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A (GREEN) SWITCH IN TIME SAVES NINE: ASSESSING THE ENVIRONMENTAL DAMAGE OF THE EUROPEAN TRUCK CARTEL

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A (Green) Switch in Time Saves Nine: Assessing the Environmental Damage of the European Truck Cartel*

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Abstract

This study examines how the cartel of European truck manufacturers coordinated the timing of compliance with emission standards, generating additional air pollution without violating environmental regulations. Although firms formally complied with environmental law, collusion restricted competition over cleaner technologies, highlighting that anticompetitive agreements can have significant environmental and health consequences. First, we quantify the volume of particulate emissions attributable to cartel behavior by constructing two plausible counterfactual scenarios for truck fleet composition, identifying substantial excess emissions of approximately 119 thousand tonnes of fine particulate matter ($PM_{2.5}$). Second, we estimate the health impact of traffic-related $PM_{2.5}$ emissions on infant respiratory outcomes using a panel of 199 European subregions observed over an 18-year period. To address endogeneity concerns, we exploit exogenous variation in EURO emission standards through a shift-share instrumental-variable strategy. The resulting elasticity allows us to compute the number of infant respiratory hospital admissions attributable to the cartel under counterfactual competitive conditions. We estimate that earlier, competition-driven adoption of cleaner technologies could have reduced average yearly infant hospital admissions by 12–18 cases per 1,000 births at the NUTS 2 level.

Keywords: Air pollution, Truck cartel, Anticompetitive agreement, Environmental damage, EURO standards, European Commission.

JEL Codes: I18, K21, L41, Q51, Q52.

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"Competition policy is not going to take the place of environmental laws or green investment. The question is rather if we can do more, to apply our rules in ways that better support the Green Deal." - Margrethe Vestager.[1]

1 Introduction

Achieving ambitious environmental objectives poses challenges that extend beyond the design of environmental regulation alone. Environmental outcomes also depend on how other public policies shape firms' incentives and market behavior. In this context, sectoral and competition policies must be at least consistent with environmental objectives and can play an important complementary role in supporting them. This challenge is particularly prominent within the European Union (EU), as it strives to achieve carbon neutrality by 2050 as part of the European Green Deal.[2] The EU recognizes that the transformation towards a sustainable economic and administrative model requires collective efforts from public authorities to address climate challenges effectively. In this context, competition policy plays a pivotal role in shaping a regulatory framework that fosters business practices aligned with environmental objectives.[3] This perspective has been repeatedly emphasized in official communications and scholarly analyses issued by the European Commission and by international organizations such as the Organisation for Economic Co-operation and Development (OECD).[4]

This paper illustrates the importance of these issues through an original case study: the European Commission's dismantling of the truck manufacturers' cartel (case AT.39824 - Trucks).[5] In this cartel, manufacturers colluded to delay the availability of European

[1]Conference organized by the DG Competition at the European Commission on how competition rules and sustainability policies can work together (February 4, 2021).

[2]See Commission, The European Green Deal COM (2019) 640 final.

[3]See Holmes (2024) for an overview on the alignment of competition policy with sustainable goals.

[4]See, for instance, Margrethe Vestager, "The Green Deal and Competition Policy" (22 September 2020); European Commission, "Call for Contributions: Competition Policy Supporting the Green Deal" (13 October 2020); European Commission, "Competition Policy Contributing to the EU Green Deal" (4 February 2021); Inge Bernaerts, "Competition Policy in Support of the Green Deal" (10 September 2021); and Julian Nowag, "Sustainability and Competition", OECD Background Note (2020). In October 2025, European Commissioner for Competition Teresa Ribera emphasised the need to better integrate environmental externalities into competitive dynamics: "A big challenge here is how we can ensure a fair competitive economy based on real costs being reflected in the final price, including externalities. Environmental economists have been working on this for decades through taxation and regulation, and we may need to step up how we tackle that from a competition policy perspective." Closing remarks at the Lisbon Conference on Competition Law and Economics (24 October 2025).

[5]A cartel is only one form of horizontal agreement. Other horizontal agreements can also have environmental impacts. For instance, Schinkel & Spiegel (2017) and Schinkel et al. (2022) study sustainability agreements

emission standards (EURO), thereby avoiding competition on the environmental quality of their trucks. The case is distinctive in that it presents a paradox. While truck manufacturers formally complied with European regulations on engine emissions, a substantial amount of emissions could have been avoided had they not restricted competition on environmental performance. Addressing these "avoidable emissions" fell squarely within the purview of the competition authority, as the manufacturers were not in breach of environmental laws. Although the Commission acknowledged the environmental implications of the cartel in its decision, these considerations were not reflected in the calculation of the fine.

The main objective of this study is to quantify the environmental impact of the European truck cartel by (i) estimating the volume of avoidable emissions of particulate matter ($PM_{2.5}$) and (ii) assessing their consequences for human health. Transport emissions account for around 20%-30% of sources of $PM_{2.5}$ in Europe (Grange et al. 2021, Pultz et al. 2022), which are known to be harmful to human health (WHO, 2021).[6] A large body of literature documents the relationship between reductions in transportation emissions and associated health benefits, mainly driven by environmental policies (Reynaert 2021, D'Haultfœuille et al. 2016, Inoue et al. 2020, Andersson 2019). The health impacts of air pollution fall into three categories: direct health effects (Williams & Phaneuf 2019, Deryugina et al. 2019, Godzinski & Castillo 2021), productivity losses (Leroutier & Ollivier 2025, Kögel 2022), and adaptation behaviors aimed at reducing exposure (Deschenes et al. 2017). This article focuses on direct health effects and is closely related to the work of Alexander & Schwandt (2022), who exploit nitrogen oxides (NOx) fraud in the Dieselgate case as a natural experiment to evaluate the impact of emissions on infant health.[7] Extensive evidence documents the adverse effects of $PM_{2.5}$ on infant health and early-life outcomes (Currie & Walker 2011, Almond et al. 2018, Lagravinese et al. 2014, Bekkar et al. 2020), underscoring the importance of considering the most vulnerable populations in studies of air pollution and health.

First, we estimate the volume of avoidable particulate emissions attributable to cartel behavior by comparing observed emissions to counterfactual scenarios that simulate competitive adoption of cleaner technologies. Following a standard but-for approach in competition

in this context. Furthermore, an emerging literature examines how environmental policy affects horizontal mergers in polluting industries and can change both the timing (Benchekroun et al. 2019) and magnitude of emissions (Choi et al. 2022), with potentially ambiguous effects on environmental goals.

[6] $PM_{2.5}$ is the pollutant most consistently linked to severe health impacts, including cardiovascular, respiratory, and early-life outcomes. It is the only pollutant regulated under the EURO emission standards that is simultaneously classified by the WHO as a major pollutant of concern for health.

[7] In the field of epidemiology, Kelly et al. (2025) assess the health consequences of Dieselgate in Europe and estimate that the scandal led to 205,000 premature deaths and 152,000 new cases of childhood asthma over the period.

policy, we construct the factual evolution of truck fleet emissions using registration data for vehicles sold between 1997 and 2011, combined with official emission factors by vehicle weight class and EURO standard from the European Environmental Agency. Fleet composition is approximated using assumptions on average vehicle lifetime and annual mileage. We then develop two plausible competitive counterfactuals reflecting earlier adoption of emission-reduction technologies: one based on evidence from the European Commission's cartel investigation, and a second based on observed market entry dates of EURO-compliant trucks documented by the European Environmental Agency. The gap between observed and counterfactual emissions yields an estimate of excess emissions that can be associated to collusion of approximately 119 thousand tonnes of $PM_{2.5}$.

Second, we estimate the causal effect of traffic-related particulate emissions on infant respiratory health. We use a two-way fixed-effects panel model at the NUTS 2 regional level over the period 2000–2018, relating $PM_{2.5}$ emissions from heavy-duty vehicles to hospital admissions for respiratory conditions among infants. To address concerns about endogeneity arising from local economic activity, transport demand, or measurement error in emissions, we exploit exogenous regulatory variation in EURO emission standards through a shift-share instrumental-variable strategy. The instrument combines predetermined regional exposure to road traffic, measured by pre-2000 highway density, with EU-wide regulatory tightening of emission standards, generating exogenous variation in pollution exposure across regions and over time. Our main specification shows that 1% increase in traffic-related emissions increase the number of hospital admission of infant by 0.116%. This relationship remains robust across a wide range of alternative specifications and robustness checks.

Finally, we quantify the health consequences of cartel-induced excess emissions by applying the estimated $PM_{2.5}$-health coefficient to the counterfactual emission scenarios. This approach allows us to compute the number of infant respiratory hospital admissions that can be associated to the cartel. Our results indicate that earlier, competition-driven adoption of cleaner technologies could have reduced infant respiratory hospital admissions by between 12 and 18 cases per 1,000 births each year at the NUTS 2 level over the period considered.

This article makes several contributions to the existing literature. First, this study is the first to explore the implementation of an emissions standard in the context of anti-competitive practices. Several dismantled European cartels have been identified as having environmental impacts (Monti 2020). Within eco-industries, these impacts generally stem from price-fixing, which undermines the adoption of green products.[8] However, limiting eco-innovations can also be a cartel's objective, as illustrated by the German car cartel.[9] In this case, the cartel coordinated on a suboptimal technology for limiting NOx emissions from cars, namely the size of Diesel Exhaust Fluid (DEF) tanks, in order to avoid additional costs and preserve trunk space. Alé-Chilet et al. (2025) show that this collusive restriction led to systematically undersized tanks (on average about 16 liters instead of the roughly 30 liters required for compliance), brought the entire industry into on-road non-compliance, and tripled actual NOx emissions. The authors estimate that coordinated non-compliance substantially reduced expected penalties for the firms while generating significant environmental and welfare losses, amounting to €1.6-5.6 billion. The truck manufacturers' cartel we study offers a distinct and novel perspective.[10] While formally complying with environmental regulations, manufacturers coordinated to adopt more stringent emission standards only when legally required, never anticipating them. Our results show that this restriction of competition led to additional air pollutant emissions.

Second, to isolate the specific impact of the cartel, we construct two counterfactual emission scenarios that deviate from the actual timeline. These scenarios are based on the identification of dates when more efficient technologies became available to truck manufacturers. This approach allows us to estimate the environmental damage directly attributable to the cartel under plausible scenarios. By adapting the 'but-for' methodology to the context of emissions, this study introduces a novel perspective on anticompetitive practices, particularly by highlighting environmental externalities that remain largely overlooked in such cases.[11]

Third, our study is among the few to demonstrate that anticompetitive practices can have measurable consequences for both emissions and public health. This finding is particularly significant given the scale of the case under investigation.

Lastly, we focus on the heavy-duty vehicle sector, whereas most of the literature on the health effects of emissions from specific vehicles concerns passenger cars. For example, previous studies have examined the effects of car emissions on maternal health near highway toll booths, as well as the link between infant mortality and weekly traffic variations (Currie & Walker 2011, Knittel et al. 2016). However, research into air pollution caused by heavy-

[8] See, for instance, the car battery recycling cartel (Case AT.40018 - *Car battery recycling* (2017)).

[9] Case AT.40178 - *Car Emissions* (2019).

[10] §50 - Case AT.39824 *Trucks*, Commission decision of July 19, 2016.

[11] Alé-Chilet et al. (2025) adapt this method to evaluate the anti-competitive conduct of the German car cartel (see above), adjusting it to the characteristics of the products (in this case, the size of the DEF tanks).

duty vehicles is much more limited, as these vehicles are subject to different regulatory frameworks, operational targets, and exposure pathways for emissions. Their greater weight and engine size mean that they emit disproportionately high volumes of pollution per vehicle. Given their dominant role in European freight transport, neglecting the truck sector leaves an important gap in the literature.

The paper is structured as follows. Section 2 sets out the institutional and conceptual background of the European truck cartel and explains how the counterfactual scenarios are constructed. Section 3 describes the data. Section 4 outlines the empirical strategy and discusses identification. Section 5 presents the estimation of the health effects of particulate emissions. Section 6 examines the health consequences of cartel-induced excess emissions based on the counterfactual scenarios. Section 7 concludes.

