}_t + \text{S}_i + \text{X}_{it} + \text{u}_{it} \tag{1}$$

Where :

- Y_{it} :
  - For air pollution: logarithm of the daily concentration of the pollutant studied for each municipality i at each time t.
  - For vehicle fleet: share of vehicle by fuel type for each municipality i at time t (annual).
- LEZ_i = dummy variable indicating whether a municipality i is inside or outside an LEZ.
- PostLEZ_t= dummy variable indicating whether the time period is before or after the implementation of the LEZ.
- $\text{LEZ}_i \times \text{PostLEZ}_t$ = the actual treatment, equal to 1 when a municipality is inside an LEZ after its implementation. τ is the coefficient of interest.
- A_t = annual fixed effects
- S_i = municipality fixed effects
- X_{it} = control variables:
  - For air pollution: logarithm of population, employment rate, weather covariates (average daily temperature, minimum temperature, maximum temperature, precipitation quantity in mm, average wind speed, max wind speed, average pressure), a dummy variable indicating weekend versus weekdays and seasonal dummies.
  - For vehicle fleet: logarithm of population, and employment rate.

Annual fixed effects (A_t) enable to control for any time trend that would not be captured by the model, whereas municipality fixed effects (S_i) take into account differences in the level of the outcome variable between municipalities that are time-invariant.

6.1 OLS assumptions

Since the DiD estimator is estimated through an OLS regression, assumptions specific to this method must be considered. These assumptions are the following:

1. The model is linear in its parameters: $Y = \beta_1 + \beta_2 X + u$
2. The error term has an expectation of zero
3. The error term is homoscedastic
4. The error term has a normal distribution
5. The values of the error term have independent distributions
6. The error term is distributed independently of the independent variables
7. There are no exact linear relationships among the independent variables (no multicollinearity)

These are strong assumptions that may not always be respected, thus sometimes requiring adjustments to the model.

Firstly, the independence assumption is addressed by Bertrand et al. (2004) who focus on the issue of serially correlated outcomes in differences-in-differences estimations. They illustrate that not accounting for autocorrelation leads to a downward bias in standard deviation, thus leading to wrong t-statistics and significance levels. They suggest clustering standard errors to solve this issue. Cameron and Miller (2015) discuss in their paper over what standard errors should be clustered. They conclude that there is no general solution. Indeed, as the size of the clusters increases, resulting in fewer clusters, bias diminishes while variability increases. This method of clustering the standard errors has since then been used in several studies. In studies on LEZ using the DiD method, Wolff (2014), Margaryan (2021), and Gehrsitz (2017) cluster the standard errors at the city level. However, other cluster sizes are possible, as an example, Zhai and Wolff (2021) use station-level clusters and Sarmiento et al. (2023) cluster at the municipality level. In the framework of this research, standard errors are clustered at the municipality level, allowing serial correlation within municipalities. Robustness checks will be performed by clustering at the larger scale of agglomerations, using the first two digits of the INS[4] code as a way of grouping municipalities. However, this approach results in only one cluster for the treated group, which is why it was not chosen as the primary method but used for robustness checks.

Secondly, plotting the residuals of the regression against their fitted values enables the verification of additional assumptions, such as whether the error term has a zero expectation, by observing whether the residuals are dispersed around zero. This plot also provides information about the homoskedasticity of the error term and the linearity assumption. Formal tests for heteroskedasticity exist, such as the Breusch-Pagan or White test. However, these tests are not compatible with the ”reghdfe” command used in Stata for performing the regression.

The analysis of these graphs, both with and without the logarithmic transformation, indicated that the OLS assumptions for the air pollution analysis are more consistently met when the logarithm is applied.

[4]Developed by Statbel, the INS code is a numerical code assigned to each municipality to facilitate statistics.

The graphs, available in appendix 4, show residuals distributed around zero with the fitted line at zero, confirming the assumption of zero expectation. The graph for O_3 appears less homogeneous compared to the others, but for consistency in interpretation, the logarithmic transformation will still be employed. Moreover, several studies on air pollution use the logarithm of the pollutant as their dependent variable (Wolff, 2014; Zhai & Wolff, 2021). Furthermore, with the clustered standard errors, heteroskedasticity can be allowed within a cluster, while maintaining the homoskedasticity assumption between clusters [5]. As for the DiD on mobility, non-logarithmic values will be used.

6.2 Differences-in-Differences assumptions and specifications

The differences-in-differences method relies on two specific assumptions. The first and often implicit one is that there are no contemporary shocks that differently affect the two groups, other than the one studied. As mentioned in section 3.3, Brussels implemented its Good Move plan in August 2022, which might influence the outcome studied for the end of 2022 and 2023. There is unfortunately no way of differentiating between the two policies, this will have to be kept in mind when looking at the results. The second assumption, the main one, is the parallel or common trends assumption. The identification of the treatment effect, the treatment being the inclusion in a LEZ, relies on this assumption. It posits that, in the absence of treatment, the treated and the control groups would have evolved similarly. Importantly, only the evolution, and not the level, is relevant. Testing this assumption would require being able to observe the evolution of the outcome in both the treated and the control group, in the absence of the treatment, which is not feasible. While not sufficient, evidence can nevertheless be provided by looking at pre-treatment data.

For the parallel trends assumption to hold, control units need to be carefully chosen. In the context of air pollution analysis, the choice was made to focus on the population to determine municipalities belonging to the control group, as done by Margaryan (2021). Indeed, municipalities with a larger population are expected to exhibit more pollution. After several analyses, it has been decided that all municipalities with population in the highest quartile of the distribution during the year preceding the implementation of Brussels' LEZ would be selected as control units. This criterion is similarly applied to the analysis of mobility. A robustness check will be realized by lowering this threshold.

Furthermore, given that pollution does not stop at Brussels' borders, municipalities on the outskirts of the city are likely to be influenced by the LEZ. Specifically, if drivers opt for a longer route to circumvent the restricted area, these municipalities may experience negative spillover effects. Conversely, the benefits of the policy could extend beyond the concerned area, resulting in positive spillover effects for nearby municipalities (Margaryan, 2021). The same applies to the LEZs of Antwerp and Gent. As a result, these outskirts municipalities have been removed from the control group and will be further analyzed when looking at spillover effects. Including them would violate the implicit Stable Unit Treatment Value Assumption (SUTVA), which states that there cannot be interference between individuals and their respective treatment statuses. To be more specific, the 17 outskirts municipalities within a ten-kilometer radius from the center of Brussels' region have been removed from the control group. This selection will be subject to variation by studying a larger group. Appendix 5 represents these municipalities.

In the end, the control group is constituted of 104 municipalities, and the treatment group of the 19 municipalities of Brussels. A map representing the control units is available in appendix 6.

[5] In addition, Stata does not allow the cluster and the robust option to be used simultaneously.

6.2.1 Testing the parallel trends assumption

This assumption being paramount for the analysis, a thorough examination is conducted to determine under which conditions it appears to be best respected. At first, the simplest way to assess this assumption is by visualizing the raw evolution of the outcomes in both the treatment and control groups. However, to gain a clearer view of potential differences in pre-trends, event study graphs are most commonly used to assess the assumption. The β coefficients and their 95% confidence intervals are retrieved from the following regression, which consists of computing a coefficient for each year, using the year preceding the policy implementation as a reference:

$$log(y_i t) = \alpha + \sum_{t=2013, t\neq 2017}^{2023} \beta_t \mathrm{LEZ}_i \times \mathrm{year}_t + \mathrm{A}_t + \mathrm{S}_i + \mathrm{X}_{it} + \mathrm{u}_{it} \tag{2}$$

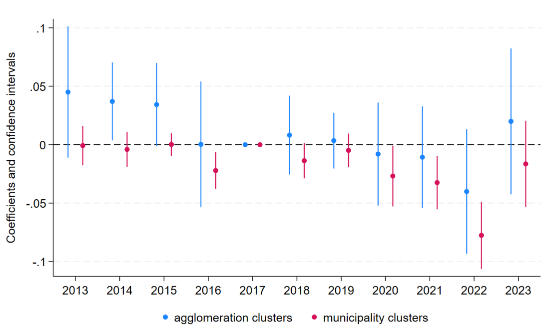
This method allows controlling for variables as specified in equation (1), thus testing conditional parallel trends, and enables the introduction of specifications such as fixed effects or clustered standard errors in the regression. As previously stated, standard errors will be clustered at the municipality level. However, to illustrate the fact that the cluster level influences inference, the coefficients of equation (2) have been computed using the municipality level (in pink) and the agglomeration level (in blue), the latter being applied in the robustness checks. If the parallel trends hypothesis is verified, the pre-reform coefficients should not be statistically different from zero.

Preliminary analysis: air pollution

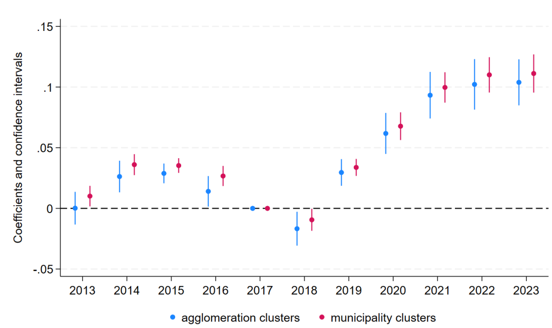
Raw parallel trends for air pollution are available in appendix 7 Although a difference in the level of pollution between the control and treatment groups is consistently observed, as it was already indicated in table 3, this does not constitute an issue for the validation of the hypothesis, as it is the evolution that matters. It appears that the trends in the levels[6] of PM_{10} and $PM_{2.5}$ are not consistently parallel. For instance, a slight divergence in the trends is observed in 2017 for $PM_{2.5}$, compared to 2015 and 2016 where the trends appear parallel. The underlying causes of this divergence are undetermined and could potentially be attributed to a change in behavior in anticipation of the policy. Figure 4 hereunder, following equation (2), provides a more precise view of the hypothesis.

Figure 4: Conditional parallel trends for air pollution

(a) Conditional parallel trends NO_2

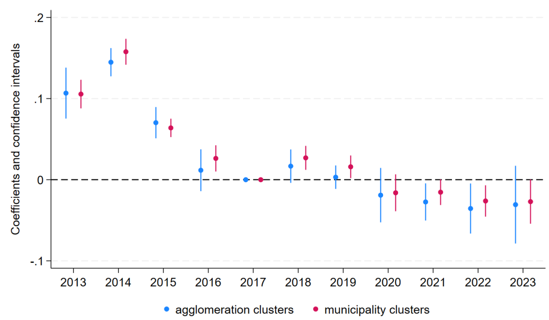


(b) Conditional parallel trends O_3

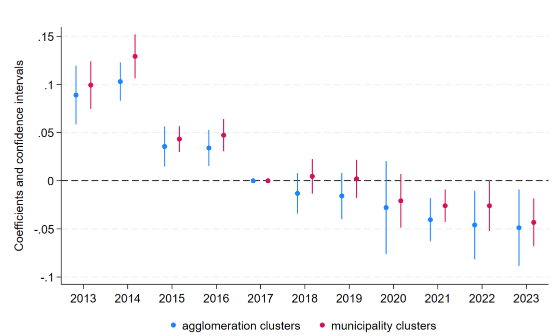


[6] In light of the observations made in the previous section, logarithmic values will be used for the rest of the analysis.

