UNIVERSITAT DE BARCELONA

Essays on Mobility and Environmental Policies in the Metropolitan Area of Barcelona

Jordi Rosell i Segura

PhD in Economics

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Agraïments

Aquesta tesi no hagués estat possible sense una pila de gent que m’ha ajudat, recolzat i animat, en tots els sentits, tant a la universitat com a casa. A tots ells vull dedicar-los aquest treball i, sobretot, agrair tot el que han fet per mi.

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quantitat i qualitat de punts de vista que poden tenir les petites coses. Una tesis sense aquestes petites visions mai és una tesis.

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CHAPTER 1

INTRODUCTION

1 Agglomerations: mobility and environment

Europe's cities – home to 70% of the EU population and generating over 80% of the Union's GDP - are connected by one of the world's best transport systems (European Commission Regional Policy, 2011). However, urban mobility is still heavily reliant on the use of conventionally-fueled private cars in some cities; only slow progress is being made in shifting towards more sustainable modes of urban mobility. Besides, in the European Union, transport is responsible for around a 25% of GHG emissions.

In the developed cities there is a key difference between the European and US traditions. In European cities there is a history of strategic planning and control of the land development process from the national to the local level, with clear priorities, responsibilities and controls. This strong planning environment has allowed cities to retain many of their historic structures and suburbanization has also been controlled; this has in turn meant that public transport has had a stronger role to play. In the US, many of the local zoning regulations restrict density levels and the mixing of land uses, and this presents a significant barrier to compact development (TRB, 2009). After all, how citizens organize themselves on the ground through institutions is the key factor determining how much fossil fuel they burn.

In Europe, there are no mega-cities, the possible exception being London, where the city region has a population of 15 million (25% of the UK population) and 40% of GDP. The key concerns here are quality and sustainability that cover equality of opportunity, and access to services and facilities, as well as high environmental standards. Here, the growth rates are modest and there are strong governance structures that encourage order, priority for people, a polycentric urban form (London is a city of villages), and the full integration of land use and transport (Banister, 2011).

National, regional, and local governments implement mobility policies on their territories. They combine different sets of policies to pursue more sustainable economic growth. The potential role of institutional structures to redirect mobility policies towards the mitigation of climate change are important, jointly with a broader governance environment and the resources and politics involved in transport policy (Marsden and Groer, 2016). Haarstad (2015) points to urban mobility as the intersection of a particular technology, actors and a site-specific infrastructure. These aspects entangle urban mobility in the global trajectories of technological development, regional cultures shaping what is regular transport behavior, and local histories of city-building and land use.

An efficient equilibrium is defined as a situation in which marginal social costs are equal to marginal social benefits. If the price is set equal to the marginal social benefits, we will get a socially efficient amount of improvement. So negative externalities are a form of market failure, which means that the market is incapable of reaching an efficient equilibrium. Negative externalities in the road transport sector include congestion, air pollution, accidents, noise and climate change effects. To correct these externalities, policy-makers rely on different instruments, such as taxes, subsidies, tradable permits or regulation, among others. Designing mobility policies should contribute to decreasing these externalities.

2 Barcelona metropolitan area and mobility-related characteristics

The Barcelona metropolitan area is home to 3 million people and occupies a surface area of 636 km^2. This makes it one of the largest metropolitan areas in Europe, while it ranks eighth in terms of population. The metropolitan area presents two forms of urbanization and land use. In the metropolitan core, Barcelona and its neighboring municipalities constitute a continuous, compact urban area with mixed urban functions, whereas its periphery is characterized by low population density, a less compact urban area and a less specialized urban function by zone. Population density decreases as we move out from Barcelona and away from the highways (Garcia-López, 2010), whereas housing affordability increases with greater distance from the Barcelona city center. Only 20% of inhabitants rent their dwellings; the others are owner-occupiers. There is low residential mobility for purposes of work.

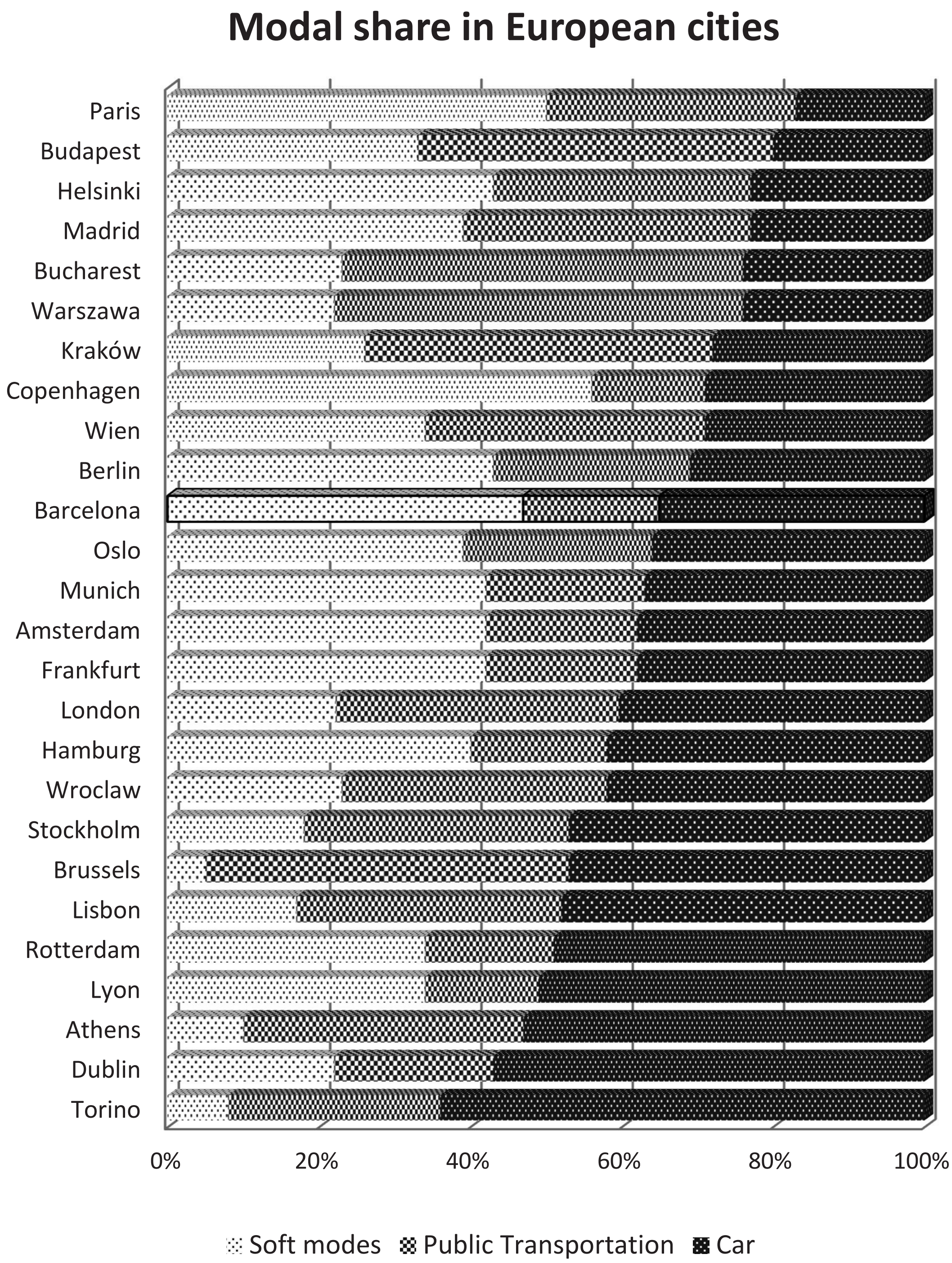
The metropolitan area is structured around a radial transport network, with the principal agglomerations of population and employment connected to the center of Barcelona by means of railway lines and metropolitan highways. There are nine motorway access roads to the inner city. Three of these are tolled, and in one case there is a toll-free alternative route. The other six motorways are toll-free. Tolls are fixed by taking various financial factors into account, including the concessionaires' investment, maintenance expenses and the recovery of other costs. Environmental considerations played no role in the tolls charged some years ago, although discounts are in force nowadays. Overall, the main factors accounting for the mobility carbon footprint in Barcelona are population density and accessibility, while the average municipal family income and job ratio are not as important (Muñiz and Galindo, 2005). Public transport fares are not linked to distance and most of the population lives in the same fare zone, with a very small percentage living in the second fare zone. A single ticket allows passengers to switch from one mode of transport to another to reach their required destination.

Each municipality regulates its own parking policies. However, land use parking policies are the responsibility of the broader territorial entity, the *Àrea Metropolitana de Barcelona* (AMB). In the case of all new housing developments, one parking space must be provided for every apartment unit until nowadays. Curbside parking regulations are implemented in most cases in the city centers of each municipality. Barcelona has operated curbside parking since the 1980s. This has expanded over the years, and today is combined with residential permits (Gragera and Albalate, 2016). In the metropolitan area as a whole, off-street parking amounts to two-thirds of total supply – split between private (2/3) and public (1/3) access. The remaining third is made up of curbside and 'free' on-street parking.

In a comparison between European cities, Barcelona has a good position in low emitter transport modes, since soft modes and public transport account for 64% of total travels (Figure 1). However, one of the characteristics of Barcelona is a high density of cars in the main city (Figure 2). While the number of cars per inhabitant is consistent with the average of the large European conurbations, on the geographical side, Barcelona is one of the leaders in numbers of cars per unit of surface. Despite the high density of vehicles, Barcelona does not lead the rankings of European urban congestion areas, although it is the first in Spain (TomTom, 2016). One of the factors that allow a daily significant volume of motor-vehicle journeys in such a dense space is that a large percentage of internal journeys are made by motorized two-wheelers. PTWs account for 17.4% of mobility in private vehicles in the inner city. However, most private journeys between the inner city and the metropolitan region, are made by car.

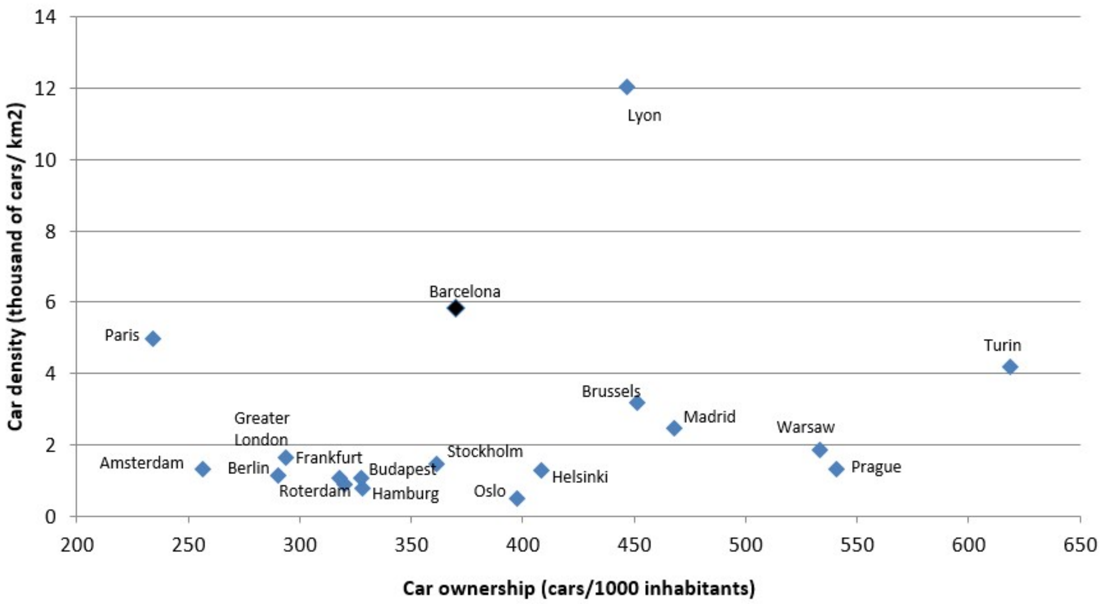
This high density of cars has resulted in the city of Barcelona being one of the most polluted cities in Europe. In the Council Directive 1999/30/EC, an annual mean limit value for nitrogen dioxide of 40 mg NO_2/m^3 has been set for the protection of human health, while an hourly limit value of 200 mg NO_2/m^3, not to be exceeded more than 18 times per calendar year, has also been set. In Barcelona, 5 of 13 NO_2 stations surpass the limit value, while the hourly limit value has not been exceeded anytime during 2016. On the PM_{10} side, a limit value of 40 mg/m^3 has also been set while 50 mg/m^3, (24-hour average) is not to be exceeded more than 35 times per calendar year. In PM_{10}, both limits were surpassed in 2016.

Figure 1. Modal share in principal EU agglomerations



Source: Own elaboration from EPOMM and EMTA (2014)

Figure 2. Car density and car ownership in major European cities

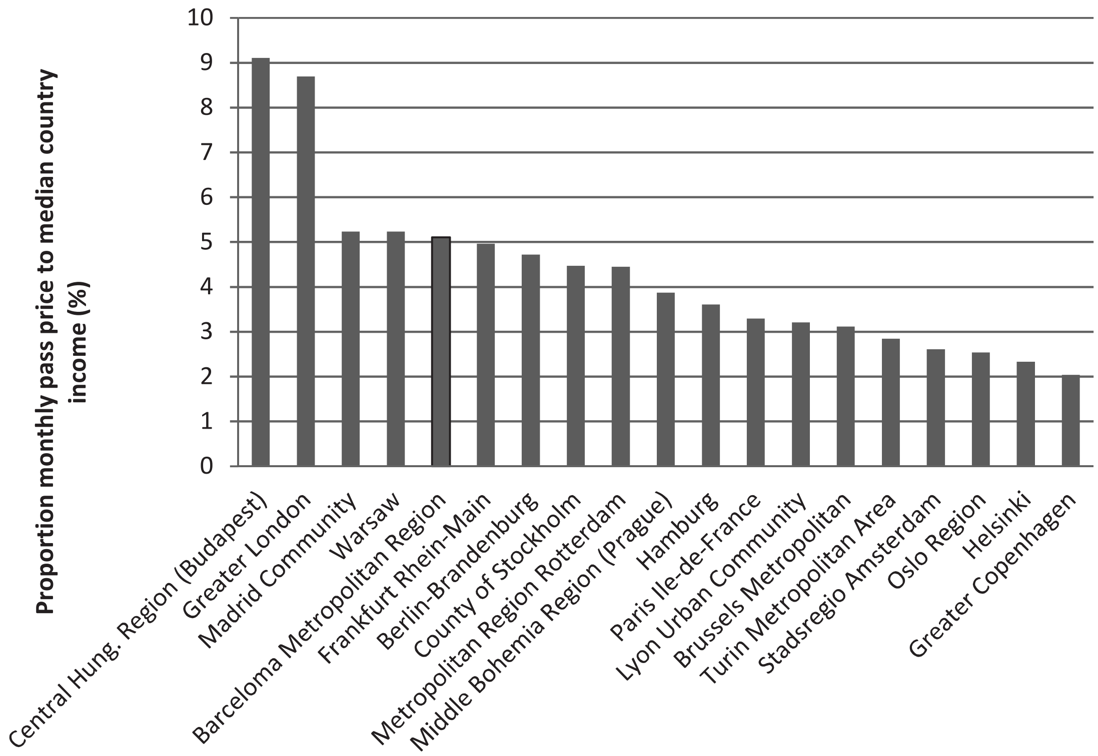


Source: Own elaboration from Eurostat

The new Directive 2016/2284, on the reduction of national emissions of certain atmospheric pollutants, advocates for a 40-60% reduction in NO_x levels in Spain by 2020, based on 2005 levels. In order to fulfil their emission reduction commitments, Spain should implement a national program for the reduction of air pollution. According to the Spanish Constitution, environmental competences are transferred to lower governments. Regional and local governments should play a central role in finding solutions. Recent trends suggest a stagnation or even decline in car ownership in some of the highest-income countries (Klein and Smart, 2017).

The second rationale is that lower public transit fares discourage automobile use, thereby reducing external costs from traffic congestion, local and global air pollution, and traffic accidents. This is a second-best argument, since it assumes that these external costs cannot be internalized through appropriate road pricing. Lowering transport fares might not imply large increases in public transport demand (Cats et al., 2014). Figure 3 shows an approximation of the relative cost of a monthly travel pass in a city, related to the median income in the country. We choose the median rather than the average income due to public transport being more frequently used by low-median income households, and the median income being normally less than the average. In Budapest and London, public transportation costs are relatively higher, around 9% of salaries. However, there is a group of five cities, including Barcelona, where that proportional cost is 5%.

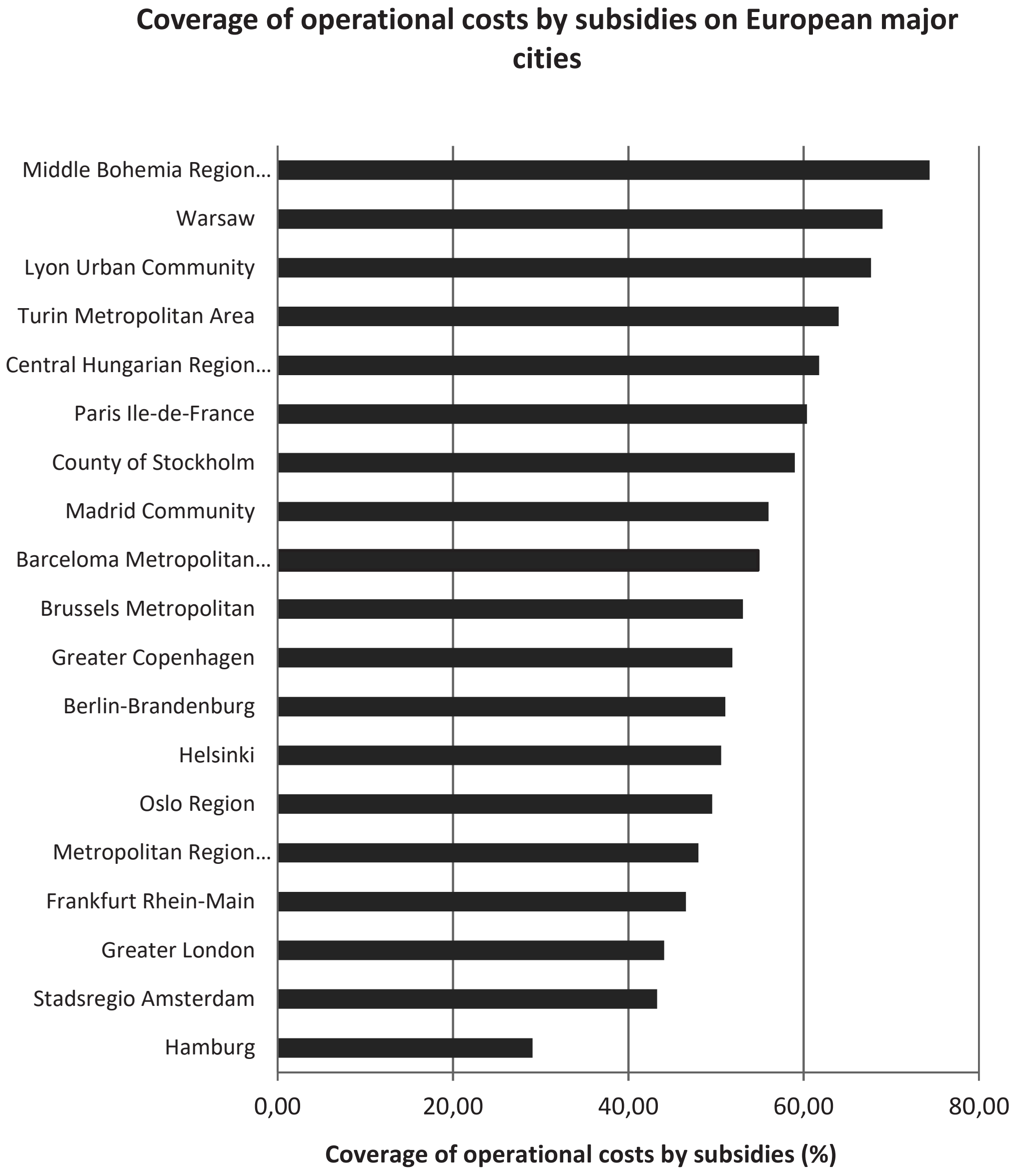
Figure 3. Price of monthly public transport pass related to median income



Source: Own elaboration from Eurostat and EMTA (2014)

Passenger fares for public transportation in most developed cities are half-subsidized. Nowadays, governments' fiscal stress can be a reason to raise passenger fares to balance the subsidy against public transport revenues. There is a wide debate about whether or not to increase the subsidy from government. The crucial matter for all is the externalities caused by road transport: congestion, air pollution and traffic. Parry and Small (2009) find that extending fares subsidies beyond 50 percent of operating costs is welfare-improving at a margin across modes, periods and cities. The Association of European Metropolitan Transport Authorities (EMTA) publishes the operational costs covered by public subsidies. Comparisons between cities can be complicated, and the authority may prescribe different scales of distance, since as they move further from the city centers, transportation systems tend to be more subsidized. Still, it is an introductory approach that allows us to put Barcelona in the European context (Figure 4). In 2012, there was an average of 45.6% covered by public subsidies, and 48.2% by fare revenues. The Barcelona Metropolitan Region has a 55% subsidy, which can be considered in the middle range. Trying to ensure that public transport remains as competitive as possible would help in keeping subsidies under control.

Figure 4. Subsidy proportion of public transport costs



Source: Own elaboration from EMTA (2014)

3 An overview of Public Policy Evaluation

Woodrow Wilson, Princeton University professor and former President of the US, led a movement in the late nineteenth century to separate politics and public administration, and to implement a scientific approach to the latter, to evaluate public sector performance. But it was not until the 1960s that public policy analysis proliferated in the US federal government. The Planning, Programming, and Budgeting System (PPBS) established in the Department of Defense, sets the foundation for the regular use of policy analysis (Bellinger, 2015). This evaluating culture was spread to other federal agencies after the Kennedy administration, requiring them to prepare planning documents and issue analysis papers for the Bureau of Budget. The federal social action programs under Johnson's presidency were a failure in terms of US federal evaluation. Cost-benefit analysis was required by the federal government in the 1970s and 1980s for all proposed regulations that were to be issued by agencies.

The wide variety of attitudes to evaluation policy across European countries is conditioned by their administrative and legal cultures (Furubo and Sandahl, 2002), although stable institutional characteristics of the political system and in specific features of the socioeconomic situation of a country, are always desirable (Wagner and Wollmann, 1986).

In some countries, evaluation may be deeply ingrained within the administrative culture; in others, it might be rooted only within particular departments or policy areas; whereas in still other cases, evaluation is viewed as having little significance at all (Bachtler et al., 2000). European Union countries can be divided into different groups, depending on their commitment to evaluation.

The first group includes the northwestern European countries, such as Germany, the Netherlands, Sweden or the UK, in which evaluation plays an important part in policy culture. Evaluation emphasizes value for money and accountability, and the UK was the front-runner. The National Audit Office (2013) of the UK reviewed nearly 6,000 analytical and research documents published between 2006 and 2012 on government department websites. Only 305 of these were impact evaluations; and of these, only 70 made an assessment of cost-effectiveness.

A second group of countries, with evaluation policies that were developed intermittently in the nineteen-nineties, is made up of Finland, Austria, Ireland and Luxembourg. And the third group, countries using the continental administrative and legal systems, were not favorable at first to the emergence of an evaluation culture. This group includes France and southern countries, such as Italy, Portugal and Spain. So, northern European countries have the longest tradition and culture in this field, dating back to the 1970s, while in other parts of the continent, especially the south, evaluation did not attract interest until the 1990s. Ultimately, the public policy evaluation catalyst in most EU countries has been the obligation to evaluate EU structural funding (Bachtler et al., 2000), particularly for regional policies (Polverari and Bachtler, 2004).

With regard to Spain, the practice and the degree of institutionalization of evaluation has been one of the lowest in the OECD (Furubo and Sandahl, 2002;

Varone et al., 2005; Viñas, 2009). The traditional concern of civil servants has been with the legality of procedures rather than with paying attention to results, and that has conditioned the Spanish evaluation culture.

The political democratic transition in the mid-1970s strengthened accountability in the public sector, but it was Spain's entry into the European Union in 1986 that was the turning point in the evaluation of public programs and policies, due to the mandatory evaluation of structural funds from the European Union (Feinstein and Zapico-Goñi, 2010).

The first expansion of evaluation culture in Spain was in the nineteen-nineties, and it focused on the redrafting of procedural manuals rather than on introducing this culture into the administrative body. Central government departments' evaluation reforms were more formalistic than functional. Legal and hierarchical administrative traditions in Spanish administrative bodies are so strong that belief in the success of evaluation programs relies more on establishing new norms rather than in evaluating purposes. So, in reality, evaluation in Spain, until 2007, was based on European Union co-founded programs and the evaluation of development cooperation programs. The State Agency for the Evaluation of Public Policy and Service Quality (AEVAL) was established in 2007, as a landmark along Spain's road towards creating an evaluation culture.

However, country-specific analysis should be balanced with other countries' state-of-the-art evaluation culture. Jacob et al. (2015) evaluated nineteen developed countries, awarding scores for all of them. The vast majority (79%) of the countries included in the sample showed a high degree of evaluation culture maturity while Spain is in the third to last position, followed by Italy and Ireland. An observed characteristic of Spain is that discussions based on evaluation findings rarely take place in Parliament. In 2011, the differences between countries were less contrasted than in the past; evaluation culture has matured over the last decade. An encouraging result in Spain is the profoundly better performance in evaluation culture from 2001 to 2011, where ratings changed from 5 to 11.3.

Nowadays, Spain does not have a single body or a single evaluation system. Both from the sectoral point of view and the territorial, and even from both simultaneously, the responsibility for evaluation is clearly segmented. Since the evaluation function at the central level of government is entrusted to AEVAL, some regional evaluating agencies arise. The most developed and noticeable is in Catalonia, where an Institute for the Evaluation of Public Policies, Ivalua, was created in 2009, with representatives from the regional and local governments, universities and public and private foundations.

4 Dissertation structure

This thesis, is structured into four chapters and this introductory one. While chapter two analyze mobility characteristic in the Barcelona Metropolitan Area with policy implications, chapter three and four pursue to evaluate specific mobility policies. On chapter 5 concludes and add policy recommendations.

On chapter 2 we analyze the factors influencing CO_2 emissions from daily urban mobility. Concerns about the unequal distribution of greenhouse gas emissions attributable to mobility are gaining increasing attention in scholarly analyses as well as in the public policy arena. Factors influencing on different emitters are largely unknown; and the influence is assumed to be the same for all emitters, be them low or high emitters. We use a household travel survey in the metropolitan area of Barcelona to differentiate the factors that result in different rates of emission. As a results, we find that top ten per cent emitters produce 49% of total emissions, while non-daily emitters account for 38.5% of the sample. We adopt a quantile regression approach, which allows us to find significant differences between groups. Gender, income and home-municipality type are influential in accounting for CO_2 emissions for all groups. Educational level appears to be less significant, and occupation shows no significance at all. We confirm that socioeconomic factors have different influences on different emitting groups; these characteristics do not impact equally across all the population.