2 Institutional and Conceptual Background

This section outlines the institutional and conceptual background for our analysis. We first describe the structure and operation of the European Truck Cartel, emphasizing its role in delaying the adoption of cleaner technologies despite available emission-reduction innovations in Section 2.1. We then motivate our counterfactual scenario in Section 2.2, which assumes that in a competitive market, firms would have had incentives to adopt stricter emissions standards earlier.

2.1 Cartel conduct and delayed compliance

The cartel was initiated on January 17, 1997, and involved six major manufacturers: DAF, Daimler, Iveco, MAN, Scania, and Volvo/Renault. Together, these firms accounted for nearly 90% of the European truck market, operating in a highly concentrated industry with substantial barriers to entry and long-established distribution networks. The cartel was thus not all-inclusive, and encountered competition from several manufacturers across America and Asia. The collusion lasted until January 18, 2011, when MAN applied for leniency to the European Commission. Its unusually long duration reflected the stability of market participants and the absence of significant competitive disruptions. In 2017, the European Commission imposed a record fine of €3.8 billion, sanctioning the participants for price coordination.[12]

According to the Commission's decision,[13] the cartel involved three main types of collusion constituted a single and continuous infringement of Article 101(1) of the Treaty on the Functioning of the European Union (TFEU) and Article 53(1) of the Agreement on the European Economic Area: (i) coordination on gross list prices across the European Economic Area; (ii) agreement on the timing of the introduction of emission-reduction technologies; and (iii) coordination on the intention to pass compliance costs related to environmental standards onto customers. However, the fines imposed addressed only the first dimension,[14] and did not account for the environmental consequences of the second infringement. Although the Commission acknowledged the environmental implications of the collusion, these were not integrated into the sanctioning process.[15]

This second infringement of the cartel is the core of our purpose. Instead of competing through innovation and early compliance with environmental standards, firms agreed on a timing for the technological adoption of successive EURO standards. These standards, progressively implemented from the early 1990s onward, set increasingly strict limits on emissions of pollutants such as NOx, carbon monoxide (CO), hydrocarbons (HC), and $PM_{2.5}$ from heavy-duty vehicles.[16] In other words, they agreed to delay the supply of EURO III, EURO IV, EURO V and EURO VI compliant trucks.[17]

Despite the availability of relevant technologies, manufacturers deliberately delayed the rollout of EURO III to VI compliant trucks to synchronize launch dates and preserve profit margins.[18] According to the Commission, these delays were discussed in confidential meetings, often concealed within trade fairs or through private email exchanges. This strategy not only signaled reluctance to advance environmental standards but also revealed a coordinated

[12] A summary of fines by manufacturers is available in Table A1 in Appendix. See also Figure B1, Figure B2 and Figures B3 for market share distributions.

[13] §50 - Case AT.39824 *Trucks* Commission decision of July 19, 2016.

[14] In determining the amount of the fines imposed, the Commission took into account the sales of medium and heavy trucks of each company in the European economic area, as well as the gravity of the infringement, the high combined market share of the companies, the geographic scope and the duration of the cartel.

[15] In a welfare assessment, Beyer et al. (2020) estimate that the European truck cartel reduced consumer surplus by 5-7%.

[16] Details about the texts giving the dates of application of the new limit values are in Table A2 and Figure B4 in the Appendix.

[17] Article 1 (Case AT.39824-Trucks, 19 July 2016) "By colluding on [the timing] for the introduction of emission technologies for medium and heavy trucks required by EURO 3 to 6 standards, the following undertakings [truck manufacturers'] infringed Article 101 TFEU and Article 53 of the EEA Agreement [...]". §52 "They agreed not to offer EURO 3 standard compliant trucks before it was compulsory to do so", "[...] discussions took place concerning, [...] the introduction of EURO 4 standard compliant trucks, similar to the discussions that had previously been held concerning the EURO 3 standard".

[18] The drafting of the Commission directives (EC, 2017) involved a consultation process in which truck manufacturers and industry associations commented on technical feasibility.

effort to control market dynamics by postponing cleaner technologies. Compliance involved costly upgrades, including particulate filters and selective catalytic reduction systems.

By aligning compliance timelines and agreeing to pass regulatory costs onto consumers, cartel members likely delayed technological adoption and weakened incentives for environmental innovation. While this paper does not assess the legal merits of the Commission's findings, it seeks to quantify the environmental and health damages linked to this coordinated delay. Although companies continued to formally comply with EU environmental legislation, their collusion resulted in a significant volume of emissions that could have been avoided, highlighting the role of the competitive policy.

2.2 Constructing the counterfactual

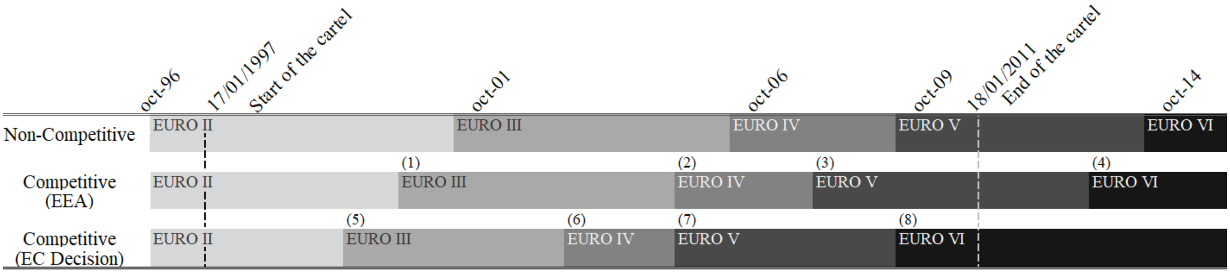
In order to assess the environmental consequences of the anti-competitive agreement among truck manufacturers, we must establish plausible counterfactual scenarios reflecting market dynamics in the absence of collusion (Section 2.2.1). This construction is based on two fundamental assumptions (Section 2.2.2).

2.2.1 Simulating emissions without the cartel

We construct two counterfactual scenarios, inspired by the "but-for" approach commonly used in competition policy, to assess damages (Veljanovski 2023, Alé-Chilet et al. 2025).

The EC counterfactual. The first source we used is the European Commission's investigation, which is the factual basis for manufacturers' collusive behavior regarding EURO standards. This source provides factual evidence that the cartel companies sought to influence the timing of the introduction of EURO standards, despite having the technical capacity to implement them earlier. For each standard introduced during the cartel period (i.e. over 14 years), we identified elements in the Commission's findings referring to the availability of technologies meeting EURO requirements, as well as efforts to conceal this availability whenever possible. The relevant references are as follows: EURO III compliance is addressed in §52 of the July 19, 2016 decision; EURO IV in §§54, 115, 130, 135, and 141; and EURO V and VI in §§135, 141, 180, and 181 of the September 27, 2017 decision. This counterfactual scenario is referred to as 'EC' for the remainder of the paper.

The EEA counterfactual. The second source is the 2019 European Environmental Agency report, which is a comprehensive reference for calculating air pollutant emissions from road vehicles. It outlines the methodology used by emissions assessment centers across Europe for heavy and medium commercial vehicles. The report determines the introduction date of compliant vehicles based on their year of first registration in Europe, as defined by EU legislation.[19] The report allowed us to identify the date when compliant trucks entered the market, including those produced by non-cartel manufacturers and imports, providing a useful benchmark for assessing the pace of adoption in a competitive environment. This counterfactual scenario is referred to as 'EEA' for the remainder of the paper.



Standard	Implementation		Simulation from EEA		Simulation from EC-Decision
EURO III	oct-01	(1)	jan-00	(5)	apr-98
EURO IV	oct-06	(2)	jan-05	(6)	apr-03
EURO V	oct-09	(3)	jan-08	(7)	may-05
EURO VI	oct-13	(4)	jan-13	(8)	sept-09

Figure 1: Timing of the implementation of EURO standards for trucks in the factual scenario C, Counterfactual *EEA* and *EC*

Figure 1 summarizes the timelines for introducing vehicles equipped with these standards to the market between October 1996 and October 2014. The first timeline shows the situation during the cartel. The next two timelines show the counterfactual scenarios in which the technologies were introduced earlier.

2.2.2 Two fundamental assumptions

Assumption #1: Competition means earlier adoption of EURO standards. Following the European Commission's view,[20] our first assumption is that, under competition, firms would have had incentives to differentiate by launching EURO-compliant vehicles ahead of regulatory deadlines. While not directly testable, this assumption rests on both theory and empirical evidence.

[19]See the detailed implementation dates in Table A3 in the Appendix.

[20]Commission Decision of 27 September 2017, paragraph 286: "In a competitive environment, competitors strive to outperform one another by introducing more advanced technology."

First, the Air Pollutant Emission Inventory Guide 2019 report by the European Environment Agency (EEA) provides information on the introduction of EURO standards. It confirms that some trucks, including imports and models from manufacturers outside the cartel, were compliant with the new EURO standards before they became legally binding. This indicates that forward-thinking companies were already using cleaner technologies. One of the sources used in our counterfactual analysis (see below) draws on these real-world cases of early adoption, providing tangible evidence of competition based on environmental performance, even in the absence of regulatory constraints.

Second, a growing literature shows that product market competition spurs green innovation, especially when driven by environmentally minded consumers or policy signals. Aghion et al. (2023) find that competitive pressure and green preferences jointly raise innovation rates in the global auto industry. While freight operators are the consumers in our context, competitive forces could still encourage early adoption, especially under regulatory or reputational pressure orienting their purchasing policies towards green consumption.

Third, even without strong customer preferences,[21] competition in oligopolies can lead to environmental differentiation under external pressure. Thornton et al. (2009) show that large, visible firms in concentrated markets respond strongly to social expectations and institutional norms. The European truck manufacturers involved in the cartel fit this profile: they are global brands. For such firms, being perceived as laggards on emission performance entails reputational risks, including negative publicity and pressure from civil society. In the absence of collusion, these reputational concerns, combined with policy-driven incentives, would therefore likely have pushed manufacturers to improve the environmental performance of their trucks.

Fourth, cleaner technologies often improve fuel efficiency, which is a key factor in fleet purchasing decisions (Gallagher & Muehlegger 2011). The Commission itself notes the central role of fuel consumption (as "operating costs") in customer choices (EC Decision, 2016). Because EURO compliance typically involves innovations that cut both emissions and fuel use, firms had a clear incentive to pursue vertical product differentiation. This link is discussed in industry sources,[22] policy reports,[23] and academic work in economics (Reynaert

[21] That cartel members explicitly agreed not to compete on environmental performance suggests such preferences may have been weak.

[22] See, e.g., specialist press: https://www.commercialmotor.com/knowledge-hub/article/euro-7-for-trucks

[23] See ICCT: https://theicct.org/sites/default/files/publications/ICCT_G20-briefing-paper_Jun2015_updated.pdf

2021, Lin & Linn 2023) and engineering (e.g., Keramydas et al. 2019).

Fifth, the directives' drafting process (EC, 2017) involved consultations with manufacturers and industrial associations on technical feasibility. Their participation, without major objections, indicates that the required innovations were already feasible (or nearly so) before the deadlines.

Overall, the convergence of concerns about fuel costs, reputational dynamics, policy signals and available technology makes it highly plausible that cleaner trucks would have been introduced earlier without the cartel, resulting in reduced emissions.

Assumption #2: Same number of trucks sold (pure quality effect). A second assumption is required to build the counterfactual scenario, as the number of trucks that would have been sold under competitive conditions is unknown. One could speculate that sales might have been higher, since truck prices would likely have been lower without collusion. This argument dates back to Buchanan (1969) who emphasized that market power could reduce emissions by restricting quantities.

However, economic consulting reports indicate that truck prices continued to rise during and after the cartel period (Fideres Partners 2023). It should also be noted that anticipating stricter emission standards for trucks would have likely had a positive impact on their selling price. Meanwhile, data from the European Automobile Manufacturers' Association (ACEA) show that truck registrations declined from 2012 to 2015 (ACEA 2012), before increasing again from 2015 to 2019 (ACEA 2019). To test for a structural shift, we compared average registrations in 2007-2010 and 2011-2015 using a Student' t-test and found no statistically significant difference ($p > 0.10$).