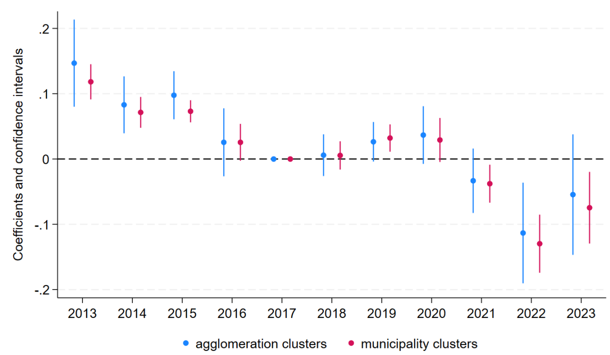
(c) Conditional parallel trends PM_{10}



(d) Conditional parallel trends $PM_{2.5}$



(e) Conditional parallel trends BC



Source: CELINE, Statbel, RMI, own computations and representations

The assumption appears to be approximately respected for NO_2 and O_3, despite some slight deviations. The impact of these deviations on the robustness of the results will be further discussed in 7.2.2. It also appears that the impact post-reform increases over time, which will be studied by looking at the different phases of the LEZ. However, the graphs for PM_{10}, $PM_{2.5}$, and BC do not indicate the validity of the hypothesis. Nonetheless, examining the coefficients for the last two years prior to the policy implementation shows that they are either equal or nearly equal to zero. As the analysis will be realized at a more detailed aggregation level, the parallel trends focusing on 2016 and 2017 can be assessed at a smaller level, such as monthly. Appendix 8 displays this, using the last month of 2017 as the baseline. Unfortunately, the hypothesis does not seem validated.

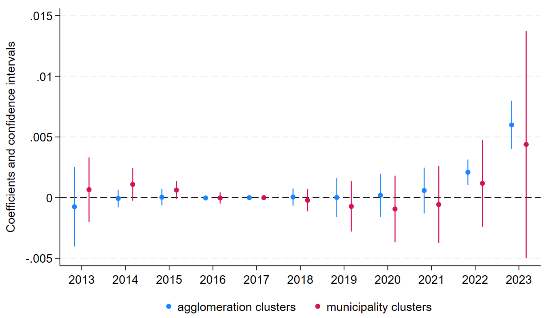
The analysis will therefore focus on two outcomes to measure pollution: the logarithms of NO_2 and O_3, from 2013 to 2023, COVID years excluded.

Main analysis: mobility

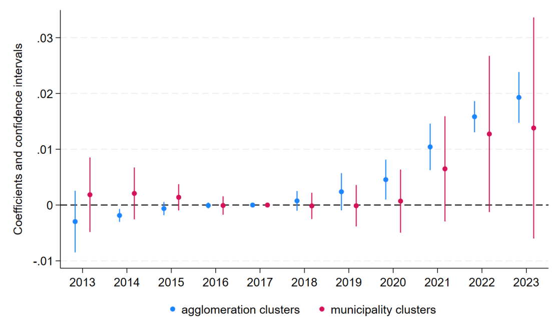
The same approach can be applied to the analysis on mobility. The graphical representations of the raw trends, available in appendix 9, seem to support the parallel trends assumption for the share of AFV and hybrid vehicles. As represented in this appendix and by table 3, the level between the two groups in the pre reform period is very similar. Some convergence is however observed in the trends of the share of gasoline vehicles, potentially due to an anticipation of the policy. Once again, event study graphs can be generated following equation (2) to assess whether the assumption is more accurately maintained with conditional parallel trends. Figure 6 displays evidence that leads to assuming the validity of the parallel trends assumption based on pre-trends for the share of AFV and hybrid vehicles. Although there are minor deviations observed for the share of gasoline and diesel vehicles, the assumption appears to be mainly upheld when standard errors are clustered at the municipality level.

Figure 5: Conditional parallel trends for mobility

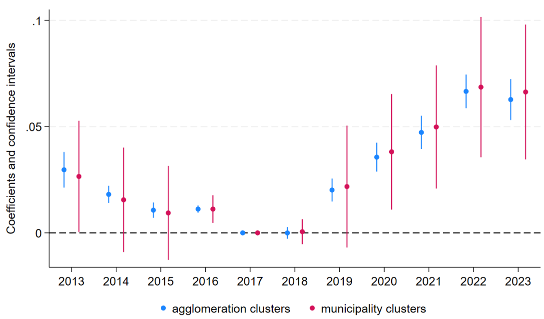
(a) Conditional parallel trends AFV



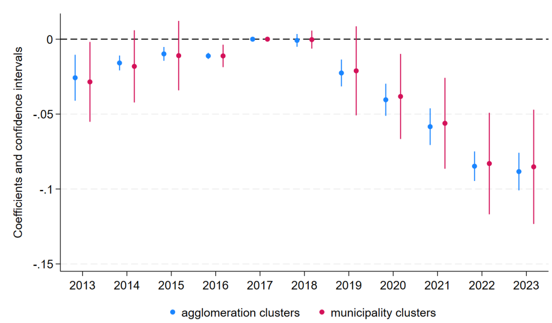
(b) Conditional parallel trends hybrid vehicles



(c) Conditional parallel trends gasoline vehicles



(d) Conditional parallel trends diesel vehicles



Source: Statbel, RMI, own computations and representations

6.2.2 Sensitivity analysis

Recent developments in the literature have examined various aspects of the differences-in-differences method, relaxing some components of the classic DiD setup. This classic form of DiD is a 2x2 setup, two time periods and two groups, following the parallel trends assumption.

The first way this classic form can be relaxed is when there are more than two time periods, with units being treated at different times. It seems that this staggered policy implementation could lead to biased estimates (Goodman-Bacon, 2021), due to already treated units potentially acting as controls. One of the solutions to this issue has been proposed by Callaway and Sant'Anna (2021), who developed an estimator specifically designed for such situations. In the LEZ literature, Sarmiento et al. (2023), focusing on the case of Germany, confirmed the existence of this bias in the case of LEZ based on the method of Goodman-Bacon (2021) and implemented the DiD suggested by Callaway and Sant'Anna (2021). Margaryan (2021) took into account the observation made by Goodman-Bacon (2021) and excluded previously treated cities from the estimation, but did not use the CS estimator. The studies of Wolff (2014) and Gehrsitz (2017) being written before the aforementioned papers, do not take such considerations into account. The results of Sarmiento et al. (2023) were in line with other studies on the case of Germany realized with DiD (Margaryan, 2021; Wolff, 2014) regarding PM_{10} and NO_2, but found an increase in ground-ozone level, which was not studied in the other papers.

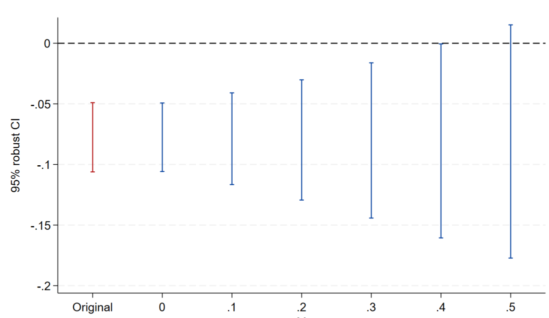
In the context of this thesis, the case studied adopts a 2x2 setup, focusing on the single low emission zone of Brussels. This contrasts with the studies referenced in the preceding paragraph, which examine multiple LEZs implemented across Germany. Consequently, the classic DiD appears to be appropriate for this analysis.

A second strand of the literature focuses on the parallel trends assumption. As realized in this research, the parallel trends are often tested by looking at pre-treatment differences in trends, through event study plots. This technique is however associated with several issues. First of all, the validity of the assumption for pre-treatment data does not imply that parallel trends necessarily hold (Roth et al., 2023). Moreover, the low power of the test might be unable to detect differences in pre-trends. Roth (2022) indicates that these conventional tests would detect linear violation of the parallel trends only 50% of the time. To address these potential violations of this main assumption, Rambachan and Roth (2023) develop the honest DiD package, allowing to conduct a sensitivity analysis for the post-reform coefficients. The idea is to assess the significance of the post-reform coefficients when faced with a violation of the parallel trends, which could be caused for instance by a contemporary shock affecting the two groups. One approach to achieve this involves bounding the post-treatment trend violation to be no more than a constant M larger than the maximum violation of parallel trends observed in the pre-treatment period.

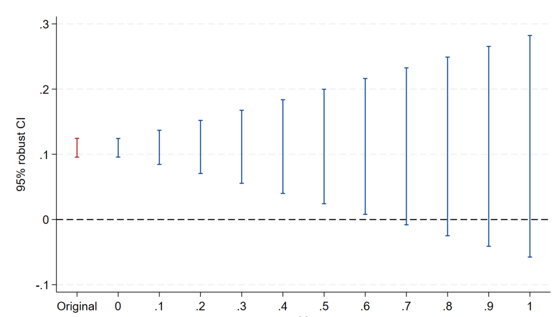
Applying this method to all coefficients would be quite tedious. Therefore, it was chosen to test the robustness of the coefficient's significance for the year in which the greatest impact is observed, corresponding most of the time to 2022. Figures 6 and 7 indicate that the 2022 coefficients for NO_2 and O_3 remain robust respectively up to 0.4 and 0.6 times the maximum violation observed in pre-reform, and up to 0.3 and 0.8 for the share of gasoline and diesel vehicles. Although these margins are not substantial, they indicate a degree of tolerance for deviations from the parallel trends assumption[7].

Figure 6: Sensitivity analysis of conditional parallel trends for selected pollutants

(a) Sensitivity analysis for 2022 coefficient NO_2



(b) Sensitivity analysis for 2022 coefficient O_3



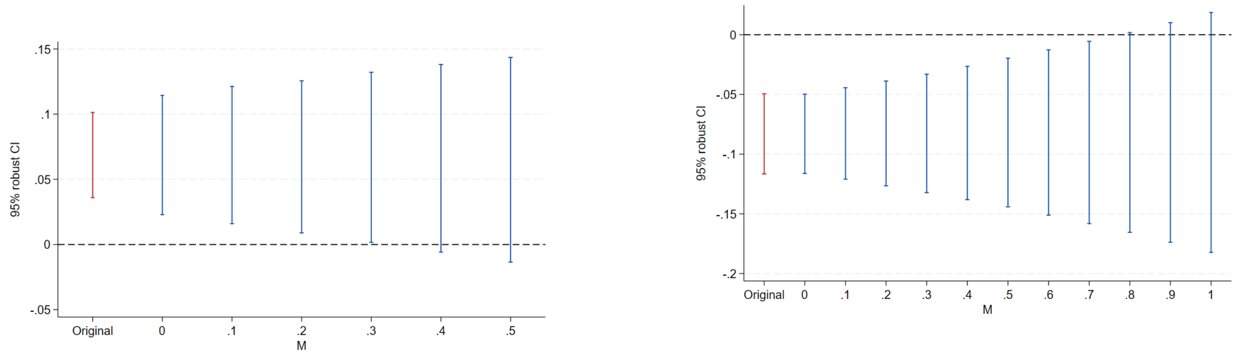
Source: CELINE, Statbel, RMI, own computations and representations

[7]Sensitivity analysis was not performed for the share of AFV and hybrid vehicles as there is no deviation from the parallel trends in figure 5 (a) and (b).