In the public policy design, the study confirms an inadequate toll policy implementation in the area, while parking and public transport policy are well designed to reduce CO_2 emissions. The application of quantile regression using mobility survey data from different cities would provide evidence useful to further improve the design of urban mobility policies.

This chapter has been published under the title *The impact of socioeconomic characteristics on CO_2 emissions associated with urban mobility: Inequality across individuals* in Energy Economics, 64, pp. 251-261.

On chapter 3 two speed management policies – a variable speed system and an 80 km/h speed limit – have been implemented on Barcelona’s urban motorways to mitigate NO_x and PM_{10} air pollution. In 2008, the maximum speed limit was reduced from 120 and 100 km/h to 80 km/h and, in 2009, a variable speed system was introduced on some metropolitan motorways. Chapter 2 evaluates whether such policies have been successful in promoting cleaner air, not only in terms of average pollutant levels but also during high and low pollution episodes. To do so, we use difference-in-differences methodology on the average and on different quantiles for fixed effect panel data, which allows us analyzing different scenarios. We find that the variable speed system improves air quality with regard to the two pollutants considered here, being most effective when nitrogen oxide levels are not too low and when particulate matter concentrations are below extremely high levels. However, reducing the maximum speed limit from 120/100 km/h to 80 km/h has no effect - or even a slightly increasing effect -on the two pollutants, depending on the pollution scenario.

This chapter has been published in two different articles. The first article is *Effects of the 80 km/h speed limits on air pollution in the metropolitan area of Barcelona*, in Transportation Research Part D: Transport and Environment, 23, pp. 90-97. The second article corresponds to *The environmental effects of changing speed limits: A quantile regression approach* in Transportation Research Part D: Transport and Environment, 36, pp. 75-85.

On chapter 4 we compare the relative merits of public and private delivery within a mixed delivery system. We study the role played by ownership, transaction costs, and competition on local public service delivery within the same jurisdiction. Using a stochastic cost frontier, we analyze the public-private urban bus system in the Barcelona Metropolitan Area. Academics and policy makers are increasingly shifting the debate concerning the best form of public service provision beyond the traditional dilemma between pure public and pure private delivery modes, because, among other reasons, there is a growing body of evidence that casts doubt on the existence of systematic cost savings from privatization, while any competition seems to be eroded over time. We find that private firms have higher delivery costs than those incurred by the public firm, especially when transaction costs are taken into account. Furthermore, tenders tend to decrease delivery costs.

The debate on the reform of public service delivery has most frequently focused on the dilemma between pure public and pure private delivery modes. Our analysis suggests that mixed delivery (as long as economic conditions for service fragmentation exist) creates a framework in which policy makers and regulators can employ different tools to pursue different objectives.

This chapter has been published under the title *Public and private production in a mixed delivery system: regulation, competition and costs*, in Journal of Policy Analysis and Management, 35(3), pp. 533-558.

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CHAPTER 2

THE IMPACT OF SOCIOECONOMIC CHARACTERISTICS ON CO_2 EMISSIONS ASSOCIATED WITH URBAN MOBILITY: INEQUALITY ACROSS INDIVIDUALS

1 Introduction

Transport is a major contributor to various environmental externalities, including most notably greenhouse gas (GHG) emissions and local air pollution. Indeed, the transport sector is reported to be responsible for a mostly quarter of energy-related carbon dioxide – CO_2 – emissions (IEA, 2015). This being the case, it is essential to understand what motivates people to employ (or otherwise) private vehicles, public transportation, and non-motorized modes, respectively. Having obtained this knowledge, appropriate policies can then be designed to enhance transport sustainability. Thus, from a policy perspective, it is important to understand how the distribution of CO_2 emissions is related to individual characteristics, since this provides fundamental insights into the distributional implications of climate change mitigation policies. Critically, travel patterns are known to vary in line with socio-economic characteristics, and more specifically with lifestyle characteristics combined with personal preferences and attitudes (Anable, 2005). This makes behavioral change a key factor in reducing the weight of transport in CO_2 emissions in relation to other sectors, particularly in the short term (Chapman, 2007).

Increasing attention is being paid to the analysis of international inequality in energy consumption and per capita CO_2 emissions (Duro and Padilla, 2006, 2011; Mussini and Grossi, 2015). Furthermore, in recent years a discussion has emerged on how transport emissions are distributed very unequal in developed countries (Brand and Boardman, 2008; Brand and Preston, 2010). Thus, some studies have used econometric techniques to differentiate between groups. For example, Ko et al. (2011) identify a group of "high emitters" and, more recently, Büchs and Schnepf (2013) described a group of "low emitters" (although in this case without performing an econometric analysis for this group). Allinson et al. (2016) advocates more work to understand better what causes emissions in households with high total emissions.

This unequal distribution of GHG emissions should have an impact on policy intervention. Changes in aggregate carbon intensity for personal transportation in some OECD countries between the seventies and nineties show the importance of both fuel price and governmental policies in order to contain CO_2 emissions (Greening, 2004).

However, most of the research effort to date has focused on the average household, or on high emitter profiles. Thus, attempts at identifying different groups of emitters have been largely neglected and the results describing the distribution of emissions remain inconclusive. For this reason, there continues to be a considerable dearth of knowledge regarding the full implications of policy measures, such as fuel taxes, parking fees, or congestion charges.

This paper seeks to contribute to the literature by examining the impact of a series of individual characteristics on CO_2 transport emissions. While most papers to date have analyzed this impact in terms of the average emitter (with a few focusing on the top per cent of high emitters), we analyze the level of CO_2 emissions for different population groups. To the best of our knowledge, this is the first econometric analysis using quantiles of transport emitters. To do so, we adopt a quantile regression approach. Our analysis of a sample of 24,605 individuals confirms the unequal distribution of CO_2 transport emissions. The top 10% of high emitters are responsible for 49% of total emissions, while the top 20% of pollutants contribute 74%. These are two of the highest rates reported to date in the literature.

We use an institutional household travel survey conducted in the metropolitan area of Barcelona (Catalonia) to identify the factors that allow us to differentiate groups of emitters. This survey includes socioeconomic, demographic, residential, and transport characteristics. After briefly reviewing the related literature in the next section, we report the details concerning our data in section three. We then discuss the econometric methodology of quantile regressions. Thereafter we present the results obtained by using this methodology, comparing it with logistic regressions to check the robustness of our analysis. Then, we conduct the quantile analysis for different mobility policies in a population subsample. Finally, we discuss our main results, examine their policy implications, and draw the main conclusions.

2 Literature review

Within the literature on mobility and CO_2 emissions, the determination of the type of emitters has gained increasing interest in recent years. Several econometric studies have analyzed transport CO_2 emissions and the impact on these of socioeconomic, demographic, geographic and household characteristics. Using econometric methods to analyze how socioeconomic factors influence CO_2 transport emissions has gained ground, with the aim to measure the impact of those factors on the amount of CO_2 for the average emitter on Quebec City (Barla et al. 2011). Descriptive statistics in Brand and Preston (2010) show that there is an unequal CO_2 emissions distribution among individuals, but their econometric analysis is performed only on the average emitter for Oxford (UK) citizens. Brand et al. (2013) aim to separate the effect of socioeconomic factors in CO_2 emissions, but they just differentiate according to the purpose of the trip, without any differentiation between types of emitter on Oxfordshire (UK). With a similar methodology, and for the UK countrywide, descriptive statistics in Büchs and Schnepf (2013) show differences between high and low emitters, but they conduct OLS regression only on the mean.[1] Ko et al. (2011) clearly discern on the type of emitter using an econometric approach. By means of logistic regression, significant differences are found between the top 10% CO_2 emitters and the rest of emitters in the Seoul metropolitan area.

Table 1 displays the studies focusing on socio-demographic factors that affect mobility carbon emissions, summarizing their main characteristics and results. On the gender issue, most studies report a positive relation between being male and CO_2 emissions: the proportion of high emitters among males being much greater than among females. The impact of income has been widely analyzed in the literature, with the relation always being reported as significant and positive. For example, Büchs and Schnepf (2013) find an elasticity of income-CO_2 emissions with respect to income of 0.9. A number of studies (Brand et al., 2013; Büchs and Schnepf, 2013) find a positive relation between CO_2 emissions and higher education; yet, higher education can also imply a CO_2 reduction, given a greater awareness of environmental issues among the better educated. This might account for the fact that Barla et al. (2011) find higher education not to have a significant relationship with CO_2 emissions. In the case of age, most studies find an inverse U-shaped curve; however, age trends are not clear. While adults emit more CO_2 than teenagers-children and retirees, there are marked divergences between these two groups in terms of their emissions. Likewise, it is unclear whether those aged between 18 and 30 emit more or less than the next cohort (up to the age of 45). For example, in the Seoul metropolitan area, more than 75% of high-emitters belong to the 40- to 59-year-old age group (Ko et al., 2011).

In relation to factors of mobility, the results are conclusive: holding a car license and, more specifically, owning a car imply more CO_2 emissions in all studies.

[1] Another study worth noting is that by Xiao et al. (forthcoming) discern the level of emissions and income on Beijing citizens, but also without separating them on the multilevel regression analysis, being their focus on spatial level. A moderate sample finds that the top 20 percent of emitters are responsible for 65 percent of total emissions. However, because this study uses a survey on energy expenditure, it is not comparable to the studies included in Table 1.

Indeed, the majority of studies show a positive relation between car ownership and income with private car split. Patterns of urban transportation systems and of travel behavior vary widely, even among countries with similar urbanization and per-capita income levels. Santos et al. (2013) find the number of students in universities and higher education to be positively associated with the use of all modes of transport, but the car. Unsurprisingly, they also find that GDP per capita is positively associated with car sharing. Several studies also included the geographic factor as a determinant of emissions related to car dependency. Thus, Büchs and Schenpf (2013) find that downtown or high density areas are associated with lower CO_2 emissions. In contrast, lower population densities imply more GHG emissions (Barla et al, 2011), due to an increasing car dependency. However, Brand and Preston (2010) and Brand et al. (2013) find no geographic effects for Oxford and Oxfordshire, respectively. This might be attributed to the fact that people seem to have the need to be involved in particular activities, which do not depend significantly on their immediate spatial environment, defined in terms of relative location vis-à-vis the city center and the nature of the public transportation system (Timmermans, 2003).

Table 1 Impact of different household socio-demographic variables on CO_2 transport emissions in previous econometric studies

Authors	Zone	Method	Groups analyze	Gender	Income	Education	Age	Car ownership	Geographic zone
Brand and Preston (2010)	Oxford (UK)	Regression model	All	...	+, but only extremes		... except negative on retirees	+	....
Barla et al. (2011)	Quebec city	Regression model	All	+	+, but only extremes	....	Less emissions 50-64 years than 35-49.		+ for non-downtown
Ko et al. (2011)	Seoul metro area	Tree-based regression and logistic model	High emitters (top 10%) and non	+	+		∩	+	+ for metropolitan cities
Brand et al. (2013)	Oxfordshire (UK)	Regression model and logistic regression	All and high emitters (top 20%)	+	+	+	∩	+	....
Büchs and Schnepf (2013)	UK	Regression model	All		+	+	∩		+ rural

Note: +: Positive relation; ...: not significant; ∩: significant for all ages with a maximum on adults; blank: not considered in the analysis

Findings for occupational status are clear: those in employment are associated with higher emissions, while being either a homemaker or a retiree implies fewer emissions. However, the impact of being unemployed is less clear – findings suggesting lower emissions or no impact. Another factor taken into account by Büchs and Schnepf (2013) is the composition and number of people in the household, finding that the first child reduces transport emissions, while the second child increases them. All in all, effects assessed in the previous literature commonly work in the same direction, although sometimes the techniques used or the areas researched may cause some variable to lose significance.

Quantile regression has been recently introduced in transport and CO_2 empirical analyzes. Qing Su (2012) analyzes the extra utilization of vehicles due to improved fuel efficiency (rebound effect); Hammoudeh et al. (2014) investigate the impact of changes in crude oil prices, natural gas prices, coal prices, and electricity prices on the distribution of the CO_2 emission allowance prices in the United States; Bel et al. (2015) measure the impact of speed limits on environmental pollutants. With respect to object of study in this paper, the closest use of the quantile regression methodology is Han et al. (2015) analysis of how household characteristics differ in their associations with household embedded carbon emissions. Without taking into account transport carbon emissions, they find that sociodemographic factors affect GHC emissions differently. It is worth noting that these studies find that independent variables across quantiles have a significant but differential impact on dependent variable. Quantile regression is in a better position to capture the impact of socioeconomic characteristics than OLS. This is particularly relevant when the central location of conditional distribution and tails vary significantly with covariates.

It is worth noting that not all these studies provide the same potential for comparability with Barcelona.[2] The urban structure and mobility characteristics of the metropolitan area of Barcelona are different from those in Oxford (and Oxfordshire) and Quebec, areas with much lower population and less dense mobility. Also, a study such as Büchs and Schnepf (2013), as it is conducted for a whole country –the UK-, offers little room for comparability. Contrariwise, the Seoul metropolitan area (studied by Ko et al., 2011) is more suitable for comparisons with Barcelona. The density in both central cities is very similar (Seoul 16,500 inhabitants/km^2; Barcelona, 16,000 inhabitants/km^2). Both central cities (and corresponding metro areas) are wealthier and have more income than

[2] These studies tend to look at the socioeconomic characteristics in a moment of time. An exception to this pattern is that done by Zahabi et al. (2015), conducted for Montreal from 1998 to 2008, paying special attention to geographic characteristics -rather than socioeconomic ones-. They find emissions reduction over time, due to distance travelled reduction, and efficiency improvement of the fleet.

the rest of the respective countries. Finally, it is also worth noting that the Household Travel Surveys used in Ko et al. (2011) was conducted in 2006, which is precisely the year when the household Survey for the area of Barcelona was conducted.

Overall, the existing evidence points to inconclusive trends and the lack of specific attention to different groups. Therefore, this paper seeks to contribute to the literature by comparing the impact of various individual characteristics on CO_2 transport emissions by different emitters. By doing so, we find evidence that not all socioeconomic variables impact equally to all individuals, as well as these variables may have different effects depending on the city analyzed.

3 Data

3.1 Data source

Our main data base is the household travel survey (HTS) conducted for the entire metropolitan region of Barcelona in 2006 by the Metropolitan Transport Authority (*Institut d'Estudis Regionals i Metropolitans de Barcelona*, 2006). This data base includes a sample of individuals reporting their previous day's (daily) trips, including origin and destination, journey time, day and hour, transport mode, and trip purpose. The survey employs a computer-assisted telephone interviewing (CATI) technique in contacting with a representative sample of the population. Using a multistage stratified sampling, individuals are selected by applying sex and age quotas. The interview comprises four blocks of questions: the first block concerns household composition and is used to select the individual; the second block of questions gathers details about all the previous day's trips; the third block comprises questions about the individual's socioeconomic characteristics; and, the fourth block gather personal details related to the individual's mobility. The individual characteristics gathered in the third block are gender, age, educational level, family income, and occupational status. The fourth block gathers financial information, including monthly expenditure on public transport, fuel, tolls, and parking away from home. We compute all journeys within the metropolitan area of Barcelona, that is, all metropolitan journeys that have their origin and destination inside the area. The total number of journeys is 93,864, and the total number of travelers is 24,605. All types of journey (be they for leisure, work, shopping, etc.) are included.

3.2 Emissions estimation procedure

CO_2 intensity is a measure of emissions per unit of activity, and is calculated using the estimation procedure employed by the International Transport Forum (2009). For private cars, we also use the emission factors, corrected by the proportion of gasoline and diesel vehicles making up the fleet. The regional government provides different emission factors according to three average speeds: 21 km/h, 70 km/h, and 107 km/h. We obtain different emission factors based on road type (urban, interurban, or motorway), time slot, and city of origin and destination. The emission factor is corrected if the individual is accompanied by a traveling companion, depending on the metropolitan occupancy rate. CO_2 emissions are calculated using equation 1:

$$CO_2 private\ mode = Emission\ time\ x\ emission\ factor\ x\ occupancy \quad (1)$$

We conduct the same procedure for motorbikes, except that there are no diesel vehicles among this mode of transport. Emissions from soft modes, cycling and walking, are categorized as zero. In the case of taxis, we have information on the emission factor of each vehicle type and the composition of the fleet, and we also correct for traffic conditions. For subways, tramways, interurban buses, and national and regional trains, we use the official emission factors (for passenger-kilometers) applying equation 2:

$$CO_2\ public\ transport = Distance\ x\ emission\ factor \quad (2)$$

For intra-city bus journeys, we only have information about journey time, so we apply equation 1. Therefore, to obtain the corresponding CO_2 emissions we discount walking (transit access) time and apply an average speed.

The first limitation we encounter (and one that we need to take into account when interpreting our results) concerns those individual commuters that switch transport modes when completing the same trip. The HTS does not indicate where these changes of transport mode occur; thus, it is not possible to quantify their CO_2 emissions accurately. There is no way of overcoming this limitation, so we opted to omit these individuals.[3] Additionally, information on several variables is unavailable – this is the case of car and motorbike ownership, and household size (in particular, the number of children and their ages).

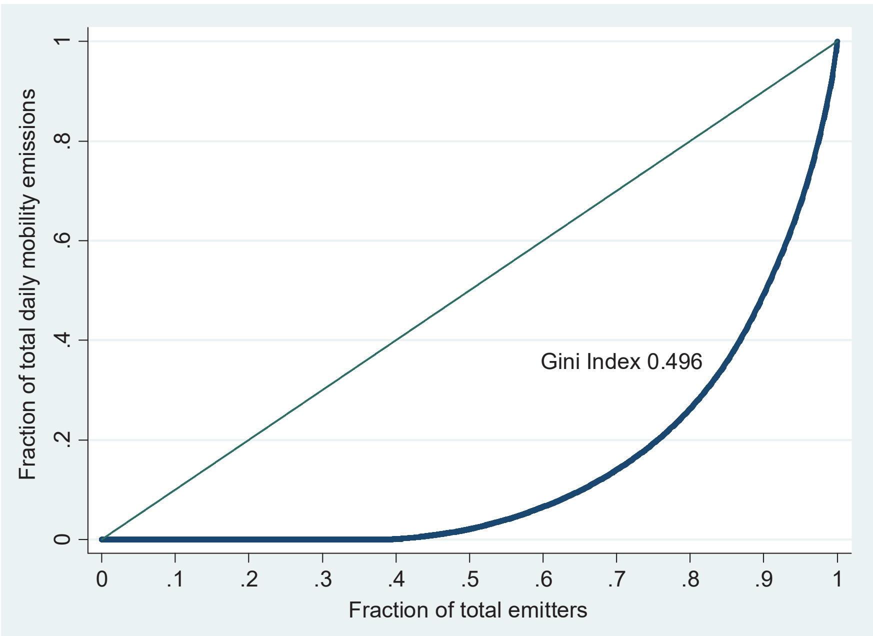
[3] It is worth noting that there are not differences between individuals in both samples, with respect to socioeconomic characteristics.

3.3 Emission results

Based on the available information and making the computations outlined above, we find that the individual, daily average emission is 1,738 g CO_2. The top 10% of pollutants are responsible for 49% of total emissions, that is, 8,961 g CO_2 per day, while the top 20% of pollutants contribute 74% of total emissions (Figure 1). These results are consistent with the literature: top ten per cent of emitters produce 43 and 63% of emissions in Oxford and Seoul, respectively. If we focus on the highest quintile only, we find they produced 62 and 82% of total emissions in Oxford and Seoul, respectively (Brand and Preston, 2010; Ko et al., 2011).

In Barcelona, 38.5% of individuals do not produce CO_2.[4] These findings point to considerable inequality in daily mobility emissions, presenting a coefficient of 0.496 on the Gini index. If we compare this with the income-related Gini index for 2006 for the same area (recorded at 0.296), the inequality for mobility emissions is twenty points higher and, as such, is much more pronounced. The Jarque-Bera test indicates that individual emissions do not follow a normal distribution (p-value equal to zero). On average, 86% of the emissions attributable to an individual emitter are produced by private vehicles. Average emissions per journey in a private vehicle are 1,258 g CO_2, while emissions per journey on public transportation are 439 g CO_2.

Figure 1. Cumulative distribution of total daily CO_2 mobility emissions



[4] The low proportion of non-emitters is due to the sampling method. The travel survey only asks about the trips undertaken the previous day.

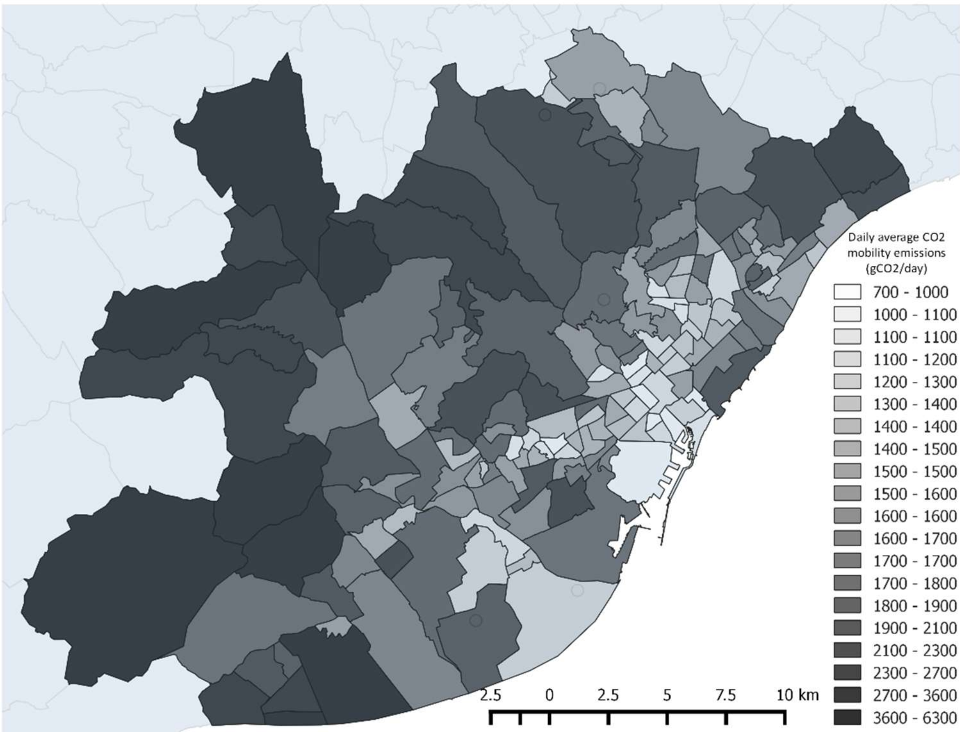
Table 2 shows the socioeconomic traits considered: sex, monthly family income, educational level, occupation status, size of hometown, and monthly expenditure on different transportation alternatives: public transportation, car fuel, tolls, and parking away from home (i.e., all parking expenditure, except home parking). The 10th and 25th percentiles cluster the non CO_2 emitters; the 50th percentile corresponds to the median emitter and 0.69 to the average emitter – hence, the 75th quantile can be interpreted as such.

Table 2. Mean and standard deviations of socioeconomic characteristics

Family variables	Variable	Levels	Quantile 0.25	0.5	0.75	0.9
Demo geographic	Gender	Male	0.421	0.407	0.476	0.627
			(0.4936)	(0.4915)	(0.4996)	(0.4839)
		Female	0.579	0.593	0.524	0.373
			(0.4936)	(0.4915)	(0.4996)	(0.4839)
		4-29	0.311	0.458	0.312	0.199
			(0.4629)	(0.4984)	(0.4635)	(0.3991)
	Age	30-44	0.166	0.203	0.318	0.446
			(0.3724)	(0.4021)	(0.4660)	(0.4973)
		45-64	0.252	0.215	0.267	0.301
			(0.4340)	(0.4107)	(0.4426)	(0.4589)
		Above 65	0.271	0.124	0.102	0.054
			(0.4445)	(0.3301)	(0.3033)	(0.2261)
	Hometown Inhabitants	Barcelona	0.392	0.486	0.384	0.320
			(0.4882)	(0.5000)	(0.4865)	(0.4667)
		< 10,000	0.027	0.056	0.069	0.048
			(0.1620)	(0.2304)	(0.2541)	(0.2131)
		10,000-50,000	0.152	0.138	0.161	0.178
			(0.3588)	(0.3448)	(0.3675)	(0.3826)
		>50,000	0.429	0.319	0.386	0.454
			(0.4950)	(0.4666)	(0.4871)	(0.4981)
Economic	Family monthly income	Less than 1000 €	0.371	0.216	0.136	0.073
			(0.4830)	(0.4117)	(0.3429)	(0.2611)
		1000-2000 €	0.404	0.400	0.419	0.394
			(0.4907)	(0.4903)	(0.4936)	(0.4889)
		2000-3000 €	0.151	0.240	0.271	0.311
			(0.3578)	(0.4276)	(0.4446)	(0.4631)
		3000-4000 €	0.050	0.077	0.114	0.135
			(0.2179)	(0.2660)	(0.3180)	(0.3420)
		4000-5000 €	0.015	0.042	0.041	0.046
			(0.1209)	(0.2015)	(0.1974)	(0.2089)
		> 5000 €	0.010	0.025	0.019	0.041
			(0.1006)	(0.1549)	(0.1392)	(0.1977)
	Educational level	No studies	0.145	0.068	0.033	0.012
			(0.3523)	(0.2530)	(0.1784)	(0.1087)
		Primary studies	0.480	0.373	0.319	0.264
			(0.4996)	(0.4838)	(0.4662)	(0.4407)
		Second studies	0.238	0.340	0.3854	0.397
			(0.4253)	(0.4738)	(0.4869)	(0.4895)

		Tertiary studies	0.137	0.219	0.263	0.327
			(0.3441)	(0.4139)	(0.4404)	(0.4693)
	Occupation status	Scholar	0.251	0.348	0.168	0.055
			(0.4332)	(0.4766)	(0.3739)	(0.2277)
		Housekeeper	0.122	0.064	0.055	0.029
			(0.3276)	(0.2465)	(0.2284)	(0.1689)
		Retiree	0.304	0.145	0.121	0.069
			(0.4601)	(0.3521)	(0.3267)	(0.2537)
		Employed	0.268	0.408	0.602	0.803
			(0.4428)	(0.4916)	(0.4897)	(0.3979)
		Unemployed	0.055	0.034	0.054	0.044
			(0.2286)	(0.1814)	(0.2252)	(0.2044)
Mobility expenditure		Public transport	13.22	24.42	22.75	14.76
			(16.026)	(20.959)	(24.328)	(25.838)
		Fuel	47.86	47.82	59.60	84.00
			(44.076)	(54.001)	(80.087)	(74.316)
		Tolls	10.37	13.21	11.16	14.36
			(20.134)	(24.014)	(22.907)	(31.950)
		Parking away home	5.22	7.44	8.38	15.49
			(16.195)	(17.92)	(22.135)	(33.665)
Number of daily journeys			3.74	3.57	3.862	4.05
			(1.819)	(1.650)	(1.772)	(2.055)

Figure 2. Geographical distribution of average daily CO_2 mobility emissions



The average daily CO_2 mobility emissions per capita are unequally distributed geographically (Figure 2). The lowest emission are recorded in the city of

Barcelona and the contiguous area, whereas the highest emissions are recorded in the municipalities located furthest from the inner city, characterized by relatively low population densities and poor public transport networks. This is consistent with the findings of Muñiz and Galindo (2005), who studied the ecological footprint in the metropolitan area of Barcelona and found that municipalities with low-density levels located in the outer periphery have higher per capita daily mobility emissions than municipalities located in denser, more central areas.