These mixed findings suggest caution when speculating on potential quantity effects. We therefore assume that the number of trucks sold under competition would have remained unchanged. This allows us to isolate the cartel's environmental impact arising from the 'composition effect' of the truck fleet with respect to the EURO emission standards, independent of its 'volume effect' (Asker et al. 2024).[24]

[24] Asker et al. (2024) distinguish between the volume and composition effects of a cartel in their analysis of OPEC's market power in oil extraction, where the composition effect reflects the allocation of production across oil fields with different carbon intensities.

3 Data

The analysis relies on a panel of European NUTS 2 regions, corresponding to sub-national administrative units within European countries, observed annually over the period 2000-2018. Owing to data availability, the number of regions included in the sample gradually increases over time. It rises from 63 regions in 2000 to 199 regions per year from 2010 onwards. Overall, the final dataset consists of 2,924 region-year observations covering 199 distinct NUTS 2 regions.

Section 3.1 describes the estimation of the $PM_{2.5}$ emissions volume, Section 3.2 presents the health outcome variables, and Section 3.3 details the set of control variables included in the analysis.

3.1 Trucks data

3.1.1 Fleet composition

To capture the annual inflow of new trucks during the cartel period and beyond, we rely on registration data from the ACEA.[25] The dataset distinguishes between two main weight categories: Medium Commercial Vehicles (CV), weighing between 3.5 and 16 tonnes, and Heavy Commercial Vehicles (HCV), exceeding 16 tonnes. It covers annual registrations from 1993 to 2018 and is based on manufacturers' declarations. The European truck market exhibits a high degree of concentration and stability (Bovin & Bos 2023), with relatively constant market shares across major manufacturers, as illustrated in Figure B1 in the Appendix.

To approximate the age distribution of the circulating truck fleet, we draw on the French National Road Committee's (CNR) long-haul transport survey (2006-2010),[26] which reports an average operational lifespan of 6.8 years and an average annual mileage of 113,800 km per truck. Based on this, we assume that over the entire period covered by the agreement, the circulating fleet includes approximately seven overlapping cohorts, each conforming to the EURO emission standard in force at the time of registration. The resulting fleet composition by standard and year is presented in Figure 2.

[25] Available at `https://www.acea.auto`.

[26] The CNR 'Long Distance' surveys provide detailed information on operating conditions and vehicle characteristics. Conducted annually among a representative panel of 220 companies, these surveys focus on freight transport.

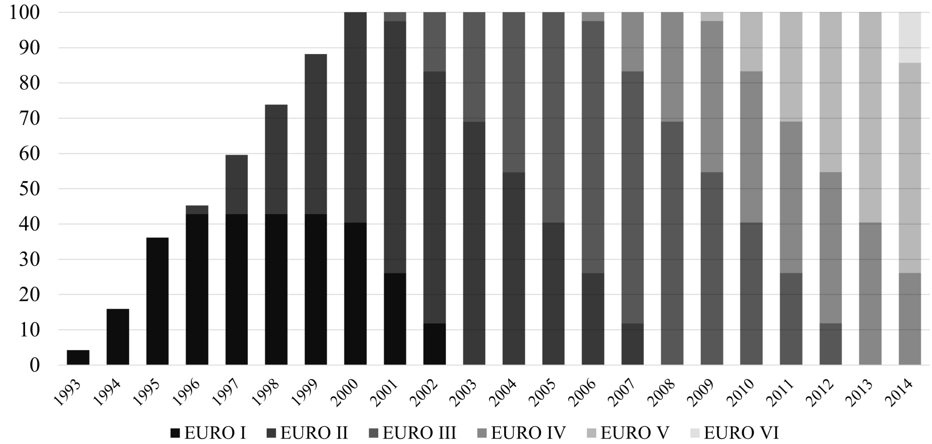


Figure 2: Composition (in %) of the truck fleet by EURO standard, 1993-2014

Consistent with the findings of the European Commission's infringement decisions (19 July 2016 and 27 September 2017), which established that cartel members deliberately delayed the introduction of cleaner technologies, we assign each vehicle to the EURO standard legally required at the time of its registration.

3.1.2 Emissions factors and regulatory thresholds

For our analysis, we compiled the official implementation dates for each EURO standard[27] and type of vehicle. Before a vehicle model or group of similar models can be placed on the market, truck manufacturers are required to commission an independent testing body to measure the emissions of each vehicle type in real-world conditions. These measurements are collected by the EEA in their Transportation Emissions Assessment Report (EEA, 2019).[28]

To link these emissions to the registration data, we follow the classification provided by the United Nations Economic Commission for Europe, which specifies that heavy-duty vehicles (excluding buses) fall into categories N2 (for CV) and N3 (for HCV). The EEA report provides emission measurements broken down by driving conditions (urban, rural, highway). Table 1 reports the average recorded emissions per kilometer, by vehicle weight category and by pollutant regulated under the EURO standards. These figures are used to estimate total emissions during the cartel period using highway driving condition.

[27]See Table A2 in the Appendix

[28]This guide aims to provide the methodology, emission factors and activity data required to calculate exhaust emissions from road vehicles.

Table 1: Average recorded emission scores of a single truck by weight and standard in highway driving conditions

Weight	Standard	NOx (g/km)	CO (g/km)	HC (g/km)	$PM_{2.5}$ (g/km)
3.5 - 16 t	EURO I	4.34	0.8385	0.2595	0.165
	EURO II	4.495	0.7195	0.165	0.0825
	EURO III	3.465	0.778	0.152	0.07235
	EURO IV	2.145	0.059	0.0065	0.01335
	EURO V	1.2215	0.059	0.0065	0.01335
	EURO VI	0.2355	0.059	0.0065	0.00065
>16 t	EURO I	8.28	1.725	0.4795	0.3275
	EURO II	8.635	1.535	0.308	0.1745
	EURO III	6.85	1.64	0.293	0.1405
	EURO IV	4.22	0.113	0.011	0.02535
	EURO V	2.405	0.113	0.011	0.02535
	EURO VI	0.4645	0.113	0.011	0.00125

Note: This analysis adheres to the Tier 2 methodology as outlined in the EEA (2019) report. It involves quantifying emission factors for a set of vehicles and technologies in grams per vehicle-kilometer in highway driving condition. These factors are derived from average European datasets, encompassing variables such as typical driving speeds, and ambient temperatures. Our calculations use data from Tables 3-21 and 3-22 of the EEA report (2019). To ensure comprehensive coverage of the entire weight range of medium and heavy trucks, we interpolate between adjacent weight classes, following the recommendations for Tier 2 inventory methodologies.

To estimate total emissions per vehicle, we multiply the values in Table 1 (average exhaust emissions in g/km)[29] by the vehicle's annual mileage and operational lifetime. Specifically, we assume an average yearly mileage of 113,793 km and a lifespan of 6.8 years, based on survey data previously described. The result is an estimate of cumulative lifetime emissions for each vehicle type, by weight and EURO standard. Table A4 in the Appendix A summarizes these lifetime emissions, in tonnes of pollutant per truck. We then aggregate yearly emissions by combining fleet composition data (from Section 3.1.1) with per-vehicle lifetime emissions, disaggregated by weight class and standard.

3.1.3 Emission calculation based on the observed fleet

This analysis primarily focuses on $PM_{2.5}$ emissions, given their well-documented adverse effects on human health. This section quantifies the factual emissions generated by trucks registered during the cartel period, relying exclusively on observed fleet composition and emission factors. This factual scenario is hereafter referred to as C.

[29] We do not include other forms of emissions from road transport, such as evaporation, road wear caused by vehicle movement, and wear and tear on road vehicle tires and brakes, as emission standards do not take these forms into account.

As described in Subsection 3.1.1, we use truck registration data to construct a monthly database of the truck fleet. For each month m, the data records the number of heavy-duty trucks and medium-duty trucks registered under each EURO emission standard $i \in \{0, \ldots, VI\}$, denoted N_{im}^{HCV} and N_{im}^{CV}, respectively.

Emissions are computed by combining average measured emission factors per vehicle, differentiated by weight class and EURO standard, with assumptions on vehicle usage and lifetime. Starting from emission estimates expressed in grams per kilometer and the estimated average distance travelled, we derive monthly emission levels per vehicle. We denote monthly emissions per truck for heavy and medium vehicles by E_i^{HCV} and E_i^{CV}, respectively.

We define $m = 0$ as the beginning of the cartel period (January 1997) and $m = 168$ as its end (January 2011). At the start of the cartel, the fleet largely consisted of trucks introduced up to 6.8 years earlier (October 1990, $m = -75$). Given the assumed average vehicle lifetime, trucks sold in the final cartel month ($m = 168$) continued to generate emissions until October 2017 ($m = 249$).

In each month, newly registered trucks enter the fleet, while previously registered trucks, typically meeting older EURO standards, remain in operation and continue to emit pollutants. For example, trucks sold in January 2001 ($m = 48$) complied with the EURO III standard but operated alongside EURO I and EURO II vehicles. Total fleet emissions in month m are therefore obtained by summing emissions across all active EURO standards and vehicle types. Specifically, in month $m = 48$, emissions in scenario C are given by:

$$P_{48}^{C} = N_{I48}^{HCV} E_{I}^{HCV} + N_{I48}^{CV} E_{I}^{CV} + N_{II48}^{HCV} E_{II}^{HCV} + N_{II48}^{CV} E_{II}^{CV} + N_{III48}^{HCV} E_{III}^{HCV} + N_{III48}^{CV} E_{III}^{CV}$$

Total emissions under the observed fleet evolution are obtained by summing monthly emissions over time. The complete set of equations is provided in Appendix C, Equation (6).

Over the period considered, the truck fleet undergoes ten distinct configurations, reflecting the overlapping presence of vehicles complying with different EURO standards. At the outset of the cartel, compliance with the EURO II standard was already mandatory, resulting in a fleet composed of EURO 0, EURO I, and EURO II vehicles. Subsequent introductions of EURO III, EURO IV, EURO V, and EURO VI standards progressively altered the fleet composition until the cartel's dissolution in January 2011. Emissions from these vehicles persist for up to 6.8 years after sale, extending the impact of the cartel until October 2017.

Finally, monthly emission estimates are aggregated to annual levels to match the temporal resolution of the health outcome data.

3.1.4 Regional disaggregation of emissions by European sub-regions

Assigning truck emissions to specific regions is inherently challenging, particularly for long-haul freight vehicles whose environmental impact extends well beyond their place of registration. Truck emissions are generated along the entire route travelled and therefore affect all regions crossed, rather than only the origin or destination areas.

This issue has been widely documented in the literature. For example, Janic (2007) emphasizes the importance of accounting for transit countries when evaluating the social costs of freight transport networks. Similarly, Levy et al. (2010) show that traffic-related environmental and health impacts should be attributed to locations where vehicles are actually used and where congestion occurs, rather than solely to registration areas.[30]

To account for this spatial complexity and avoid misattributing emissions to vehicle registration locations, we adopt a spatial disaggregation approach known as *gridding*, following the methodology recommended by the EMEP/EEA Guidebook (2019, Section 4.6). This method redistributes aggregate emissions to smaller spatial units (such as NUTS 2 regions) using spatial proxies that better reflect the geographic distribution of pollutant-generating activity in the road transport sector.

The gridding procedure combines a top-down allocation of total emissions with bottom-up spatial information. In our application, annual truck emissions are distributed across European regions in proportion to each region's share of the total highway network. This infrastructure-based proxy is consistent with EEA guidance when detailed traffic flow data are unavailable and reflects the assumption that regions with greater highway length are more likely to host or transit freight traffic.

Formally, emissions attributed to NUTS 2 region r in year t are given by

$$P_{r,t}^{C} = h_r \times P_t^{C},$$

where P_t^C denotes total truck emissions in year t under scenario C, and h_r is region r's share

[30] Luxembourg provides a clear illustration of this issue. Despite having few registered trucks relative to its highway density, it is likely to experience substantial emissions from foreign-registered trucks using its dense highway network. Its highway length per surface area (53 km per 1,000 km^2) is more than three times the European average (15 km per 1,000 km^2).

of the European highway network, defined as

$$h_r = \frac{L_r}{\sum_R L_R},$$

with L_r representing total highway length in region r and the denominator summing over all regions R.

By explicitly accounting for the transboundary nature of freight transport, this gridding-based approach provides the spatial granularity required for downstream analyses of pollutant exposure and health impacts, particularly along international freight corridors.