Figure 7: Sensitivity analysis of conditional parallel trends for the share of gasoline and diesel vehicles

(a) Sensitivity analysis for 2022 coefficient gasoline

(b) Sensitivity analysis for 2022 coefficient diesel



Source: CELINE, Statbel, RMI, own computations and representations

7 Results

Results realised with the previously described method are now presented, allowing to see how the Brussels LEZ influences air pollution and mobility, for areas inside and in the surrounding of the zone. Table 4 and Table 6 present the results from the regression in equation (1) and with the specifications detailed in the previous section, respectively for air pollution in a first time, and then for mobility. Various iterations of equation (1) are estimated, ranging from the most basic model without controls, fixed effects, or clustered standard errors, to the most comprehensive specification. For the sake of clarity, the coefficients of the control variables do not appear in the table. A complete version with these coefficients is available in appendix 10 and appendix 11.

The overall effect of the LEZ is represented in Panel A, representing the coefficient τ of equation (1). However, the implementation of the policy declines into several phases, each with different levels of stringency regarding banned vehicles. The different phases are detailed in table 2. As a reminder, four distinct phases were introduced to this date. At the beginning of the LEZ in 2018, only diesel vehicles with Euro norms 0 and 1 were banned. In 2019, this extended to Euro 2 diesel and Euro 0 and 1 gasoline vehicles. Then Euro 3 and Euro 4 diesel vehicles were banned respectively in 2020 and 2022. At this date, the policy does not target heavy-duty vehicles, and only weak limits are set regarding gasoline vehicles. Further strengthening is planned in 2025, which should include heavy-duty vehicles and strengthen the rules for gasoline vehicles.

To gain further insight into the effects of the LEZ and the severity of its rules, the impact of these phases is detailed as well, modifying equation (1) as follows:

$$\mathrm{Y}_{it} = \alpha + \beta \mathrm{LEZ}_i + \sum_{k=1}^{4} \lambda_k \mathrm{phase}_k + \sum_{k=1}^{4} \tau_k \mathrm{LEZ}_i \times \mathrm{phase}_k + \mathrm{A}_t + \mathrm{S}_i + \mathrm{X}_{it} + \mathrm{u}_{it} \qquad (3)$$

The results of this equation are represented in Panel B of table 4. As the third phase occurred in 2020 and 2021, years removed from the analysis due to the COVID-19 pandemic, the impact of this phase is not examined. Since only annual data are available for the mobility analysis, the low number of observations does not allow a phase-specific study.

Additionally, as highlighted in the literature review, the success of policies such as LEZ largely depends on the reactions of road users. Drivers may upgrade their vehicle in case where it is non-compliant, or take longer routes to circumvent the zone (Lurkin et al., 2021). To determine whether the LEZ yield spillover effects and, if so, whether these effects are positive or negative, equation (1) can be adapted by analyzing a set of municipalities just outside the zone, defined in the following equation by ”outskirt”. Two groups are studied, a smaller and a larger one, as defined in appendix 5.

$$\mathrm{Y}_{it} = \alpha + \beta \mathrm{Outskirt}_i + \lambda \mathrm{PostLEZ}_t + \tau \mathrm{Outskirt}_i \times \mathrm{PostLEZ}_t + \mathrm{A}_t + \mathrm{S}_i + \mathrm{X}_{it} + \mathrm{u}_{it} \qquad (4)$$

The results of this equation are available in table 5 and table 7.

7.1 Preliminary analysis: impact on air pollution

The findings indicate a statistically significant overall reduction in NO_2, but an overall increase in the ozone levels. Model (3), corresponding to equation (1) in the methodology section, is the most accurate. It incorporates control variables, yearly fixed effects, municipality fixed effects, and clustered standard errors at the municipality level. According to this model, the implementation of Brussels' low emission zone resulted in a decrease of 2.1% in NO_2 levels and an increase of 5.5% in O_3 levels for municipalities within the LEZ. However, the effect on NO_2 is only significant at a 10% level. These effects differ from what is observed with model (1) and (2), highlighting the importance of using detailed specifications.

Concerning the effect of the different phases, it is important to note that the coefficients represent the impact of each phase relative to the pre-implementation period of the policy, rather than indicating an incremental effect. The last phase of the implementation has the greatest influence on NO_2 levels, while the second phase has the least impact on this pollutant. Additionally, the initial phase brought about a reduction in ozone levels by approximately 3%, which was then counterbalanced by increases in subsequent phases, particularly in phase 4, ultimately resulting in an overall elevation of ozone concentrations.

Table 4: Impact of LEZ on air pollution

	log(NO2)			log(O3)		
	(1)	(2)	(3)	(1)	(2)	(3)
Panel A: overall effect						
LEZ overall (τ)	-0.026***	-0.018***	-0.021*	0.056***	0.054***	0.055***
	(0.005)	(0.003)	(0.012)	(0.005)	(0.003)	(0.006)
Panel B: effect by phases						
phase 1 (τ_1)	-0.024***	-0.028***	-0.018***	-0.035***	-0.029***	-0.029***
	(0.007)	(0.005)	(0.005)	(0.008)	(0.005)	(0.003)
phase 2 (τ_2)	-0.011	-0.008*	0.002	0.021**	0.029***	0.027***
	(0.007)	(0.005)	(0.008)	(0.008)	(0.005)	(0.004)
phase 4 (τ_4)	-0.038***	-0.030***	-0.038*	0.121***	0.112***	0.124***
	(0.006)	(0.004)	(0.020)	(0.006)	(0.004)	(0.010)
Observations	391,018	391,018	391,018	391,018	391,018	391,018
control	No	Yes	Yes	No	Yes	Yes
fixed effects	No	No	Yes	No	No	Yes
cluster	No	No	Yes	No	No	Yes

Sources: CELINE, Statbel, RMI, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly and municipality fixed effects; standard errors clustered at the municipality level. Phase 1 = 2018, Phase 2 = 2019, Phase 3 = 2020 and 2021, Phase 4 = 2022 and 2023. See Table 2 for more details.

These findings can be compared to the literature. Unfortunately, the 2022 report on Brussels LEZ (Bruxelles environnement, 2023) does not study NO_2 and O_3, making a direct comparison with this research impossible. Their findings for other pollutants are however much larger than the magnitude observed in this study, with a reduction of 31% for NO_x, 19% for PM_{10}, and 30% for $PM_{2.5}$ between 2018 and 2022 (Bruxelles environnement, 2023). As previously discussed, their methodology appears to be a comparison of the before and after levels, while the differences-in-differences approach allows for isolating the pollution reduction truly attributable to the policy.

Studies investigating the impact on NO_2 conclude in no significant effect (Margaryan, 2021) or small effect around 3% (Ma et al., 2021), aligning with the results of Table 4. Moreover, Gehrsitz (2017) indicates a greater reduction in air pollution as the rules tightened, as it is observed in this case with a reduction of almost 4% in NO_2 levels during the 4th phase. This last phase banned Euro 4 diesel vehicles, which according to Bruxelles environnement (2023) emit six times more fine particles than authorised diesel vehicles, which could explain this greater decrease in NO_2.

Lastly, the increase in O_3 levels has been observed in one other case studied by Sarmiento et al. (2023). The authors provide an explanation for this seemingly surprising result. As developed in the literature review, ozone is the result of the interaction of two other pollutants, NO_x and VOC. The relationship between these pollutants can be complex. Urban regions are prone to being NO_x-saturated areas due to the intensity of traffic, which creates a situation where NO_x can decrease ozone. Therefore, policies aimed at reducing NO_x levels can lead to an increase in ozone levels, as observed in Brussels.

The complete table with the coefficients for control variables, available in appendix 10, provides information on the significance of these variables. The negative coefficients for weekends indicate lower pollution levels compared to weekdays, which is intuitive. However, these coefficients are positive for O_3. This can be explained by the previously mentioned phenomenon involving ozone and NO_x: NO_x emissions are likely reduced during weekends due to decreased traffic, leading to an increase in ozone levels. The coefficients for seasons show positive signs for winter and negative ones for summer for NO_2, suggesting lower pollution levels in summer than in autumn, which is the baseline. In contrast, coefficients for ozone are positive across all seasons, with higher values observed in summer and spring compared to winter. This can be attributed to the formation of ozone in sunlight, explaining these seasonal differences.

7.1.1 Spillover effects

Results of table 6 indicate that the LEZ generates significant spillover effects on both pollutants. First, a decrease of 4.8% in NO_2 levels is noted for municipalities in the periphery of Brussels, which is larger than for municipalities located within the city. It seems that the implementation of the policy reduces pollution more in municipalities located just outside the zone than in those within it. Moreover, when widening the studied area, the effect is amplified. These results suggest that drivers do not bypass the LEZ by taking a longer route, which would have resulted in an increase in pollution in the periphery of Brussels. As for ozone, which as previously addressed behaves differently from the other pollutants, municipalities on the outskirts face an increase in ozone levels of 1.6%, which is smaller than for the units within. This effect does not remain when expanding the studied area, indicating that the impact is primarily localized within the zone concerned by the policy.

Table 5: Air pollution impact of LEZ on outskirt municipalities

	log(NO2)	log(O3)
Panel A: group 1		
LEZ on outskirt municipalities (τ)	-0.048***	0.016**
	(0.013)	(0.007)
Panel B: group 2		
LEZ on outskirt municipalities (τ)	-0.064***	0.011
	(0.010)	(0.007)
Observations (1)	385,216	385,216
Observations (2)	414,245	414,245

Sources: CELINE, Statbel, RMI, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effects and municipality fixed effects; standard errors clustered at municipality level. Group 1: represented in appendix 5 by the green area; group 2: represented in appendix 5 by the green and white areas.

7.2 Main analysis: impact on mobility

To study the impact of the policy on mobility, the vehicle fleet composition according to fuel types is examined. Fuel types have been grouped into four main categories: alternative fuel vehicles, which include electric, natural gas, bioethanol, and hydrogen vehicles; hybrid vehicles, which combine electric or gas with gasoline or diesel; gasoline vehicles; and diesel vehicles. Table 6 displays the result of the DiD following various specifications of equation (1). Due to the smaller number of observations available, as only annual data are accessible, the impact per phase is not studied.

Table 6: Impact of LEZ on the vehicle fleet composition

	share AFV			share hybrid			share gasoline			share diesel		
	(1)	(2)	(3)	(1)	(2)	(3)	(1)	(2)	(3)	(1)	(2)	(3)
LEZ overall (τ)	0.001	0.001	0.002	0.011***	0.009***	0.008*	0.026**	0.023**	0.018	-0.038***	-0.035***	-0.029**
	(0.001)	(0.001)	(0.002)	(0.004)	(0.004)	(0.005)	(0.011)	(0.011)	(0.012)	(0.012)	(0.012)	(0.013)
Observations	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107
Control	No	Yes	Yes	No	Yes	Yes	No	Yes	Yes	No	Yes	Yes
Fixed Effects	No	No	Yes	No	No	Yes	No	No	Yes	No	No	Yes
Cluster	No	No	Yes	No	No	Yes	No	No	Yes	No	No	Yes

Sources: SPF Economie, Statbel, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly and municipality fixed effects; standard errors clustered at the municipality level.