4 Methods

We use quantile regression to investigate the characteristics of individuals depending on their CO_2 emission levels. Quantile regression was first introduced by Koenker and Bassett (1978) as an extension of the notion of ordinary quantiles (or "percentiles") in a location model. In this way, the regression model can be extended to conditional quantiles of the response variable. Quantile regression is especially useful when the rate of change in the conditional quantile, expressed by the regression coefficients, depends on the quantile. Thus, we can study the whole distribution of the collected data rather than simply the mean. This makes it particularly valuable for applications in which extremes are important or which differ markedly from the mean. Quantile-based estimators are more robust and more efficient than mean estimators when distributions have fat tails: quantiles estimations are preferred than OLS ones Equations are designed to estimate the relation of covariates. Two important features of the estimation are that quantile regression is more robust to non-normal errors as well as to outliers.[5]

The linear model is defined as:

$$Q_{Y_i}(\tau) = \beta(\tau)X_i + \theta_i \qquad (3)$$

where $Q_{Y_i}(\tau)$ is the quantile function at confidence level. The model in (3) allows the influence of covariates X_{it} to depend on the quantile level τ.[6] As proposed in Koenker (2004), we want to estimate the parameters in model (1) simultaneously for all quantiles under study, $\tau_q,\ q = 1, \ldots, Q$. Following Koenker (2004) this implies solving:

$$\min_{(\beta,\gamma,\theta)} \sum_{q=1}^{Q} \sum_{i=1}^{n} \sum_{t=1}^{T_i} w_q\, \rho_{\tau_q}\big(Y_{it} - \beta(\tau_q)X_{it} - \theta_i\big), \qquad (4)$$

5 As discussed by Deaton (1997), quantile regression is most useful when the errors are heteroscedastic.

6 See Bel et al. (2015) for a detailed explanation of the quantile regression method.

where $\rho_\tau(\cdot)$ is a function defined by Koenker and Bassett (1978) as:

$$\rho_\tau(u) = \begin{cases} \tau|u|, & u \geq 0 \\ (1-\tau)|u|, & u < 0 \end{cases} \quad (5)$$

The terms w_q are weights. They control the influence of the quantiles on the estimation of the fixed effects. We assume that the weights are the same for all the quantiles we analyze. We opted here to regress the 25^{th}, 50^{th}, 75^{th} and 90^{th} quantiles. Equations are designed to estimate the relation between socioeconomic characteristics and mobility expenditure and the CO_2 individual emission, conditional on quantiles of CO_2 individual emission.

5 Results

5.1 Quantile regression

In this model, all the explanatory variables are categorical variables; thus, one class of each variable was designated as a reference. As we are aware that multicollinearity problems might affect some of these variables, we conducted the VIF test on the simple model, and found three – secondary education, retirees, and employed – with values between five and eight. Indeed, the results for the variables related to occupational status presented considerable instability in all estimations. In distribution tests of the dependent variable, normality is always rejected; therefore quantile regression is preferred to OLS models. We performed the Machado-Santos Silva (2000) test to detect heteroscedasticity. The assumption that residuals are normally distributed is violated owing to the multiple cases of non-emitters in our sample, as such the presence of heteroscedasticity is confirmed. We performed quantile regression and found robust standard errors and t-statistics that are asymptotically valid under heteroscedasticity using the Machado et al. (2011) package. Additionally, we include the results from the OLS regression so that we can compare the sign and the significance of the variables obtained with each methodology.[7]

First, when using quantile regression, the explanatory power of the estimations for the groups of higher emitters increases. In fact, our results are poor for the groups of lower emitters. We omit the 25^{th} quantile from Table 3 as the pseudo-R^2 was below 0.02 and because of the poor performance of most explanatory variables (results available upon request). In contrast, we obtain a relatively high

[7] As the presence of heteroscedasticity is confirmed, we use robust (White) standard errors when conducting the OLS regressions.

explanatory power of the estimations for the highest quantiles ranges between 0.14 and 0.16.[8] All in all, socioeconomic variables have a limited ability to capture the variability of individual CO_2 emissions. Indeed, the model based on these socioeconomic variables fails to discern with sufficient precision between those who are high emitters and those who are not.

Demo-geographic variables: Males produce more CO_2 mobility emissions than females in all quantiles. This is an extremely robust result: the emissions for being male increase over the quartiles. Age is a significant and positive factor for people aged between 30 and 44, while in the other population groups there are no significant differences. Emitters resident in small municipalities emit significantly more (followed by individuals in medium and large municipalities) than do those resident in Barcelona, as Figure 2 also shows. Individuals living in highly populated areas produce less daily mobility emissions, but their emissions from long-distance trips are greater than those produced by individuals living in rural areas, as Reichert et al. (2016) report. In terms of median values, there are no statistical differences between municipalities above 10,000 inhabitants and Barcelona (although OLS indicates a significant effect).

Table 3. Quantile regression

Family variables	Variable (reference level)	Levels	OLS	0.50	0.75	0.90
Demo geographic	Gender (Female)	Male	909.3*** (49.08)	227.9*** (42.0)	1,067.3*** (87.6)	1,661*** (149.9)
	Age (Under 30)	30-44	328.8*** (85.08)	93.06* (56.3)	604.8*** (154.0)	876.4*** (235.1)
		45-64	53.07 (88.91)	-59.61 (54.1)	44.7 (156.8)	313.3 (223.4)
		Above 65	16.34 (108.60)	-59.61 (56.1)	44.7 (162.8)	108.9 (245.2)
	Hometown	< 10,000	1,289*** (133.11)	696.0*** (122.6)	1,510*** (233.1)	2,708*** (465.1)
	Inhabitants	10,000-50,000	597.4*** (73.71)	1.39e-11 (25.1)	313.8*** (100.2)	989.8*** (184.5)
	(Barcelona)	>50,000	588.5*** (51.20)	1.73e-11 (18.0)	403.1*** (68.1)	770.7*** (110.2)
Economic	Family monthly income (<1000 €)	1000-2000 €	267.1*** (53.66)	1.33-11 (17.5)	255.5*** (73.7)	496.2*** (136.4)
		2000-3000 €	655.0*** (76.29)	317.8*** (48.4)	894.0*** (135.4)	1,470*** (240.3)

[8] As mentioned above, besides the influence of socioeconomic characteristics on travel patterns, the latter are known to vary greatly according to lifestyle characteristics combined with personal preferences and attitudes (Anable, 2005; Chapman, 2007).

	3000-4000 €	787.7***	519.2***	1,397.1***	1,591***
		(106.35)	(95.0)	(172.7)	(266.0)
	4000-5000 €	1,303***	647.5***	1,727.2***	3,262***
		(191.69)	(156.4)	(224.7)	(514.0)
	> 5000 €	1,517***	1,277.13***	2,509.4***	2,997***
		(200.78)	(428.2)	(406.2)	(435.6)
Educational level (no studies)	Primary studies	236.6***	1.18e-10	3.05e-10	634.5***
		(50.51)	(14.6)	(38.6)	(81.8)
	Second studies	580.2***	133.6***	460.1***	1,510***
		(71.68)	(29.1)	(88.2)	(191.7)
	Tertiary studies	457.0***	1.70e-10	218.6**	1,170***
		(85.25)	(40.3)	(100.9)	(210.3)
Occupation status (student)	Housekeep-er	-216.5*	-292.9***	-258.6	-730.6**
		(129.21)	(74.7)	(199.2)	(347.4)
	Retiree	-393.5***	-292.9***	-258.6	-646.6*
		(137.38)	(76.6)	(200.2)	(358.0)
	Employed	955.6***	723.0***	1,736***	2,219***
		(114.07)	(70.0)	(165.0)	(313.1)
	Unem-ployed	-51.35	-204.5***	-35.41	-58.6 (372.9)
		(140.12)	(77.4)	(201.5)	
Observations		16,409	16,409	16,409	16,409
Pseudo R^2		0.146	0.1259	0.1405	0.1574
Machado-Santos Silva test			1630.23	1100.73	577.73

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent (standard errors are presented in parentheses)

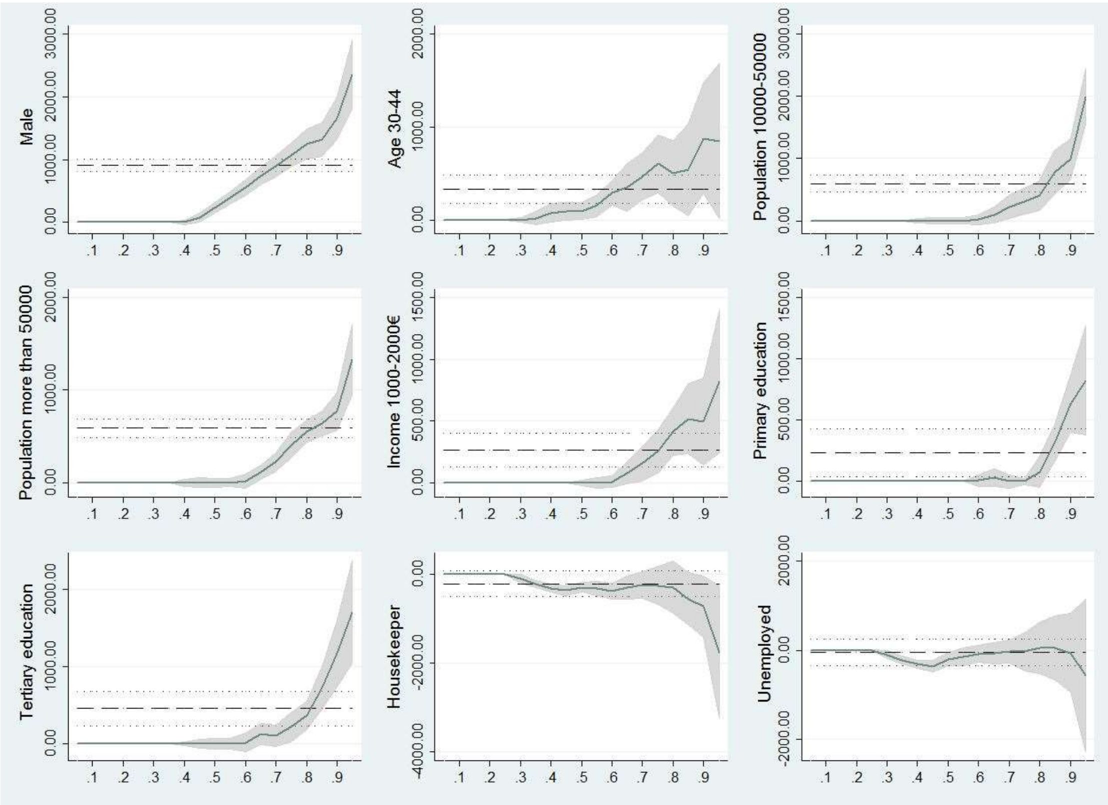
Economic variables: Income is highly significant in all quantiles and for almost all categories. The level of education seems to follow an inverted U-shaped curve in some quantiles, which is consistent with findings in Santos et al. (2013) for European cities, where the more highly educated tend to be associated with a higher proportion of low emitting modes and with greater public transport use. A divergence is found between OLS and quantile outcomes in relation to the impact of primary education: OLS identifies a positive effect of primary education vs no-education, while at the 0.5 and 0.75 quantiles there is a non-effect.

In Figure 3 we plot the estimated coefficients for different quantiles and variables, and their 95% confident interval.[9] We only plot variables whose quantile and OLS estimates differ. Superimposed on the plot we represent the ordinary least squares estimation of the mean effect (dashed line), and the 90% per cent confidence interval (dotted lines). The graph illustrates how the impact on daily CO_2 emissions of the demo-geographic and economic variables vary over quantiles, and how the

[9] Stata software does not allow us to perform robust standard errors. Figure 2 includes standard errors obtained using the simultaneous interquantile procedure, with the same weight for each quantile.

magnitude of these effects at various quantiles differ considerably from the corresponding OLS coefficient, even in terms of the confidence intervals around each coefficient. Note also that these coefficients are significantly different from zero for most quantiles, especially for the highest ones, while these coefficients are zero for the low quantiles. By way of example, this means being male, living in a medium-sized municipality and having tertiary education impacts differently. Thus, a policy design that focuses solely on higher educated individuals or males would be erroneous because these are not differentiating factors. Note that the quantile regression estimates lie at some point outside the confidence interval for OLS, suggesting that covariate impacts are not the same for all emitters. These results support our hypothesis that the majority of socioeconomic factors do not have an equal impact on individuals' emissions and emitter types need to be separated. This finding is typically ignored when using models that only consider average pollutant households or high emitters. In contrast, by using quantile regression we are able to analyze these differential impacts.

Figure 3. Selection of estimated socioeconomic parameters by quantile with 95% confidence limits



Furthermore, for most of the variables and the top quantiles we obtain higher confidence bands. As such, high emitters appear to be less predictable than other groups of emitters. In general, and as we have discussed above, differential impacts are found when we compare the extreme quantiles and the average emitter. Here again, the use of quantile regression enables us to take these differences into account and to analyze their dimensions.

5.2 Logistic regressions

We regress logit models in investigating the characteristics of individuals in order to check the robustness of our previous results. We sort the individuals according to their daily CO_2 transport emissions and then classify them as either high emitters (top 10%) and non-high emitters (other 90%). Following the method used by Ko et al. (2011) for high emitters in the Seoul metropolis area, we create a dichotomous variable with a value of 1 if the individual is a high emitter and 0 otherwise. The top 10% of emitters are responsible for 49% of total CO_2 mobility emissions, producing more than 5,532 g CO_2 per day. Additionally, we undertake a second classification creating a second dummy variable: non-emitters (bottom 38.5%) given a value of one, and zero otherwise (remaining 61.5%).While the logistic regression conducted on the high emitters serves as a robustness check, the non-emitter logistic regression adds new information as the lower quantiles were not previously considered.

A binary logistics model allows us to examine the way in which socioeconomic characteristics affect an individual's probability of being a high emitter and, in the other model, their probability of being a non-emitter. Table 4 shows the impacts of the demo-geographic and economic variables on individual mobility emissions. Note the pseudo-R^2 value of 0.118-0.122 cannot be considered low, as we conduct logistic regressions. In addition, the Hosmer-Lemeshow statistic for 10 groups suggests that the model fits the model specification satisfactorily.

High emitters: The odds of a male being a high emitter are 112% higher than those of a female. Age only presents a clear pattern in the case of the 30- to 44-year olds, who present a higher probability of being high emitters. For those older than 65, the probability of being a high emitter decreases in comparison with those under the age of 30. Living outside the city of Barcelona increases the probability of being a high emitter. Furthermore, individuals in Barcelona present a lower probability of being high emitters. Family monthly income presents a clear pattern, with the probability of being a high emitter increasing with income. For a top income family, the odds of being a high emitter are four times higher than those for a family with less than 1000€ per month. The probability of being a high emitter also increases with level of education. However, overall this factor appears to present an inverted U-shaped curve, since those with secondary education are more likely to be high emitters of CO_2 than those with tertiary education. In relation to occupational status, being employed significantly increases the probability of being a high emitter compared to the reference group of students, but no other significant differences are found (except being a homemaker decreases the probability).

Non-emitters: The results for non-emitters are clearer than those for high-emitters, with the outcomes for most variables being the inverse of those obtained for high emitters. Thus, being male decreases the probability of being a non-emitter, while those in the upper age levels are more likely to be non-emitters. A higher income and a higher level of education are associated with a lower probability of being in this group. Homemakers and retirees have odds of 150% of being non-emitters compared to the reference group of students. However, having a job reduces the probability of being a non-emitter by 25%.

Table 4. Logistic regressions on high emitters

Family Variable	Variable (reference level)	Levels	High emitters		Non emitters	
			Coefficient (Standard error)	Odds-ratio	Coefficient (Standard error)	Odds-ratio
Demo geographic	Gender (Female)	Male	0.750*** (0.054)	2.117	-0.202*** (0.039)	0.817
	Age (Under 30)	30-44	0.184** (0.075)	1.202	0.0170 (0.065)	1.017
		45-64	0.0374 (0.083)	1.038	0.128* (0.068)	1.136
		Above 65	-0.395** (0.193)	0.674	0.209** (0.096)	1.233
	Hometown inhabitants (Barcelona)	< 10,000	0.938*** (0.101)	2.554	-0.708*** (0.098)	0.493
		10,000-50,000	0.497*** (0.078)	1.643	0.0182 (0.055)	1.018
		> 50,000	0.655*** (0.061)	1.925	0.0301 (0.041)	1.031
Economic	Family monthly income (< 1000 €)	1000-2000 €	0.449*** (0.109)	1.566	-0.276*** (0.051)	0.759
		2000-3000 €	0.715*** (0.114)	2.043	-0.543*** (0.063)	0.581
		3000-4000 €	0.821*** (0.127)	2.272	-0.632*** (0.085)	0.532
		4000-5000 €	1.018*** (0.155)	2.767	-0.784*** (0.132)	0.456
		> 5000 €	1.372*** (0.161)	3.942	-0.906*** (0.153)	0.404
	Educational level (no studies)	Primary studies	0.802*** (0.251)	2.23	-0.382*** (0.075)	0.682
		Secondary studies	1.122*** (0.253)	3.072	-0.716*** (0.082)	0.489

	Tertiary studies	1.007*** (0.256)	2.737	-0.708*** (0.088)	0.493
Occupation status (student)	Housekeeper	-0.367* (0.214)	0.692	0.901*** (0.113)	2.462
	Retiree	-0.302 (0.200)	0.74	0.913*** (0.116)	2.491
	Employed	0.761*** (0.133)	2.14	-0.297*** (0.093)	0.743
	Unemployed	0.0169 (0.185)	1.017	0.475*** (0.113)	1.608
Observations		16409		16448	
Pseudo R^2		0.122		0.137	
Cox-Snell R^2		0.081		0.164	
Nagelkerke		0.165		0.225	
Hosmer-Lemeshow statistic (10 groups)		11.04 (p-value 0.199)		13.61 (p-value 0.09)	

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent (standard errors are presented in parentheses)

It was suggested above that the Seoul and Barcelona metropolitan areas have certain similarities in terms of their population density and economic performance. However, the respective results obtained in relation to high emitters differ somewhat. While Ko et al. (2011) report a pronounced U-shaped curve for age, we did not find the same effect. As for income, while both studies report the same pattern, the impact is much greater in our study. Likewise, the impact of the gender variable is much greater in the Barcelona area. As such, cultural factors combined with country- and city-specific citizen characteristics impact differently depending on the geographical zone.

5.3 Mobility expenditure impacts

The daily mobility survey seeks to provide a reliable sample of citizens' mobility patterns in the Barcelona metropolitan area and, to this effect, the results reported in the previous section and the impacts described do not contain any relevant bias. However, the observations available in this section correspond to a non-random selection, given that the share of citizens providing information about their mobility expenditure (1) corresponds to just a quarter of the above, and, more importantly, (2) the respondents are affected by some selection bias.[10]

[10] We are aware that there is a potential problem of endogeneity between mobility expenditure and CO_2 emissions. However, we believe this should not be a serious concern in this study. As explained above in section 3.1, neither fuel prices, tolls or parking pricing policies are decided in relation to CO_2 emissions (or that of any other pollutant). Moreover, the vast majority of the survey population live in an area that is subject to an integrated mass transit fare, which is unrelated to emissions generated by the particular trip. Finally, in Spain, taxation on fuel is

Table 5. Descriptive statistics

	Sample		Subsample for mobility expenditure variables	
	Mean	Standard deviation	Mean	Standard deviation
Emission household (g CO_2/day)	1582.9	2870.65	2704.1	3231.28
Income (dummies) (2 between 1000 and 2000 €)	2.27	1.125	2.74	1.163
Age (dummies) (4 between 30 and 44 years)	4.08	1.617	4.39	0.935
% employed	0.427	0.4947	0.713	0.4525
% students	0.246	0.4308	0.044	0.2045
% housekeepers	0.087	0.2813	0.033	0.1793
Number of observations	16409		4002	

In Table 5, we report the mean and standard deviation for several variables. It can be readily seen that this subsample has a higher proportion of high income earners, respondents are older than those in the overall whole sample, and the proportion of employed people is higher – in contrast to the frequency of students and homemakers. All these characteristics of the subsample are consistent with an upward bias in the proportion of high emitters – while 10% of individuals in the whole sample were high emitters, 17.3% are in this subsample. Hence, in the subsample high emitters are overrepresented. This bias cannot be corrected; thus, we need to be extremely cautious when interpreting our results.[11]

Bearing in mind these caveats, the results in Table 6 suggest that expenditure on public transportation is a good measure of CO_2 emissions in the case of the mobility variables: the higher the spending on public transport, the lower the carbon emission rates.[12] This holds for almost all cases across all quantiles. Results for expenditure on car fuel and parking are always positive and significant for all quantiles: the higher the spending on fuel and parking, the higher the emissions of CO_2. Furthermore, we find that lower emission rates are associated with higher expenditure on tolls. This result might seem counterintuitive at first sight, but we believe it to be a logical outcome in the case of the metropolitan area of Barcelona. While it might indeed be surprising if all motorways accessing the inner city of

determined by the central government, and so there are no inter-territorial differences. Besides, central government has always been reluctant to introduce a carbon tax.

11 It is worth noting that this type of bias is not exceptional among the surveys used in the studies described herein. For instance, the sample Ko et al. (2011) use in their study of the Seoul metropolitan area contains 54.7% males, a figure that is higher than the actual percentage of males in the population.

12 Table-A1 reports a logistic regression for high- and non-emitters for the expenditure variables.

Barcelona were tolled, in reality only two out of eight access motorways are tolled. This means our result is consistent with a lower frequency of private car trips on access corridors served by the tolled motorways. Indeed, users of vehicles that are obliged to pay tolls are subject to a monetary disincentive to use their private vehicles, whereas travelers that use toll-free roads do not face the same monetary disincentive.

Table 6. Quantile regression with mobility expenditure variables

Family variables	Variable (reference level)	Levels	OLS	0.25	0.50	0.75	0.90
Demo geographic	Gender (Female)	Male	344.2***	146.7***	263.4***	388.2**	468.9*
			(103.03)	(56.9)	(96.8)	(152.7)	(305.2)
	Age (Under 30)	30-44	63.03	-196.2*	-157.2	63.2	7.4
			(148.44)	(101.4)	(160.7)	(246.7)	(407.1)
		45-64	-65.38	-207.8**	-307.1*	-108.9	-25.59
			(156.81)	(104.5)	(159.8)	(260.5)	(376.2)
		Above 65	-193.7	-180.7	-336.8*	-407.3	-1,038
			(253.92)	(118.5)	(197.6)	(397.4)	(862.3)
	Hometown Inhabitants (Barcelona)	< 10,000	820.8***	245.7*	400.0*	1,141***	1,988**
			(259.70)	(140.9)	(221.3)	(405.8)	(858.5)
		10,000-50,000	135.2	48.06	-29.72	-38.67	406.2
			(140.38)	(73.8)	(110.3)	(190.2)	(365.6)
		>50,000	388.0***	56.75	173.2*	376.7**	636.1*
			(108.67)	(50.4)	(91.1)	(161.0)	(348.3)
Economic	Family monthly income (<1000 €)	1000-2000 €	347.5**	75.11*	344.4***	755.4***	437.5
			(154.34)	(51.7)	(109.3)	(191.7)	(547.1)
		2000-3000 €	541.5***	153.9**	365.0***	970.5***	821.7
			(175.87)	(69.9)	(133.8)	(235.7)	(605.8)
		3000-4000 €	380.6*	197.4*	464.6***	1,008***	19.94
			(198.48)	(104.2)	(177.8)	(287.0)	(625.1)
		4000-5000 €	754.7**	189.9	415.4	1,343**	1,817**
			(314.48)	(167.5)	(270.0)	(537.7)	(875.2)
		> 5000 €	693.2**	382.6	908.6***	1,294***	683.1
			(312.92)	(321.8)	(310.2)	(379.1)	(926.9)
	Educational level (no studies)	Primary studies	148.5	1.56e-11	118.7	-85.81	-88.41
			(281.74)	(83.9)	(152.5)	(567.7)	(858.2)
		Second studies	548.3	87.03	399.1**	372.9	563.1
			(293.86)	(95.7)	(174.0)	(564.1)	(797.8)
		Tertiary studies	333.0	25.59	220.1	-46.77	129.0
			(299.77)	(102.8)	(187.3)	(577.1)	(805.8)
	Occupation status (student)	Housekeeper	-1,287***	-401.0*	-787.2***	-1,296***	-3,153***
			(347.3)	(206.1)	(266.0)	(474.9)	(885.8)
		Retiree	-1,403***	-586.6***	-1,154***	-1,259***	-1,946*
			(332.4)	(205.5)	(265.8)	(474.1)	(1033.7)
		Employed	-327.6	-90.74	21.97	233.5	-762.5
			(267.5)	(192.4)	(223.1)	(372.7)	(789.2)
		Unemployed	-1,213***	-511.5***	-955.9***	-868.1**	-1,926**
			(324.3)	(198.3)	(258.6)	(440.4)	(928.9)
Mobility monthly expenditure	Public transport (0 €)	(0-20] €	-294.1**	-89.9	-285.5**	-364.5	-1,003**
			(145.2)	(74.0)	(129.2)	(264.3)	(426.8)
		(20-40] €	-1,180***	-318.1***	-1,030***	-1,638***	-2,232***
			(163.2)	(84.2)	(147.1)	(278.7)	(561.6)
		(40-60] €	-1,428***	-408.4***	-1,264***	-1,965***	-2,596***
			(241.9)	(123.8)	(213.7)	(360.9)	(662.9)

	> 60 €	-980.3**	-323.8*	-921.9***	-1,770***	-2,173*
		(473.7)	(163.1)	(254.9)	(485.2)	(1,254.7))
Car fuel	(0-50] €	537.6***	89.9	395.5***	743.3***	1,048
(0 €)		(177.6)	(59.00)	(111.44)	(210.7)	(800.4)
	(50-100] €	1,651***	378.0***	1,342***	2,295***	3,071***
		(204.6)	(86.3)	(162.2)	(264.6)	(853.4)
	(100-150] €	2,470***	716.3***	2,331***	3,582***	4,903***
		(293.5)	(196.0)	(455.8)	(400.66)	(1,018)
	> 150 €	2,021***	665.5***	1,418***	2,876***	3,786***
		(337.9)	(227.4)	(342.8)	(522.1)	(1,387)
Toll	(0-20] €	-350.5***	-95.7*	-259.7***	-675.6***	-752.2**
(0 €)		(110.6)	(50.6)	(87.7)	(159.5)	(311.9)
	(20-40] €	-520.3***	-131.5*	-414.2***	-955.0***	-815.9**
		(166.2)	(74.3)	(129.8)	(214.6)	(374.7)
	(40-60] €	-693.9***	-203.9	-572.3**	-916.4	-1,139*
		(265.6)	(152.2)	(229.6)	(750.5)	(625.0)
	> 60 €	-593.9*	-353.7	-547.3**	-760.5	-451.1
		(355.5)	(283.9)	(268.0)	(491.2)	(1,083.8)
Park far home	(0-20] €	587.7*	56.8	476.5***	1,065***	1,528***
(0 €)		(109.2)	(52.7)	(116.7)	(174.19)	(321.0)
	(20-40] €	760.8***	412.9*	1,071***	900.0*	1,446**
		(257.6)	(218.9)	(228.7)	(505.5)	(987.3)
	>40 €	1,503***	630.7**	1,460***	2,294***	2,385***
		(299.7)	(302.6)	(460.9)	(557.1)	(509.7)
Observations		4002	4002	4002	4002	4002
Pseudo R^2		0.163	0.144	0.157	0.160	0.155
Machado-Santos Silva test			496.02	322.71	145.78	65.77

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent (standard errors are presented in parentheses)

Generally, we find similar patterns when comparing the OLS and quantile estimation results. However, a number of interesting differences emerge, especially when we examine the results for the mobility expenditure variables. In the case of expenditure on tolls, OLS values show a significant negative effect for all levels of expenditure, while quantile estimations indicate that non-emitters (0.25) present little or no effect of tolls, reflecting the fact that these individuals probably make little use of private vehicles. For the highest level of expenditure on tolls, we find a very limited reaction for all groups of emitters, with the exception of the median group. The fact that the highest emitters with the highest toll expenditure show no significant reaction to tolls might reflect a lower demand-toll elasticity for the wealthiest private car users, which is consistent with the results linking the highest emitters with the highest levels of monthly income.