3.2 Child health data

Air pollution profoundly impacts health, particularly the respiratory system, by exposing individuals to harmful particulate matter.[31] The complexity of adult health determinants, which include a wide range of lifestyle and behavioral factors (including smoking habits, physical activity levels, and dietary preferences) motivates the focus on child health in this study. These confounding factors pose problems in isolating the direct effects of emissions in adult cohorts for epidemiological research (Black et al. 2007, Almond et al. 2018). Conversely, child health provides a cleaner setting for analysis, as this population are less exposed to lifestyle-related factors.

Furthermore, the vulnerability of infants to air pollution is a growing concern due to their unique physiological and developmental characteristics. Their respiratory and immune systems are still maturing, which increases their vulnerability to damage from $PM_{2.5}$ and other pollutants. Infant respiratory health has therefore become an important variable in health research because it reflects not only the health of the mother and child, but also the quality of the environment during pregnancy. Recent studies have shown a significant association between prenatal exposure to traffic-related air pollution and the incidence of low birth weight. As a result, this variable has been widely used in research to assess the effects of air pollution on human health through infants (Alexander & Schwandt 2022, Jones & Goodkind 2019, Currie & Walker 2011, Black et al. 2007, Almond et al. 2018).[32] These

[31] This exposure can lead to respiratory tract irritation, asthma, and chronic obstructive pulmonary disease, and it intensifies cardiovascular risks by inducing inflammation and vascular damage, thereby increasing the likelihood of heart attacks, hypertension, and strokes (Godzinski & Castillo 2021).

[32] For example, an expansive study conducted in 2013 analyzed data from three million births across nine nations, identifying a clear connection between $PM_{2.5}$ exposure and low birth weights (Pedersen et al. 2013, Bekkar et al. 2020).

findings support the use of child health outcomes as indicators of the health effects of truck emissions.

For this study, we used regional health data at the NUTS 2 level from Eurostat's Healthcare Non-Expenditure Statistics database, focusing on the total number of hospital admissions due to respiratory conditions (these conditions are classified by the International Classification of Diseases (ICD-10) as codes J00-J99[33]). The dataset includes three age groups: infants (under one year old), young children (1 to 4 years old), and the general population (between 0 and 65 years old). These data provide insight into the healthcare burden associated with respiratory diseases.

To construct a most consistent panel, we addressed missing regional data as follows: If one region's data was missing, we used national totals minus the sum of admissions of available regions. If several regions had missing values, we allocated the remainder of the national total proportionally based on population shares.

The panel is unbalanced, reflecting the staggered availability of regional health data across countries and years.[34] This unbalanced structure does not pose a threat to identification for two reasons. First, all empirical specifications include NUTS 2 fixed effects, which absorb time-invariant regional characteristics and ensure that identification exploits within-region variation over time. Second, year fixed effects control flexibly for common shocks affecting all regions in a given year, including aggregate trends in health outcomes, economic conditions, and regulatory changes. As a result, the estimates are identified from deviations around region-specific means rather than from cross-sectional differences or changes in sample composition.

3.3 Demographic and economic control variables

Our econometric model evaluates the impact of truck traffic emissions on child health, while controlling for regional demographic and economic characteristics to account for intra-national heterogeneity and potential confounding factors.

[33]The ICD-10, developed by the World Health Organization, is an international classification system in which each code corresponds to a disease. 'J' code stands for respiratory diseases that include: acute upper respiratory infections (J00-J06), influenza and pneumonia (J09-J18), chronic lower respiratory diseases such as chronic obstructive pulmonary disease and asthma (J40-J47), and other respiratory conditions such as pleurisy, respiratory failure and lung diseases (J60-J99).

[34]Early years in the sample contain fewer observations, with coverage expanding gradually over time as additional regions enter the dataset. By the mid-2000s onward, coverage stabilizes, and the majority of regions are observed consistently through to the end of the period.

To account for regional economic conditions, we use GDP per capita, expressed in euros at current market prices.[35] Economic prosperity may simultaneously influence pollution levels and healthcare quality, as wealthier regions generally invest more in environmental protection and healthcare infrastructure, potentially leading to better health outcomes.

We complement our data with the total stock of registered vehicles reported at the NUTS 2 regional level,[36] as this variable captures regional transport intensity. Regions with higher transport intensity likely experience higher overall emissions, potentially affecting respiratory health outcomes independently of truck-specific emissions.

We control for median age to account for differences in demographic structure across regions, which directly influence baseline health risks and hospital utilization. We also include the number of recorded births to adjust for cohort size effects in infant-related outcomes, ensuring that estimated pollution impacts reflect changes in health risk rather than differences in population scale. These variables are fully described in Table 2.

Table 2: Statistical portrait of the database

Names	Unit	Obs	Mean (Std. Dev.)	Min	Max	Kurt.
Hospital Adm. (0-1 YO)	Number of adm.	2,924	1,477.17 (1,623.201)	1.00	15,382.00	18.69
Hospital Adm. (1-4 YO)	Number of adm.	2,052	2,497.96 (2,508.079)	1.00	18,329.00	11.4
Hospital Adm. (0-65 YO)	Number of adm.	2,640	25,900.47 (20,712.9)	5.00	123,067.00	6.54
Infant Mortality Rates	Percentage	2,924	3.69 (1.346)	0.00	13.00	8.48
Births	Number of births	2,924	21,005.07 (21,634.51)	267.00	191,813.90	24.7
GDP per capita	(€)	2,924	28,239.01 (16,639.82)	3,100.00	164,061.50	16.26
Median Age	Year	2,924	41.54 (3.248)	30.90	51.50	3.06
Stock of vehicles	Number of vehicles registered	2,924	936,503.62 (148,710.7)	24,362.0	6,606,202	15.49

Source: Eurostat. "Adm." is short for "admission". The observations cover a period of 18 years and 199 NUTS 2 regions. This represents 23 countries, with an average of 17 NUTS 2 regions per country.

[35] Data obtained from Eurostat's *Regional Economic Accounts* under the code nama_10r_2gdp. Swiss regional data were supplemented using information from the Federal Statistical Office of Switzerland.

[36] Data from Eurostat's database "Vehicle fleet by category by NUTS 2 region", including all vehicles except trailers and motorbikes.

4 Empirical Strategy and Identification

This section presents our empirical strategy for assessing the health impact of trucks-related emission. The baseline model is in Section 4.1. We motivate and elaborate the shift-share instrument in Section 4.2 in order to estimate a causal impact of truck-related emissions on infant health.

4.1 Baseline empirical model

We study the relationship between truck-related emissions and infant health outcomes using a panel of European NUTS 2 regions observed annually between 2000 and 2018. Our baseline outcome of interest is the logarithm of the number of hospital admissions for respiratory causes among infants under one year old. This population group is widely considered to be particularly vulnerable to air pollution exposure due to physiological immaturity and limited capacity for avoidance behavior.

Our starting point is a fixed-effects panel regression of the following form:

$$\text{Hosp}_{r,t} = \alpha_r + \beta P^C_{r,t} + \beta_X X'_{r,t} + \lambda_r + \lambda_t + u_{r,t} \tag{1}$$

where $\text{Hosp}_{r,t}$ denotes respiratory hospital admissions in region r and year t, $P^C_{r,t}$ is the truck-related emissions of $\text{PM}_{2.5}$, and $X'_{r,t}$ is a vector of time-varying regional controls, including GDP per capita, median age, the size of the heavy-vehicle fleet, and the number of births recorded. Region fixed effects (λ_r) absorb time-invariant differences in baseline health, geography, infrastructure, and healthcare capacity, while year fixed effects (λ_t) capture aggregate shocks common to all regions, such as medical progress, macroeconomic fluctuations, and policy changes. All variables are expressed in logarithms. The estimated coefficients should be interpreted as the local average treatment effects resulting from the interaction between EU-wide regulatory tightening and past level of exposure to road traffic.

Estimating Equation (1) by ordinary least squares (OLS) is unlikely to recover a causal effect of emission on infant health for two main reasons. First, truck emissions may be endogenous to local economic activity, transport demand, and urban development. Regions experiencing economic growth may simultaneously generate more freight traffic while improving healthcare access, leading to potentially offsetting biases. For instance, populations in poor health might avoid polluted areas or pressure for stricter environmental enforcement (see Moretti & Neidell 2011, Deschenes et al. 2017). As highlighted by Lagravinese et al. (2014), failing to account for such sources of endogeneity can lead to biased estimates due to omitted variables and simultaneity. Second, emissions are not directly observed at a fine spatial scale and are constructed using engineering-based estimates, which makes them susceptible to measurement error. Classical measurement error would attenuate the estimated coefficient toward zero, while non-classical error may introduce additional bias. These concerns motivate an instrumental variable strategy.

4.2 Shift-share instrument based on EURO emission standards

To identify the causal effect of truck-related emission on infant health outcomes, we exploit exogenous regulatory variation induced by the successive tightening of European vehicle emission standards (EURO) using a shift-share instrumental variable strategy, following the recommendations of Borusyak et al. (2025). This approach combines predetermined regional exposure to road transport infrastructure (the share) with EU-wide regulatory shocks affecting emissions per vehicle (the shift), generating plausibly exogenous variation in pollution exposure across regions and over time.

4.2.1 Construction of the shift-share instrument

The share component captures long-run regional exposure to road traffic and is defined as pre-2000 highway density, measured as the length of highways relative to regional surface area.[37]

This variable reflects historical infrastructure decisions driven by geography, national planning priorities, and long-term transport policy, most of which were determined decades before the introduction of modern EURO emission standards. Because it is measured prior to the study period, it is fixed over time and cannot respond to contemporaneous changes in air pollution, health outcomes, or regulatory intensity.

Regions with higher pre-existing highway density are therefore mechanically more exposed to road traffic and experience larger changes in air pollution when vehicle emission intensity is altered by regulation.

The shift component consists of successive EURO standards that progressively tightened

[37]Data sourced from `trans_r_net`, Eurostat Regional Transport statistics. A descriptive map in the Appendix (Figure B6) displays regions above and below the median of pre-2000 highway density.

particulate emission limits for heavy-duty vehicles at the EU level. These regulations were introduced according to a common regulatory timeline and applied uniformly across countries and regions.

Notably, the EURO standards affect emissions per vehicle rather than traffic volumes, congestion, commuting patterns, or spatial population distribution. As a result, they generate plausibly exogenous variation in air pollution that operates independently of local economic or demographic dynamics.

We define the shift-share instrument ($SS_{r,t}$) as the interaction between predetermined regional exposure to road traffic and the successive implementation of EURO emission standards:

$$SS_{r,t} = \sum_{i \in \{0,\ldots,V\}} \text{RoadDensity}_r \times \mathbb{1}\{\text{EURO}_i \text{ in force at } t\}, \tag{2}$$

where RoadDensity_r denotes the pre-2000 highway density of region r, and $\mathbb{1}\{\text{EURO}_i \text{ in force at } t\}$ is an indicator equal to one if emission standard EURO i is in force at year t.

4.2.2 Instrumental variable specification

We instrument truck-related particulate emissions using the shift-share variation described above. The first-stage equation is:

$$\hat{P}^C_{r,t} = \pi\, SS_{r,t} + \gamma_X X'_{r,t} + \lambda_r + \lambda_t + v_{r,t}, \tag{3}$$

where $\hat{P}^C_{r,t}$ denotes truck-related $PM_{2.5}$ emissions in region r and year t.

The second-stage equation corresponds to Equation (1), where $P^C_{r,t}$ is replaced by its instrumented counterpart.

All specifications include region and year fixed effects. Conditional on these fixed effects, only the interaction between road density and EU-wide emission regulation generates identifying variation, operating through emissions per vehicle. Consistent with the shift-share literature (Borusyak et al. 2025), inference should reflect the level at which common shocks operate. EURO standards are EU-wide and time-varying, while exposure differs across regions. Standard errors are clustered at the regional level because identification in our shift-share specification arises from cross-sectional differences in exposure across regions.

First-stage validity tests in Table 3 confirm that the shift-share instrument strongly predicts truck-related particulate emissions. First-stage coefficients estimated are in Table A5 in the Appendix. The Sanderson-Windmeijer F-statistic exceeds conventional Stock-Yogo thresholds, and the Kleibergen-Paap LM test strongly rejects underidentification. Weak-instrument-robust Anderson-Rubin and Stock-Wright tests further support the validity of inference.