Significant coefficients are observed with the more basic models (1) and (2) for all categories except the share of AFV. However, with the more comprehensive model (3), the policy only has significant effects on diesel and hybrid vehicles. The share of diesel vehicles decreased by 2.9% following the implementation of the LEZ, while the share of hybrid vehicles increased by 0.08%. This last number is however only significant at the 10% level. Consequently, although the policy decreases the proportion of diesel vehicles, it provides little or no incentive to switch to less polluting vehicles.

Diesel vehicles are the most heavily targeted ones, as highlighted in Table 2, explaining the result in this category. Gasoline vehicles are also targeted, but the absence of a significant impact suggests that the current level of stringency is insufficient to achieve a noticeable decrease in the proportion of gasoline cars.

7.2.1 Spillover effects

As in the case of air pollution, the effects of the LEZ on the vehicle fleet may extend beyond the limit of the zone. To verify this, spillover effects can be examined, through equation (4). Two sizes of perimeter are considered as well. In the more restrained area (panel A), results are similar to what is observed inside the zone, although with a smaller effect on the share of diesel cars. Effects are amplified when extending the studied area (panel B), resulting in a significant increase of 0.4% in the share of alternative vehicles and a decrease of 1.2% and 1.5% in the share of hybrid and diesel cars.

Although these results remain small, they suggest that the policy provides more incentive for individuals living in the surrounding municipalities than the ones living inside the city to opt for greener vehicles. People living further from the city center are more likely to need a vehicle to commute to the city than the ones living there, which could be a possible explanation for this result. The effect on diesel vehicles is however smaller in the periphery of Brussels, and the effect on the proportion of gasoline cars remains non significant.

Table 7: Mobility impact of LEZ on outskirts municipalities

	share AFV	share hybrid	share gasoline	share diesel
Panel A: group 1				
LEZ on outskirt municipalities (τ)	0.001	0.010***	-0.001	-0.011 *
	(0.002)	(0.004)	(0.004)	(0.006)
Panel B: group 2				
LEZ on outskirt municipalities (τ)	0.004***	0.012***	-0.000	-0.015***
	(0.001)	(0.002)	(0.005)	(0.005)
Observations (1)	1,089	1,089	1,089	1,089
Observations (2)	1,170	1,170	1,170	1,170

Sources: SPF Economie, Statbel, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effects and municipality fixed effects; standard errors clustered at municipality level. Group 1: represented in appendix 5 by the green area; group 2: represented in appendix 5 by the green and white areas.

As a reminder, it is recognized in the literature that low emission zones have a positive impact towards less polluting vehicles. However, research usually analyses this impact based on the Euro norms (Margaryan, 2021; Wolff, 2014) rather than the type of fuel used. Nevertheless, the paper of Peters et al. (2021) takes interest in the evolution of the car fleet by fuel type. It concludes that the transition was predominantly towards alternative fossil fuel-powered vehicles and plug-in hybrid electric vehicles, which are not the most efficient to reduce CO2 emissions. On the other hand, the impact on electric cars seemed to be limited (Peters et al., 2021). The findings in the case of Brussels align with this conclusion. There is a noted, although small, increase in the share of hybrid vehicles, while the effect on electric and gas-powered vehicles remains negligible. Data on the vehicle fleet with respect to their Euro norm is unfortunately not available at the municipality level, preventing a direct confirmation of the conclusion drawn by the previously mentioned studies.

Furthermore, the 2022 study by Bruxelles environnement (2023) on the Brussels LEZ observed a nearly 50% reduction in the proportion of diesel vehicles, accompanied by a 25% increase in the share of gasoline vehicles and only a slight increase in the number of electric vehicles. The results in Table 6 exhibit similar signs, albeit with significantly smaller magnitudes.

There is limited literature focusing on other aspects of mobility, besides Ding et al. (2023) studying the impact on public bike sharing and Moral-Carcedo (2022) focusing on traffic and congestion. As it was discussed in the data section, no historical data were available to study the impact on or the influence of other aspects, such as public bike, car sharing, or public transport. Despite this limitation, this research aims at providing the most detailed information possible that could explain the results just described. Therefore, further analysis will be realized in the discussion to seek potential explanations for what influences the impact of the LEZ.

8 Discussion

Further information is provided in this section on the factors influencing the results and an in-depth examination of Brussels municipalities, concluding with robustness checks and the limitations and extensions of this research.

8.1 Heterogeneity tests

Several heterogeneity tests are conducted to examine the influence of various variables on the policy's impact on pollution and mobility. The objective is to address the following questions: did the wealthiest and most unequal municipalities derive the greatest benefits from the low emission zone? Did political ideology influence the outcomes? To do so, three variables are examined: the per capita income, the interquartile difference, and the political ideology[8]. Dummy variables have been created based on each variable, indicating respectively municipalities with an above-median per capita income, an above-median interquartile difference, and municipalities with a majority (>50%) of the vote for left-wing political parties. These represent the "hetero var" in the following regression:

$$\begin{aligned} \text{Y}_{it} = \alpha + \beta_1 \text{LEZ}_i \times \text{PostLEZ}_t + \beta_2 \text{LEZ}_i \times \text{hetero var}_i + \beta_3 \text{PostLEZ}_t \times \text{hetero var}_i \\ + \tau \text{LEZ}_i \times \text{PostLEZ}_t \times \text{hetero var}_i + \text{A}_t + \text{S}_i + \text{X}_{it} + \text{u}_{it} \end{aligned} \tag{5}$$

Table 8 illustrates that the policy's impact is more pronounced in municipalities with higher-than-median levels of inequality and income per capita. In these municipalities, a reduction in O_3 levels is observed, contrasting with the overall trend of increased O_3 levels. As a reminder, the concept of environmental justice was discussed in the literature review, highlighting that low socio-economic groups are often exposed to higher levels of pollution (Hajat et al., 2015) and that low emission zones tend to benefit the most to the wealthiest (Poulhès & Proulhac, 2021). In the specific case of Brussels, a negative relationship between air pollution and income was indicated by Verbeek and Hincks (2022). The findings of this research allow to confirm this, indicating that for the specific case of Brussels, it seems that the wealthiest individuals indeed benefit the most from the policy. On the other hand, the political ideology does not influence the effect of the policy on air pollution.

Looking now at the heterogeneity tests for mobility in table 9, only the political ideology significantly influences all fuel categories. Municipalities where individuals voted in majority for left-wing parties have experienced an increase in the share of AFV, hybrid, and gasoline vehicles and a decrease in the share of diesel vehicles. Finally, the income per capita and the inequality measure only affect the influence of the policy on gasoline and diesel vehicles. Municipalities with a higher income per capita and inequality level experienced a decrease in the share of gasoline cars and an increase in the share of diesel cars, compared to other municipalities, which is the opposite of the general trend.

[8] Per capita income data are for the year 2017, while interquartile difference and political ideology data are from 2019, for availability reasons.

Table 8: Heterogeneity tests on air pollution

	log(NO2)	log(O3)
income per capita		
LEZ × PostLEZ	-0.029**	0.061***
	(0.011)	(0.007)
LEZ × PostLEZ × income pc	-0.038*	-0.028**
	(0.020)	(0.013)
Inequality - interquartile difference		
LEZ × PostLEZ	-0.013	0.065***
	(0.010)	(0.006)
LEZ × PostLEZ× interquartile diff.	-0.046**	-0.035***
	(0.023)	(0.013)
Political ideology		
LEZ × PostLEZ	-0.048***	0.048***
	(0.017)	(0.008)
LEZ × PostLEZ× left vote	0.038	-0.007
	(0.025)	(0.012)
Observations	391,018	391,018
Clusters with above-median income		
inside LEZ	3	3
outside LEZ	48	48
Clusters with above-median inequality		
inside LEZ	5	5
outside LEZ	34	34
Clusters with >50% left-wing votes		
inside LEZ	12	12
outside LEZ	30	30

Sources: SPF Economie, Statbel, Datastore.Brussels, SPF mobilité, IBZ, own computations. Notes: standard errors in parentheses. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects. Total number of clusters: 19 inside the LEZ, 104 outside.

Table 9: Heterogeneity tests on mobility

	share AFV	share hybrid	share gasoline	share diesel
Income per capita				
LEZ $\times$ PostLEZ	0.004	0.013**	0.028**	-0.045***
	(0.002)	(0.006)	(0.013)	(0.015)
LEZ $\times$ PostLEZ $\times$ income pc	-0.004	-0.011	-0.038 **	0.053**
	(0.003)	(0.007)	(0.017)	(0.018)
Inequality - interquartile difference				
LEZ $\times$ PostLEZ	0.003	0.010	0.031*	-0.045**
	(0.002)	(0.007)	(0.016)	(0.017)
LEZ $\times$ PostLEZ$\times$ interquartile diff.	-0.003	-0.006	-0.047**	0.057***
	(0.003)	(0.008)	(0.021)	(0.021)
Political ideology				
LEZ $\times$ PostLEZ	-0.001	0.002	-0.004	0.003
	(0.001)	(0.003)	(0.007)	(0.006)
LEZ $\times$ PostLEZ$\times$ left vote	0.009***	0.018**	0.042*	-0.069***
	(0.003)	(0.008)	(0.022)	(0.022)
Observations	1,107	1,107	1,107	1,107
Clusters with above-median income				
inside LEZ	3	3	3	3
outside LEZ	48	48	48	48
Clusters with above-median inequality				
inside LEZ	5	5	5	5
outside LEZ	34	34	34	34
Clusters with >50% left-wing votes				
inside LEZ	12	12	12	12
outside LEZ	30	30	30	30

Sources: SPF Economie, Statbel, Datastore.Brussels, SPF mobilité, IBZ, own computations. Notes: standard errors in parentheses. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects. Total number of clusters: 19 inside the LEZ, 104 outside.

8.2 Study by Brussels municipalities

The analyses conducted thus far do not provide information on which municipalities in Brussels are most affected by the policy. This section aims to address this gap by conducting a descriptive analysis for all previously used variables. Additionally, some mobility data are only available for the Brussels region, making them incompatible with the prior econometric methods. However, it is still valuable to explore potential links between these mobility indicators and other variables. Specifically, three mobility indicators are examined, as outlined in the data section: the number of public bike (Villo!) attachment points, the number of Cambio car-sharing stations, and the number of electric vehicle charging stations. These data are only available for the year 2023. Table 10 presents a summary of the variables used in this research for each municipality.