In the case of expenditure on parking away from the home, we also find interesting differences. Non-emitters present a lower reaction to parking expenditure, which is only highly significant for high levels of expenditure. Likewise, the coefficient (intensity) of the reaction to parking expenditure increases sharply when we consider groups of high emitters. These differences – as well as those related to toll expenditure – which cannot be observed from the OLS results, have interesting implications for public policy, as parking prices and tolls are two policies that local/regional authorities can regulate.

6 Discussion and policy implications

Based on the information obtained from the quantile and logistic regressions, we can define the socioeconomic traits of the different emitters. Non-emitters tend to be female, retirees, homemakers and/or unemployed. Similarly, non-emitters tend to be older and to live in the inner city of Barcelona or in large neighboring towns. However, educational status is much more difficult to link to a specific pollutant profile. The profile of the low emitter is very similar to that of the non-emitter. In contrast, high emitters tend to be male, middle-aged, employed and residents of the smallest cities relatively far from the city of Barcelona.

In keeping with our expectations – taking into account the characteristics of our data, including outliers, skewed distribution, etc., the quantile regressions performed better than OLS regression, although we need to exercise some caution given the upward bias for high emitters in the mobility expenditure subsample. The coefficient signs tend to be similar for the different variables, but it is worth noting that the statistical significances differ when using quantile regressions. Interestingly, this is the case for toll expenditures and for spending on parking away from home. Likewise, notable differences are found in the intensity of the coefficients of these (as well as other) variables.

By employing quantile regression, we observe that impacts differ considerably across individuals. These impacts follow an increasing or decreasing trend (according to different socioeconomic traits), with few structural changes between quantiles, but with different impacts across them. Most socioeconomic factors do not have an equal influence on pollutant emitters of different levels; however, conventional methodologies are unable to assess this, as they only analyze average pollutant emitters or top emitters.

If we focus specifically on our mobility expenditure results, we find, in general, that expenditure on car fuel and parking is associated with higher CO_2 emissions. In contrast, using public transportation is associated with lower emissions. The same is true of commuting on toll roads. Recall that just two of the eight inner city access motorways are tolled; hence, commuting on a toll road is likely to be associated with a lower usage of private vehicles. Beyond these general patterns, our quantile estimations reveal differences in significance and intensity (coefficients) between quantiles and regressions. Travelers that spend more on tolls and who are high emitters seem to be more reactive to tolls, although this does not hold for the highest level of expenditure. Similarly, the response to spending on parking away from home increases for travelers that spend more and who emit more.

These results should be of interest in devising more effective policies for the metropolitan area of Barcelona. While car fuel prices lie outside the control of local and metropolitan authorities, other policies can be implemented at the metropolitan level in relation to parking fees, tolls, and the supply and pricing of public transportation. Increasing parking charges in the inner city of Barcelona would help reduce emissions and the impact of this measure would be greatest among individuals that spend more on parking, and among those in the top emitting quantiles, as the results from our quantile regressions show. In the same vein, extending tolls to all motorways accessing the inner city should reduce the use of private vehicles and, thus, have the potential to reduce emissions. Our results suggest that this might not, however, be significant for the group of travelers that spend more on tolls and emit more; yet, decreasing congestion should also relieve the emissions of these travelers that use their private vehicles and who show little sensitivity to tolls.

Clearly, suggesting that increasing parking away from home costs and introducing tolls on all access motorways would reduce emissions is unsurprising, as this outcome has been reported in many cities that have implemented measures of this type. Having said that, however, the results from our quantile regressions point to the particularly intense effect of such measures in the Barcelona metropolitan, given the coefficients reported for the high emitters' quantiles for expenditure on parking and (at least, until very high levels) on tolls. Furthermore, if the net revenues from these cost-increasing policies on parking and tolls were devoted to improving public transportation supply, this would further help reduce CO_2 emissions.

Within this framework, we suggest that the application of the quantile regression methodology is of interest not only for scholarly analysis, but also for policy making –particularly policies designed to have long-term effects. In practice, household travel surveys are available for most large conurbations, which means the CO_2 emissions of each trip can be calculated and the data treated with quantile regression so that specific analyses can be undertaken for all areas. This means quantile regression is a methodology that governments can use to improve their understanding of the socioeconomic profile of different types of emitter in different population groups and this information should help them design effective and specific transport policies to mitigate the greenhouse effect.

7 Conclusion

Cutting CO_2 emissions attributable to urban mobility has become a challenge for large cities. This paper has sought to address a gap in the literature by comparing different emitters of carbon dioxide and in terms of their socioeconomic characteristics. An examination of the personal factors affecting the modal share has been undertaken in recent years, but conventional models encounter difficulties in explaining travel patterns and behavior, since the latter vary according to lifestyles, personal preferences and attitudes. Indeed, certain characteristics are significant in accounting for factors that affect CO_2 mobility emissions.

Our research contributes to the literature by adopting an innovative methodology – that of, quantile regression – to explain the relation between socioeconomic variables and transport emissions in urban areas according to different levels of emission. In so doing, we contribute to filling the gap in the knowledge of differences between CO_2 high emitters, average emitters, low emitters and non-emitters.

Based on our analysis of socioeconomic characteristics across quantiles of pollutant individuals, and the impact we report for variables related to mobility expenditure, we have been able to confirm the potential impact of several measures that could be highly effective in reducing emissions in the case of Barcelona, as well as to gain a better understanding of their effects on different groups of emitters. As such, quantile analysis appears to be a useful tool for analyzing the behavior of groups of emitters in different urban conurbations. We recommend that governments analyze available data from travel surveys using quantile regressions to verify whether their mobility policies are being effectively implemented, and to obtain a clearer picture of their potential impacts on different groups of emitters.

The household travel survey data used here have several shortcomings. First, we have insufficient information to compute the CO_2 emissions of commuters that use more than one travel mode and so we were forced to omit these journeys. This reduced the available number of observations by 20%. Indeed, future surveys would be greatly improved if they asked respondents to indicate where they changed modes of transport. Second, data are unavailable for some socioeconomic characteristics, including private vehicle ownership and household size. And third, the sample bias with regard to mobility expenditure means high-emitters – that is, males, in employment and with high incomes – are over- represented. However, it is our belief that these shortcomings have not seriously affected our main findings and conclusions.

The type of analysis conducted here should be an effective tool for analyzing mobility behavior in different metropolitan areas. Indeed, if other cities can begin to fill this information gap, they should be able to make more accurate and more correct policy decisions. Indeed, we are aware, as shown by a sizable body of evidence in the literature, that most of the CO_2 emissions related to mobility are attributable to factors and attitudes intrinsic to each person, which means an individual's socioeconomic characteristics only account for a part of this variability. Mobility patterns differ from one city to another as do the socio-economic characteristics of their respective citizens.

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APPENDIX

Table A1. Logistic regressions with mobility expenditures

Family variables	Variable (reference level)	Levels	High emitters		Non emitters	
			Coefficient (Standard error)	Odds-ratio	Coefficient (Standard error)	Odds-ratio
Demo geographic	Gender (Female)	Male	0.247** (0.100)	1.281	0.061 (0.090)	1.063
	Age (Under 30)	30-44	-0.0157 (0.135)	0.984	0.189 (0.136)	1.208
		45-64	-0.102 (0.146)	0.903	0.179 (0.144)	1.196
		65 and >	-0.193 (0.322)	0.825	0.238 (0.218)	1.269
	Hometown inhabitants (Barcelona)	< 10,000	0.655*** (0.185)	1.925	-0.209 (0.199)	0.811
		10,000-50,000	0.0434 (0.140)	1.044	0.052 (0.119)	1.053
		> 50,000	0.412*** (0.109)	1.509	0.0923 (0.092)	1.096
Economic	Family monthly income (less than 1000 €)	1000-2000 €	0.0758 (0.200)	1.079	-0.504*** (0.130)	0.604
		2000-3000 €	0.322 (0.206)	1.380	-0.630*** (0.145)	0.533
		3000-4000 €	0.269 (0.228)	1.309	-0.570*** (0.173)	0.566
		4000-5000 €	0.441 (0.276)	1.554	-0.746*** (0.249)	0.474
		> 5000 €	0.654** (0.285)	1.923	-0.761*** (0.279)	0.467
	Educational level (no studies)	Primary studies	0.221 (0.495)	1.247	-0.259 (0.264)	0.772
		Secondary studies	0.446 (0.496)	1.562	-0.583** (0.270)	0.558
		Tertiary studies	0.168 (0.501)	1.183	-0.487* (0.277)	0.615
	Occupation status (student)	Housekeeper	-0.959** (0.423)	0.383	1.485*** (0.348)	4.413
		Retiree	-0.709** (0.344)	0.492	1.659*** (0.324)	5.253
		Employed	-0.0313 (0.239)	0.969	0.608** (0.285)	1.837
		Unemployed	-0.902*** (0.339)	0.406	1.460*** (0.315)	4.304
Mobility monthly expenditure	Public transport (0€)	(0-20] €	-0.0462 (0.122)	0.955	0.322*** (0.117)	1.380
		(20-40] €	-0.695*** (0.161)	0.499	0.348** (0.139)	1.416
		(40-60] €	-1.095*** (0.299)	0.334	0.346 (0.219)	1.413
		> 60 €	-0.868** (0.376)	0.420	-0.013 (0.338)	0.987
	Car fuel (0 €)	(0-50] €	0.0263 (0.329)	1.027	-0.589*** (0.190)	0.555

	(50-100) €	0.892*** (0.330)	2.441	-0.876*** (0.201)	0.416
	(100-150] €	1.376*** (0.349)	3.958	-1.106*** (0.256)	0.331
	> 150 €	0.985*** (0.366)	2.677	-1.311*** (0.297)	0.270
Toll (0 €)	(0-20] €	-0.228** (0.106)	0.795	0.278*** (0.0907)	1.321
	(20-40] €	-0.347** (0.162)	0.707	0.309** (0.139)	1.362
	(40-60] €	-0.393* (0.227)	0.675	0.239 (0.223)	1.270
	>60 €	-0.552** (0.234)	0.576	0.561** (0.238)	1.752
Park away home (0 €)	(0-20] €	0.537*** (0.101)	1.711	-0.204** (0.090)	0.816
	(20-40] €	0.494** (0.194)	1.638	-0.565*** (0.219)	0.568
	>40 €	0.882*** (0.184)	2.415	-0.476** (0.235)	0.621
Observations		4005		4009	
Pseudo R^2		0.118		0.10	
Cox-Snell R^2		0.103		0.104	
Nagelkerke		0.171		0.154	
Hosmer-Lemeshow statistic (10 groups)		7.56 (p-value 0.48)		5.3 (p-value 0.725)	

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent (standard errors are presented in parentheses)

CHAPTER 3

EFFECTS OF THE 80 KM/H AND VARIABLE SPEED LIMIT ON AIR POLLUTION

1 Introduction

The European Union sets air pollution limits in order to prevent, avoid or reduce harmful effects on its inhabitants and the environment. Air quality status must be maintained where it is acceptable or improved where limits are exceeded. Following EU guidelines and mandates, different policies have been implemented by EU member states to reduce pollution in large urban agglomerations. Depending on the internal distribution of power in each country, national or regional governments have been responsible for formulating and implementing plans to improve air quality. In July 2007 the regional government of Catalonia, Spain, passed the Action Plan for Improving Air Quality in the Barcelona Metropolitan Region to reduce air pollution.

The most important and newsworthy measure was to limit the maximum speed on motorways in the Barcelona metropolitan area to 80 km/h. The 80 km/h speed limit affects 19 municipalities with around 1.35 million inhabitants. The new limit did not actually affect the city of Barcelona because it had no motorways with speed limits above 80 km/h even before the measure was implemented. Before the new limit was imposed, 63.2% of the affected roads had a speed limit of 100 km/h and 20.4% of the motorways had a previous limit of 120 km/h. According to the regional government, the expected effects of the measure were a reduction in pollution and a subsequent increase in life expectancy, and compliance with European legislation on air quality and noise reduction.[13] In February 2011 the newly-elected regional government abolished the 80 km/h maximum speed limit, thus fulfilling its electoral commitment. Most roads returned to speed limits similar to those existing in 2007, although a central government regulation issued in March 2011 reduced the maximum speed from 120 km/h to 110 km/h on all motorways in Spain.

The second policy measure consisted of implementing variable speed limit systems on sections of the C31 and C32 access routes in January 2009 (southern motorways). On these two roads, the speed limit was to be changed according to traffic congestion situations, possible incidents involving road safety (accidents,

[13] And, in different areas to ours, fewer accidents and improved traffic flows. This study does not deal with the potential effects on accidents and congestion.

incidents, road works, maintenance work, etc.), situations of pollution or poor weather conditions (rain, fog, wind, etc.). This system adjusted the limit in 10km/h increments from a maximum of 80 km/h to a minimum of 40 km/h and communicated the limits to drivers via variable message boards placed approximately every kilometre. The limits were enforced by means of radar detection and fines. In addition to the aim of improving environmental conditions, the variable speed limit system also looks after reducing stop and go. The newly elected regional government did not abolish this policy. In short, after canceling the 80 km/h speed limit, the new regional government announced the expansion of the variable speed system to other highways in the Metropolitan Area of Barcelona in the coming years.

Speed reduction policies have been analyzed on interurban and rural motorways around the world. In the case of Spain, a temporary measure of a speed limit on interurban transit (from 120 km/h to 110 km/h) was applied in 2011. This measure did imply gasoline consumption savings around 1.5-3% (Asensio et al. (2014) and Castillo-Manzano et al. (2014)).

Policymakers have different options at hand to improve the air quality in a metropolitan area, and optimizing driving speeds is one of the available strategies to reduce fuel consumption and reduce negative environmental effects from traffic (Krutilla and Graham, 2012). Limiting the maximum speed to 80 km/h or establishing variable speed systems are examples of them, and obtaining accurate evaluations of the effectiveness of these type of policies becomes extremely important. In this particular case, the speed restrictions (particularly the fixed speed limit) imposed increased travel times for vehicle users, thereby decreasing utility for road commuters (and general road users) and increasing travel costs, most notably during the hours of the day when congestion is absent. Reducing pollution could, of course, eventually offset the extra costs for car users deriving from increased travel times. But this cannot be taken for granted *a priori*. Besides, there are other alternatives for regulating car usage that have been shown to be very effective in reducing pollution, such as congestion charges, which have been implemented in recent years in important European cities, resulting in significant decreases in pollution: in London emissions of NO_x decreased by 13.4% and emissions of PM_{10} decreased by 16% (Transports for London, 2008); in Stockholm NO_x decreased by 8.5% and PM_{10} by between 10% and 14% (Eliasson et al., 2009); and in Milan emissions of NO_x decreased by 17% and emissions of PM_{10} by 23% (AMMA, 2008; Rotaris et al., 2010). Therefore, because policymakers have different options available for reducing pollution, the effects of alternative policies need to be accurately evaluated and compared.[14]

This study empirically evaluates two policies. We aim to evaluate whether lowering the speed limit from 120 km/h and 100 km/h to 80 km/h brought about improvements in air quality in terms of pollutant concentrations of nitrogen oxides (NO_x) and particles less than 10 µm (PM_{10}) generated by road traffic, and also the environmental effects from variable speed in a particular area. We contribute to the literature in several ways. First, whereas most of the previous studies on the environmental effects of reducing speed limits on metropolitan motorways are made up of computations based on theoretically established parameters, we conduct an empirical analysis for both policies using actual data on air quality and checking for meteorological factors. Second, our empirical evaluation includes urban and interurban areas subject to the speed reduction together with areas where the legal speed limit was unchanged. Third, in this study we use actual data (travel speeds as well as air quality pollutants and other related data) for a much longer period than previous studies (five years). In this way we aim to contribute to the literature by providing a more robust and accurate policy evaluation. Fourth, we combine 80 km/h speed limit and variable speed limit, both pursuing similar targets, in the same geographical area using differences in differences. To our knowledge, this is the first study that compares two speed policies that pursue pollution abatement. Fifth, all previous econometric approaches have been conducted on the mean, and so overlook the fact that policy impacts might vary with levels of pollution. Here, the quantile regression approach allows us to analyse the effects of speed limit policies at different pollution concentrations. As such, the main contribution of this paper is determining whether different atmospheric scenarios have a differentiated impact on a given speed limit policy. To the best of our knowledge, this is the first time that this methodology has been applied to such an analysis.

2 Related literature

It is widely accepted that lower speed means lower emissions from road traffic, following the well-known U curve type of relationship between traffic emissions and average speed (Litman and Doherty, 2009), especially under constant speeds (Eerens et al., 1993; LAT, 2006). However, when accelerations and decelerations are introduced into the model, gains from lower emissions due to the reduction in

[14] Emissions of CO_2 also decreased in all cities: by 16% in London, 16% in Stockholm and 14% in Milan. See Albalate & Bel (2009) for a review of the effects of congestion-charging policies.

speed are much lower (Int Panis et al., 2006). For identical distances, emissions are larger for traffic stops and starts than if the vehicles are moving at a constant speed. Because of this, speed-reduction policies could eventually be counterproductive as far as pollution emissions are concerned. Reducing urban speeds from 50 km/h to 30 km/h, for instance, would not reduce emissions (Int Panis et al., 2011). Simulations conducted show that tailpipe vehicle emissions of PM10 are higher at speeds below 40 km/h due to incomplete combustion. NOx emissions, on the other hand, increase above 100 km/h because of high temperature combustion. The lowest NO_x emissions are to be found between 60 and 100 km/h (LAT, 2006).

Most previous studies on the environmental effects of speed limits reduction on metropolitan areas are made up of computations based on theoretically established parameters. Keller et al. (2008), which relies on modeled effects according to changing scenarios, is an example of such approach. The simulation is conducted for Switzerland, and analyses the effects on ozone levels of vehicle speed reduction from 120 km/h to 80 km/h. They use an air quality model package and different emission factors depending on speed. The reference scenario refers to the regular maximum speed limit of 120 km/h on motorways, while the limited scenario is based on a reduced maximum speed limit of 80 km/h. The modeled effects imply a 1% reduction in concentration of ozone, which translates into an equivalent increase of about 4% in NO_x emissions.

While Keller et al (2008) is done for interurban motorways, Keuken et al. (2010) study the effect of speed limit reduction in Dutch cities. They adopt two different approaches: (a) one based on air quality monitoring in combination with dispersion modeling, and (b) one based on applying relevant emission factors relating to the change in traffic dynamics. Both methods are applied to data from Amsterdam and Rotterdam. Keuken et al. conduct a linear regression of the measured and modeled contributions for NO_x and PM_{10} for periods both with and without implementation of the 80 km/h zone. For speed limits, the study found no relevant change in PM_{10} emissions and a reduction in NO_x of between 30% and 32%, depending on the city. [15] When using traffic dynamics analysis both cities show a reduction in both pollutants of between 16% and 24%. Overall they find that reducing traffic dynamics – thereby decreasing congestion – is more important than reducing

[15] It is widely accepted that the NO_x come from anthropogenic sources, particularly the urban traffic. For PM_{10} differences may appear depending on the area, but for the city of Barcelona for the years 1999 and 2000, 54% come from anthropogenic sources (Querol et al., 2001). The fact that a much larger fraction of the automobile fleet is made of diesel cars can explain why PM_{10} has a stronger relationship with traffic in Europe than that discussed by van Benthem (2012) for the US.

average speed. Moreover, they find that the impact of the measure is more significant if a high proportion of heavy vehicles use the highway.

Computations based on simulations exist also for the metropolitan area of Barcelona, for which our own analysis is conducted. Gonçalves et al. (2008) make simulations to compare emissions of vehicles moving at different speeds, and their effects on air quality improvement. They use an emission model and include a photochemical pollution episode (17-18 June 2004). The model checks for meteorological variables, emissions from a variety of industrial and energy facilities, domestic and commercial fossil fuel use, domestic and commercial solvent use, road transport, ports, airports and biogenic emission. When the 80 km/h limit is simulated, the 24-h average NO_2 concentration over the Barcelona area decreases by 0.7% to 0.8% on the selected day. Largest reductions are observed in areas directly affected by the reduced speed limit, while results are more modest for the centre of Barcelona, with reduction of 0.1% to 0.3% for NO_2 and of 0.1% to 0.2% for PM_{10}. Still for Barcelona, Baldasano et al. (2010) conduct an evaluation using data on vehicle speeds and daily traffic in 2007 and 2008, year in which the speed limit reduction to 80 km/h was applied. They use emissions modeling to find the change in air quality. NO_x emissions decreased by 10.98% and PM_{10} emissions by 10.99% in areas where the 80 km/h speed limit was enforced, whereas both pollutants decreased by 4% over the entire metro area. Looking at the air quality levels, the study found a reduction of between 5% and 8% in NO_x concentrations in the most affected areas, whereas for PM_{10} the reduction was 3%.

Very few studies have made use of actual data on emissions before and after the speed limits where changed. Dijkema et al. (2008) study for Amsterdam aims to assess whether lowering the maximum speed limit on the city ring motorway (A10) from 100 to 80 km/h had reduced traffic-related air pollution in a neighborhood near the highway. This study used data on emissions (daily mean concentrations of NO_x and PM_{10}) from the year before and the year after the limit was introduced, data on traffic volumes and traffic congestion, and data on wind direction. They conducted linear multivariate regressions and found no significant changes for NO_x air quality improvement, while PM_{10} concentrations had decreased by 2.20 ppm.

To our knowledge, the most complete study so far is that by van Benthem (2015), who uses differences in differences methodology to analyze the speed limit changes introduced by the US federal government US in 1987, and subsequent changes in 1996. Van Benthem study focuses on intercity motorways along the west coast states, Washington State, Oregon and California. He uses actual data for three years before and after the changes of speed limits on pollution, traffic, weather and other variables from a large network of monitoring stations, and finds that NO_2 and NO_X decrease by 9 i 16% respectively, while for PM_{10} the measures were not significant. Van Benthem's work and ours share many similar methodological characteristics in the analysis of the effects of speed limits changes on pollution. The most important differences are that van Benthenm focuses on interstate motorways, while we focus on speed limits in a metro area, where congestion and environmental problems usually have bigger dimension. Furthermore, our analysis also considers the effects of variable speed regulations, which were not yet considered as a measure in the years for which van Benthem conducts his analysis.[16]

All the studies discussed above are concerned with the way in which a policy shift might impact mean pollution levels. However, it is debateable that the effectiveness of a policy is ever homogeneous across all levels of the pollution concentration distribution and that the policy is equally effective for all pollutants. If the shape of the distribution of pollution indicators changes after introducing a speed policy, the change in the mean pollution concentration is not representative of all pollution levels and, as such, cannot be generalized. Using the mean to characterize highly skewed distributions is obviously dangerous; yet, pollutants present an asymmetric statistical distribution (many days present only small to moderate concentrations and only a few present large to extreme levels) as we show in the data section. In this regard, the quantile regression methodology allows us to model policy impacts at all pollution levels.

In this regards, our work joins efforts by some researchers that in recent years have begun to evaluate the variable speed limit on real motorway data (Papageorgiou et al., 2008). The positive impact of variable speed limits on traffic safety is due to speed reduction and speed homogenization. The implementation of a system of variable speed limits generally entails an improvement in uniformity of traffic flows and congestion, has a positive effect in the treatment of shock waves, the average speed decreases and so does its variability, and therefore the number of lane change manoeuvres, and reduces the percentage of intervals too short (Hegyi et al., 2005). In addition, the homogenization of speeds decreases the number of accelerations and decelerations.

One of the key issues related to the real effect of speed limiting measures is that of enforcement by means of radar control. Indeed, radar speed control on interurban

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[16] Other additional and interesting issues analyzed in Van Benthen (2012) are, for instance, effects on fatalities. This is not a relevant issue in our area, because of the small dimension and variability of this problem in the area analyzed.

roads – accompanied by penalties, either administrative or economic – has been recommended because it produces benefits for society (Chen et al., 2006). The largest single benefit is the value of prevented injury, death and property damage. The largest single cost, on the other hand, is the value of lost travel time, followed by the direct costs of implementing the program. In the Chen et al. evaluation, radar speed control brings about net savings, although the potential environmental costs are not taken into account. It is worth bearing in mind that the presence of radars can produce accelerations and decelerations on a particular section of the motorway and this can raise pollutant emissions with respect to moderately higher constant speeds. Because of this, Coelho et al. (2005) suggest that it is better to control speed limits over distance rather than at specific points. This is relevant to our case, because the area we are studying has a very large number of speed cameras per unit area – as well as in absolute numbers– and checks are made for specific points rather than for a long section of the motorway.[17]

The dieselization process has taken place in Spain in the past years, and Gonzalez-Marrero and Marrero (2012) find that the rebound effect on CO_2 emissions caused by dieselization has been more important than its direct, technology-efficiency impact. In the province of Barcelona, in 2006, 52% of heavy and light vehicles used diesel, while in 2010 this percentage increased to 58%. However, it is worth noting that emissions of NO_x and PM from modern diesel vehicles are dramatically lower than those from older vehicles (Wallington et al., 2013).

Finally, we need to take into account the causal relationship between the growth of the economy and pollution. Chay and Greenstone (1998) demonstrates that average total suspended particulates in the United States falls not only after Clean Air Act Amendment (1970) was implemented, but also during 1981-82 recession. Changes in traffic volumes present a positive relationship with the evolution of GDP. Elasticity per ton of freight per kilometer with respect to GDP was greater than 1 in the periods analyzed (Tapio, 2005). Specific studies for Spain have shown a short-term elasticity of demand of 0.89 with respect to GDP for interurban motorways (Matas et al., 2003). Indeed, the studies on pollution in the metropolitan

[17] The reduction of atmospheric pollution from traffic can be carried out with other policies. Currie et al. (2011) evaluates the introduction of electronic toll collection, (E-ZPass), which greatly reduced both traffic congestion and vehicle emissions by comparison with stopping to manually pay a toll. Gallego et al. (2011) analyze different policies in Santiago de Chile to persuade drivers to give up their cars in favor of public transport. They find that Hoy-No-Circula (HNC) policy and the reform of public transport do not involve long-term benefits. Auffhamer, Bento and Lowe (2009) measure the 1990 Clean Air Act Amendments effects on PM_{10} background concentrations between 1990 and 2005. They find subregional differences in ambient air pollution; counties in non-attainment of pollution standards, PM_{10} abatement plans have bigger effects.

area of Barcelona mentioned above controlled for traffic levels. Baldasano et al. (2010) take into account that traffic volume decreased by 3.3%, and that the economic crisis was the main cause. As regards statistics for the city, there is an increase in traffic from 2005 to 2007, then from that year on the amount of traffic falls again. This happens on the city streets, the main roads of the city and the ring roads (Barcelona City Council, 2011). Apart from traffic reduction, the activity of other sources of pollutants such as NO_x and PM_{10} (i.e. industrial production, energy production) are correlated with economic activity, providing additional sources of changes in pollutant emissions other than changes in speed due to the speed limit policy.