Overall, this strategy isolates plausibly exogenous variation in $PM_{2.5}$ emission driven by regulatory changes interacting with predetermined infrastructure, allowing a causal interpretation of the estimated effects on infant health outcomes.

Table 3: Instrument validity tests

Instrument Strength Tests	
Kleibergen-Paap LM stat	15.08 (p = 0.001)
SW F-stat (excluded inst.)	10.23 (p = 0.000)
Cragg-Donald F-stat	39.04
Stock-Yogo critical values:	
10% maximal IV bias	9.08
5% maximal IV bias	13.91
Weak-IV robust tests:	
Anderson-Rubin F	4.01 (p = 0.008)
Anderson-Rubin Chi^2	12.19 (p = 0.007)
Stock-Wright LM	30.62 (p = 0.000)

Note: The Kleibergen-Paap LM test rejects underidentification, and the Kleibergen-Paap and Sanderson-Windmeijer F-statistics exceed the Stock-Yogo critical values, indicating that the instruments are not weak. Anderson-Rubin and Stock-Wright tests provide weak-IV-robust inference and support the joint significance of the instrumented coefficient.

4.3 Threats to identification and exclusion restriction

A remaining concern in this setting is whether the regulatory variation exploited here may affect health outcomes through channels other than air pollution. While our identification strategy exploits regulatory variation in emission standards interacted with regional exposure, several alternative channels could, in principle, link EURO regulations to health outcomes beyond air pollution. We briefly address the most plausible candidates. First, traffic noise is unlikely to confound our estimates, as EURO standards regulate exhaust emissions but are unlikely to be materially affected by changes in vehicle noise levels. Traffic noise is primarily driven by mechanical and rolling factors and subject to distinct regulatory frameworks. Moreover, our health outcome focuses exclusively on hospital admissions for respiratory diseases, which limits the scope for noise-related channels (such as stress or sleep disturbance) to drive the estimated effects. Second, road safety is not a concern in our context: by focusing exclusively on hospital admissions for respiratory diseases, our outcome variable explicitly excludes health shocks related to traffic accidents. Third, congestion effects are unlikely to drive the results. Our counterfactual analysis assumes constant vehicle volumes (see Section 2.2.2), and there is no institutional or empirical evidence that the introduction of EURO standards led to discrete changes in traffic density. Any gradual congestion trends are absorbed by region and time fixed effects. Finally, while regulatory compliance may affect vehicle costs, general price effects would operate slowly and diffusely across regions, and are unlikely to align with the timing and spatial structure of the regulatory variation exploited here. Taken together, while no instrumental variable strategy can entirely rule out all alternative channels, these considerations suggest that the estimated effects are unlikely to be driven by non-pollution mechanisms.

5 Estimating the Health Effects of Particulate Emissions

5.1 Baseline estimates: OLS vs IV

Table 4 presents our baseline estimates of the impact of truck-related particulate emissions on respiratory hospital admissions among infants under one year old, comparing fixed-effects OLS estimates to our instrumental-variable (IV) strategy. The objective of this table is twofold. First, it illustrates why an IV approach is required to estimate the health effects of road-related pollution. Second, it provides direct evidence of the bias level affecting naïve OLS estimates.

Table 4: OLS and IV estimates of pollution on infant respiratory admissions

Estimator	OLS (1)	IV (2)
Emission	0.014	0.116**
	(0.006)	(0.053)
GDP per capita	-0.045	-0.275
	(0.111)	(0.213)
Median Age	-0.229	-0.505
	(0.603)	(0.644)
Stock Vehicles	-0.244***	-0.472***
	(0.090)	(0.131)
Number of Births	0.703***	0.811***
	(0.111)	(0.164)
Observations	2,924	2,924
R-squared	0.086	0.05
FE (r, t)	Yes	Yes
Number of NUTS 2	199	199

Note: This table reports in (1) coefficients from fixed-effects OLS estimates without instrumenting emissions and in (2) IV coefficient of emission. Standard errors are clustered at the NUTS 2 level. Year and region fixed effects are included. The coefficient on emission is small and statistically insignificant, consistent with attenuation bias due to measurement error.

Column (1) reports OLS estimates. The coefficient on emissions is small and statistically insignificant, suggesting small association between air pollution and infant health when emissions are not instrumented. This result is consistent with attenuation bias arising from measurement error in estimated emissions, as well as potential simultaneity between economic activity, traffic intensity, and health outcomes.

Column (2) presents IV estimates using the shift-share strategy. The coefficient increases substantially and becomes statistically significant, indicating that higher truck-related emissions raise respiratory admissions among infants. The stark contrast between OLS and IV estimates points to a downward bias in OLS estimates. These findings motivate the use of the IV strategy for the remainder of the analysis, which we adopt in all subsequent specifications.

5.2 Inference robustness

Table 5 examines whether inference about the effect of traffic-related air pollution depends on the way standard errors are computed. This issue is particularly important in a shift-share setting, where residual correlation within regions over time may lead conventional standard errors to understate true uncertainty.

Table 5: Sensitivity of IV-FE estimates to robust and clustered standard errors

	(1)	(2)	(3)
Emission	0.116***	0.116***	0.116**
	(0.030)	(0.030)	(0.053)
GDP per capita	-0.275***	-0.275**	-0.275
	(0.098)	(0.108)	(0.213)
Median Age	-0.505	-0.505	-0.505
	(0.458)	(0.412)	(0.644)
Stock Vehicles	-0.472***	-0.472***	-0.472***
	(0.084)	(0.080)	(0.131)
Number of Births	0.811***	0.811***	0.811***
	(0.101)	(0.087)	(0.164)
Observations	2,924	2,924	2,924
R-squared	0.05	0.05	0.05
Number of NUTS 2	199	199	199
Robust SE	No	Yes	Yes
Error clustered at NUTS 2 level	No	No	Yes
EF (c, t)	Yes	Yes	Yes

Note: The table reports 2SLS fixed-effects estimates of the impact of traffic-related emission on hospital admissions for respiratory causes among children under age 1. The endogenous regressor Emission is instrumented with the shift-share variables based on EURO emission standards. All specifications include NUTS 2-region fixed effects, year fixed effects, and control for log GDP per capita, log median age, log stock of lorries, and log cohort size. Column (1) reports conventional standard errors; column (2) reports robust standard errors (SE); column (3) reports robust standard errors clustered at the NUTS 2 level. ***, *, and * denote significance at the 1, 5, and 10 percent levels, respectively.

Columns (1)-(3) report the same IV-FE specification while successively moving from conventional standard errors, to heteroskedasticity-robust errors, and finally to standard errors clustered at the NUTS 2 level. Across all three specifications, the estimated coefficients on emission are identical in magnitude and remain statistically significant, although precision decreases when clustering is introduced.

We conduct several additional checks to assess the validity of the shift-share design and the robustness of the main results. First, we test for differential pre-trends by interacting pre-determined road exposure with a linear time trend in the pre-regulation period. The coefficient is mainly equal to zero and statistically insignificant, providing no evidence that more exposed regions followed different health trends prior to the introduction of EURO standards. This supports the exogeneity of the share component.

Secondly, we assess the sensitivity of the estimates to extreme hospital admissions by progressively excluding regions based on their number of hospital admissions. As expected, removing the top 1%, 5% or 10% of the distribution leaves the sign and magnitude of the estimated pollution effect largely unchanged, while maintaining precision. coefficients for emission remain positive and of a similar magnitude across all specifications. These are reported in Table 6. We also replicate this sensitivity verification for regions with extreme road density; the coefficients are given in Table A7 in the Appendix. The results are similar but slightly less precise.

Table 6: Sensitivity of IV estimates to regions with the most hospital admissions

	(1)	(2)	(3)	(4)
Outliers dropped	None	1%	5%	10%
Emission	0.116**	0.118**	0.118**	0.117**
	(0.053)	(0.055)	(0.055)	(0.055)
GDP per capita	-0.275	-0.305	-0.305	-0.335
	(0.213)	(0.224)	(0.224)	(0.233)
Median Age	-0.505	-0.163	-0.163	-0.063
	(0.644)	(0.664)	(0.664)	(0.682)
Stock Vehicles	-0.472***	-0.449***	-0.449***	-0.438***
	(0.131)	(0.143)	(0.143)	(0.146)
Number of Births	0.811***	0.830***	0.830***	0.829***
	(0.164)	(0.172)	(0.172)	(0.178)
Observations	2,924	2,879	2,749	2,646
R-squared	0.057	0.057	0.057	0.054
FE(r,t)	Yes	Yes	Yes	Yes
Number of NUTS 2	199	198	189	180

Note: This table reports the robustness of the main IV estimate to excluding regions with the most hospital admissions of infants. Columns (2)-(4) progressively remove regions in the top 1%, top 5%, top 10% of the distribution. All specifications use the full IV model with fixed effects, robust and NUTS 2-clustered standard errors. Coefficients on emission remain positive and broadly stable across specifications, indicating that results are not driven by a small number of extreme regions. ***, *, and * denote significance at the 1, 5, and 10 percent levels, respectively.

Third, we also verify that the instrument does not significantly predict either the number of births or GDP per capita. To do so, we replace emission in the model by the shift-share instrument, and the coefficient were insignificant. This suggests that it operates through emissions rather than broader economic channels.

Fourth, we assess the sensitivity of our results to differential long-run trends across NUTS 2 regions. Allowing for NUTS 2-specific linear time trends proves too demanding for our shift-share design, as it absorbs most of the identifying time variation and leads to under-identification. We therefore consider country-specific linear time trends as an alternative robustness check. Results remain positive and of similar magnitude, though less precisely estimated. The corresponding estimates are reported in the Electronic Supplementary Material (Table 1).

5.3 Instrument choice robustness

A key concern in shift-share designs is whether the estimated effect is driven by a specific policy episode or by the cumulative structure of the instrument. To address this concern, Table 7 compares our baseline IV specification using the full set of EURO-based instruments (EURO III, IV, and V) with a more conservative specification relying solely on the earliest regulatory shock (EURO III).

Table 7: Robustness of IV estimates: Full Instrument set vs. Conservative Instrument

	Full set (1)	One Instrument (2)
Emission	0.116**	0.079*
	(0.053)	(0.043)
GDP per capita	-0.275	-0.200
	(0.213)	(0.179)
Median Age	-0.505	-0.415
	(0.644)	(0.623)
Stock Vehicles	-0.472***	-0.398***
	(0.131)	(0.110)
Number of Births	0.811***	0.776***
	(0.164)	(0.146)
Observations	2,924	2,924
R-squared	0.05	0.024
EF (c, t)	Yes	Yes
Number of NUTS 2	199	199

Note: This table compares two instrumental-variable specifications. Column (1) uses the full set of excluded instruments (shift-share for EURO III, EURO IV and EURO V), while Column (2) uses only the most conservative instrument (shift-share for EURO III). All models include NUTS 2 fixed effects, year fixed effects, and the full set of controls. Standard errors are clustered at the NUTS 2 level. *, **, *** denote significance at the 10%, 5%, and 1% levels, respectively.

Column (1) reports results using the full instrument set, while Column (2) instruments emissions exclusively with the shift-share instrument which captures the introduction of EURO III standards. This first regulatory tightening constitutes the most conservative instrument for two reasons. First, it is temporally distant from later policy revisions and therefore minimizes concerns related to anticipation effects or endogenous regulatory responses. Second, it precedes much of the broader environmental policy agenda in Europe, reducing the risk that it correlates with other contemporaneous health or environmental interventions.

The estimated effect of traffic-related emissions on infant respiratory admissions remains positive and statistically significant in both specifications. The magnitude of the coefficient is stable across columns, ranging from 0.08 to 0.12, indicating that the estimated effect is not driven by a single regulatory episode. As expected, using the full instrument set improves statistical precision by exploiting additional exogenous variation in emissions generated by later EURO standards. These results confirm that the signal is robust to alternative instrument choices and does not rely on any specific policy shock.