Air pollution decreased the most in Auderghem, Uccle, and Watermael-Boitsforts. However, these municipalities do not experience a large variation in the vehicle fleet composition. As a reminder, appendix 3 showed that before the implementation of the policy, these same municipalities were the ones with the lowest level of pollution, with the exception of O_3 levels. Table 10 indicates that these municipalities are also the ones with the highest per capita income. Even though Auderghem is not highlighted in the top 20%, its per capita income is still above the third quartile. Similarly, they also have some of the highest interquartile differences, indicating higher inequality. The relationship between income and inequality with the LEZ was emphasized in the heterogeneity tests and appears here as well. Finally, these municipalities do not have an infrastructure regarding sustainable mobility much developed.

Regarding the vehicle fleet, the most significant changes occurred in Brussels and Saint-Josse-ten-Noode. Brussels has the most advanced infrastructure for public bike, car sharing, and electric car charging stations, and is the only municipality that experienced a reduction in the proportion of gasoline cars. This can be attributed to its status as the city center and its substantial population. Conversely, Saint-Josse-ten-Noode, while not exhibiting high mobility indicators, recorded the highest rate of left-wing votes in the 2019 election. Heterogeneity tests indicated that while political ideology has limited influence on air pollution, it significantly affects the vehicle fleet.

To gain more insight into the influence of the three mobility indicators that are public bike use, car sharing, and electric car charging stations, correlations are computed. Appendix 12 displays these correlations between mobility indicators and air pollution levels as well as the composition of the vehicle fleet in 2023.

Ozone is negatively correlated with all the mobility indicators. This negative correlation indicates that municipalities with more developed infrastructure regarding public bike, car sharing, or electric charging stations also have lower ozone levels. On the contrary, results demonstrated that the LEZ has a positive impact on ozone. Developing mobility infrastructure could therefore compensate for this effect. Further analyses would be necessary to confirm this, as correlation does not imply causation. Moreover, it seems that municipalities with more public bike, car-sharing, and electric charging stations have a lower percentage of gasoline cars and a higher share of AFV and hybrid vehicles.

Table 10: Indicators for Brussels municipalities

city	Δ NO_2	Δ O_3	Δ AFV	Δ hybrid	Δ gasoline	Δ diesel	income pc	population	interquartile	% green	% left	% right	villo	cambio	electric
Anderlecht	-44.1	33.7	.92	5.2	21.9	-28.1	11,815	118,241	16,913	16.3	45.7	23.8	729	14	17
Auderghem	-53.7	33.6	1.6	7.1	14.7	-23.4	18,903	33,313	29,704	25	15.8	29.7	369	6	15
Berchem-Sainte-Agathe	-45.3	35.1	1.3	6.9	18.7	-26.9	15,678	25,396	24,701	15.6	32.3	29.5	174	7	10
Bruxelles	-43.4	41.6	6.5	19.9	-3.1	-22.4	13,053	176,545	19,114	19.3	42.2	20.4	1,940	40	60
Etterbeek1	-47.4	40.1	1.6	8.6	18.0	-28.9	14,976	47,414	23,542	26.7	.22.8	31	260	20	50
Evere	- 42.3	48	1.1	8.4	42.3	-53.2	13,950	40,394	21,030	16.2	36.9	26	285	7	15
Forest	-44.4	36.1	1.3	6.2	19.2	-26.8	14,935	55,746	20,963	27.4	35.2	20.2	394	13	14
Ganshoren	-44.7	37.1	0.8	5.4	18.7	-25.0	15,235	24,596	20,040	16.2	30.1	29.1	171	8	8
Ixelles	-45.4	38.8	2.4	10.7	9.2	-22.2	16,950	86,244	24,053	31.2	25.2	27	694	31	12
Jette	-44.4	38.7	0.7	5.5	20.1	-26.6	15,099	51,933	21,408	19.2	32.4	25.5	310	21	18
Koekelberg	-43.7	39.3	0.9	5.1	20.9	-26.9	12,821	21,609	19,055	16.5	45.7	23.7	164	8	6
Molenbeek-Saint-Jean	-41.8	36.9	0.5	3.9	24.4	-28.9	10,488	96,629	16,428	13.9	53.7	18.3	463	9	26
Saint-Gilles	-38.2	39.5	1.2	4.5	19.2	-25.2	13,364	50,471	17,757	30.3	42.8	15.4	299	21	16
Saint-Josse-ten-Noode	-.43.2	40.4	8.8	21.5	17.9	-47.9	9,317	27,115	14,559	18.3	60	10.4	166	2	4
Schaerbeek	-43.8	44.5	2.9	8.2	21.2	-32.4	12,174	133,042	19,264	23	41.1	16.1	890	37	40
Uccle	- 48.6	33.7	1.7	7.5	11.2	-20.5	20,463	82,307	28,813	23.8	17	39.3	452	15	26
Watermael-Boitsfort	-51.8	33.9	1.8	6.8	11.9	-20.5	20,286	24,871	29,317	31.4	18.8	28.6	143	15	22
Woluwe-Saint-Lambert	-48.5	41.4	2.2	11.5	14.2	-28	17,758	55,216	28,221	19.6	13?7	29.8	306	10	33
Woluwe-Saint-Pierre	- 49.3	40.4	2.4	7.9	10.3	-20.8	20,397	41,217	33,454	19.6	9.9	41.4	225	13	21

Sources: CELINE, Statbel, RMI, Datastore.Brussels, SPF mobilité, IBZ, own computations. Notes: standard errors in parentheses. In red, the 20% highest values for each variable, except variation in O3 and in the share of gasoline vehicles for which the lowest values are highlighted. Pollution in µg/m^3 and vehicle fleet in percentage. % green, % left, and % right correspond to votes for main political ideologies at the 2019 federal election.

$\Delta\text{PM10} = (\frac{\text{PM10}_{2023} - \text{PM10}_{2017}}{\text{PM10}_{2017}}) \times 100$ and similarly for the other variables.

8.3 Robustness checks

Tables 11 to 14 display robustness checks on the results for the overall effect of the LEZ as well as on the spillover effects. The base model corresponds to model (3) of Table 4 and Table 6, which is the model with all control variables, yearly and municipality-fixed effects, and standard errors clustered at the municipality level.

When changing the level of clustering for standard errors, results for air pollution remain mostly similar. It does however change the significance of the coefficients from the mobility. When standard errors are clustered at the agglomeration level, the impact of LEZ on all categories of vehicles is significant: the LEZ increases the share of AFV, hybrid, and gasoline vehicles and decreases the share of fossil fuel. This effect remains negligible for the first two categories. As previously discussed, it is difficult to know which level of cluster is most appropriate. However, clustering at the agglomeration level results in only one cluster for the treated group, and diminishes the validity of the parallel trends assumption for the diesel and gasoline categories, as demonstrated by Figure 5 (c) and (d).

Including the COVID years, 2020 and 2021, only increases the magnitude of the effect on air pollution while it barely impacts the analysis on mobility. Then, restricting the time studied for NO_2 and O_3 analysis generates no significant overall effect for NO_2 and spillover effects for O_3. Results for mobility are barely impacted by a change in the time period studied, except for the impact on the gasoline category that becomes significant. Finally, results are robust to a widening of the control group.

Table 11: Robustness checks: overall impact of LEZ on air pollution

	log(NO2)	log(O3)
(1) Original	-0.021*	0.055***
	(0.012)	(0.006)
(2) Agglomeration-level clusters	-0.029**	0.056***
	(0.014)	(0.008)
(3) COVID included	-0.027**	0.070***
	(0.013)	(0.006)
(4) Change time (2016 to 2023)	-0.000	0.045***
	(0.011)	(0.006)
(5) Control group using mean	-0.021*	0.057***
	(0.012)	(0.006)
Observations 1 and 2	391,018	391,018
Observations 3	478,632	478,632
Observations 4	238,856	238,856
Observations 5	449,351	449,351

Sources: CELINE, Statbel, RMI, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects except for (2)

Table 12: Robustness checks: spillover impact of LEZ on air pollution

	log(NO2)	log(O3)
(1) Original	-0.048***	0.016**
	(0.013)	(0.007)
(2) Agglomeration-level clusters	-0.067***	0.022**
	(0.021)	(0.009)
(3) Covid included	-0.057***	0.023***
	(0.013)	(0.007)
(4) Change time (2016 to 2023)	-0.034***	0.013
	(0.011)	(0.009)
(5) Control group using mean	-0.045***	0.016**
	(0.013)	(0.007)
Observations 1 and 2	385,216	385,216
Observations 3	471,415	471,415
Observations 4	254,226	254,226
Observations 5	443,549	443,549

Sources: CELINE, Statbel, RMI, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects, except for (2). Impact on the smaller group of peripheral municipalities.

Table 13: Robustness checks: overall effect of LEZ on mobility

	share AFV	share hybrid	share gasoline	share diesel
(1) Original	0.002	0.008*	0.018	-0.029**
	(0.002)	(0.005)	(0.012)	(0.013)
(2) Agglomeration-level clusters	0.002***	0.009***	0.021***	-0.032***
	(0.001)	(0.002)	(0.003)	(0.004)
(3) COVID included	0.001	0.008*	0.021	-0.030**
	(0.002)	(0.005)	(0.013)	(0.014)
(4) Change time (2015 to 2023)	0.001	0.008*	0.029**	-0.038***
	(0.002)	(0.005)	(0.013)	(0.013)
(5) Control group using mean	0.001	0.008	0.019	-0.029**
	(0.002)	(0.005)	(0.012)	(0.013)
Observations 1 and 2	1,107	1,107	1,107	1,107
Observations 3	1,353	1,353	1,353	1,353
Observations 4	847	847	847	847
Observations 5	1,269	1,269	1,269	1,269

Sources: SPF Economie, Statbel, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects

Table 14: Robustness checks: spillover impact of LEZ on mobility

	share AFV	share hybrid	share gasoline	share diesel
(1) Original	0.001	0.010***	-0.000	-0.11*
	(0.002)	(0.004)	(0.004)	(0.006)
(2) Agglomeration-level clusters	0.003*	0.015***	0.003	-0.023**
	(0.002)	(0.005)	(0.008)	(0.011)
(3) COVID included	0.001	0.009***	-0.002	-0.008
	(0.002)	(0.003)	(0.004)	(0.006)
(4) Change time (from 2015-2023)	0.000	0.008***	0.007*	-0.016**
	(0.002)	(0.003)	(0.004)	(0.006)
(5) Control group using mean	0.001	0.009**	0.002	-0.011**
	(0.002)	(0.004)	(0.005)	(0.006)
Observations 1 and 2	1,089	1,089	1,089	1,089
Observations 3	1,331	1,331	1,331	1,331
Observations 4	847	847	847	847
Observations 5	1251	1251	1251	1251

Sources: SPF Economie, Statbel, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects. Impact on the smaller group of peripheral municipalities.

8.4 Limits and extensions

To conclude this discussion section, the limits and the potential extensions of this research are going to be discussed.