3 Data

This research is based on a compilation from different entities of the regional government of Catalonia. Measurement of ambient pollutants is reported by 15 air quality surface stations in the Barcelona metropolitan area. The time period studied is from 2006 to 2010, with 1826 temporal observations being made. Earlier periods are not included due to the low number of air pollutant stations and traffic stations available. 2010 is the last year in which the 80 km/h measure was in force. Speed limits change in 2011, so we cannot extend the analysis. The network of pollutant stations is the limiting variable in our study, despite the fact that the metropolitan area of Barcelona has one of the densest networks of air quality stations in Europe (Baldasano et al., 2010). After choosing the pollutant station, we choose the traffic station in the same town. The weather station could be found outside the municipality as long as the difference in altitude between the two stations does not exceed 200 meters or the two stations are not more than 10 kilometers (Figure 1 and 2).

Two dependent variables are used: nitrogen oxide (NO_x) and particles of 10 µm or less in aerodynamic diameter (PM_{10}). These pollutants are designated as priorities; we focus our attention on them.

For NO_x, measurements are taken every hour. The daily average was calculated from all the daily observations provided by the station. The PM_{10} has a daily manual sampling, so few measurements during weekends and holidays exist. The data are provided by the Monitoring Service and Air Control of the regional government. Table A1 (appendix) shows the annual average concentration for the pollutants per year and area.

Figure 1. 80 km/h and variable speed zones

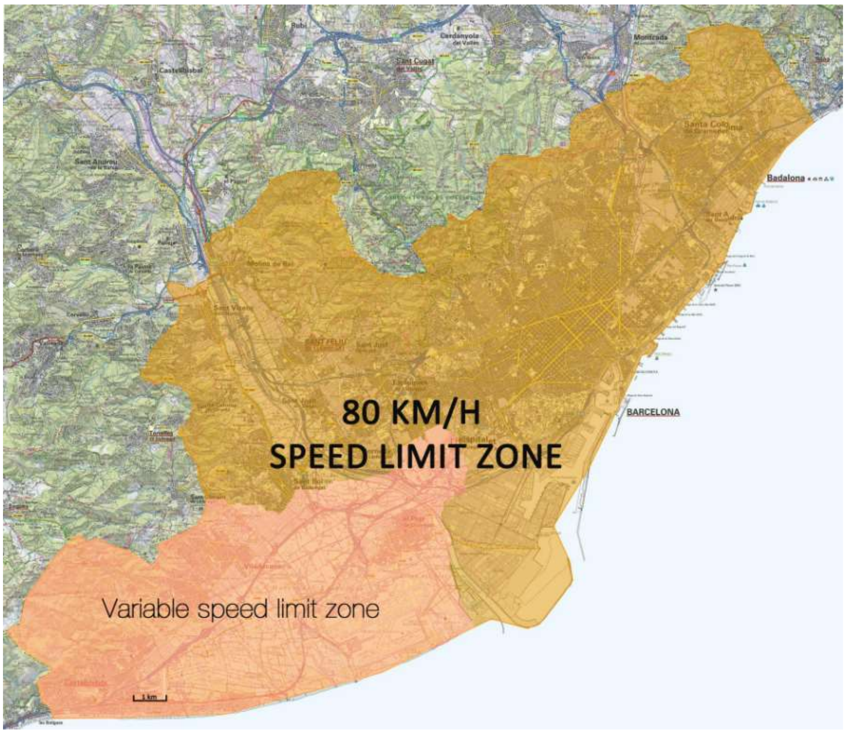
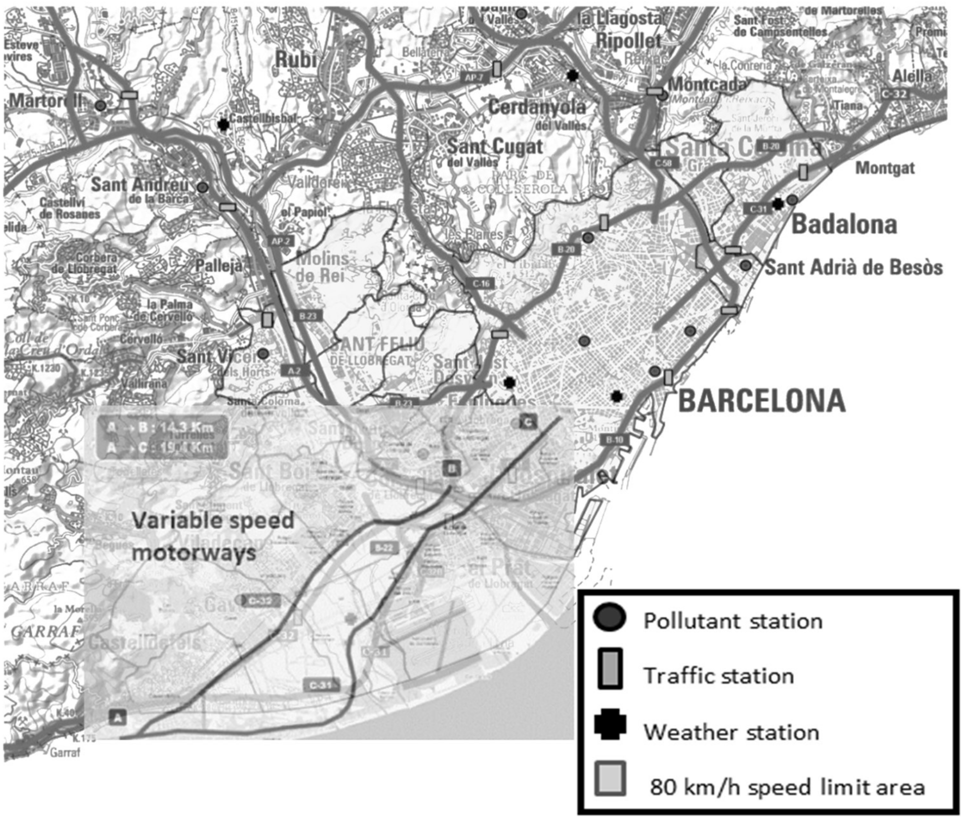


Figure 2. Pollutant, weather and traffic station locations in the Barcelona metropolitan area



The explanatory variables proposed aim to collect the pollutant source variability and its transport, sedimentation and/or reaction. Table 1 shows the explanatory variables used and their most important descriptive statistics. Traffic data includes the number of vehicles passing a specific kilometric point as close as possible to the station measuring pollution on the motorway. Measurements are taken every hour in both directions, with 48 measurements obtained daily. If we do not have these 48 measurements per day, we consider it an observation without data.

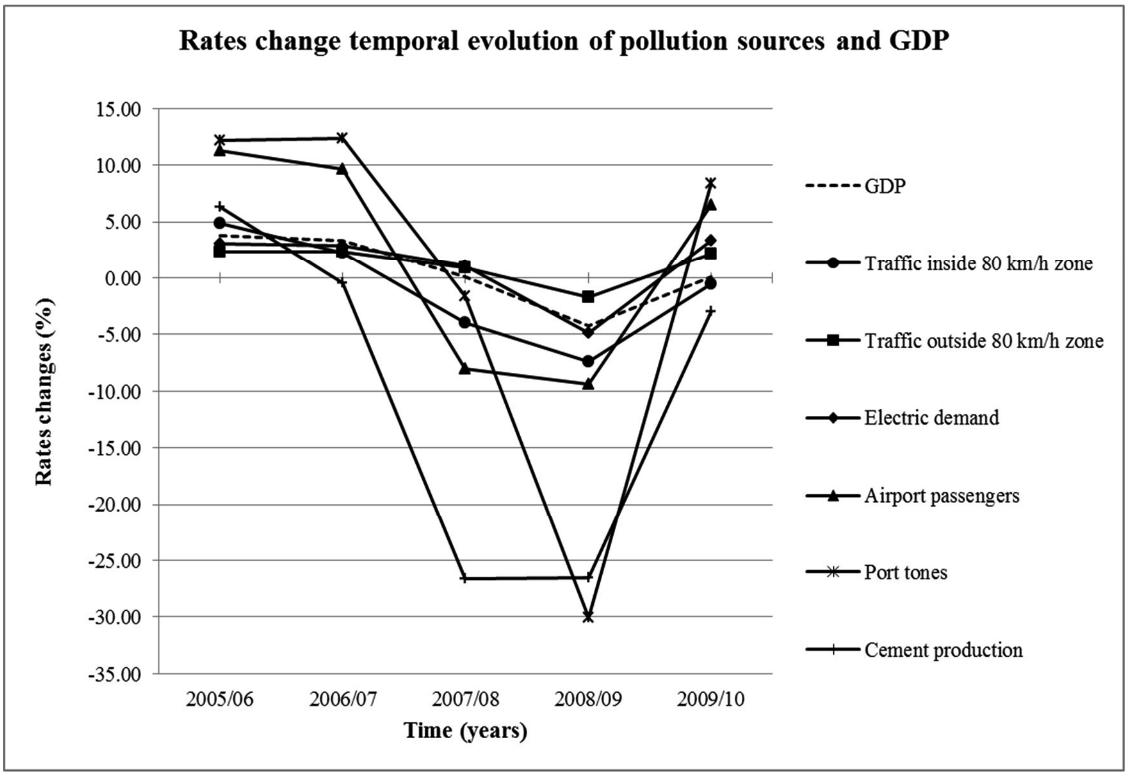
Table 1. Explanatory variables and descriptive statistics

Variables	Description	Mean	Standard deviation	Average observations per pollutant station
NO_x	Nitrogen oxide daily average concentration (µg/m^3)	84.46	59.95	1743
PM_{10}	Particulate matter daily average concentration with less than 10 µm (µg/m^3)	40.7	19.11	626
NO_x(-1) and PM_{10}(-1)	One period lag variables (1 day)			
80km/h speed limit zone	Binary variable: 1 if 80km/h speed limit is implemented. 0 otherwise	0.47	0.50	2191
Variable speed	Binary variable: 1 if variable speed is implemented. 0 otherwise	0.07	0.25	2191
Traffic	Daily vehicles on both ways (taken in logarithms)	11.352	0.43	1500
Temperature	Daily average temperature (ºC)	16.51	6.32	1472
Relative humidity	Daily average relative humidity (%)	66.85	11.65	1472
Precipitation	Daily rainfall (mm)	1.56	5.86	1473
Wind speed	Daily average wind speed (m/s)	3.298	2.74	1020
Atmospheric pressure	Daily average atmospheric pressure (hPa)	1014.8	25.42	1035

Data are provided by the Traffic Service of the regional government. As described earlier, the evolution of traffic enables us to capture the effect of the other sources that emit pollutants. Figure 3 shows the evolution of the rates of variation susceptible to pollution sources: the rate of change in the Catalonian GDP, the variation in the number of vehicles on roads inside and outside the speed limit area, the variation in electricity demand in Spain, the variation in the number of passengers carried in Barcelona airport, the tons of containers transported in the port of Barcelona, and the tons of cement produced in Catalonia. The graph shows how growth rates start to fall slightly from 2007 on. In 2009 they reach the minimum, this being between -2% and -10%, except for maritime shipping tons and cement production, which have fallen about 25-30%. As would be expected, the slowdown of the economy reduced road traffic, electricity demand, extraction activity, the number of passengers carried in Barcelona airport, and tons transported in the port of Barcelona.

The implementation of the 80 km/h speed limit and variable speed system on high capacity roads can mean that many drivers might choose to take urban secondary roads (spillage effects). We take into account that most of these city roads have a speed limit of 50 km/h. Municipalities in the metropolitan area of Barcelona have promoted policies to divert vehicles of their urban areas, such as promoting 30 km/h speed limits or increasing traffic lights. Not only are few secondary roads leading into Barcelona, but also no increases in traffic from 2008 (Barcelona City Council, 2011).

Figure 3. Change in rates of temporal evolution of pollution sources and GDP



The atmosphere is not watertight; there are many interactions. The meteorological variables aim to collect this variability. (a) Contaminants can be transported, so we include the *average daily wind speed*. (b) Pollutants are not only transported, they also undergo reaction processes. Temperature influences the reaction rate, so the *average daily temperature* is included. This is closely related to solar irradiance, which affects the reaction balance. (c) Water can bring a reactive change in the equilibrium or increase sedimentation, so we include *relative humidity* and *daily* *rainfall*. (d) Atmospheric pressure means movements of air masses ascending or descending and wind formation. At each station *atmospheric pressure* has been transformed into atmospheric pressure at sea level in order to establish comparisons. All the above meteorological variables except rainfall are daily averages calculated by the meteorological station.

The variable of central policy interest is the dummy *80 km/h speed limit area*. It takes value 1 for stations closest to roads where the 80 km/h speed limit came into force on 1 January 2008 and for stations in Barcelona city throughout the entire period of study. The variable takes value 0 in stations for the period before the measure came into force and for those stations outside the 80 km/h limit area over the entire period. For our estimations, we consider 1 January 2008 as the date the policy came into force in the affected areas.[18] The variable speed variable takes value 1 for stations closest to roads where the variable speed limit came into force on 15 January 2009 and 0 otherwise. Only 3 of 15 stations are under the influence of the variable speed policy. Stations with variable speed limit, also have a 1 in 80 km/h variable. Figure 1 above shows the geographical location of these areas.

4 Empirical strategy

This study aims to estimate the atmospheric concentration of pollutants in different municipalities in the Barcelona metropolitan area for the period 2006-2010 in order to assess the effects of the speed related policies. In the absence of a randomized trial, the method chosen is a slight extension of the differences-in-differences estimation procedure specified as a two-way fixed effects model that takes the following form

$$Y_{it} = \beta X_{it} + \gamma Z_{it} + \theta_i + \delta_t + \varepsilon_{it} \qquad (1)$$

where Y_{it} is the chosen dependent variable (air pollutant concentration), X_{it} contains the vector of time-varying control covariates, and Z_{it} are the policy dummy variable to be evaluated. As usual, θ_i and δ_t are municipal-specific and time-specific fixed effects and ε_{it} is a mean-zero random error. Municipal fixed

[18] The measure was actually introduced in December 2007, but it was introduced with no penalty for drivers who exceeded the 80 km/h limit. It was from 1 January 2008 when drivers exceeding 80 km/h were punishable by means of radar control. Therefore we consider 1 January 2008 as the date when the measure was effectively enforced.

effects control for time-invariant municipal-specific omitted variables, and time fixed effects control for municipal trends.

The key element of this differences-in-differences model is parameter γ, which measures the difference between the average change in the pollutant air concentration for the treatment group (zones with an 80 km/h speed limit or zones with variable speed limit) and the average change in the pollutant air concentration for the control group (zones with a speed limit exceeding 80 km/h or without the implementation of variable speed). Specifically,

$$\gamma = [E(\,Y_B/G = 1) - E(Y_A/G = 1)] - [E(\,Y_B/G = 0) - E(Y_A/G = 0)] \qquad (2)$$

where Y_B and Y_A denote the pollutant air concentration before and after the change in law and G=1 and G=0 denote treatment and control group observations respectively.

We also pretend to analyse the policy impacts on different polluted levels. The classical linear model evaluates the influence of the covariates, the policies and the individual effect on the mean of the dependent variable and supposes that the influence is constant in the domain of the distribution of the dependent variable. However, there are models where a constant influence will not necessarily be true. For example, policies are unlikely to be equally effective when pollution levels are high or when they are low. In a classical approach (differences-in-differences), a policy is effective if it reduces $E(Y_{it})$ significantly; thus, testing whether γ is negative and significantly different from zero leads to the conclusion that a policy action is successful. Here, we study the reduction from a much wider perspective.

When working with panel data an alternative to the linear model with individual fixed effects is the quantile regression with fixed effects, which is defined as (see Koenker and Bassett, 1978, and Koenker, 2004):

$$Q_{Y_{it}}(\tau) = \beta(\tau)X_{it} + \gamma(\tau)Z_{it} + \theta_i \qquad (3)$$

where $Q_{Y_{it}}(\tau)$ is the quantile function at τ confidence level. The model in (3) allows the influence of covariates X_{it} and Z_{it} to depend on the quantile confidence level τ. Koenker (2004) proposes estimating the parameters in model (3) simultaneously for all quantiles under study, $\tau_q,\ q = 1, \dots, Q,$ and to do so proposes solving:

$$\min_{(\beta,\gamma,\theta)} \sum_{q=1}^{Q} \sum_{i=1}^{n} \sum_{t=1}^{T_i} w_q\, \rho_{\tau_q}\big(Y_{it} - \beta(\tau_q)X_{it} - \gamma(\tau_q)Z_{it} - \theta_i\big) \qquad (4)$$

where $\rho_\tau(\cdot)$ is a function defined by Koenker and Bassett (1978) (see also Koenker, 1984) as:

$$\rho_\tau(u) = \begin{cases} \tau|u|, & u \geq 0 \\ (1-\tau)|u|, & u < 0 \end{cases} \tag{5}$$

The terms w_q are weights and they control the influence of the quantiles on the estimation of the fixed effects. In our case we assume that the weights are the same for all the quantiles analysed.

An important feature of the estimator based on minimizing (4) consists of being a robust estimator, i.e. Gaussian condition and classical hypothesis related to a random error term are not necessary. The major difficulty is in itself the minimization of expression (3). We use the approach discussed in Koenker and Ng (2003), available on *rqpd* package of R, and obtain our results without any computational problem.

The quantile regression specified in (2) can be interpreted as a quantile generalization of difference-in-differences model which, given the control variables, measures the difference between the change in the quantile of air pollutant concentration for the treatment group (i.e., zones with an 80 km/h speed limit or zones with a variable speed limit, G=1) and the change in the same quantile in control group (i.e., zones neither with 80 km/h speed limit nor with variable speed, G=0). For a given confidence level, these differences are:

$$\left[Q_{Y_{B|G=1}}(\tau) - Q_{Y_{A|G=1}}(\tau)\right] - \left[Q_{Y_{B|G=0}}(\tau) - Q_{Y_{A|G=0}}(\tau)\right], \tag{6}$$

where Y_A and Y_B are the pollutant concentration variable after (A) and before (B) the implementation of the policy.

The choice of stations to measure pollution has been driven by the consideration that the more stations and the longer the time series, the better the analysis. We assume that obtaining measures of pollution in the 80 km/h area and outside follows a random process.

There are several reasons that led us to choose the differences-in-differences method. First, we do not need to know all the variables that affect the pollutants concentration because we assume that those variables remain constant before and after the policy came into force. Second, it solves the problem that missing data creates for both participants and non-participants in the periods before and after the measure came into force, for air quality modeling throughout the study area. Third, it allows us to discern how much of the change is due to the policy impact and how much would have happened even if the speed limit policy had not been implemented. Finally, it allows us to avoid selection bias of unobservable factors affecting the model.

One of the most basic assumptions of differences-in-differences models is that the temporal effect in the two areas is the same in the absence of intervention. This is called the fundamental identifying assumption and is described as the equality between average changes in the two groups in the absence of intervention. As in Galiani, Gertler & Schargrodsky (2005), we test for the equality between average changes in the two groups in the pre-treatment period to assess the plausibility of the fundamental identifying assumption. This endogeneity test is important even if it is usually forgotten in the differences-in-differences applied literature.

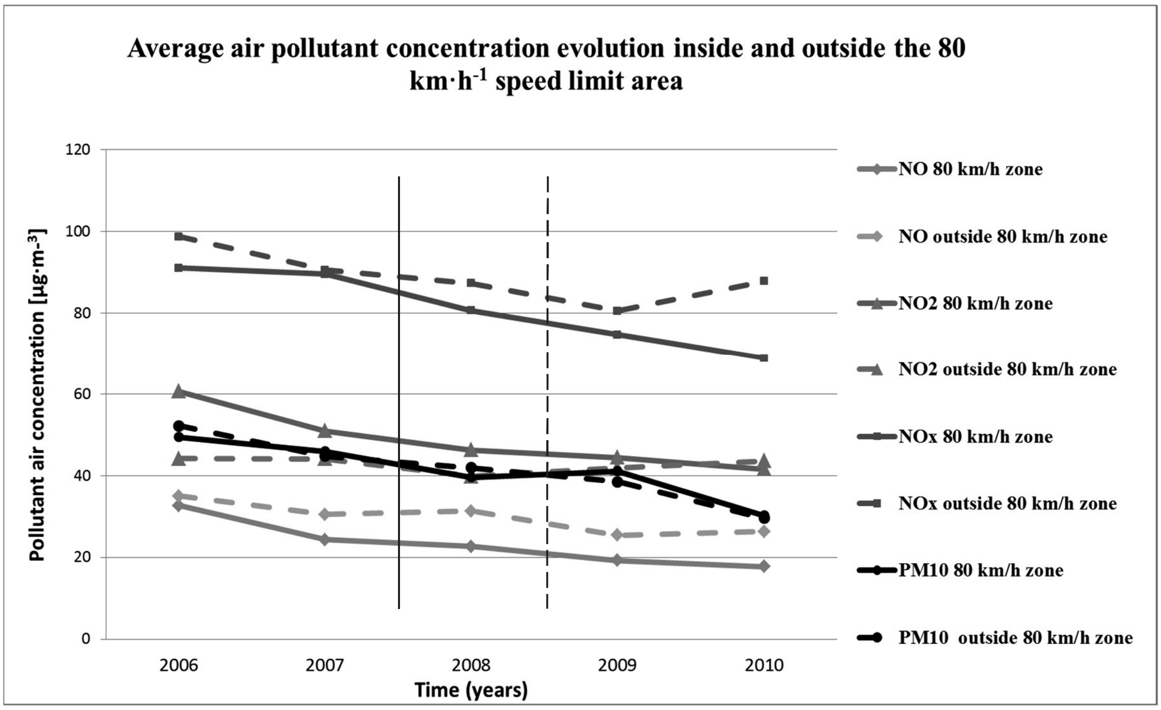
We follow the testing procedure as in Galiani et al. (2005). We generate a dummy that separates those stations that were affected by the limitation on January 2008 from those that were not affected. We regress equation 1 by replacing the 80 km/h speed limit variable with this dummy that differentiates between control groups and treated groups. Our null hypothesis H_0 is equality in the trend between treated groups and control groups. We cannot reject the null hypothesis for NO_x and PM_{10} at a 95% confidence level. These results imply non-biased estimations on NO_x and PM_{10} air quality levels.

Most problems related to endogeneity can usually be avoided by using differences in differences (Bertrand et al., 2004). However, it may be that the implementation does not follow a random scheme (Besley et al., 2000), which would be troublesome regarding endogeneity issues. Related to our own work, between 2006 and 2007 we find decreases in annual average concentrations of many pollutants, especially in what would be the 80 km/h speed limit area from 2008 on (Figure 4). This suggests that implementation of the policy does not respond to air quality deterioration in the years immediately preceding the coming into force of the speed reduction measure. Thus this enables endogeneity problems that could bias the policy effects to be avoided. It should also be noted that none of the political parties in government had proposed establishing an 80 km/h speed limit in their election programs, which would have conditioned the adoption of the measure.

Unobservable factors should not be underestimated in evaluations such as ours. They can be classified into two types: those that are fixed throughout the program and those that vary over time. One of our assumptions is that unobservable factors are constant over time. The high number of time periods we use enables the problems this type of assumption could pose to be minimized.

When using differences in differences, regressions must be done with fixed effects: the correlation between the error component of station θ_i and the explanatory variables is different from 0. We conduct the Hausman test, and we can reject the null hypothesis at 99% significance, our fixed effects assumption being correct. Panel data errors from different stations may be correlated (contemporary correlation), and within each unit there may be temporal correlation (autocorrelation or serial correlation). Also, if the error variance is not constant, we may find heterocedasticity.

Figure 4. Average evolution of pollutant concentrations inside and outside the 80 km/h speed limit area



To detect autocorrelation we use the Wooldridge test. We find an autocorrelation scheme. To find heterocedasticity we use the modified Wald test for heterocedasticity. We reject the constant variance null hypothesis. The contemporary correlation means that non-observable features in some municipalities are related to non-observable characteristics of other municipalities. One problem we have is that there is spatial correlation between pollution stations, although our pollution stations have followed a random selection. To test spatial dependence we use the Pesaran test (Pesaran, 2004). While this test introduces distortions when N is large and T is finite, our case is the opposite. Applying the test for fixed effects, we obtain spatial dependence for the four dependent variables. We use the Breusch-Pagan test, this being the null hypothesis of cross-sectional independence, which is rejected.

Once all this is taken into account, we have autocorrelation, heterocedasticity and contemporary correlation. We can solve those problems with Feasible Generalized Least Squares (FGLS) or Panel Corrected Standard Errors (PCSE). It has been demonstrated that PCSE obtain more accurate standard errors (Beck, 2001). We estimated by Ordinary Least Squares (OLS), allowing for heterocedasticity, contemporary correlation and following an AR1 autocorrelation scheme (Beck and Katz, 1995). Regarding the recommendation that the time periods should be greater than the sampling stations, it should be remembered that we have 2191 periods against 15 stations.

The equation we estimate for different dependent variables (NO_x, and PM_{10}) is the following:

$Y_{it} = \beta_0 + \beta_1 80\_speed_{it} + \beta_2 variable\_speed + \beta_3\ ltraffic_{it} + \beta_4\ pollutant\_lag_{it} + \beta_5\ temperature_{it} + \beta_6\ humidity_{it} + \beta_7\ rainfall_{it} + \beta_8\ wind\_speed_{it} + \beta_9\ atmospheric\_pressure_{it} + \beta_{10}\ year2007 + \beta_{11}\ year2008 + \beta_{12}\ year2009 + \beta_{13}\ year2010 + \theta_i + \lambda_t + \epsilon_{it}$

5 Results

5.1 Difference-in-differences on the average

The results of the pollutant estimations are reported in Tables 2, 3 and 4. The Wald test shows the variables to be jointly significant at 1%, while the R^2 model fit value ranges between 0.50 and 0.38 depending on the pollutant[19].

Tables 2 and 3 each show the results from four different regression models and both adhere to the panel corrected standard errors methodology. The introduction of the 80 km/h speed limit resulted in an increase in NO_x concentrations of between 1.9 and 1.3 $\mu g/m^3$. If we take the average NO_x concentrations in the speed limit area for the year 2007, the enforcement of the new limit meant an increase in NO_x concentrations of between 1.7 and 2.5%. In the case of PM_{10} specification, we found an increase of between 5.4 and 5.7%. By contrast, the variable speed system variable is highly significant and negative for all pollutants. We found an NO_x reduction of between 16 and 17.1% and a PM_{10} reduction of between 14.7 and 17.3%. Here, our reference is the pollution levels in 2008, the year before the measure was introduced.