Next, we further assess the sensitivity of our IV estimates to potential violations of the exclusion restriction using the plausibly exogenous instrument framework of Conley et al. (2012). Instead of assuming that the EURO-based shift-share instrument affects health outcomes exclusively through particulate emissions, this approach allows for a small direct effect of the instrument on hospital admissions. Allowing for violations of up to 25% of the reduced-form effect, the resulting union of confidence intervals for the emission coefficient is [0.006, 0.268]. This interval remains largely positive and economically meaningful, implying that substantial violations of the exclusion restriction would be required to overturn our baseline IV results.

5.4 Extension to other health variables

In this section, we aim to underline interpretation of the main results and to assess the robustness and plausibility of the estimated average effect. These results reinforce the interpretation that traffic-related emission has a meaningful impact on infant health.

A key advantage of our setting is the ability to assess whether the estimated effects of traffic-related emission vary across population groups with different biological vulnerability. In particular, infants under one year of age are known to be especially sensitive to air pollution due to lung immaturity, higher breathing rates, and limited capacity for avoidance behavior. Table A6, in the Appendix, reports IV fixed-effects estimates separately for three age groups: infants aged 0-1, children aged 1-4, and the overall population aged 0-65. The results reveal a clear pattern of heterogeneity. Emissions have a positive and statistically significant effect on respiratory hospital admissions for infants under one year old, with a coefficient of 0.116. In contrast, the estimated effects are small and statistically insignificant for both older children

(1-4 years) and the broader population. This concentration of effects among infants supports a biological interpretation of the results. If the estimated relationship reflected changes in hospital utilization behavior, reporting practices, or health system congestion, similar effects would be expected across child age groups. Instead, the absence of effects for less vulnerable populations suggests that the estimated impact operates through genuine health deterioration among the most at-risk individuals.

Table A8, in the Appendix, extends the baseline analysis beyond hospital admissions to examine whether traffic-related emission also affects the birth mortality rate. Column (1) restates the baseline effect on infant respiratory admissions, showing a positive and significant impact of emissions. Column (2) considers infant mortality,[38] a rare but clinically severe outcome that is less influenced by hospital capacity or admission practices. The estimated coefficient remains positive, with a magnitude similar to admissions (0.069 vs. 0.116), although it is less precisely estimated (p-value = 0.11), which is expected given the relative rarity and multifactorial nature of infant mortality. Importantly, the mortality results do not contradict the main findings and instead suggest consistency across distinct health outcomes.

6 The Health Consequences of the Truck Cartel

This section examines the health consequences of excess emissions resulting from the delayed implementation of cleaner technologies, caused by the anti-competitive agreement of truck manufacturers. Using our estimate of the effect of $PM_{2.5}$ emissions on infant respiratory hospital admissions, we simulate alternative health trajectories under competitive scenarios corresponding to our two counterfactuals and compare the resulting outcomes with the observed data. This enables us to quantify the number of hospital admissions associated with the cartel-induced excess emissions.

6.1 Counterfactual emissions under early adoption scenarios

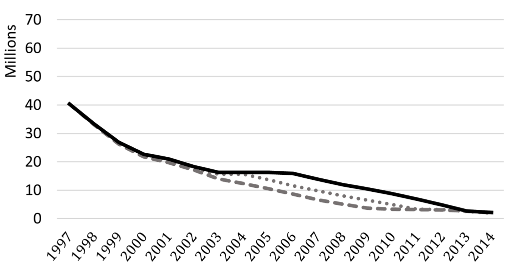
$PM_{2.5}$ emissions are recalculated using alternative adoption dates,[39] while maintaining the original methodology (see Section 3.1.3). This procedure yields the level of emissions that would have been avoided had the EURO standard been implemented according to the earlier

[38] The variable 'demo_r_minfind' comes from the Population change database (Eurostat).

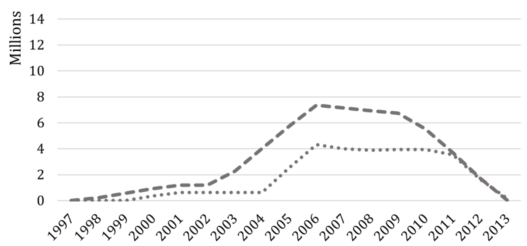
[39] See Appendix C for a detailed presentation of the emission calculations under the alternative scenarios.

dates proposed by the EEA and the EC. The difference between actual emissions (P^C) and counterfactual emissions (P^{EC} or P^{EEA}) thus provides a plausible estimate of the excess emissions attributable to the cartel.

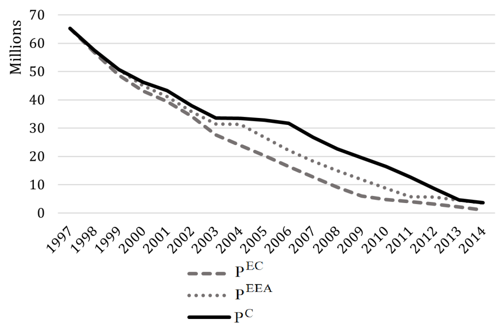
Table A9, in Appendix A, reports the P^C, P^{EC}, and P^{EEA} emission volumes for PM2.5, together with their breakdown by truck category. The last column shows the resulting excess emissions. Summing the emissions from both vehicle segments indicates that the cartel generated additional PM2.5 emissions ranging from 83.81 to 155.33 thousand tonnes (with a midpoint of 119 thousand tonnes).



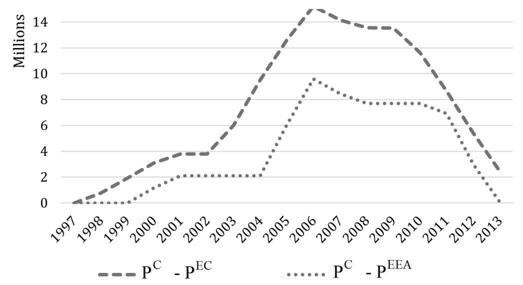
(a) Emission of $PM_{2.5}$ by scenario for CV



(b) Emission Difference between the Status Quo and Alternative Scenarios for CV



(c) Emission of $PM_{2.5}$ by scenario for HCV



(d) Emission Difference between Status Quo and Alternative Scenarios for HCV

Figure 3: Estimated quantity of $PM_{2.5}$ (in tonnes) emitted during the cartel agreement and the alternative Scenarios

The three scenarios differ markedly in the total air pollution emitted by the truck fleet, reflecting the distinct timelines for introducing vehicles complying with successive EURO emission standards. Figure 3 compares the emission trajectories across scenarios by truck weight (Figures 3a and 3c) and highlights the gaps between each counterfactual scenario and the factual situation (Figures 3b and 3d).

While the underlying fleet composition differs across scenarios, the key takeaway is that relatively modest changes in the timing of adoption of newer EURO standards substantially reduce emissions, reflecting the sharply declining emission limits across successive standards.[40]

Additionally, the emissions effect of the cartel extends beyond its formal end in 2010. Given the average truck lifespan of 6.8 years (see Section 3.1.1), decisions made during the cartel period continued to affect air quality until the end of 2017 at least.

6.2 Health implications of cartel-induced excess emissions

This subsection combines the counterfactual emission volumes derived in Section 6.1 with the emission-health elasticities estimated in Section 5. As such, the resulting health impacts should be interpreted as conditional on the estimated coefficient and the maintained assumptions underlying the empirical model. To do so, we first compute, for each region and year, the share of emissions attributable to the cartel under each counterfactual scenario:

$$\%Excess^{j}_{r,t} = \left(1 - \frac{P^{j}_{r,t}}{P^{C}_{r,t}}\right) \quad \text{where} \quad j \in \{EC, EEA\} \tag{4}$$

We then multiply the predicted number of respiratory hospital admissions ($H\hat{o}sp_{r,t}$), based on column (3) of Table 5 by the cartel-induced excess-emission share to obtain the number of admissions associated with it:

$$NH^{j}_{\text{Cartel},r,t} = H\hat{o}sp_{r,t} \times \%Excess^{j}_{r,t} \quad \text{for} \quad j \in \{EC, EEA\} \tag{5}$$

While the implied aggregate number of avoidable admissions over the 2000-2018 period is large, its interpretation is more informative when expressed relative to cohort size. On average, traffic-related $PM_{2.5}$ emissions are associated with 72.5 respiratory hospital admissions per 1,000 births per year at the NUTS 2 level. Under the counterfactual scenarios of earlier competitive adoption, this figure would have been reduced to around 60 admissions per 1,000 births in the EEA scenario and to about 54 admissions per 1,000 births in the EC scenario.

Figure 4 illustrates these dynamics by plotting the annual average admission rates per 1,000 births associated with observed emissions and with the two counterfactual scenarios. The narrowing gap over time reflects the progressive implementation of EURO standards and the associated decline in excess emissions.

[40] The distribution of truck sales by EURO standard and the corresponding stringency differences are shown in Appendix Figures B5 and B4.

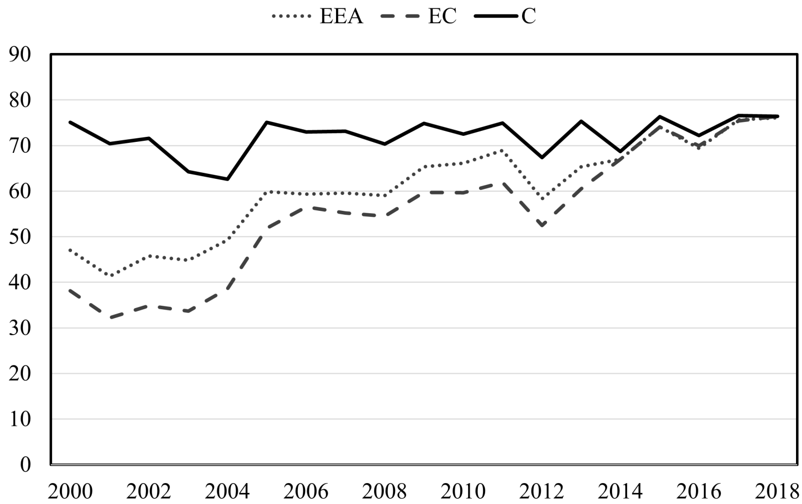


Figure 4: Estimated number of infant respiratory admissions per 1,000 births under our emission scenarios for 199 European NUTS 2 regions between 2000-18

To complement this Figure, Table A10 in the Appendix reports the annual average number of infant respiratory hospital admissions per 1,000 births that can be associated to cartel-induced excess emissions, aggregated at the country level under the two counterfactual scenarios. Averaged across countries, the estimates amount to 137.69 admissions per 1,000 births per year under the EEA scenario and 235.25 admissions per 1,000 births per year under the EC scenario. When viewed over the full 2000-2018 sample period, these figures provide an order-of-magnitude indication of the potential cumulative health impacts associated with the cartel-induced delay in adopting cleaner technologies. The table also highlights substantial cross-country heterogeneity, with larger values observed in countries more exposed to heavy-duty vehicle traffic.

7 Conclusion

This article examines the public health consequences of the collusion among truck manufacturers to delay the deployment of cleaner engine technologies. Our analysis suggests that, in addition to traditional anticompetitive harm, significant societal costs should be considered when evaluating the economic impact of a cartel. In this paper, we find that the cartel's actions resulted in up to 119 thousand tonnes of $PM_{2.5}$. We applied a fixed-effects panel analysis and estimated that, in a competitive situation, between 12 and 18 hospital admissions for respiratory disease per 1,000 births at NUTS 2 level could have been avoided.

While striking, these figures should be interpreted with caution. Some limitations of our approach deserve to be highlighted. First, the counterfactual scenarios rely on two key assumptions regarding the speed at which firms would have adopted cleaner technologies under competitive conditions and the assumption of a constant number of trucks sold. Although grounded in institutional evidence and technological feasibility, these assumptions cannot be directly validated. Accordingly, these scenarios should not be interpreted as uniquely identified predictions of competitive equilibrium outcomes. Rather, they represent plausible paths of early adoption under competition. Consequently, the calculated exceeding emissions and health consequences can be interpreted as upper bound estimates of the environmental and public-health damages associated with the cartel-induced excess emissions. Second, the emission calculations draw on average emission factors and a gridding procedure that allocates national emissions to regions based on highway length. This inevitably abstracts from heterogeneity in traffic intensity, driving patterns, and non-exhaust emissions. Third, our health analysis focuses on $PM_{2.5}$-related infant respiratory hospital admissions. We do not account for other diseases (e.g., cardiovascular conditions), even though a growing literature highlights the multiple pathways through which $PM_{2.5}$ affects health. We also focus only on immediate health effects. As a result, the health burden we quantify represents only a subset of the overall impact.[41] Finally, as in any shift-share design, the identifying assumption cannot be directly tested, and the estimates should be interpreted as causal under the maintained assumption that no other contemporaneous shocks interact systematically with pre-existing road infrastructure.