Several limitations were encountered when realizing the analysis. The primary obstacle to a complete examination of the situation is the availability of data regarding the study on mobility. Specifically, the lack of historical data for other mobility indicators prevents the inclusion of more control variables in the DiD regression. Consequently, additional analyses have been realized in this section to seek more complete results. For the DiD approach in general, the internal validity of the model depends on a series of hypotheses, as described in the methodology section. The primary issue could come from the potential violation of the parallel trends assumption. Nevertheless, the outcome variables were selected to maximize the likelihood that this assumption holds true, and sensitivity analyses have shown that the results are robust up to a certain degree to deviations from the common trends assumption. As discussed, it is also challenging to isolate the impact of the LEZ from other policies targeting the Brussels region, such as the 30 km/h speed limit zone. However, the latter, implemented in 2022, only potentially influences the last two years of the results. Finally, it has been demonstrated by the literature review that several elements can influence the results of an LEZ from one city to another. Therefore, the external validity of the study is limited by the specific characteristics of the city and the design of the LEZ. The results can be generalized, but only to cities with similar features.

Some other points that did not enter into the framework of this research would be worth developing. It was addressed in the literature review and the theoretical framework that an alternative or complementary policy is the congestion charge. Bernardo et al. (2021) state in their study that the LEZ is more efficient when pollution is more important than congestion, and inversely. Therefore, with data on traffic, it would be interesting to study the impact of the LEZ on congestion, and eventually conduct a comparison to a city with similar features that implemented the congestion charge instead of, or in addition to, an LEZ to see which combination of policy is best suited. The availability of historical and detailed data regarding sustainable transport modes would also allow an econometric analysis to determine how the LEZ affects these variables.

Moreover, the differences-in-differences approach appeared to not be feasible on PM_{10}, $PM_{2.5}$, and BC. To have a more comprehensive analysis, it would be interesting to use a different approach to study the impact of those pollutants. Finally, the inequality aspect of the policy was highlighted in the literature review, and studied in section 8.1., showing that municipalities with a higher income and inequality experienced a bigger reduction in pollution. Studying the impact of policies aiming at reducing inequality could allow to determine which type of policies can counterbalance this effect. For instance, the Bruxell'air initiative, encouraging Brussels inhabitants to switch to sustainable transport modes, could be a subject of analysis.

9 Conclusion

This research aimed to evaluate the effectiveness of Brussels' low emission zone in reducing pollution and its impact on mobility through the greening of the vehicle fleet, both within the zone and in the surrounding areas.

The findings reveal nuanced impacts on both air pollution and mobility. While the introduction of the policy was efficient in reducing the share of diesel vehicles both inside (3.9%) and in the periphery of the zone (1.5%), it did not substantially encourage individuals living in the concerned area to switch to greener vehicles, only slightly increasing the share of hybrid vehicles (0.8%). It did however influence the greening of the vehicle fleet for surrounding municipalities, increasing the share of alternative fuel vehicles and hybrid vehicles by respectively 0.4% and 1.2%. In conclusion, although the literature highlighted that LEZs led to a decrease in the share of non-compliant cars (Margaryan, 2021; Wolff, 2014), it appears that in the case of Brussels, it did not encourage or have very little influence on the use of less polluting cars such as electric ones. Peters et al. (2021) reach a similar conclusion in the case of Madrid, highlighting that the LEZ has a significant effect on the AFV registrations, but usually towards non-zero emission vehicles.

Before the LEZ implementation, the EU limits for NO_2 were already being met, indicating that the policy was not necessary for compliance with these standards. Nonetheless, the introduction of the LEZ resulted in a modest reduction of 2.1% in NO_2 levels within the zone, which align with the literature (Ma et al., 2021; Margaryan, 2021). The most significant reductions were observed during the last phase of the LEZ, which banned Euro 4 diesel vehicles. However, this reduction was offset by a 5.5% increase in O_3 levels, a phenomenon explained by the mechanism of formation of O_3. In NO_x-saturated areas like city centers, a decrease in NO_x can lead to an increase in ozone levels (Sarmiento et al., 2023). Therefore, a policy resulting in a decrease in NO_x would increase O_3 levels, as it was most likely the case in Brussels. Additionally, municipalities surrounding Brussels saw even larger reductions in NO_2 levels, approximately 5%, with a slight increase in O_3 levels. These results mirroring the ones inside the zone, it suggests that road users did not react to the policy by bypassing the zone to avoid it.

It has also been emphasized that municipalities with higher income per capita and inequality, such as Auderghem, Uccle, and Watermael-Boitsfort, experienced more substantial reductions in air pollution. Heterogeneity tests also indicated that municipalities with a majority of the vote for left-wing political parties experienced a more important greening of the vehicle fleet, while municipalities with higher income per capita and inequality experienced an increase in the share of diesel vehicles and a decrease in the share of gasoline vehicles as compared to other municipalities.

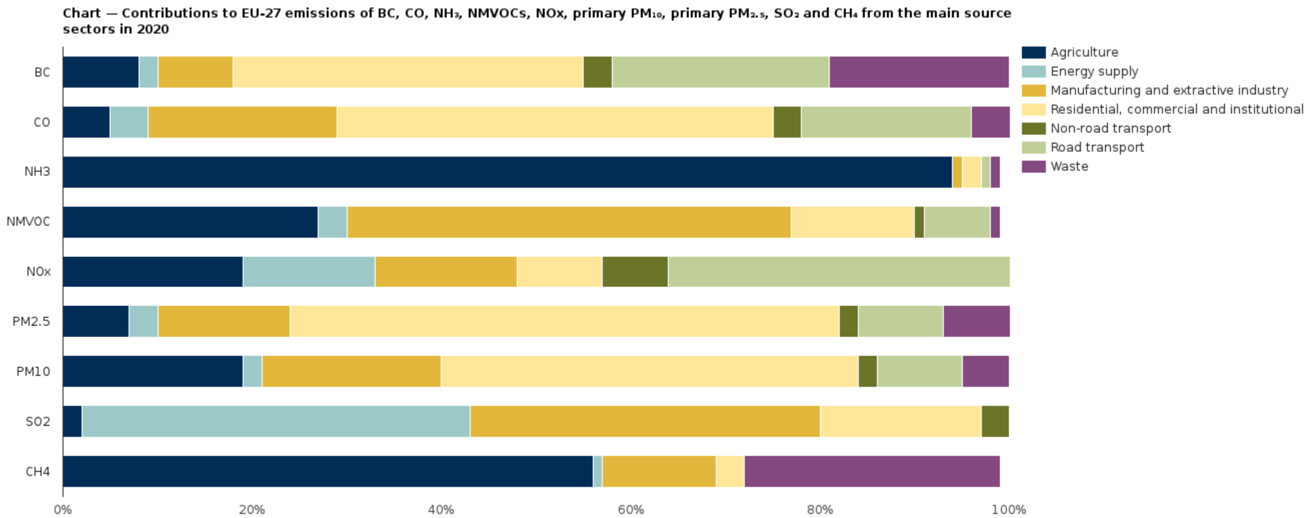
In terms of policy implications, these findings suggest that a way of dealing with these rising ozone levels should be thought of. Furthermore, the lack of significant effect on the share of gasoline vehicles and the negligible effect on AFV and hybrid vehicles could indicate that the stringency of the regulation is too weak at the moment to generate a substantial greening of the vehicle fleet. In addition, as discussed in the theory framework, although the LEZ has the potential to affect the greenness of the vehicle fleet, it does not influence car usage, as the quantity of kilometers driven is not targeted. Therefore, a strengthening of the rules of the LEZ could improve the greenness of the vehicle fleet, but would still not address vehicle usage and could have economic and social repercussions. Consequently, complementary policies should be considered. Correlations indicate that municipalities with more developed sustainable mobility infrastructures have a greener vehicle fleet and lower ozone levels, suggesting that investing in sustainable mobility could be beneficial.

Further research on alternative policies, such as congestion charges, should be considered and studied in depth. Additionally, a complementary econometric method should be developed to enable a more comprehensive analysis of air pollution, by incorporating a wider range of pollutants. Moreover, since the benefits of the LEZ primarily accrue to wealthier individuals, policies addressing the inequality aspects should be investigated.

Finally, some limitations of this research should be noted, such as the lack of historical mobility data that limited the ability to control for additional variables in the vehicle fleet analysis and the effects of other policies implemented in the same area that could not be isolated.

10 Appendices

Appendix 1: contribution of various sectors to air pollution



Source: European Environment Agency (2022)

Appendix 2: WHO recommendations regarding pollution

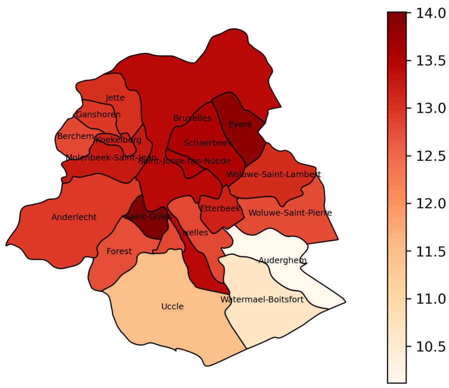
Pollutant	Averaging time	Interim target				AQG level
		1	2	3	4	
$PM_{2.5}$, µg/m³	Annual	35	25	15	10	5
	24-hour[a]	75	50	37.5	25	15
PM_{10}, µg/m³	Annual	70	50	30	20	15
	24-hour[a]	150	100	75	50	45
O_3, µg/m³	Peak season[b]	100	70	–	–	60
	8-hour[a]	160	120	–	–	100
NO_2, µg/m³	Annual	40	30	20	–	10
	24-hour[a]	120	50	–	–	25
SO_2, µg/m³	24-hour[a]	125	50	–	–	40
CO, mg/m³	24-hour[a]	7	–	–	–	4

[a] 99th percentile (i.e. 3–4 exceedance days per year).
[b] Average of daily maximum 8-hour mean O_3 concentration in the six consecutive months with the highest six-month running-average O_3 concentration.