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[19] In line with Currie and Walker (2011), we also estimated both models for SO_2 emissions. We found no significance of the policy variables introduced, and the explained variability of the model was below 8%.

Table 2. Least-squares estimation with Panel Corrected Standard Errors (NO_x)

Dependent variable: NO_x	**Specifications**			
80 km/h speed limit	1.887**	1.308*	-	-
Variable speed	-10.462***	-	-9.811***	-
Traffic	17.240***	17.108***	17.431***	17.245***
NO_x temporal lag	0.375***	0.375***	0.375***	0.375***
Temperature	-1.065***	-1.060***	-1.053***	-1.052***
Humidity	0.248***	0.246***	0.239***	0.240***
Rainfall	-0.100*	-0.099*	-0.096***	-0.097*
Wind speed	-2.153***	-2.142***	-2.181***	-2.163***
Atmospheric pressure	0.349***	0.348***	0.338***	0.340***
Year 2007	-11.165***	-10.870***	-10.307***	-10.276***
Year 2008	-8.264**	-7.850**	-6.940**	-6.932
Year 2009	-5.473	-6.101*	-4.459	-5.355
Year 2010	1.797	2.117	2.940	2.912
R^2	0.38	0.38	0.38	0.38
Nº observations	9159	9159	9159	9159
Joint significance	4825.4	4807.7	4861.5	4832.2

Notes: Standard errors are reported in parentheses. Each model also includes spatial and time fixed effects, and a constant term. * Statistically significant at the 10 %; ** at 5 %; and *** at 1 %.

Table 3. Least-squares estimation with Panel Corrected Standard Errors (PM_{10})

Dependent variable: PM_{10}	**Specifications**			
80 km/h speed limit	2.594***	2.469***	-	-
Variable speed	-6.196***	-	-5.272***	-
Traffic	1.087***	1.102***	0.864**	0.808**
PM_{10} temporal lag	0.572***	0.574***	0.584***	0.585***
Temperature	-0.096***	-0.085***	-0.081**	-0.073**
Humidity	-0.057***	-0.058***	-0.069***	-0.070***
Rainfall	-0.318***	-0.315***	-0.318***	-0.316***
Wind speed	-0.254***	-0.249***	-0.223***	-0.220***
Atmospheric pressure	0.371***	0.369***	0.339***	0.339***
Year 2007	-2.283***	-2.265***	-2.314***	-2.297***
Year 2008	-6.601***	-6.478***	-4.525***	-4.508***
Year 2009	-5.255***	-5.329***	-3.437***	-3.577***
Year 2010	-8.909***	-8.759***	-7.010***	-6.963***
R^2	0.50	0.50	0.50	0.50
Nº observations	1910	1910	1910	1910
Joint significance	5442.7	4941.1	5167	4784.4

Notes: Standard errors are reported in parentheses. Each model also includes spatial and time fixed effects, and a constant term. * Statistically significant at the 10 %; ** at 5 %; and *** at 1 %.

Traffic volume is significant in accounting for the concentration of all pollutants with the exception of some PM_{10} specifications. The sign is always positive, indicating that a greater traffic volume increases pollutant emission levels, which is consistent with available evidence. The coefficient's significance and the expected sign highlight the importance of traffic volume as a proxy for emissions from sources other than from traffic, and show how these correlate with the economic cycle.

The pollutant concentration in the previous period is significant in all regressions. Thus, 37% of NO_x levels are explained by the previous day's concentration, while this rises to 57-58% in the case of PM_{10}. On days with rain we found a significant reduction in PM_{10} and NO_x concentrations, whereas the concentration of all pollutants increased with higher atmospheric pressure, indicative of a more stable atmosphere. An increase in average wind speed resulted in lower pollutant concentrations for all pollutants, the wind acting as a dispersant in the Barcelona metropolitan area. The average temperature and average humidity variables also have an important explanatory capacity in the model, being significant for most regressions. However, these specific variables impacted the pollutant equilibrium equations, and any interpretation of these equations falls outside the scope of our analysis.

The year dummies are highly significant for PM_{10}. We record a significant reduction in pollutant concentration over time, which may be attributed to an economic effect uncorrelated with traffic volumes or other variables such as technological change or alternative pollution abatement policies. In the case of the NO_x specification some year dummies are also highly significant and negative, but here a decreasing trend over time cannot be identified.

Table 4 shows the policy impact using panel corrected standard errors and Driscoll and Kraay standard errors. We present different specifications according to the econometric method employed, the temporal data selected and when combining the two policy variables or omitting one or the other. There is no evidence that the two policy variables are affected by the econometric method adopted. The significance level and the sign are maintained, while the policy impact does not present a marked difference. Differences do arise, however, when we omit the month of August, a period of the year characterized by atypical traffic volumes due to the marked seasonal conditions in the metropolitan area of Barcelona (and throughout Spain). The speed policy variables remain significant and maintain their sign, but their impact is not as great. NO_x concentrations fall by half for some specifications, above all for the variable speed policy, while PM_{10} concentrations are maintained across the specifications.

Table 4. Least-squares estimation with Panel Corrected Standard Errors and Driscoll-Kraay standard errors

Pollutant	Methodology	All months		All months except August	
		80 km/h speed limit	Variable speed	80 km/h speed limit	Variable speed
NO_x	PCSE	1.887**	-10.461***	1.692***	-5.720***
		1.308*	-	1.377**	-
		-	-9.811***	-	-5.138***
	Driscoll-Kraay standard errors	2.365**	-10.022***	1.593**	-5.230***
		1.794	-	1.304*	-
		-	-9.227***	-	-4.687***
PM_{10}	PCSE	2.594***	-6.196***	2.702***	-6.181***
		2.469***	-	2.567***	-
		-	-5.272***	-	-5.219***
	Driscoll-Kraay standard errors	2.573***	-6.167**	2.665***	-6.134***
		2.449***	-	2.532***	-
		-	-5.256**	-	-5.193**

Notes: Standard errors are reported in parentheses. Each model also includes spatial and time fixed effects, and a constant term. * Statistically significant at the 10 %; ** at 5 %; and *** at 1 %.

Table 5. Policy impacts through least-squares estimation with Panel Corrected Standard Errors and Driscoll-Kraay standard errors

Pollutant	Methodology	All months		All months except August	
		80 km/h speed limit	Variable speed	80 km/h speed limit	Variable speed
NO_x	PCSE	2.5%	-17.1%	2.3%	-9.3%
		1.7%	-	1.8%	-
		-	-16.0%	-	-8.4%
	Driscoll-Kraay standard errors	3.2%	-16.4%	2.1%	-8.5%
		2.4%	-	*1.7%*	-
		-	-15.1%	-	-7.7%
PM_{10}	PCSE	5.7%	-17.3%	5.9%	-17.2%
		5.4%	-	5.6%	-
		-	14.7%	-	14.5%
	Driscoll-Kraay standard errors	5.6%	-17.2%	5.8%	-17.1%.
		5.3%	-	5.5%	-
		-	-14.6%	-	-14.5%

Table 5 shows the impact of policies on the pollution levels from the OLS estimations shown in Table 4. The 80 km/h speed limit policy increased NO_x

concentrations by between 1.7 and 3.2%, while the variable speed system reduced them by between 7.7 and 17.1%, the range reflecting the model, method or data employed. In the case of PM_{10}, the 80 km/h speed limit policy also increased air concentrations by between 5.3 and 5.9%, while the variable speed limit policy reduced particulate matter concentrations by between 14.5 and 17.3%. If the impact of a reduction in pollution can be expressed in monetary terms, policymakers can make their decisions more easily. A number of studies seek to do just that, including Michiels et al. (2012), while in the case of Barcelona, Pérez et al. (2009) estimate both the health and economic benefits of improving PM_{10} air quality in the metropolitan area. Identifying a fall in particulate matter from 50 to 40 $\mu g/m^3$, they report a reduction in mortality and morbidity equivalent to 2,300 (1,200-2,700) million euros per year at a 95% confidence interval. When applying a proportional reduction in policy impacts on the target population, we obtain mean savings of 262 (125-333) million euros in the case of the variable speed system, and mean losses of 592 (294-726) million euros in the case of the 80 km/h maximum speed limit policy. Doubtless, these figures reflect significant (and contradictory) impacts of the measures evaluated here on human health in the metropolitan area of Barcelona.

5.2 Difference-in-differences on quantiles

In Figure 5 we show transformed kernel density estimations[20] (see Alemany et al. 2013, Bolancé et al., 2008 and Bolancé, 2010) associated with NO_x and PM_{10} variables, we compare the probability distribution functions with and without policy actions. We have three cases: (i) without 80 km/h speed limit and without variable speed, (ii) with 80 km/h speed limit and without variable speed and (iii) with 80 km/h speed limit and with variable speed. From the Figure 1 we can conclude that when policies are in force, they do not only shift the mean of the distribution but they also affect the shape of the distributions, especially when the effect of speed is added, i.e., the effect of the speed policy is not only reflected in a shift in the distribution.

In Table 6 we show the estimated quantiles with confidence levels τ ranging from 0.05 to 0.95 for two pollutant concentrations, namely NO_x and PM_{10}. The first column in each case shows the behaviour for observations when no policy is implemented, the second column considers observations corresponding to cases with 80 km/h speed limit and without variable speed, the third column includes

[20] Transformed kernel density estimator is a nonparametric method which outperforms the classical kernel density estimation when the statistical distribution of the variable is right skewed.

observations for 80 km/h speed limit and with variable speed and the fourth column is all data together. We observe that the risk of high pollution is reduced more steeply when the policy that relies on "variable speed" is incorporated. However, the values presented in Table 3 are crude, so there is no control for exogenous conditions such as temperature, traffic, relative humidity and so on.

Figure 5. Density estimations for pollution data under different policies.

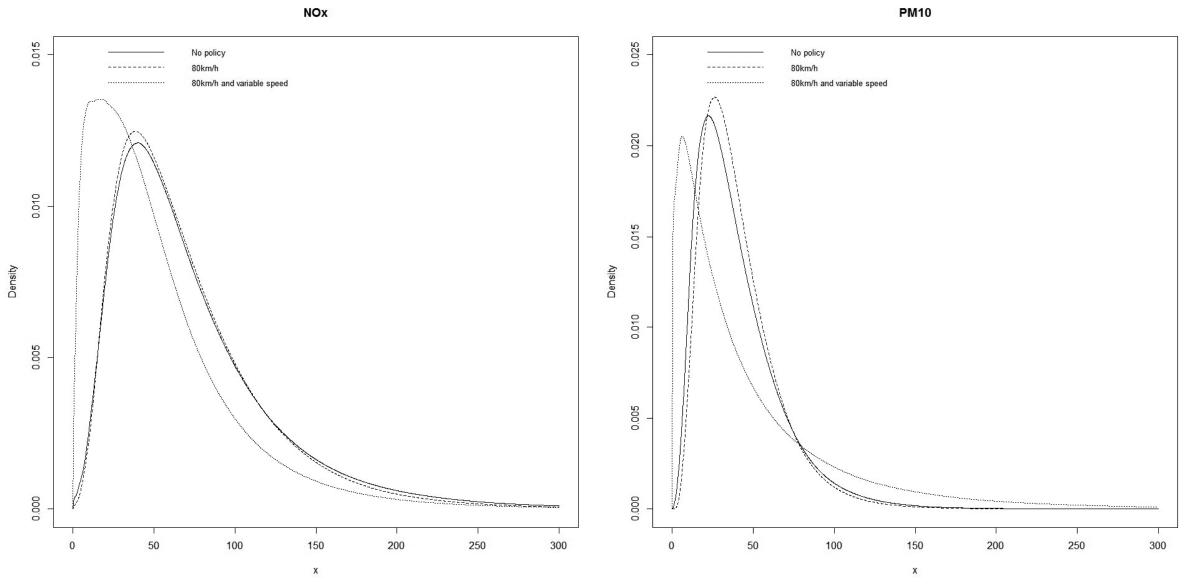

Table 6. Empirical quantiles of pollutant concentration statistical distribution for all data and under different policies by confidence level τ.

	NO_x				PM_{10}			
τ	**No policy**	**80 km/h**	**80 km/h and variable speed**	**All data**	**No policy**	**80 km/h**	**80 km/h and variable speed**	**All data**
0.05	19.1	21.7	22.7	21.0	18.0	17.5	18.9	17.7
0.10	26.1	26.6	26.6	26.4	21.0	21.0	21.0	21.0
0.25	39.8	38.3	37.1	38.7	27.0	27.1	28.0	27.1
0.50	60.0	57.9	51.1	58.2	35.1	35.0	35.0	35.0
0.75	84.6	86.2	68.7	85.3	45.9	44.7	46.7	45.0
0.90	111.5	114.0	104.0	113.1	58.0	56.3	57.0	57.0
0.95	134.1	134.9	125.5	134.4	65.8	64.1	62.2	64.8

When no policy is implemented, there is a 10% risk that pollution (NO_x) is above 111.5 and a 5% risk that pollution is above 134.1. If 80 km/h speed limit and variable speed are in force, the pollution level is lower than 104.0 with 90% probability, and there is a 95% probability that it is under 125.5. Similarly, when looking at the pollution in terms of PM_{10}, if no policy is implemented, there is a 10% risk that pollution is above 58.0 and a 5% risk that pollution is above 65.8%, whereas if 80 km/h speed limit and variable speed are in force, the pollution level is larger than 57.0 with a 10% risk, and there is a risk of 5% that PM_{10} is above 62.2.

In Table 7 we show the estimated parameters associated with the policy variables for different confidence levels τ. The results of all the estimated parameters are shown in the Appendix. The results for NO_x indicate that the 80 km/h speed limit does not improve NO_x air quality while variable speed system is clearly effective. The fixed speed system at 80 km/h is not significantly effective in the majority of the quartiles and, moreover, the policy effect is significant for some quality with a positive coefficient meaning that the 80 km/h speed limit policy is counterproductive because it involves an increase in the 90% quantile concentration of nitrogen oxide. The variable speed system is effective in all quartiles except on lower quantiles of NO_x quantiles, so the results indicate that variable speed system is effective to reduce NO_x pollution from traffic except on low polluted scenarios.

Table 7: Estimated coefficients of both policies on the pollution distribution quantiles of pollutants NO_x and PM_{10}. Significance (p-value) levels are given in brackets (n=sample size)

	NO_x (n=9159)		**PM_{10}** (n=1910)	
τ	**80 km/h speed limit**	**Variable speed**	**80 km/h speed limit**	**Variable speed**
0.05	2.847 (0.056)	1.125 (0.259)	3.201* (0.027)	-8.892** (0.000)
0.10	1.760 (0.265)	-1.134 (0.254)	3.119 (0.070)	-9.124** (0.000)
0.25	0.567 (0.713)	-4.203** (0.006)	2.335 (0.181)	-10.085** (0.000)
0.50	1.249 (0.495)	-9.042** (0.000)	1.945 (0.197)	-4.022** (0.000)
0.75	3.748 (0.080)	-17.277** (0.000)	1.714 (0.189)	-3.009* (0.011)
0.90	6.408* (0.016)	-16.168** (0.000)	1.813 (0.190)	-4.292** (0.000)
0.95	4.568 (0.249)	-21.258** (0.000)	2.101 (0.139)	-2.307 (0.062)

Quantile regression coefficients control for traffic, temperature, relative humidity, precipitation, windspeed, atmospheric pressure, Sahar desert dust and fire. * Significant at 5% level and ** significant at 1%

A similar pattern is reproduced on PM_{10} impacts. The 80 km/h speed limit does not show an impact except on the lowest quartile, where again it is counterproductive, because the coefficient is positive and significant, so the 5% quantile is increase in case this policy is implemented. Interestingly, the variable speed system implies a clear diminishing effect in all quartiles except the highest when looking at the PM_{10} concentration. The impact is clearly higher on lower-medium quartiles than on medium and higher ones.

Table 8 shows the impact of the policies in terms of the estimated coefficients by quantile regression reported in Table 7 and in terms of the pollution levels, some of them reported in Table A1. Once non-significant impacts are omitted, we obtain a neat effect of variable speed policy and the 80 km/h speed limit policy on the quantiles of NO_x and PM_{10}, concentration distributions. Model coefficients with significance levels between 10% and 5% are not taken into account due the large number of observations. The magnitude of the impact presented in Table 8 shows a clear decreasing effect on PM_{10} of the variable speed policy as the quantiles of concentration increase. In the 25th quantile, the implementation of the variable speed policy reduces the quantile by 56.3%) where as in the 75th quantile, the policy implementation reduce the quantile by 6,3% only. When considering the reduction of NO_x quantile when the variable speed policy is implemented the policy effect does not decrease so sharply as in the PM_{10} case, but it seems that that there is a maximum effect between the 25th quantile (11.7% reduction) and the 75th quantile (20.1% reduction) of NO_x. The effect if the variable speed limit policy is negligible for the quantiles below 25th quantile of NO_x distribution and it is also non-significant for extremely high quantiles above the 90th quantiles of the PM_{10} distribution.

Table 8. Significant policy impacts on distribution quantiles.

	NO_x		PM_{10}	
τ	**80 km/h speed limit**	**Variable speed**	**80 km/h speed limit**	**Variable speed**
0.05	-	-	24.1%	-68.4%
0.10	-	-	-	-56.3%
0.25	-	-11.7%	-	-38.1%
0.50	-	-16.6%	-	-11.7%
0.75	-	-20.1%	-	-6.3%
0.90	3.9%	-9.4%	-	-5.1%
0.95	-	-10.7%	-	-

Only 5% and 1% level

This certain abatement effect on both pollutants as concentration increases could provide from two main reasons: pollutants origin and policies impact limitation. NO_x origin is mostly anthropogenic and transport is the main source. However, PM_{10} can have a non-negligible natural origin, with an important influence of certain Sahara dust or salt coastal episodes that entail the presence of powder in the air.

The pollutant lag coefficients for all regressions are significant at 1% levels (Table A2). The interpretation is that in the first quantile, 25.6% of NO_x concentration was explained by the day before concentration, while on PM_{10} this effect is 18.6%. As the concentration increases, the pollutant lag effect increases, reaching 69.6% on the last NO_x quantile and 91.5% on PM_{10} last quantile. This highlights the fact that in high polluted environments, the next day pollutant concentration is going to be heavily influenced by the current situation, giving little margin for other factors, such as active policies. The policies impact can have their effects (or not), but in any case, but their effect is somehow limited by the previous-day conditions.

Due to the possible interaction of the two policies, we pursue to analyze them separately, trying to isolate both policies separately. We can only analyze the 80 fixed speed limit separately from variable speed one because the variable speed limit policy overlaps on time and range with the 80 km/h speed one. In order to obtain the fixed speed limit impact, we conduct the same analysis but we only take into consideration from 2006 to 2008, before the variable speed limit policy was in force. The main result is that for the seven quantiles analyzed and for both pollutants, there is only a positive and significant effect on the 90^{th} NO_x quantile, with significance level at 5%. This result confirms our previous result that 80 km/h speed limit policy has no significant effect on the quantiles of the distribution of pollutants. So, it does not change its shape significantly.

The first advice to policy-makers is that the policy impact will not necessarily be larger on high polluted episodes. The politicians could be tempted to implement certain policies only in high polluted episodes. Moreover, it is not true that the impact on all pollution measures decreases as the level of pollution increases. Second, we have found enough empirical evidence to implement variable speed limit, regardless of whether we are in a scenario strongly or slightly polluted and the type of pollutant. This policy only fails to provide a substantial effect in very extreme highly polluted PM_{10} episodes. And lastly, the maximum speed limit imposing a reduction from 120 or 100 km/h to 80 km/h does not imply an improvement in air quality for any scenario and for the pollutants considered here. Moreover, for some cases, there is an increase in pollution. For example, in the case of highly NO_x contaminated environments and low polluted PM_{10} environments, the fixed speed limit is detrimental.

6 Conclusion and policy implications

This paper has analyzed the effects of reducing the speed limit to 80 km/h on roads accessing the city of Barcelona, and the implementation of variable speed system in the same area. We have looked especially at the impact on pollution reduction of NO_x and PM_{10}. In contrast to most previous studies, this panel has relatively long periods before the measure came into force and all the periods after it came into force. Previous speed limit experiences for 80 km/h showed limited improvements in air quality. However, the implementation of both policies in the same area and time provide a singular framework.

The results of our empirical analysis show that for NO_x, the 80 km/h speed limit has not an air quality improvement, while for PM_{10} there is a 5.6% deterioration. Variable speed reduces NO_x and PM_{10} pollution by a 14.3% and 13.5%, respectively. When we omit the month of August in our estimations, the results remain very similar and robust except in the NO_x variable speed impact.

Our results contradict those obtained previously by means of simulations in the same area, which associated reductions in pollutant concentration with the speed limit. When controlling for a wide range of factors that influence pollutant emissions, we find that the new constant speed limit has little effect – if any – on NO_x. Its effects are also in the wrong direction for PM_{10}. On the contrary, the policy of implementing variable speed has positive environmental effects. Congestion reduction in peak hours due to variable speed system can explain this positive effects by comparison to 80 km/h speed limit. However, our study only has a small area under the influence of this measure and a short period of time, being a clear limitation. Future research should overcome this limitation.

We show as the shape of the distribution associated with the pollution concentration variables changes depending on the type of speed limit policy implemented. Furthermore, we observed that the risk of extreme NO_x pollution was considerably reduced when a variable speed policy was applied. We have used a fixed effect quantile regression model for an unbalanced data panel that allowed us isolate the policy effect in the different quantiles and conclude that this effect depends on the pollution level and type of pollutant.

Our results confirm that variable speed system abates NO_x and PM_{10} rather than reducing the maximum speed limit to 80 km/h. The reducing effect is for most scenarios, but the policy impact is strongly dependent on the pollution level. Variable speed limit is especially well suited when nitrogen oxide is large and when particulate matter concentration is low, so when causes of pollution are other than weather-related. As a general rule, variable speed level has a substantial effect to reduce the risk of nitrogen oxide above the first quartile and `particulate matter below the third quartile.

Traffic did consistently decrease in the years following the measure's coming into force because of the effects of the economic slowdown. Therefore the economic recession has been an important factor in the decline in pollutant emissions in the area. The pronounced decrease in NO_x and PM_{10} air concentrations should not be attributed to the effects of the 80 km/h speed limit policy, but rather to the effects of the economic crisis and to the variable speed system. There is a 260 million euros net loss due to 80 km/h speed limit. Increasing variable speed limit area and abolishing 80 km/h should reverse this losses.

In all, our analysis provides empirical evidence that can be useful for policy makers interested in improving air quality and health in urban areas by means of decreasing pollutant emissions. In this regard, implementing variable speed systems can yield far better environmental outcomes than introducing indiscriminate fixed maximum speed limits, at least within the ranges of speeds like those we have evaluated. Also, our analysis suggests that public policy needs to be aware of the effects of technologic changes in fields in which these are intense, because measures that were presumed to have positive effects under 'old' conditions may cease to have these effects under new technologic conditions, thus potentially undermining social welfare.

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APPENDIX

Table A1. Pollutants average concentration per years and areas for 10th, 50th and 90th percentiles.

	80 km/h speed limit zone					
	NO_x (μg/m³)					
	80 zone			Outside 80 zone		
	10th	50th	90th	10th	50th	90th
2006	17.0	53.8	171.3	18.9	54.4	165.9
2007	18.9	54.0	164.9	18.7	54.4	174.3
2008	18.7	54.4	160.9	16.8	54.4	165.2
2009	15.4	54.2	162.5	16.6	54.3	165.2
2010	16.5	54.1	155.1	18.3	54.2	161.4

	80 km/h speed limit zone					
	PM_{10} (μg/m³)					
	80 zone			Outside 80 zone		
	10th	50th	90th	10th	50th	90th
2006	21.8	34.6	74.5	16.2	34.3	81.9
2007	18.9	34.6	78.6	17.5	34.6	80.6
2008	16.2	34.3	80.4	16.2	34.4	77.2
2009	15.8	34.5	73.5	15.2	34.5	70.1
2010	16.0	34.3	75.8	16.0	34.2	75.7

	Variable speed limit zone					
	NO_x (μg/m³)			***PM_{10} (μg/m³)***		
	10th	50th	90th	10th	50th	90th
2006	14.9	54.3	169.5	16.2	34.4	85.2
2007	14.8	54.3	170.5	16.2	34.4	84.8
2008	14.8	54.4	171.4	16.3	34.3	84.9
2009	14.9	54.2	170.8	16.0	34.4	85.4
2010	14.8	54.3	170.8	16.2	34.4	85.4

Table A2. Estimated pollutants coefficients lags (all significant at 1% level)

τ	80 km/h speed limit	Variable speed
0.05	0.256	0.186
0.10	0.307	0.279
0.25	0.378	0.447
0.50	0.477	0.566
0.75	0.574	0.741
0.90	0.641	0.833
0.95	0.696	0.915

CHAPTER 4

PUBLIC AND PRIVATE PRODUCTION IN A MIXED DELIVERY SYSTEM: REGULATION, COMPETITION AND COSTS ON THE URBAN BUS SYSTEM

1 Introduction

Local governments in many developed countries have introduced competition and private delivery in the production of their public services. This has given rise to much discussion among both policy makers and scholars as they examine the relative merits of pure public and pure private forms of delivery. Owing to a lack of systematic cost savings from privatization (Bel and Warner, 2008; Bel, Fageda and Warner, 2010), scholarly analyses have shown a growing interest in alternative forms of service delivery that break with the traditional dichotomy between full direct service and complete contracting out (Bel, Brown and Warner, 2014).

One alternative form of production is the mixed delivery system, under which public and private firms deliver the same service within the same local jurisdiction (Miranda, and Lerner 1995). Recall that while sometimes 'mixed delivery' is understood as public entities and private entities jointly producing a service by splitting production and delivery tasks, by 'mixed delivery system' we mean here carving up a jurisdiction into sub units and then parceling out the same production and delivery process to different entities. Mixed delivery is fairly common in the US, with around 20 percent of local services over the last two decades being provided under mixed delivery, according to the regular surveys conducted by the ICMA (Warner and Hefetz, 2008; Girth et al., 2012; Hefetz, Warner and Vigoda-Gadot, 2014). Mixed delivery is frequent in such services as solid waste collection, nursing homes, and elderly care programs, among others (Miranda and Lerner, 1995; Warner and Hefetz, 2008). In contrast, mixed delivery is less frequent in Europe (Warner and Bel, 2008). Indeed, European policy makers and scholarly literature prefer the term mixed firm, an organizational form that differs from mixed delivery inasmuch as the government and its private partners share ownership of the firm (Cruz et al., 2014), which, moreover, is typically granted the monopoly provision of services in the local jurisdiction.[21]

The benefits of mixed delivery are arguably numerous. Theoretical analyses stress that the system offers the possibility of securing failsafe delivery (Williamson, 1991; Miranda and Lerner, 1995; Brown, Potoski and van Slyke, 2006) while promoting competition for the contract in the local market (Albalate, Bel and Calzada, 2012; Girth et al., 2012; Bel, Brown and Warner, 2014). Mixed delivery also reduces asymmetric information between government and private firms by benchmarking production processes and costs (Miranda and Lerner, 1995; Brown and Potoski, 2006; Warner and Hefetz, 2008), which in turn reduces transaction costs (Brown, Potoski and van Slyke, 2008). Taking full advantage of these opportunities requires effective management, an essential factor for improving service delivery (Hill and Lynn, 2004; Knott and Payne, 2004; Meier, O'Toole and Nicholson-Crotty, 2004; Brown and Potoski, 2006). In this regard, the management requirements are dependent on whether mixed delivery is provided under competitive tender or under performance contracts, which involve more standard regulation (Albalate, Bel and Calzada, 2012).