Despite these limitations, the findings underscore the importance of recognizing the environmental and health consequences of anticompetitive practices and call for a deeper integration of such concerns into competition policy debates. The case we study highlights a situation in which the authorities' decision was based solely on economic damage, even though manufacturers, while formally respecting the EURO standards timetable, coordinated to delay the implementation of cleaner technologies. This behaviour remained outside the remit of environmental regulators but nonetheless generated avoidable emissions and associated health costs. Our analysis contributes to this broader research agenda by documenting one concrete channel through which anticompetitive conduct can have measurable health consequences. We hope it will stimulate further work at the interface of industrial

[41] More broadly, air pollution has consequences that extend beyond health outcomes, including productivity losses (Kögel 2022), adaptation behaviours, and impacts on agricultural yields and ecosystems, which are well documented in the broader literature on air pollution.

organization, environmental economics, and competition policy.

Appendices

A Additional tables

Table A1: Summary of fines imposed on the European truck cartel

	Reduction under the Leniency Notice (%)	Reduction under the Settlement Notice (%)	Fine (thousand euros)
MAN	100	10	0
Volvo/Renault	40	10	670,448
Daimler	30	10	1,008,766
Iveco	10	10	494,606
DAF	0	10	752,679
Scania	0	0	880,523
		Total	3,807,022

Source: Case AT.39824 Commission decision of July 19, 2016.

Table A2: Emission limits for commercial vehicles imposed by EURO standards

Standard	Reference texts	Application date	NOx (g/kWh)	CO (g/kWh)	HC (g/kWh)	$PM_{2.5}$ (g/kWh)
EURO 0	88/77	01/10/1990	14.4	11.2	2.4	-
EURO I	91/542 (A)	01/10/1993	9	4.9	1.23	0.36
EURO II	91/542 (B)	01/10/1996	7	4	1.1	0.15
EURO III	1999/96	01/10/2001	5	2.1	0.66	0.13
EURO IV	1999/96 2005/13/CE 2005/55/CE	01/10/2006	3.5	1.5	0.46	0.02
EURO V	1999/96 2005/13/CE 2005/55/CE	01/10/2009	2	1.5	0.46	0.02
EURO VI	Regulation (EC) N 595/2009	31/12/2013	0.4	1.5	0.13	0.01

Table A3: Implementation dates of the European emission standards (EEA, 2019)

Vehicle category	Type	EURO Standard	Start Date	End Date
Heavy Duty	All Diesel	EURO II	1996	2000
Trucks		EURO III	2000	2005
		EURO IV	2005	2008
		EURO V	2008	2013
		EURO VI	2013	2019

Note: This table −an extract from Table 2 − 2 of the "Air pollutant emission inventory guidebook 2019 (EEA)"− shows a summary of the date of implementation of the directives for trucks to be equipped with emissions reduction technologies. Vehicles complying with Economic Commission for Europe and earlier are all classified as 'conventional'. Directive 91/542/EEC, implemented in two stages, brought in two sets of reduced emission limits, valid from 1992 to 1995 (Stage 1 − EURO I) and from 1996 to 2000 (Stage 2 − EURO II). Directive 1999/96/EC Step 1 (EURO III) was valid from 2000, and introduced a 30% reduction of all pollutants relative to EURO II. The same Directive included an intermediate step in 2005 (EURO IV), and a final step in 2008 (EURO V). The EURO V standards are very strict, requiring a reduction in NOx of more than 70% and a reduction in $PM_{2.5}$ of more than 85% compared with the EURO II standards. This was achieved with engine tuning and oxidation catalysts for $PM_{2.5}$ emissions, and selective catalytic reduction for NOx emissions. The most recent emission limits, at the EURO VI level, have been enforced since 2013/14.

Table A4: Estimation of the $PM_{2.5}$ emission volume (tonnes) from a truck over its lifetime based on weight

Pollutant (in tonnes)	EURO II	EURO III	EURO IV	EURO V	EURO VI
			CV (3.5-16 T)		
	0.0638	0.056	0.0103	0.0103	0.0005
			HCV (>16 T)		
	0.135	0.1087	0.0196	0.0196	0.001

Note: Calculation from data in Table 1 and information provided by the CNR Long Distance Report (2009). This report contained information about a truck's lifetime, annual mileage in kilometers, and average speed on national roads.

Table A5: First-Stage estimates

Instrument	Coefficient	t-stat	p-value
Shift-Share for EURO III	0.319 (0.145)	2.19	0.028
Shift-Share for EURO IV	0.084 (0.098)	0.85	0.394
Shift-Share for EURO V	-0.079 (0.084)	-0.94	0.347
Control	Yes		
Year FE	Yes		
Region FE	Yes		
Clusters (NUTS 2)	199		
Observations	2924		

Table A6: Heterogeneous effects of traffic-related emission on hospital admissions by age group

	(1)	(2)	(3)
Ages:	0-1 YO	1-4 YO	0-65 YO
Emission	0.116**	-0.025	-0.052
	(0.053)	(0.063)	(0.044)
GDP per capita	-0.275	0.360*	0.350
	(0.213)	(0.187)	(0.234)
Median Age	-0.505	-0.759	-0.421
	(0.644)	(0.943)	(0.562)
Stock Vehicles	-0.472***	-0.318***	-0.265**
	(0.131)	(0.111)	(0.110)
Number of Births	0.811***	0.319*	0.168
	(0.164)	(0.186)	(0.168)
Observations	2,924	2,052	2,640
R-squared	0.05	0.322	0.071
EF (c, t)	Yes	Yes	Yes
Number of NUTS 2	199	154	180

Note: All regressions are estimated using 2SLS with region fixed effects, year fixed effects, and clustered standard errors at the NUTS 2 level. The endogenous variable Emission is instrumented using the full set of shift-share instruments. Column (1) corresponds to infants aged 0-1, column (2) to children aged 1-4, and column (3) to the full population aged 0-65. *, **, *** denote significance at the 10%, 5%, and 1% levels, respectively.

Table A7: Sensitivity of IV estimates to extreme road density

	(1)	(2)	(3)	(4)
Outliers dropped	None	1%	5%	10%
Emission	0.116**	0.101**	0.081*	0.072
	(0.053)	(0.048)	(0.035)	(0.032)
GDP per capita	-0.275	-0.240	-0.116	-0.071
	(0.213)	(0.195)	(0.148)	(0.140)
Median Age	-0.505	-0.579	-0.770	-0.641
	(0.644)	(0.638)	(0.641)	(0.672)
Stock Vehicles	-0.472***	-0.452***	-0.432***	-0.440***
	(0.131)	(0.122)	(0.107)	(0.106)
Number of Births	0.811***	0.781***	0.723***	0.700***
	(0.164)	(0.158)	(0.140)	(0.139)
Observations	2,924	2,879	2,749	2,646
R-squared	0.05	0.018	0.049	0.059
FE(r,t)	Yes	Yes	Yes	Yes
Number of NUTS 2	199	198	189	180

Note: This table reports the robustness of the main IV estimate to excluding regions with extreme road density. Columns (2)-(4) progressively remove regions in the top 1%, top 5%, top 10% of the density distribution. All specifications use the full IV model with fixed effects, robust and NUTS 2-clustered standard errors. The coefficient on Emission remains positive and broadly stable across specifications, indicating that results are not driven by a small number of extreme regions. ***, *, and * denote significance at the 1, 5, and 10 percent levels, respectively.

Table A8: Emission effects across alternative health outcome

	(1)	(2)
Dep. variable:	Hospital Admission	Birth Mortality
Emission	0.116**	0.069 .
	(0.053)	(0.045)
GDP per capita	-0.275	-0.421***
	(0.213)	(0.158)
Median Age	-0.505	0.274
	(0.644)	(0.433)
Stock Vehicles	-0.472***	-0.246*
	(0.131)	(0.127)
Number of Births	0.811***	0.647***
	(0.164)	(0.123)
Observations	2,924	2,924
R-squared	0.05	0.203
EF (c, t)	Yes	Yes
Number of NUTS 2	199	199

Note: This table reports 2SLS fixed-effects estimates of the impact of truck-related particulate emissions on one alternative health outcome. Column (1) shows respiratory hospital admissions among infants under age one,and column (2) shows infant mortality. Emissions are instrumented using shift-share variables based on EURO emission standards. All models include NUTS 2 and year fixed effects and control for log GDP per capita, log median age, log lorry stock, and log births. Standard errors are clustered at the NUTS 2 level. ' . ', *, **, *** denote significance at the 15%, 10%, 5%, and 1% levels, respectively.

Table A9: Estimated $PM_{2.5}$ emissions by truck weight

Pollutant (thousand tonnes)	P^C	P^{EC}	P^{EEA}	Excess emissions
	CV (3.5-16 T)			
	175.861	115.096	142.941	[32.920; 60.765]
	HCV (>16 T)			
	249.815	155.250	198.924	[50.891; 94.565]
Total $PM_{2.5}$	**425.676**	**270.346**	**341.865**	**[83.811; 155.330]**

Note: Excess emissions are derived from estimated emissions under each scenario relative to the cartel case. The lower bound of the interval represents the optimistic, or less severe, scenario and corresponds to the first appearance of vehicles equipped with compliant technology, as documented in the EEA report. This was calculated using the formula $P^C - P^{EEA}$. Conversely, the upper bound reflects the worst-case scenario and corresponds to the technology's initial availability, as reported by the Commission's investigation findings regarding the truck cartel. The excess was calculated using $P^C - P^{EC}$.

Table A10: Annual average number of infant respiratory hospital admissions per 1,000 births attributable to cartel-induced excess emissions under *EEA* and *EC* counterfactuals

Country	EEA	EC
Autria	121.02	217.67
Belgium	247.40	553.83
Switzerland	58.65	131.52
Cyrpus	8.46	15.42
Czech Republic	136.00	223.43
Germany	432.15	844.95
Danemark	30.16	62.15
Espagna	227.79	403.58
Finland	95.38	137.72
France	273.50	474.75
Croatia	11.85	22.76
Hungary	103.94	215.04
Irland	48.17	113.07
Italie	261.72	454.94
Liechtenstein	0.00	0.00
Lithuania	82.02	192.87
Luxembourg	3.73	6.69
Malta	0.00	0.00
Netherlands	64.48	131.81
Poland	596.57	979.03
Romania	235.71	387.67
Slovakia	60.11	103.45
Slovenia	68.00	152.31
All	137.69	253.25

Reading guide: Each entry reports the estimated annual average number of infant hospital admissions per 1,000 births attributable to cartel-induced excess emissions in a given country. For example, the value 121.02 in the EEA column for Austria indicates that, under the counterfactual scenario in which adoption occurs at the first observed introduction by a non-cartel manufacturer, the cartel-induced delay is associated with approximately 121 additional infant hospital admissions per 1,000 births per year in Austria. The corresponding value of 217.67 under the EC scenario reflects a more conservative counterfactual in which clean technologies are adopted as soon as they become technically available.