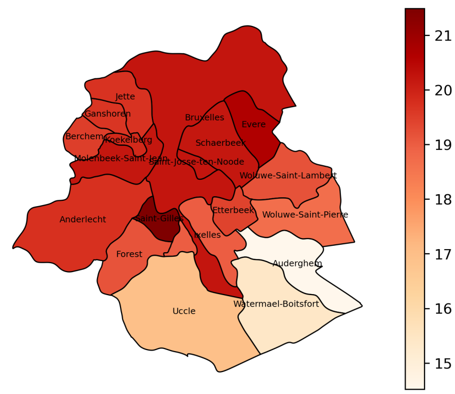
Source: World Health Organization (2022)

Appendix 3: Levels of pollution in Brussels' municipalities in 2017

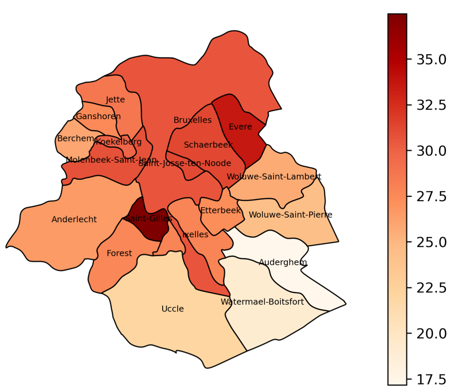
(a) Brussels $PM_{2.5}$ levels in 2017



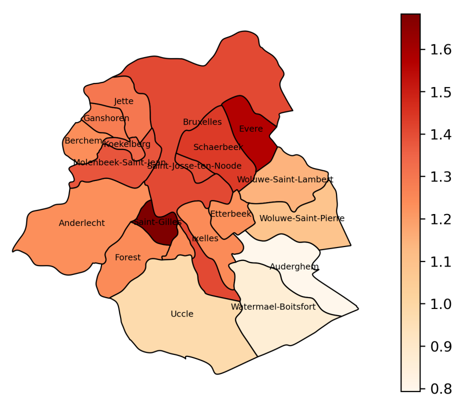
(b) Brussels PM_{10} levels in 2017



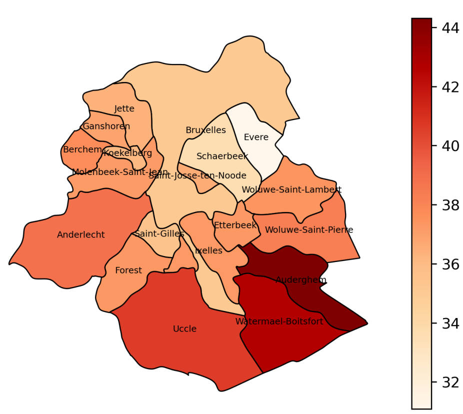
(c) Brussels NO_2 levels in 2017



(d) Brussels BC levels in 2017



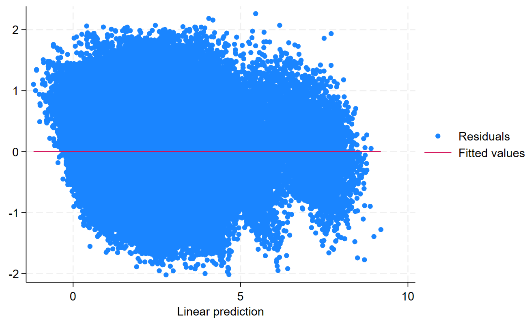
(e) Brussels O_3 levels in 2017



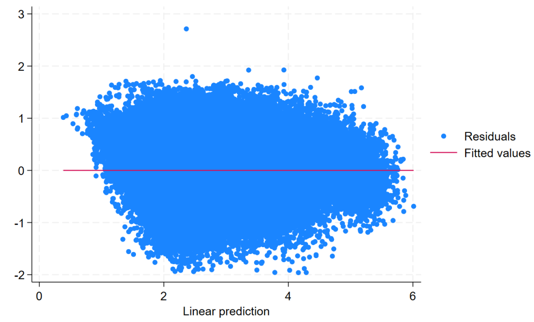
Source: CELINE, own representations

Appendix 4: residuals versus fitted values for air pollution regressions

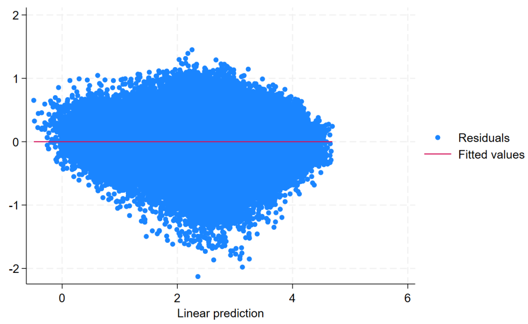
(a) Residuals of regression on log($PM_{2.5}$)



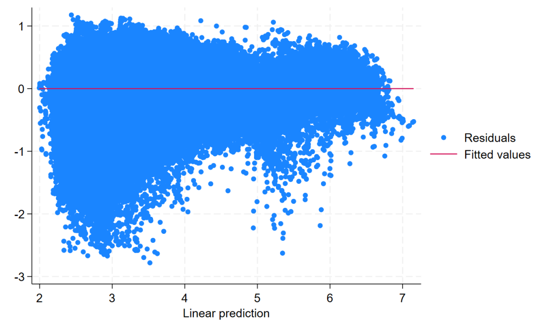
(b) Residuals of regression on log(PM_{10})



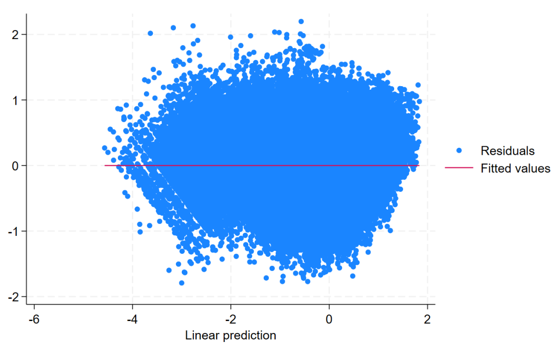
(c) Residuals of regression on log(NO_2)



(d) Residuals of regression on log(O_3)



(e) Residuals of regression on log(BC)

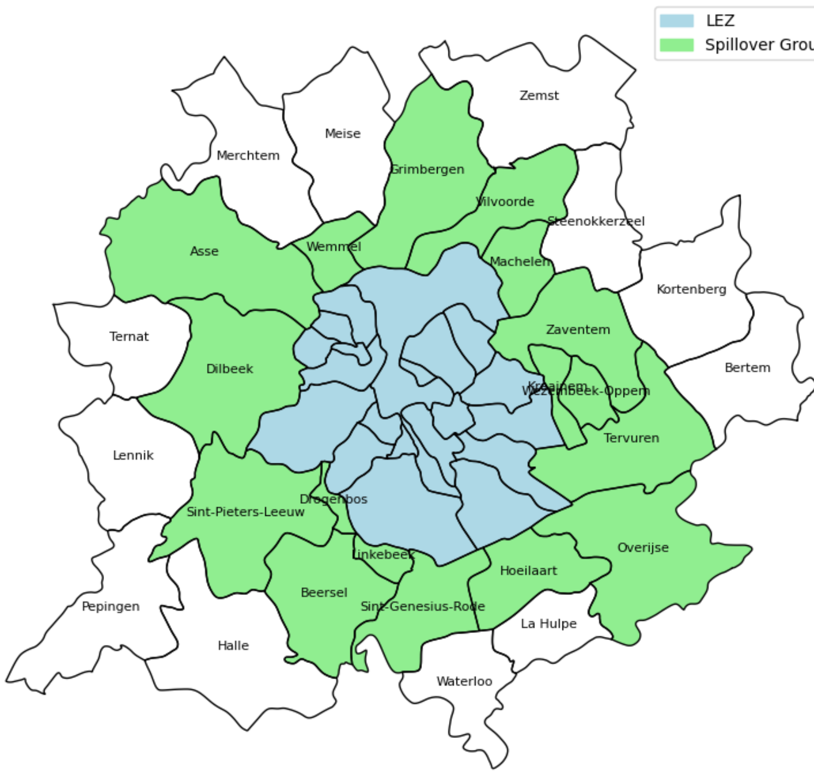


Source: CELINE, RMI, Statbel, own representations. Note: residuals from the regression:

$$\mathrm{Y}_{it} = \alpha + \beta \mathrm{LEZ}_i + \lambda \mathrm{PostLEZ}_t + \tau \mathrm{LEZ}_i \times \mathrm{PostLEZ}_t + \mathrm{A}_t + \mathrm{S}_i + \mathrm{X}_{it} + \mathrm{u}_{it}$$

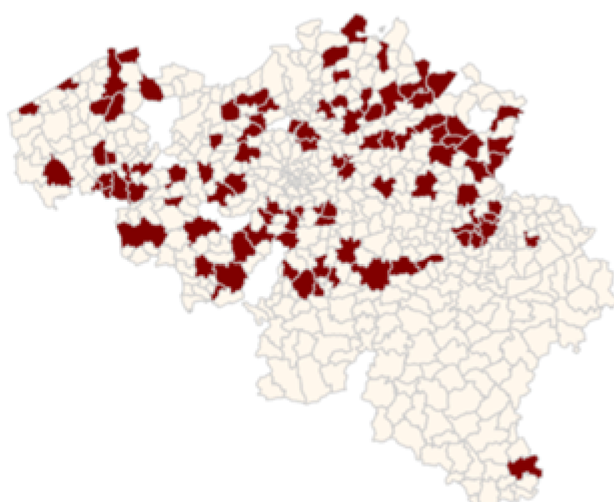
where Y is the logarithm of the daily concentrations of the pollutants, standard errors are clustered at the municipality level and yearly and municipality fixed effects are added. Control variables represented by X consist of logarithm of population, employment rate, weather covariates (average daily temperature, minimum temperature, maximum temperature, precipitation quantity in mm, average wind speed, max wind speed, average pressure), a dummy variable indicating weekend versus weekdays and seasonal dummies.

Appendix 5: municipalities for spillover study



Source: own representation. Note: the first group studied for spillover effect are municipalities represented in green, to which are added the ones in white for the second group.

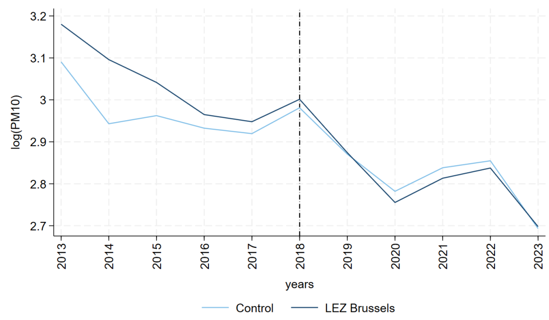
Appendix 6: map of control units



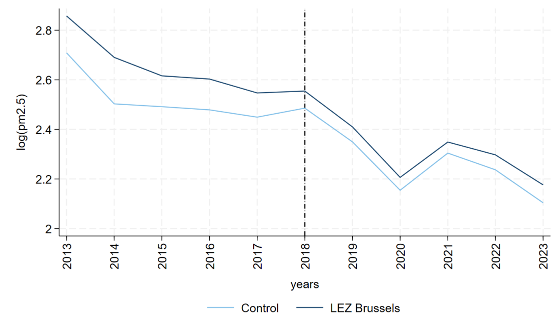
Source: own representation. Note: control units represented in red.