While several scholars have analyzed the determinants of the choice of mixed delivery (Lamothe, Lamothe and Feiock, 2008; Warner and Hefetz, 2008; Girth et al., 2012; Hefetz, Warner and Vigoda-Gadot, 2014), there is a lack of empirical evidence on the effects that mixed delivery has on costs. To the best of our knowledge, Miranda and Lerner (1995) is the only study to date to have specifically addressed this issue. The authors conduct an empirical analysis of the effects of mixed delivery (or joint contracting) on aggregate expenditures, employment, and wages in a sample of US municipalities and find that cities with a higher percentage of services provided under mixed delivery incur lower expenditures than those with a higher percentage of direct production. Miranda and Lerner (1995: 199) suggest the need for future research "to assess the impacts of benchmarking by service function to inquire whether competition within the private sector is sufficient to promote

[21] The model is also referred to as 'partial privatization' (Bel and Fageda, 2010; Albalate, Bel and Fageda, 2014)

cost savings without the need for duplication and/or overlap by instituting a production role for government".

The aim of this paper is to contribute to fill this gap in the analysis of the effects of the mixed delivery of local services on costs. More specifically, we compare the costs of public and private firms delivering the same service within the same local jurisdiction. Our analysis focuses on the Barcelona Metropolitan Area bus system, which is a notable case among the relatively few examples of the mixed delivery of local bus transit. This represents a unique opportunity to compare public and private delivery costs within the same jurisdiction by means of multivariate analysis. Additionally, and unlike most studies that compare private and public costs, our empirical analysis takes into account the transaction costs incurred when contracting out to private firms, which allows us to refine our analysis of the cost comparison. Note that a major advantage of analyzing different firms within a single jurisdiction, as opposed to studies elsewhere that draw comparisons between different cities (or even different countries), is that we do not need to rely on the assumption of homogeneity of regulators and spatial homogeneity.

2 Related literature

2.1 Public and private delivery and costs

While examples of successful service delivery contracts can be found, the outcomes of such contracts have in many cases been disappointing. In this debate, a growing body of evidence appears to cast doubts on the existence of systematic and sustainable cost savings attributable to privatization (Bel and Warner, 2008; Bel, Fageda and Warner, 2010). The main explanations offered for these mixed findings are differences in the transaction costs of contracted and direct services (Brown and Postoski 2003a, 2003b, 2005; Levin and Tadelis 2010), geographical differences in the availability of private providers (Hirsch, 1995; Warner and Hefetz 2002, 2003; Hefetz, Warner and Vigoda-Gadot 2012), and trends towards concentration and diminishing competition over time (Bel and Costas, 2006; Bel and Fageda 2011.

Transaction costs are a key issue in the debate concerning potential cost savings from privatization, and have become a regular focus in the literature on public service privatization since the seminal studies of Sappington and Stiglitz (1987) and Williamson (1991, 1999). Contracting procedures are costly, as are the monitoring and supervision of contracted services. Brown and Potoski (2003a, 2003b, 2005) show that transaction costs have a marked influence on the privatization of local services, and identify the importance of two dimensions related to transaction costs: asset specificity and ease of measurement.[22] Contracting agencies may underestimate the costs of the overall process as transaction costs are often excluded from the analysis. However, to date, transaction costs have received very little attention in the empirical analysis of public and private deliveries.

The role of the spatial dimension of competition in the provision of local public services was first taken into consideration by Warner and Hefetz (2002, 2003), who showed that different geographical areas (e.g., rural vs. urban) are characterized by differences in the respective availability of private vendors and so offer dissimilar prospects for competition. Indeed, the geographical area seems to be a critical determinant of the level of market competition. Bel and Fageda (2011), for example, report that the number of firms bidding for contracts in smaller cities is lower than that in larger cities, and also that competition is eroded over time. Furthermore, Bel and Costas (2006) show that this erosion of competition is a key factor in accounting for the lack of differences in costs between public and private delivery. In the same vein, Girth et al. (2012) find a positive correlation between the choice of delivery form and the level of competition in metro core areas. They also obtain interesting qualitative results on the dynamics of government "relational contracting": regulators tend to devote more time and resources in building and sustaining competition and so a low level of competition tends to add to transaction costs.

22 In their study of local public services, Brown and Potoski (2005) examined operation and maintenance costs and ranked bus systems 27th out of 64 services in terms of service measurability (the lower the ranking, the easier to measure the service) and 39th in terms of asset specificity (the lower the ranking, the lower the specificity). Similarly, Hefetz and Warner (2012) ranked bus system maintenance 29th out of 67 services in terms of asset specificity and 35th in terms of contract management difficulty (the lower the ranking, the lower the difficulty). All in all, Girth et al. (2012) classify bus systems as lying somewhere between monopoly and low competition services.

Mixed delivery is a production choice that escapes the classical dilemma between full direct production and complete contracting of services. Regulators can compare the firms' respective production processes and costs, while retaining direct involvement in service delivery. Mixed delivery allows a government to divide its jurisdiction in several areas, with pure public delivery being used in one or more areas and pure private production in other district(s) within the same jurisdiction (Warner and Bel, 2008). Miranda and Lerner (1995) claim that this 'redundancy' or duplication in delivery methods may in fact be efficient (as was proposed earlier by Landau, 1969), as a form of benchmarking with the private sector, and as a means of promoting bureaucratic competition in house. As such, mixed delivery can promote competition by means of introducing competitive pressures on public firms, i.e., by disciplining public managers and labor unions (Miranda and Lerner, 1995; Hatry, 1999), and by preventing private firm monopolization (Miranda and Lerner, 1995). The rise in mixed forms of delivery reflects a continuing process of change and innovation at the local government level that combines the benefits of both market and public delivery (Warner and Hefetz, 2008; Hefetz, Warner and Vigoda-Gadot, 2014): private firms are interested in profit and efficiency; the public sector is also interested in efficiency, but it is also expected to provide failsafe delivery and a higher level of public accountability and involvement.

Evidence on mixed delivery and costs is scarce, as mentioned in the introduction. With respect to the expectation that mixed delivery will provide lower costs than those associated with complete direct delivery, Miranda and Lerner (1995) obtained empirical evidence consistent with that hypothesis. Concerning our own analysis, two hypotheses can be emphasized:

> H1. Mixed delivery is expected to provide balanced costs between public and private providers, because the system disciplines the public production that is retained.
>
> H2. Mixed delivery is expected to improve competition for contracts, allowing for a reduction in costs via the tendering process.

Our paper seeks to enhance current understanding of the implications of mixed delivery for costs, and makes two major contributions to the literature. First, we analyze cost differences between public and private firms under a mixed delivery regime, and analyze whether tendering processes help to reduce delivery costs. Second, and unlike traditional cost comparisons between public and private providers, we explicitly include the transaction costs incurred by the regulatory agency (contracting procedures plus monitoring and supervision of contracted services) in the cost comparison, which allows us to refine the cost analysis for different types of service delivery.

2.2 Public and private delivery costs and tenders in local bus services

Early studies of the impact of privatization on urban bus services typically reported cost savings and greater efficiency with private delivery, for example, in the UK (Savage, 1993; White, 1997), New Zealand and Chile (Lee and Rivasplata, 2001), and Switzerland (Filippini and Prioni, 2003). While almost none of the research published before 2000 took competition into account in their empirical analyses (De Borger, Kerstens and Costa, 2002), more recent studies have controlled for competition for the contract. Leland and Smirnova (2009) compared the evolution in efficiency and effectiveness of US urban bus services, and found that privately owned and managed transit systems were no longer more efficient and effective providers than government owned agencies. They identified the lack of competition between contractors and higher transaction costs as factors. Recall that the market for bus services is imperfectly contestable (Mackie, Preston and Nash, 1995). The most common means of introducing competition in the provision of urban bus services is through competitive tendering.[23] A trans-European study has found that public firms are less productive than private ones, while firms selected through competitive tendering are more productive (Boitani, Nicolini and Scarpa, 2013). However, the most telling result is that during the second round of the tendering process (some years after the first), an increase in the gross cost is recorded in most countries. Not only are there no sustained cost savings, but there is also some deterioration in the quality of the service provided,

[23] Several studies have focused solely on regional services or on mixed urban/regional services (e.g., Fazioli, Filippini and Prioni, 1994) and so are not directly comparable to our study.

especially when the incumbent is replaced by a new entrant (Mouwen and Rietveld, 2013).

Falls in the number of bidders and in bus market concentration have been described in Sweden (Alexandersson, Hultén and Fölster, 1998), France (Yvrande-Billon, 2006) and Norway (Mathisen and Solvoll, 2008), among others. This situation is paralleled in Italy, where moreover the incumbent operator tends to win the majority of tender processes (Boitani and Cambini, 2006).24 Hensher and Wallis (2005) express a concern that tendering is open to regulatory capture by powerful monopolist bus providers. Consequently, putting this relationship out to tender, in a process that attracts few bidders and involves complex contracts and incompleteness, may not be the best solution when compared to negotiated performance-based contracts (Hensher and Stanley, 2008). Hensher (forthcoming) concludes that the gains from competitive tendering are generally illusory or overstated (outside the situation of an incumbent public firm).[25]

3 Data

The dataset is an unbalanced panel for the period 2002-2012, comprising 378 observations and four companies, one of them fully publicly-owned (TMB) and the other three completely private. Data are provided by the public bus company (TMB) and the regulator, the *Entitat Metropolitana del Transport* (EMT), the latter providing data for the private firms. The year 2002 is the first year after the integrated fare policy was introduced.

[24] In Germany there is an incumbent renewal rate of 74 percent and an average of more than five bidders per tender (Beck and Walter, 2013). The relationship between buyer and vendor evolves over time, and the dependence of one party on the other grows. Local governments tend to place greater trust in the faithfulness and honesty of their vendors when the latter party has a known reputation from prior to the relationship, strong community ties, and performs its tasks well (Lamothe and Lamothe, 2012). Thus, welfare gains from extending contract length can be relevant but accrue mostly to operators (Gagnepain, Ivaldi and Martimort, 2013).

[25] Note, however, potential unintended effects of high-powered, performance-based contracts have been found in some services (Koning and Heinrich, 2013), where these contracts can mean not providing a service to the hard-to-serve clients.

We choose private bus lines that connect Barcelona with neighboring towns and urban lines in surrounding towns in areas under concession.[26] We exclude lines that run parallel to motorways or high capacity roads, since the characteristics of these lines are not similar to those operated by the public company TMB. We choose all TMB lines that connect Barcelona with neighboring towns, as well as lines within the municipality of Barcelona but at some distance from downtown Barcelona. We exclude all night routes because they differ markedly from day routes in terms of wages paid, average speed, network length, etc.

Figure 1. Public and private bus lines in our study area

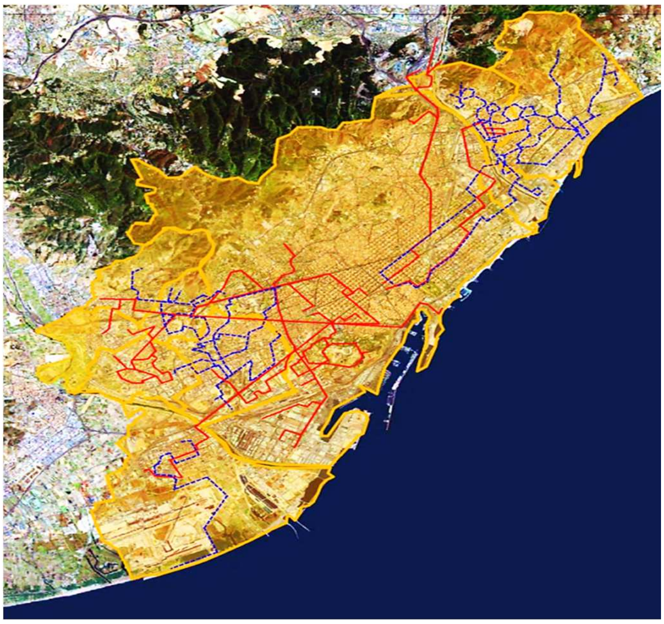

Figure 1 shows our study area. The municipalities are represented by polygons, the largest of which is Barcelona, with a length of nine kilometers

[26] We are aware that by using lines as units of observation we are assuming the strict separability of each line operated by an individual transit system. In fact, we need to use the lines as observations as we only have four firms providing services in our analysis. It would be more realistic to assume that firms choose inputs on a firm-wide basis to minimize costs, which would mean considering each company in a particular year as an observation. We use two different specifications, the second one including a dummy for each private firm, in order to control for firm's specificities. Note that our results are basically the same in both specifications (either with a dummy indicating private firm, or with a dummy for each private firm).

and a width of eight kilometers. The red lines represent public bus routes and the blue lines represent those operated by private concessionaires. The three private companies operate in three different geographical areas: the northern area is operated by Tusgsal, the western by Soler i Sauret, and the southern by Rosanbus. Note that the various lines overlap, and even though the concession areas differ, they are sufficiently similar to be compared.

Table 1 reports summary statistics for the concessionaire firms included in the analysis. The available information includes – for each concession – the number of lines analyzed, the line length, the number of passengers carried, the number of vehicles, the number of employees, and the cost and revenue per net km (thus, we do not take into account trips made without passengers from and to bus depots; hereafter, 'per km'), among others. The main differences between the concessionaires concern the number of lines analyzed, the number of passengers carried, the number of vehicles and the costs, total costs and the cost per km covered.

More than half the lines analyzed are publicly managed, while the others are distributed among the three private companies. Differences also occur between the public and private companies in terms of the cost per km, the number of passengers carried and the number of kilometers covered. Given the similarity in line length, it is clear that TMB has a higher bus frequency than that operated by the private companies. The public company also carries more passengers. However, some private companies employ a larger workforce than that employed by TMB relative to the number of kilometers covered. A further difference is the age of the fleets: the fleet renewal rate of the public company is much lower than that of the main private operators. In this regard, TMB has its own purchasing policy, while the EMT purchases the private bus fleet. Additionally, the number of employees is higher and significantly different in the public company compared to the private firms. There are marked differences in costs: the public company is the most expensive firm by line, but when we compare the total costs by km covered, it is not the most expensive. The transaction costs for the private lines remain constant each year. However, these costs can represent between 3.5 and 14 percent of their total costs.

Table 1. Mean (standard error) based on 378 bus line observations

	TMB	Barcelonès Nord (Tusgsal)	Hospitalet de Llobregat (Rosanbus)	West Barcelona (Soler Sauret)
Average speed (km/h)	12.91 (0.147)	10.72 (0.203)	12.604 (0.307)	12.58 (0.408)
Length line (km)	10.454 (0.223)	8.576 (0.496)	11.396 (0.82)	10.308 (0.364)
Net km (km)	430,924 (13,337)	293,336 (20,762)	306,643 (19,145)	123,505 (10,751)
Vehicles	10.96 (0.343)	5.48 (0.399)	6.155 (0.392)	2.63 (0.176)
Bus age (years)	6.49 (0.04)	4.47 (0.05)	5.02 (0.09)	6.23 (0.29)
Employees	32.0 (1.05)	27.02 (1.46)	16.59 (1.28)	6.25 (0.5)
Passengers	1,569,978 (69639)	883,722 (61,527)	1,129,646 (121,391)	176,540 (20,299)
Cost per net km	4.10 (0.059)	5.13 (0.116)	3.85 (0.131)	3.33 (0.183)
Revenues per net km	1.62 (0.04)	1.47 (0.06)	1.55 (0.13)	0.59 (0.04)
Total costs (€)	1,784,709 (62,653)	1,391,206 (78,470)	1,073,187 (76,330)	379,854.4 (44,492)
Transaction costs (€)	0	52,217.17 (971.9)	51,435.91 (1389.95)	51,172.17 (2556.0)
Analyzed lines (yearly average)	21	9.4	5.4	1.6

4 Empirical strategy

A firm converts inputs into output. Thus, a bus company uses employees as labor force, and fuel and other materials to obtain energy to cover kilometers and transport passengers, assuming the total variable cost to be a function of input and conditional to the level of capital provided by the authority. Beyond this specification, several authors also identify network characteristics such as the length of the line or the average speed (see De Borger, Kerstens and Costa (2002) for a detailed review). Another group of factors that influence operating costs are the form of ownership, competition and other environmental characteristics. The total variable cost frontier can be written as the following function:

$$\mathrm{TVC}_{it} = f(Y_{it}, PL_{it}, PM_{it}, K_{it}, N_{it}, O_{it}, t) \quad (1)$$

where the *total variable cost* of an urban bus line *TVC* is assumed to be a function of output *Y*, factor prices *P* (labor *L* and material and energy *M*), capital K, the network characteristics *N*, the competition and ownership *O* and the time trend *t,* among others. The model takes the urban bus line's variable cost as a dependent variable; that is, its total annual operating expenses. It is important to note that variable costs exclude fixed capital costs, i.e., the cost of vehicles and bus depots, which in many cases are funded entirely – or almost entirely – by the government and/or the regulator. This is, in fact, the case in the jurisdiction we analyze here. Thus, capital represents a quasi-fixed input. Variable costs are increasingly being used in the empirical analysis of costs in order to deal with an apparent paradox: the very frequent finding, and a common result in the literature on buses, of a negative coefficient for capital costs when using capital as an input price. This result is due to a non-cost minimization process for capital (see, for example, Cambini, Piacenza and Vannoni, 2007), because in many cases the government and/or regulators cover these costs in full or almost in full. This is the case, for instance, in France; and for this reason, Gagnepain (1998) opted to use variable costs instead of total costs. As mentioned, variable costs are increasingly being used in the literature as the dependent variable and have become the standard choice in the most recent empirical studies (e.g., Ayadi and Hammami, 2015). Therefore, in this framework, because the operators do not minimize the cost of capital, we expect a negative sign.

When data are comparable and consistent over time, the most frequently occurring functional form in the literature is a translog cost function (Coelli, 2003). This function was first proposed by Christensen et al. (1973) and was first applied to urban bus transit by Viton (1981). Subsequently, it has been applied in many empirical studies on this field.[27] The cost function equation to be estimated can be expressed in the following double log form:

$$ln\frac{TVC_{it}}{PM_{it}} = \beta_0 + \sum_{x=Y,N,SP,\frac{PL}{PM},K} \beta_x \, ln \; _{it} + \sum_{x=Y,N,SP,\frac{PL}{PM},K} \frac{1}{2}\beta'_x(\ln x_{it})^2 + \sum_{\substack{x,x'=Y,N,SP,\frac{PL}{PM},K \\ x\neq x'}} \beta_x \, ln \; _{it} \, ln \; '_{it} + +\beta_{21}TEND_{it} + \beta_{22}PRIV_{it} + \beta_{23}TUS_{it} + \quad (2)$$

[27] For instance, Farsi, Filippini and Kuenzle (2006), Filippini and Prioni (2003), Fraquelli, Piacenza and Abrate (2004), Matas and Raymond (1998), and Ottoz, Fornengo and Di Giacomo (2009). The translog cost function considers cost as a function of input prices and the production level. This flexible functional form is a second-order logarithmic approximation to any arbitrary twice-differentiable cost function. The values of the explanatory variables are normalized to the mean. We assume that input prices and output are exogenous, and that the cost function is the result of cost minimization given input prices and output.

$\beta_{24}ROS_{it} + \beta_{25}SOL_{it} + \beta_{26}BARC_{it} + \beta_{27}METR_{it} + \beta_{28}MUN_{it} + \beta_{29}TIME_t + \xi_{it}$

with $i = 1, 2, \ldots, 42 \qquad and\ t = 2002, 2003, \ldots, 2012$

where subscripts *i* and *t* denote the line and year respectively. The dependent variable is *Total Variable Costs* (TVD). The *output* variable (Y) is the vehicle-kilometers, N is the *network length*, SP is the *average speed*, *the price of labor* (PL) is related to the *price of materials* (PM) and the *capital* (K) is the number of buses per line. TEND is the *year of tender* for private concessions, PRIV is a dummy for *private* lines and TUS, ROS and SOL are dummies for *private concessions*. The control variables are BARC for lines inside the *city of Barcelona* while METR is a dummy indicating whether or not a new *metro station* has been opened near to a bus line and MUN is a dummy indicating whether or not the bus routes are within the *same municipality*. TIME is a yearly *time trend* variable. ξ_{it} is the random term divided in v_{it} (normally distributed) and u_{it}, the inefficiency term (half-normally distributed). Table 2 reports the definitions and expected outcomes for all the variables. The dependent variable and the labor input have been divided by materials input price. In addition to the standard variables of a proper cost function, we included *network length* (N) and *average commercial speed* (SP) in the model, in line with Levaggi (1994) and Gagnepain (1998).[28]

Our main objective is to analyze the ownership and competition variables. The ownership issue has been analyzed in the literature; however, the results regarding the relationship with efficiency and productivity in the private sector are inconclusive. The *year of tender* is an innovative variable for detecting the impact of a tender process as a competitive mechanism. Three control variables are included: *Barcelona*, which controls for lines inside the city of Barcelona; *municipality*, which controls for lines within a municipality other than Barcelona; and, *metro*, which controls for subway stations opened near a bus stop on a specific line.

[28] The inclusion of average speed may imply strong correlations with other explanatory variables. We obtained correlations between output and network length and average speed of 0.28 and 0.36, respectively. Both variables are included in the estimation.

Table 2. Model variables and their definitions

Grouping variables	Variable	Definition
Dependent variable	Variable Costs (TVC)	This includes labor, fuel, maintenance and other indirect costs (such as administration and coordination costs at the line level, both for public and private firms). We exclude the bus fleet costs. We evaluate these costs in two different scenarios, depending on whether we include transaction costs or not. The transaction cost is the regulator's total budget, and we distribute it equally between each private line.[a]
Output (Y)		We use *vehicle·kilometers*, a supply-related measure. Seat-kilometers would have been another suitable measure, but we have no data here for private companies. The output is expected to have a positive sign.
Price of inputs	Price of labor (PL)	The ratio of total salary expenses to the total number of hours worked.
	Price of material (PM)	This is obtained by dividing fuel and maintenance material costs by net kilometers covered. This variable divides the dependent variable and the price of labor in the equation.
Capital (quasi-fixed, K)		Capital is represented by the number of buses. It is considered a quasi-fixed factor because buses are owned by the regulator.. We expected a negative sign.
Network characteristics	Line length (N)	This variable serves as a proxy for exogenous characteristics such as public service obligations. Line length is the round trip distance divided by two. Expectations regarding the effects of *line length* on costs are ambiguous.
	Average speed (SP)	If a trip is covered in a shorter time, fewer vehicles and less labor force are required. Therefore, costs are expected to decrease with increasing network speed. Average speed is the number of kilometers divided by service hours.
Competition and ownership variables	Tender year (TEND)	This dummy variable takes a value of 1 for the year immediately following a tender process (that is, the year in which the new concession terms are applied). New contracts come into force on January 1, even though the contest has been celebrated in the previous year. If tenders were cost-minimizing artifacts, costs should decrease following a tender process.
	Private (PRIV)	This binary variable takes a value of 1 if the line is operated by a private company and 0 if it is operated by the public company TMB. Most empirical evidence for bus services indicates that private companies incur lower costs, but this evidence is not systematic across other local service sectors, such as solid waste and water.
	Concession dummies (TUS, ROS, SOL)	A dummy that clusters the same concession's bus lines up to a total of four. The private ones are TUS (Tusgsal), ROS (Rosanbus) and SOL (Soler i Sauret)

Control variables	Barcelona (BARC)	This dummy variable takes a value of 1 if the whole bus line is within the municipality of Barcelona, and 0 otherwise. In this way we control for the effect of the city of Barcelona on the cost function.
	Metro (METR)	This dummy variable takes a value of 1 if the bus line is affected by the new metropolitan subway lines opened in 2010, and 0 otherwise.
	Municipality (MUN)	This dummy takes a value of 1 if the bus route is within the same municipality (but outside Barcelona), and 0 otherwise.

[a] Recall that EMT's costs are not allocated to TMB lines, because it does not regulate TMB and does not undertake any other functional activity with respect to TMB. Both TMB and EMT are subject to political supervision by representatives of local governments in the Metropolitan Area of Barcelona.

We use cost frontiers for panel data, as first introduced by Battese and Coelli (1988). We select two different models: the Battese and Coelli model (1992) and the True Random Effects model (Greene, 2005). There are different distributional assumptions to create an empirical model from the 'composed error'. The first of these models assumes a random component term to be time variant, so that t is different from T_i (Battese and Coelli, 1992). The decay parameter –η- shows the temporal pattern of inefficiency.

$$u_{it} \equiv exp\{-\eta(t - T_i)\}u_i \qquad (3)$$

We assume that $v_{it} \sim N(0, \sigma_v^2)$ and u_{it} $N^+(0, \sigma_v^2)$. Battese and Coelli's (1992) panel data model is somewhat restrictive because it only allows inefficiency to change over time exponentially. Nonetheless, it allows exploiting better the panel structure of the data. When estimating the inefficiency measures, one of the main problems faced is avoiding the unobserved heterogeneity bias. Based on their findings, Farsi, Filippini and Kuenzle (2006) propose potential differences in cost frontier parameters and inefficiency scores. As such, True Random Effects could perform better, allowing line-specific heterogeneity to be captured. This model is particularly suited to transport industries where network and environmental characteristics (time-invariant factors) are mostly unobserved but play an important role in operating costs (Farsi, Filippini and Kuenzle, 2006). In order to calculate the inefficiency scores, the value of u_{it} needs to be positive. Indeed, the model predicts they will be positively skewed in cost frontiers. Thus, lambda $\left(\lambda = \frac{\sigma_u}{\sigma_v}\right)$ indicates the ratio of the standard errors of the inefficiency terms to the standard deviation of the stochastic term (Greene, 2005, 2008). If the estimate of λ is statistically significant, suggesting that there is, evidence of cost inefficiency in the data.

The actors affected by transaction costs are the private firms and the regulator. Private companies need to devote some of their time to preparing their tender bid and to working with the regulator if they win the contest.[29] The regulator, on the other hand, has to organize the tender process and monitor the private winner during the period of concession. However, the regulator's budget is paid for by the taxpayers, and there is no specific transfer of the regulator's costs to the private firm's services. Our paper is, to the best of our knowledge, the first to consider the regulator's transaction costs. As EMT would not need to exist if there were no private firms operating in the jurisdiction, we take the regulator's budget (which is publicly available) as our measure of the transactions costs not internalized by the firms. Then, we allocate the regulator's costs equally among all the private lines, which allows us to provide a more refined and robust cost comparison. Later, in the section dedicated to the robustness check, we consider an alternative cost allocation method: the number of passengers carried by each line.[30]

5 The Barcelona Metropolitan Bus System: Regulators and Operating Companies

The provision of public transport in the Barcelona Metropolitan Area has remained largely stable over recent decades. Local bus transit is provided under a mixed delivery system, in which one public and several private firms supply the service in different areas under the supervision of a regulator. Recall that empirical analyses of urban bus systems report U-shaped average

[29] Recall that these costs are internalized by the firms in the bid that they make. Therefore, they constitute another item in the company's costs.