B Additional figures

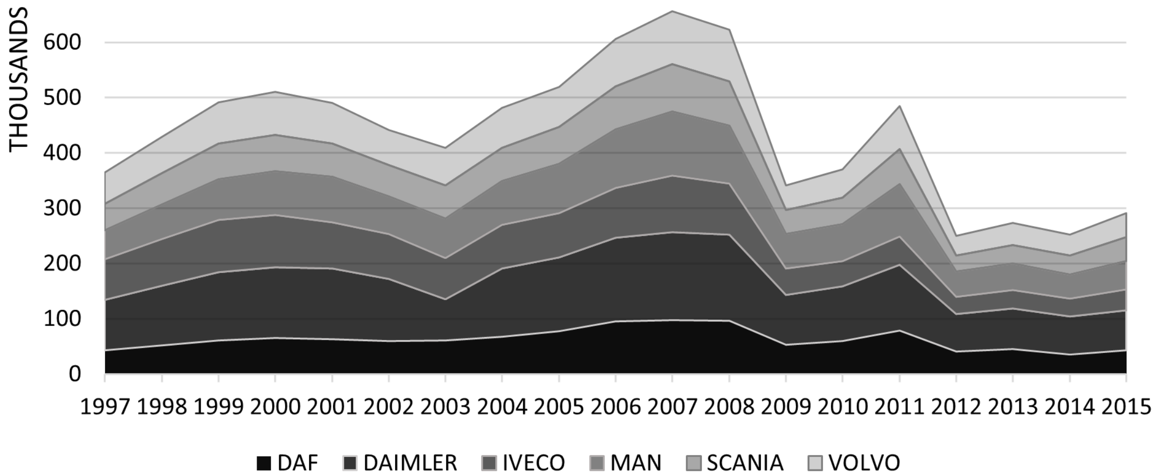


Figure B1: Registrations by manufacturer during the entire lifespan of the truck cartel, spanning from 1997 to 2015 (Source: ACEA)

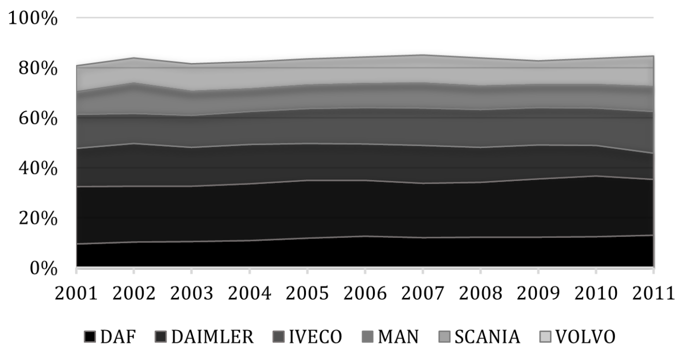


Figure B2: Market share distribution by manufacturer over the entire cartel's life on the segment between 3.5 tonnes and 16 tonnes. Source: ACEA

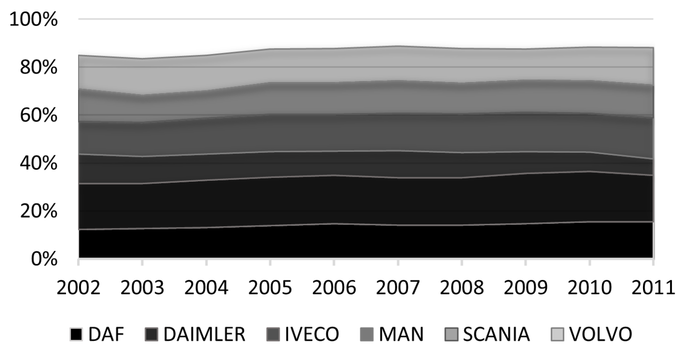


Figure B3: Market share distribution by manufacturer during the entire cartel's life on the segment over 16 tonnes. Source: ACEA

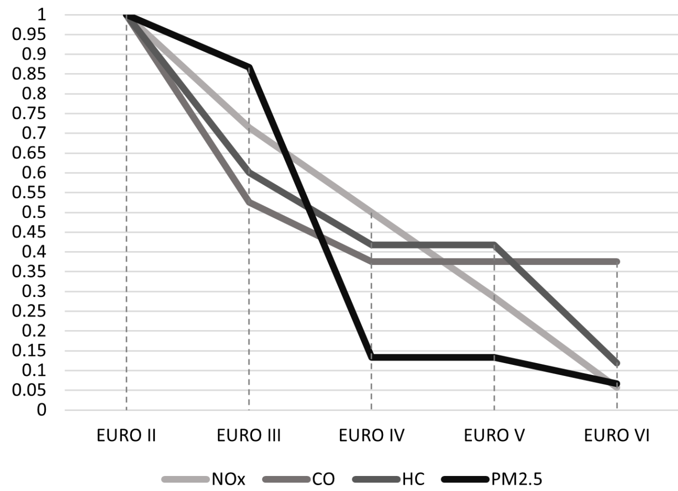


Figure B4: EURO Standard limits on the relative to EURO II

Notes: This figure shows the level of emission restriction under the EURO environmental standards relative to standard II (the standard in force at the start of the cartel). We can see that the level of restriction is not always proportional and can be drastically reduced from one standard to another.

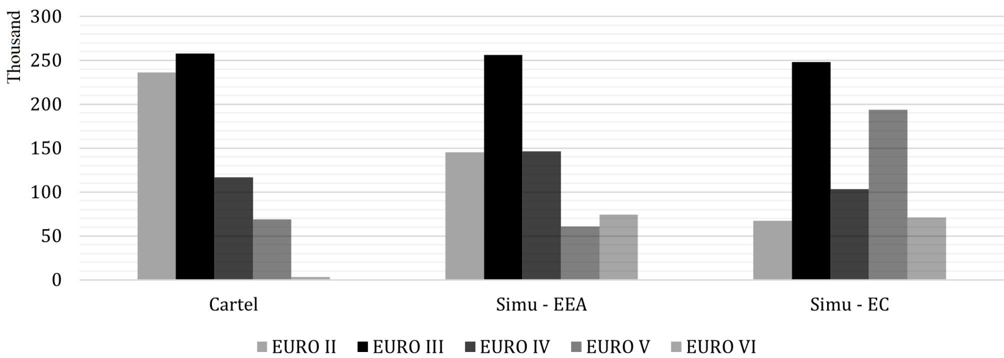


Figure B5: Distribution of total heavy goods vehicle sales over the duration of the cartel by EURO standards, comparing actual outcomes with simulated scenarios

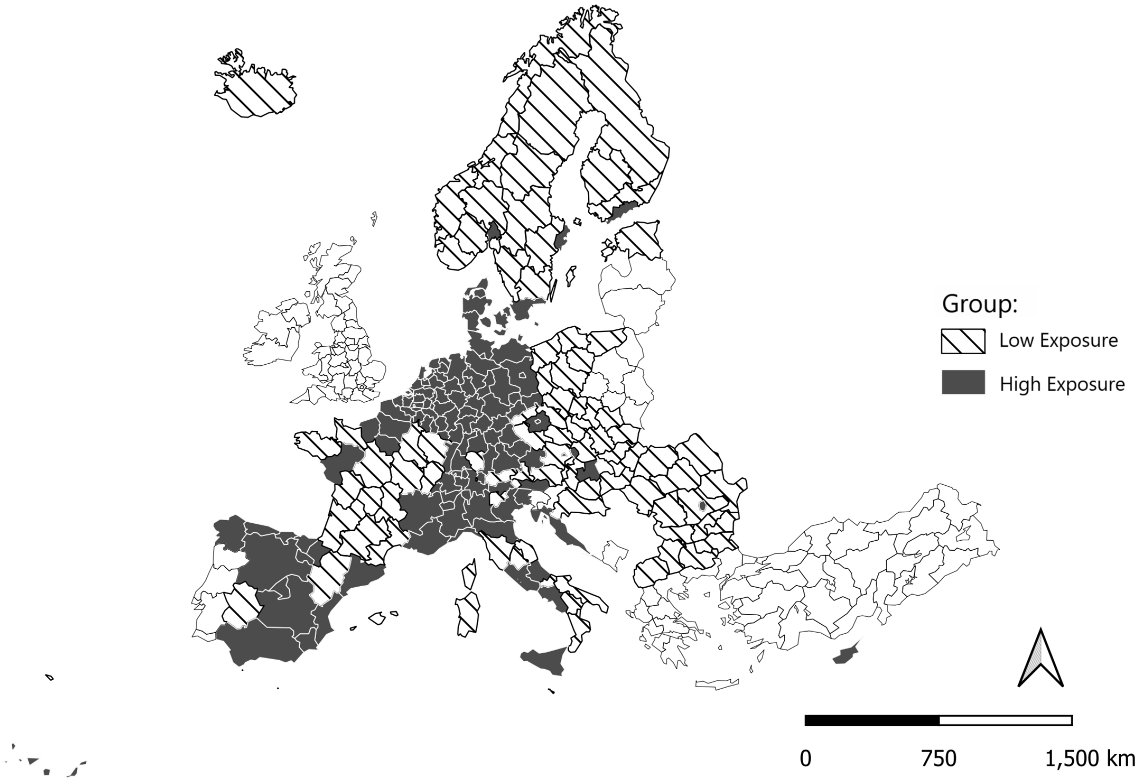


Figure B6: Distribution of exposed and non-exposed NUTS 2 regions in Europe

C Computation of total emission volume under EC and EEA scenarios

We compute the total emission volume for each scenario (P^C, P^{EC}, and P^{EEA}) using the same formula. The specific ranges of m (months) and i (EURO standards) for each scenario are defined separately. The total emission volume is calculated as:

$$
\begin{aligned}
P^C = & \sum_{m=0}^{45}\sum_{i=0}^{II}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=46}^{56}\sum_{i=I}^{II}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=57}^{81}\sum_{i=I}^{III}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=82}^{116}\sum_{i=II}^{III}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=117}^{141}\sum_{i=II}^{IV}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=142}^{155}\sum_{i=III}^{IV}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=154}^{201}\sum_{i=III}^{V}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=202}^{212}\sum_{i=IV}^{V}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=213}^{238}\sum_{i=IV}^{VI}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV}) \\
& + \sum_{m=239}^{249}\sum_{i=V}^{VI}(N_{im}^{HCV}E_i^{HCV} + N_{im}^{CV}E_i^{CV})
\end{aligned}
\tag{6}
$$

where:

- N_{im}^{HCV} and N_{im}^{CV} are the numbers of heavy commercial vehicles (HCV) and commercial vehicles (CV) in month m that comply with EURO standard i.
- E_i^{HCV} and E_i^{CV} represent the emission factors for HCV and CV, respectively, under EURO standard i.

The ranges of m and i differ across scenarios (P^C, P^{EC}, and P^{EEA}) and are specified in Tables (C1), (C2), and (C3), respectively.

Range of m	Range of i
$0 \leq m \leq 14$	$0 \leq i \leq II$
$15 \leq m \leq 45$	$0 \leq i \leq III$
$46 \leq m \leq 74$	$I \leq i \leq III$
$75 \leq m \leq 81$	$I \leq i \leq IV$
$82 \leq m \leq 99$	$II \leq i \leq IV$
$100 \leq m \leq 141$	$II \leq i \leq V$
$142 \leq m \leq 151$	$III \leq i \leq V$
$152 \leq m \leq 159$	$III \leq i \leq VI$
$160 \leq m \leq 184$	$IV \leq i \leq VI$
$185 \leq m \leq 236$	$V \leq i \leq VI$
$237 \leq m \leq 249$	$i = VI$

Table C1: Ranges of m and i for P^C.

Range of m	Range of i
$0 \leq m \leq 14$	$0 \leq i \leq II$
$15 \leq m \leq 45$	$0 \leq i \leq III$
$46 \leq m \leq 81$	$I \leq i \leq III$
$82 \leq m \leq 95$	$II \leq i \leq III$
$96 \leq m \leq 120$	$II \leq i \leq IV$
$121 \leq m \leq 131$	$III \leq i \leq IV$
$132 \leq m \leq 180$	$III \leq i \leq V$
$181 \leq m \leq 191$	$IV \leq i \leq V$
$192 \leq m \leq 216$	$IV \leq i \leq VI$
$217 \leq m \leq 249$	$V \leq i \leq VI$

Table C2: Ranges of m and i for P^{EC}.

Range of m	Range of i
$0 \leq m \leq 35$	$0 \leq i \leq II$
$36 \leq m \leq 45$	$0 \leq i \leq III$
$46 \leq m \leq 81$	$I \leq i \leq III$
$82 \leq m \leq 95$	$II \leq i \leq III$
$96 \leq m \leq 120$	$II \leq i \leq IV$
$121 \leq m \leq 131$	$III \leq i \leq IV$
$132 \leq m \leq 180$	$III \leq i \leq V$
$181 \leq m \leq 191$	$IV \leq i \leq V$
$192 \leq m \leq 216$	$IV \leq i \leq VI$
$217 \leq m \leq 249$	$V \leq i \leq VI$

Table C3: Ranges of m and i for P^{EEA}.

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