Appendix 7: raw parallel trends for air pollution

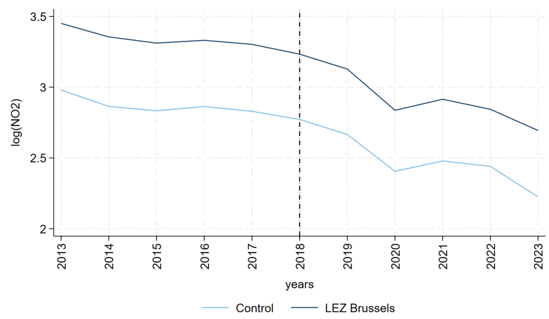
(a) Raw parallel trends PM_{10}



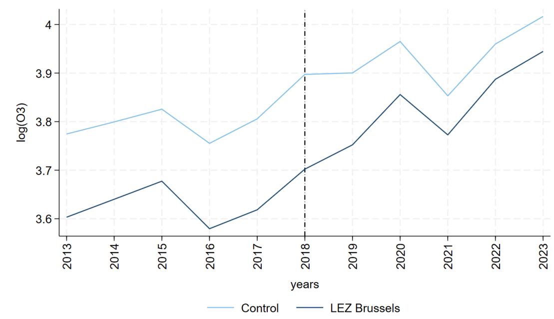
(b) Raw parallel trends $PM_{2.5}$



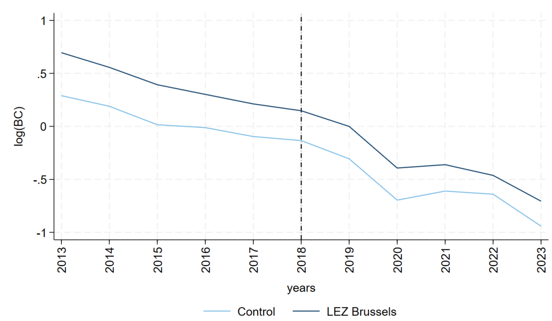
(c) Raw parallel trends NO_2



(d) Raw parallel trends O_3



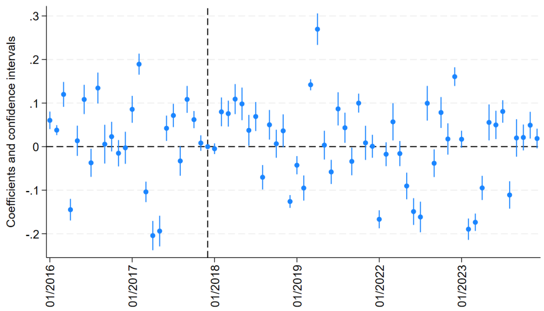
(e) Raw parallel trends BC



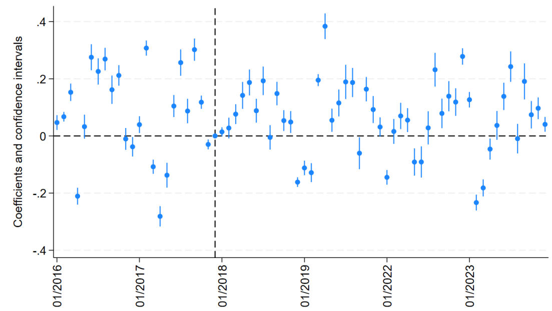
Source: CELINE, own representations

Appendix 8: monthly conditional parallel trends for PM_{10}, $PM_{2.5}$ and BC

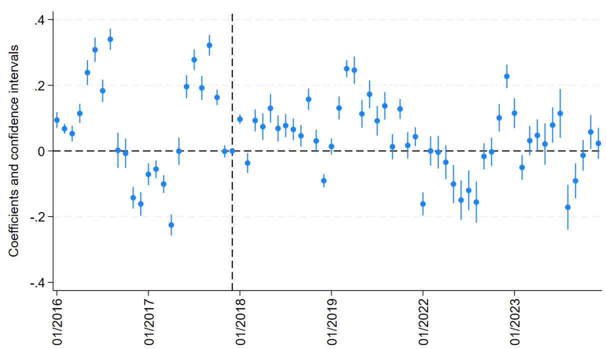
(a) Monthly conditional parallel trends PM_{10}



(b) Monthly conditional parallel trends $PM_{2.5}$



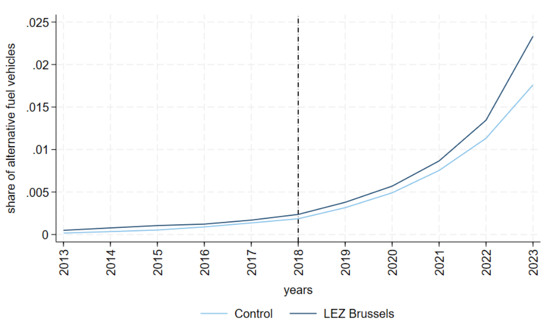
(c) Monthly conditional parallel trends BC



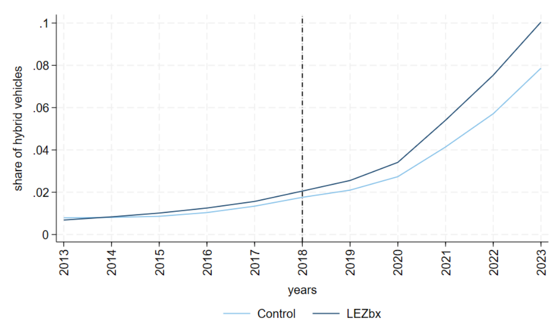
Source: CELINE, Statbel, RMI, own computations. Notes: standard errors cluster at the municipality level. Year-month and municipality fixed effects.

Appendix 9: raw parallel trends for mobility

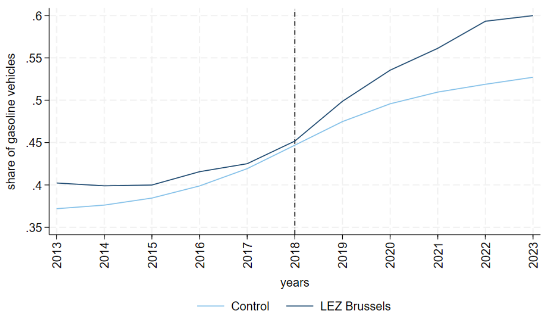
(a) Raw parallel trends AFV



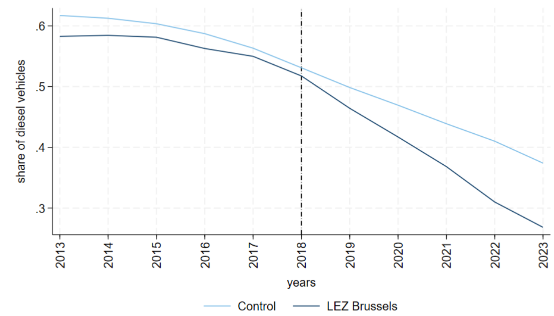
(b) Raw parallel trends hybrid vehicles



(c) Raw parallel trends gasoline



(d) Raw parallel trends diesel vehicles



Source: SPF Economie, own representations

Appendix 10: complete version of the overall impact of Brussels' LEZ on air pollution

	log(NO2)			log(O3)		
	(1)	(2)	(3)	(1)	(2)	(3)
LEZ (β)	0.493***	0.559***	/	-0.216***	-0.285***	/
PostLEZ (λ)	-0.365***	-0.342***	/	0.174***	0.111***	/
LEZ overall (τ)	-0.026***	-0.018***	-0.021*	0.056***	0.054***	0.055***
log(population)		0.107***	-0.942***		-0.044***	0.126
employment rate		-0.318***	0.241		0.404***	0.370**
weekend		-0.283***	-0.280***		0.093***	0.092***
winter		0.118***	0.140***		0.228***	0.230***
summer		-0.242***	-0.263***		0.286***	0.288***
spring		-0.045***	-0.054***		0.551***	0.558***
precipiations		0.001***	0.007***		-0.003***	-0.005***
temperature		-0.049***	-0.047***		0.037***	0.036***
min temperature		-0.009***	-0.015***		-0.022***	-0.020***
max temperature		0.030***	0.032***		0.028***	0.027***
wind speed		0.150***	0.038***		-0.128***	-0.092***
max wind speed		-0.178***	-0.124***		0.159***	0.142***
pressure		0.001***	0.001***		-0.004***	-0.004***
constant	2.761***	2.306***	12.806***	3.658***	5.988***	4.172**
N	391,018	391,018	391,018	391,018	391,018	391,018
control	No	Yes	Yes	No	Yes	Yes
fixed effects	No	No	Yes	No	No	Yes
cluster	No	No	Yes	No	No	Yes

Sources:CELINE, Statbel, RMI, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effect; standard errors clustered at municipality level. "/" means that coefficients are omitted due to collinearity.

Appendix 11: complete version of the overall impact of Brussels' LEZ on mobility

	share AFV			share hybrid			share gasoline			share diesel		
	(1)	(2)	(3)	(1)	(2)	(3)	(1)	(2)	(3)	(1)	(2)	(3)
LEZ (β)	0.001	0.003***	/	0.001	0.011***	/	0.018**	0.373***	/	-0.024***	-0.053***	/
PostLEZ (λ)	0.007***	0.006***	/	0.034***	0.029***	/	0.101***	0.095***	/	-0.143***	-0.132***	/
LEZ overall (τ)	0.001	0.001	0.002	0.011***	0.009***	0.008*	0.026**	0.023**	0.18	-0.038***	-0.035***	-0.029**
log(population)		0.001***	0.051		0.003**	0.171*		-0.001	0.297		-0.004	-0.516**
Employment rate		0.036***	-0.009		0.114***	0.084		0.192***	0.354***		-0.316***	-0.433**
constant	0.001**	-0.036***	-0.522	0.009***	-0.102***	-1.83*	0.390***	0.273***	-2.931	0.597***	0.859***	6.26**
Observations	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107	1,107
Control	No	Yes	Yes	No	Yes	Yes	No	Yes	Yes	No	Yes	Yes
Fixed Effects	No	No	Yes	No	No	Yes	No	No	Yes	No	No	Yes
Cluster	No	No	Yes	No	No	Yes	No	No	Yes	No	No	Yes

Sources: SPF Economie, Statbel, own computations. Notes: standard errors in parentheses; * p<0.1, ** p<0.05, ***p<0.01; yearly fixed effect and municipality fixed effects; standard errors clustered at the municipality level. "/" means that coefficients are omitted due to collinearity.

Appendix 12: correlations between mobility indicators and changes in air pollution and vehicle fleet

(a) Correlation air pollution in 2023 and mobility aspects in Brussels

	NO2	O3
villo stands	0.17	-0.21
cambio	0.21	-0.24
electric station	-0.015	-0.03

Source: datastore brussels, CELINE, own computations

(b) Correlation between composition of vehicle fleet in 2023 and mobility aspects in Brussels

	share AFV	share hybrid	share gasoline	share diesel
villo stands	0.41	0.43	-0.52	0.33
cambio	0.20	0.18	-0.21	0.07
electric station	0.21	0.26	-0.23	0.04

Source: datastore brussels, SPF Economie, own computations

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12 Executive summary

Low emission zones have become popular for dealing with air pollution in urban centers and mitigating its detrimental effects. This research examines the impact of this policy on the city of Brussels for which literature is scarce, on both air pollution and mobility. Through a differences-in-differences method, the effect of the LEZ can be isolated. Studying mobility through the vehicle fleet indicates that the policy is effective in reducing the share of diesel vehicles (-2.9%) but does not induce substantial changes towards less polluting vehicles. The policy influences however more individuals living on the outskirts of Brussels to switch to greener vehicles than the ones living inside. Furthermore, findings indicate that the policy reduces nitrogen dioxide levels by 2.1% inside the zone and by approximately 5% in its surroundings. It however leads to an increase in ozone levels of 5.5% in the concerned area. The study of spillover effects leads to the conclusion that individuals do not bypass the zone to avoid it. Finally, it appears that the wealthiest individuals benefit the most from the policy in terms of air pollution. Overall, these results suggest that additional policies to promote the greening of the vehicle fleet could be beneficial and that strategies to address rising ozone levels should be explored, as well as policies to address the potential inequality aspect of the LEZ.

Word count = 20,420