[30] We divided the regulator's total budget by the total number of lines (tender as well as performance-based contract lines, given that the regulator devotes effort to the performance-based contracts). When applying the new criteria for allocating transaction costs (number of passengers carried per line), we divided the regulator budget by the number of passengers. Here again we have taken into account all lines (including those with performance-based contracts).

cost functions. In other words, and as suggested earlier by Berechman (1993), economies of scale are constant or even revert to decreasing returns to scale as companies grow in size (Jørgensen, Pedersen and Volden, 1997; Matas and Raymond, 1999). This is an important outcome for the management of local bus systems, since it implies that in large cities the service can be fragmented between different firms without foregoing any of the potential benefits of scale economies.

5.1 The regulator

The EMT is responsible for regulating local bus transit in the city of Barcelona and the towns in its metropolitan area. The EMT defines the characteristics of the service offered by the concessionaires, establishes network routes, draws up bus schedules, defines quality levels, organizes the tenders, and covers the deficits run up by the private firms[31] (Albalate, Bel and Calzada, 2012). EMT has considerable experience as a management contractor and has been tendering services since the 1990s. Private companies can operate services by means of competitive tendering contracts (net-cost contracts) or negotiated performance-based contracts (gross-cost contracts), both with quality incentives. The EMT is the owner of bus fleets and bus depots, and provides vehicles and facilities to the operating companies. Recall that this reduces potential asset specificity problems, since it minimizes problems related to sunk costs, thus increasing market contestability.

5.2 The public firm: TMB

The publicly owned firm *Transports Metropolitans de Barcelona* (TMB) operates 106 lines, and in 2012 it carried around 174 million passengers. The number of passengers has remained stable since 1980, as several lines formerly operated by TMB have been privatized, and the metropolitan subway network has also been expanded. Correspondingly, supply has also remained quite stable: TMB offered 3,182 million seats-km in 2012, just a little above the 3,050 million seats-km supplied on 1989. TMB has not

[31] Another public agency, *Autoritat Metropolitana del Transport* (ATM), is responsible for setting fares for all transport modes. Since 2001 an integrated fare policy has existed for all public transport supply (including buses) with the exception of the airport shuttle bus and the city tour bus.

participated in any tender process in the Barcelona Metropolitan Area. However, in 2011 a joint venture group formed by TMB and Vectalia (a private group based in Valencia) was awarded the provision of public transport in the Perpignan Méditerranée Metropolitan Area (France). TMB enjoys freedom to design and plan its own services, and is not subject to a concession contract with the EMT. As such, TMB can be described as a corporatized company, enjoying greater managerial flexibility than that typically enjoyed by a traditional bureaucratic organization. Note that the public entity is essentially unregulated and has a much freer hand to make management decisions that the private firms, which are heavily regulated by EMT.

5.3 The private firms

Several of the private concessions overlap with the TMB delivery area as both connect Barcelona (including downtown) with the surrounding municipalities and also operate in these surrounding municipalities. The only difference is that TMB operates the intra-city routes. Bus services operated by private firms under competitive contracts represent 81 percent of the EMT's total passengers (excluding TMB). Private concessionaires that win tenders operate under net-cost contracts, with vehicle and depot facilities being supplied by the EMT.

Tender concessions follow a net-cost contract scheme or a minimum subsidy: firms receive a subsidy from the EMT that is equal to the difference between the expected revenues from passengers and the bidding cost. Private operators enjoy far less managerial autonomy than TMB. However, there are revenue incentives related to such factors as extra-passengers, punctuality, vehicle quality and passenger's perceived quality, among others. These revenues represent around 4 percent of the total revenues (excluding subsidy). The average concession length is five years, with the possibility of an extension (average length of extensions is three years).

5.4 Tendering Processes

The first tendering process took place in 1998. Table 3 displays all the tendering processes implemented up to and including 2012, and which are considered in our analysis.[32] Table 3 also includes the negotiated performance based-contracts, all of which are held by two firms belonging to the same metropolitan-based group (Baixbus). Note that we do not include negotiated performance based-contracts in our empirical analysis as the services delivered under them are of a different nature from those produced directly or tendered; and, more importantly, because the firms holding these contracts are under no obligation to report any data to the regulator. As a result, no specific firm data are available, whereas data from firms that have won tenders are available.

Table 3. Bus concessions in the Barcelona Metropolitan Area

	Concession area	Winner (Group)	Tender date	Number of bidders	Winner lowest price?
Competitive tendering	Barcelonès Nord	Tusgsal	1998	4	No
		Tusgsal	2010	2	No
	Barcelonès Nord (Night)	Tusgsal	1998	4	No
		Tusgsal	2006	3	No
	Hospitalet de Llobregat	Rosanbus (Baixbus)	2001	5	No
		Rosanbus (Baixbus)	2011	5	No
	West Barcelona	Soler Sauret	1998	4	No
		Soler Sauret	2008	3	Yes
Negotiated performance -based contracts	U1	Mohn (Baixbus)			
	U2	Oliveras (Baixbus)			
	Barcelona South Coast	Mohn (Baixbus)			
	West Barcelona (Night)	Mohn (Baixbus)			

Source: Based on documentation from the tendering processes.

The number of firms bidding to obtain a concession has varied from between two and five, but the incumbent firm has yet to be removed by a new entrant (even in the first process). In seven of the eight tenders, the incumbent did not offer the lowest price (required subsidy). This, in fact, only ever occurred in 2008 in the case of the western area concession awarded to Soler i Sauret.

[32] A more recent tender took place in 2013, but it is not included here, as the new contract did not come into operation until January 2014. However, the incumbent retained the service.

The weight of the financial proposal represents between 25 and 30 percent of the total valuation of the bid. Greater weighting is attached to characteristics of the bids other than the financial proposal (required subsidy), including experience in the sector, expected ridership, or staff and equipment used. As a result, incumbents almost always obtain higher valuations in the non-financial aspects of the bid, which allows them to retain the contract.

The majority of private firms offering services operate according to the concession awarded in the corresponding tendering process. A small number of firms concentrate the market for tendered services, even though there were several bidders in the contests. It is worth noting that no "outsiders" (that is to say, firms or groups based outside the Barcelona Metropolitan Area) have ever won a tender, even when requesting the lowest subsidy (lowest contract price). That points to the existence of relational considerations and/or rent-seeking costs; that is, the regulator has incentives to avoid changing the firm because the day-to-day relationship would change. For example, unexpected changes not provided for under the contract might be harder to implement. A further possibility is that the regulator expects the incumbent to provide a higher quality service and, so, prefers to maintain the contractual partner.

6 Results

First, we estimate a stochastic cost frontier for panel data using Battese and Coelli (1992). Table 4 shows the results (see second order coefficients in Table A-1 in the Appendix). For concession dummies we take as our reference TMB. All variables are expressed in logarithms (with the exception of the dummy variables); therefore, coefficients can be interpreted as elasticities. The original values of the monetary variables are deflated by a price index.

Half of the specifications are estimated with the dependent variable (*total variable cost*) without including the transaction costs (left-hand columns), while the other half include the transaction costs (right-hand columns). Overall no significant differences were found when using either one or other of the alternative specifications for the dependent variable, with the exception of the effect of tenders on costs.

Table 4. Cost frontier estimation with time varying effects - Battese and Coelli (1992) model (standard errors in parentheses)

	Dep. variable without transaction costs		Dep. variable with transaction costs	
	(1)	(2)	(3)	(4)
β_Y	1.290***	1.295***	0.368***	0.311***
	(0.110)	(0.104)	(0.110)	(0.106)
β_N	-0.023	-0.036	1.537***	1.557***
	(0.059)	(0.056)	(0.339)	(0.581)
β_{SP}	-0.144	-0.141	-1.644**	-1.657*
	(0.128)	(0.121)	(0.932)	(0.965)
β_{PL}	0.544***	0.563***	0.803***	0.843***
	(0.058)	(0.055)	(0.196)	(0.175)
β_K	-1.582***	-1.579***	-0.448***	-0.334***
	(0.224)	(0.212)	(0.115)	(0.092)
β_{Time}	-1E-04**	-1E-04**	-0.002**	-0.015***
	(6E-05)	(5E-05)	(0.001)	(4E-04)
Tender year	0.000	0.000	-0.013*	-0.013*
	(0.001)	(0.001)	(0.007)	(0.007)
Private	0.006***		0.103***	
	(0.001)		(0.009)	
Tusgsal		0.007***		0.104***
		(0.001)		(0.010)
Rosanbus		0.005***		0.103***
		(0.001)		(0.011)
Soler Sauret		0.005***		0.096***
		(0.001)		(0.014)
β_{BCN}	0.000	0.000	0.005	0.005
	(0.001)	(0.001)	(0.005)	(0.005)
β_{MUN}	-0.001**	0.000	-0.004	-0.003
	(4E-04)	(4E-04)	(0.005)	(0.005)
β_{MET}	0.000	0.000	0.000	0.000
	(0.001)	(0.001)	(0.005)	(0.006)
η	0.367	0.328	-0.014	-0.025
	(0.287)	(0.410)	(0.135)	(0.164)
$\lambda= \sigma_u/\sigma_v$	1.119***	0.033	1.577***	1.2156***
Log likelihood	1728.33	1734.66	944.35	944.33
Observations	378	378	378	378

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

In general, the main variables present the expected sign and are statistically significant. Furthermore, the confidents are within usual results, and have small variability throughout estimations. This is particularly so for the core variables in our study, those reflecting ownership, firm and competition, where our core contributions lie. *Output* elasticity provides the only relevant exception, as it shows large variability: between 0.311 and 1.295, implying that a 1% increase in the bus vehicle-kilometers supplied will increase total costs by just 0.311 % to 1.295%. [33] *Average speed* is negative or non-significant and our findings confirm that it is negatively related to a firm's cost performance. Therefore, public policies oriented towards increasing bus transit speeds might be desirable, because passengers (and/or taxpayers) would pay less for bus transportation. By contrast, *network length* is positive and significant or non-significant, as expected, although the result is not statistically significant in non-transaction cost estimations.

In the case of *labor input price*, which appears to be significant in all specifications, coefficients vary between 0.84 and 0.54, in line with usual results elsewhere in the literature. The coefficient for (quasi-fixed) *capital* is negative and highly significant, ranging from -1.58 to -0.33, which means that a greater availability of buses (paid for by the EMT) would bring about a reduction in total variable costs. While the available evidence shows divergent results for this variable, our results are in line with those reported, for instance, by Obeng (1985) and by Gagnepain (1998), and they have solid theoretical foundations, as discussed in Gagnepain (1998) and Croissant et al. (2013). The coefficient of the linear *time* trend is significant and negative in all models; thus, technological progress is significant in Barcelona's local bus industry. The control variables are mostly non-significant; thus, we do not find any differences between interurban lines and intra-municipal lines (Barcelona and municipality dummies). Likewise, the new subway services do not exert any competitive pressure on the bus lines (metro dummy). The value of lambda indicates the right skewness of the inefficiency term in three of the four specifications.

Our primary focus is, first and foremost, to compare the costs of public and private delivery. Interestingly, we do not find any cost savings with private delivery. In fact, *private* firms appear to have higher costs than those incurred by the public firm, this being true for all estimations, without and with transaction costs. Indeed, cost differences are more marked when transaction costs are taken into account.

[33] This wide span of results might be due to the fact that our unit of observation is 'line' rather than 'firm' or 'city' (as usually happens in other studies).

Second, we are interested in determining the effects of competition on costs. Here, the *year of tender* shows a negative and significant impact on costs, although non-transaction cost specifications show a non-significant result. In this regard, our results are relatively consistent with our second hypothesis – we expected stronger competitive pressures in the tender processes to result in cost reductions, particularly when transaction costs are taken into account.

Table 5. Cost frontier estimation - True Random Effects model (standard errors in parentheses)

	Dep. variable without transaction costs		Dep. variable with transaction costs	
	(5)	(6)	(7)	(8)
β_Y	0.813***	0.959***	-0.060	-0.202
	(0.1574)	(0.1262)	(0.8515)	(0.8701)
β_N	-0.997***	-0.343***	1.128**	0.869*
	(0.1059)	(0.0595)	(0.5109)	(0.5241)
β_{SP}	1.198***	0.087	-1.383	-1.122
	(0.2363)	(0.1388)	(0.8860)	(0.9592)
β_{PL}	1.111	0.824***	1.114**	1.394
	(0.1283)	(0.0782)	(0.4625)	(0.4277)
β_K	-0.463	-0.837***	0.583	1.017
	(0.3135)	(0.2600)	(1.7640)	(1.8111)
β_{Time}	0.000	0.000	-0.002***	-0.002***
	(0.0002)	(0.4177)	(0.0006)	(0.0006)
Tender year	-0.003*	-0.001	-0.015*	-0.015*
	(0.0017)	(0.0012)	(0.0088)	(0.0086)
Private	0.004		0.095***	
	(0.0034)		(0.0114)	
Tusgsal		0.008***		0.104***
		(0.0024)		(0.0140)
Rosanbus		0.006***		0.094***
		(0.0024)		(0.0133)
Soler Sauret		0.005***		0.089***
		(0.0026)		(0.0121)
β_{BCN}	-0.001	0.0002	0.003	0.004
	(0.0017)	(0.0014)	(0.0069)	(0.0068)
β_{MUN}	-0.001	-0.001	-0.001	-0.003
	(0.0009)	(0.0008)	(0.0043)	(0.0043)
β_{MET}	0.001	-0.001	0.003	0.001
	(0.0019)	(0.0013)	(0.0083)	(0.0083)
$\lambda = \sigma_u/\sigma_v$	1.288***	0.163	1.996***	2.879***
Log likelihood	1532.50	1765.69	951.24	953.22
Observations	405	405	405	405

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

We replicate our model using the True Random Effects model as proposed by Greene (2005). True Random Effects separate firm effects from inefficiency. Recall that one limitation of this model is that the line's inefficiency is uncorrelated with the explanatory variables and is assumed to be constant over time. In relatively long panels, and even in those cases in which the management's efficiency is constant, technical efficiency varies over time. Table 5 shows the results (see second-order coefficients in Table A-2 in the Appendix).

We obtain very similar results (sign and significance) for all variables in the equation, although estimation 5 shows somewhat contradictory results for line length and speed. Focusing on our key variables, we find, in relationship to ownership, that private lines are always more costly than public lines, except in one specification for which there is no statistical difference. In the case of the second hypothesis, we obtain cost decreases related to the tender process in three estimations, and no significant effect in one. Here, it is worth noting that we also considered the year of tender variable as a dummy variable with a value of one for all years following the tender process and 0 otherwise. Tender mechanisms lowered costs for all specifications.

One objective in fitting the frontier models is to estimate the inefficiency terms. In Table 6 we provide the inefficiency estimates for all specifications for the three different models. The interpretation of these coefficients is the excess variable costs of a line compared to the costs of the most efficient performer in our data. In all lines specifications, the inefficiency estimates fall within a realistic range. We perform the Kruskal-Wallis test for each specification by dividing the sample into public and privates lines. The main result for the majority of specifications is the statistical significance of the better performance of public as opposed to private lines. For some True Random Effects estimations, there is no statistical difference of the better performance of public compared to that of private lines. Note, however, that these specifications include ownership variables, so this variability is captured. All in all, public lines are more efficient than their private counterparts. [34]

[34] Even if that is beyond the main objectives of our analysis, we have been able to calculate scale and density economies. The parameter for scale economies does not suggest potential

Table 6. Estimated cost inefficiency scores by specification

		Statistics					Kruskal-Wallis test	
Stochastic frontier approach	Specification	Ownership	Mean	Standard deviation	Minimum	Maximum	Chi-sq	t-statistic
Battese and Coelli (1992)	1	Public	$2\cdot10^{-5}$	$2\cdot10^{-5}$	$2\cdot10^{-6}$	$1\cdot10^{-4}$	9.452	0.0021***
		Private	$1\cdot10^{-5}$	$1\cdot10^{-5}$	$2\cdot10^{-6}$	$7\cdot10^{-5}$		
	2	Public	$2\cdot10^{-5}$	$2\cdot10^{-5}$	$2\cdot10^{-6}$	$6\cdot10^{-5}$	9.446	0.0021***
		Private	$1\cdot10^{-5}$	$1\cdot10^{-5}$	$2\cdot10^{-6}$	$4\cdot10^{-5}$		
	3	Public	0.006	0.0041	0.0012	0.0195	5.714	0.0168**
		Private	0.005	0.0037	0.0027	0.0189		
	4	Public	0.007	0.0043	0.0012	0.0205	7.996	0.0047***
		Private	0.006	0.0035	0.0026	0.0176		
True Random Effects (Greene, 2005)	5	Public	0.003	0.0009	0.0012	0.0120	20.365	0.0001***
		Private	0.004	0.0023	0.0005	0.0173		
	6	Public	$3\cdot10^{-4}$	$1\cdot10^{-5}$	$3\cdot10^{-4}$	0.0004	8.791	0.003***
		Private	$3\cdot10^{-4}$	$2\cdot10^{-5}$	$1\cdot10^{-4}$	0.0004		
	7	Public	0.017	0.0064	0.0046	0.0557	0.896	0.344
		Private	0.023	0.0203	0.0021	0.1367		
	8	Public	0.018	0.0079	0.0039	0.0641	2.008	0.156
		Private	0.027	0.0261	0.0019	0.1661		

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

7 Robustness tests

In this section, we perform several robustness tests. First, we regress the model using a different approach to transaction costs in order to test different ways of allocating these costs; the definition of the model and all independent variables remain exactly the same, contrary to what happens to our dependent variable: Total variable costs (including transaction costs). Recall that in our analysis above we applied an egalitarian allocation of transaction costs (based on lines). Here, instead, we apply a non-egalitarian allocation: we allocate those transaction costs based on the different effort or time spent by the regulator in monitoring each private line. Effort and time will depend on

cost savings from extending the length of the lines. Regarding economies of density, we obtain the same trends and results. There are not differences between public values and private values; therefore, our analysis comparing public and private cost does not suffer from any distortion from this issue. Results available upon request.

the complexity of the line, and we take as an indicator for complexity the number of passengers carried by each line. Thus, for each year we allocate transaction costs based on the ridership of each line (proportional to the number of passengers carried). We use the same models as above; namely, the Battese and Coelli (1992) model and True Random Effects (Table 7).

Table 7. Robustness tests *Battese and Coelli and True Random Effects*

	Dependent variable with transaction costs by passengers			
	Battese and Coelli		True Random Effects	
	(9)	*(10)*	(11)	(12)
β_Y	0.583*	-0.063	1.278	0.323
	(0.3372)	(0.5422)	(0.9359)	(0.6878)
β_N	-0.368*	-0.001	0.035	-0.042
	(0.2176)	(0.2918)	(0.3674)	(0.2635)
β_{SP}	-1.103***	-1.043*	0.221	-0.457
	(0.3828)	(0.6280)	(1.0246)	(0.7727)
β_{PL}	0.598***	0.472*	0.315	0.330
	(0.1751)	(0.2863)	(0.5325)	(0.3817)
β_K	0.156	1.070	-1.605	0.278
	(0.6766)	(1.1030)	(1.8096)	(1.3321)
β_{Time}	-5.82e-05	-0.001***	-0.001	-0.001***
	(0.0002)	(0.0003)	(0.0006)	(0.0004)
Tender year	-0.011***	-0.014***	-0.014**	-0.013***
	(0.0024)	(0.0042)	(0.0065)	(0.0047)
Private	0.048***		0.060***	
	(0.0050)		(0.01307)	
Tusgsal		0.061***		0.057***
		(0.006)		(0.0096)
Rosanbus		0.094***		0.084***
		(0.0057)		(0.0091)
Soler Sauret		*0.061****		0.057***
		(0.0067)		(0.0104)
β_{BCN}	0.004	0.009***	0.005	0.0073**
	(0.0026)	(0.0024)	(0.0056)	(0.0037)
β_{MUN}	-0.007**	-0.016***	-0.007**	-0.013***
	(0.0033)	(0.0019)	(0.0105)	(0.0021)
β_{MET}	-0.002	-0.004	-0.007	-0.003
	(0.0019)	(0.0034)	(0.0045)	(0.0039)
η	0.063***	0.498***	-	-
	(0.0069)	(0.1156)		
$\lambda = \sigma_u/\sigma_v$	79.59	0.05	3.828***	2.015***
Log likelihood	1275.71	1127.25	1101.58	1142.51
Observations	378	378	405	405

Note: *The dependent variable includes transaction costs allocated by passengers in the line.*
Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

Table 8. Estimated cost inefficiency scores by specification

		Statistics					Kruskal-Wallis test	
Stochastic frontier approach	Specification	Ownership	Mean	Standard deviation	Mini-mum	Maxi-mum	Chi-sq	t-statistic
Battese and Coelli (1992)	9	Public	0.004	0.0021	0.0012	0.0096	172.7	0.0001***
		Private	0.034	0.0288	0.0019	0.1444		
	10	Public	0.0001	0.0006	$1 \cdot 10^{-5}$	0.0029	5.118	0.024**
		Private	0.0002	0.0006	$2 \cdot 10^{-5}$	0.0058		
True Random Effects	11	Public	0.011	0.0032	0.0039	0.0221	13.00	0.0003***
		Private	0.020	0.0212	0.0009	0.1040		
	12	Public	0.010	0.0027	0.0047	0.0221	2.07	0.1502
		Private	0.014	0.0122	0.0011	0.0634		

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

In the case of our key variables, we obtain the same results as those obtained in all the previous models for the ownership variable: private lines are more costly than public lines. In the case of the *tender* variable, a negative and significant coefficient is found for all specifications. Adopting different methods to allocate the transaction costs does not bring about relevant changes in the variables related to ownership and the tender processes. All in all, our results show remarkable stability across all the robustness tests.

The estimate of λ is statistically significant in most specifications, suggesting evidence of technical inefficiency in the data. We provide separate inefficiency measures from the Battese and Coelli and the True Random Effects models (Table 8) for the public and private lines. Inefficiency estimates stand at around 0.01-0.03 (which are results in line with the available evidence),[35] which represents a potential cost saving of between 1 and 3 per cent. The public-private comparison shows a better performance of public lines according to the Kruskal-Wallis test.

[35] In the case of Battese & Coelli without transaction costs, we obtain extremely small inefficiency scores, but we still find statistically significant differences between public and private.

Table 9. Robustness test. Battese and Coelli and True Random Effects (standard errors in parentheses)

	Dependent variable with transaction costs by passengers			
	Battese and Coelli		True Random Effects	
	(13)	(14)	(15)	(16)
β_Y	0.700**	0.045	1.404	0.438
	(0.3343)	(0.5403)	(0.9582)	(0.7086)
β_N	-0.289	0.010	0.174	-0.011
	(0.2158)	(0.2912)	(0.4101)	(0.2751)
β_{SP}	-1.094***	-1.025	0.135	-0.451
	(0.3819)	(0.6272)	(1.0935)	(0.7718)
β_{PL}	0.525***	0.419	0.226	0.275
	(0.1750)	(0.2859)	(0.5604)	(0.3937)
β_K	-0.093	0.861	-1.865	0.048
	(0.6699)	(1.0983)	(1.8565)	(1.3684)
β_{Time}	8.57e-06	-0.001***	-0.001***	-0.001***
	(0.0002)	(0.0003)	(0.0005)	(0.0004)
Tender year	-0.011***	-0.014***	-0.014***	-0.012***
	(0.0022)	(0.0039)	(0.0056)	(0.0045
Private	0.0510***		0.063***	
	(0.0050)		(0.0134)	
Tusgsal		0.061***		0.058***
		(0.0060)		(0.0097)
Rosanbus		0.094***		0.085***
		(0.0057)		(0.009)
Soler Sauret		0.067***		0.063***
		(0.0071)		(0.0107)
β_{BCN}	0.005*	0.009***	0.005	0.007***
	(0.0026)	(0.0024)	(0.0057)	(0.0037)
β_{MUN}	-0.008**	-0.016***	-0.007**	-0.013***
	(0.0034)	(0.0019)	(0.0027)	(0.0022)
β_{MET}	-0.002	-0.005	-0.007	-0.004
	(0.0018)	(0.0034)	(0.004)	(0.0038)
η	0.064***	0.496***	-	-
	(0.0071)	(0.1261)		
$\lambda = \sigma_u/\sigma_v$	3.971***	0.05	3.624***	1.988***
Log likelihood	1278.75	1127.47	1105.07	1142.82
Observations	378	378	405	405

Note: Dependent variable includes transaction costs allocated by passengers per line, and dummy equals 1 for all years after the tender. Significance levels: *10 per cent; **5 per cent; ***1 per cent

The second robustness test relates to the specification of the competition variable. In our analysis above, we specified competition as ad dummy with value 1 the year following a tender, and 0 otherwise. Here, instead, this independent variable (everything else remains equal) takes a value of 1 for all years after the tender process and 0 otherwise. We perform the same specifications (allocating transaction costs by number of passengers per line) as in the previous robustness tests. Table 9 displays the results (second-order coefficients in Table A-3).

The competition variable is also negative for all specifications. This indicates that our results are robust to different specifications. Moreover, the ownership variables are always significant; the private lines are more costly than the public lines.

8 Discussion

Throughout our empirical exercise, we find that public delivery costs are lower than private costs, and the cost advantage of public delivery further increases when transaction costs are considered, as could be expected. In this regard, our results are not in line with the expectation that public and private firms will face similar costs with mixed delivery. Furthermore, our results show the importance of considering transaction costs, as they reflect the costs paid by the citizens to maintain the regulator, and they ensure a better comparison of the total costs paid by bus users and taxpayers. The failure to consider transaction costs in previous empirical studies may well have biased the results of the cost comparisons undertaken to the detriment of public firms.

It is worth emphasizing as well that tenders for concessions tend to contribute to decrease costs. Thus, our findings provide support to expectations that mixed delivery results in more competitive tenders. Our results show that competition between private firms is effective and that private firm delivery is more costly, particularly when we take into account their transaction costs.

Governments and regulators face different challenges. First, they have to fulfill the requirements of competitive tendering that emanate from higher legal frameworks promoting contests in which there is no common carrier providing the service. Second, they are forced to seek more flexible agreements to guarantee a workable relationship based on trust with the private operator. This is particularly so if, as Heinrich and Marschke (2010) suggest, we take into account the fact that incentives should be used for agents that are responsive to them, and that procedures and regulations affecting bureaucracies or public firms can constrain them more than they constrain private firms.[36]

Competitive tendering involves greater rigidities than a negotiated contract, since it reduces government opportunities for modifying the delivery conditions. These rigidities may be weakened somewhat by changes introduced by the regulator into the contract. The big rate of new contracts awarded to incumbents -quite common, as shown in Boitani and Cambini (2006), Beck and Walter (2013), etc.- suggests a partnership of trust exists between the regulator and private operators. All this seems to indicate that competitive tendering conditions are increasingly tending to resemble a negotiated performance-based contract, as observed. In this context, regulator accountability is weakened.

However, private production (supply from private vendors) within a mixed delivery system may also be used as the basis for a credible threat for the privatization of some of publicly delivered services. Indeed, this threat has been used in Barcelona at the time of labor conflicts within the TMB (Albalate, Bel and Calzada, 2012: 97). As such, it may have contributed to cost containment in the public firm, thus helping to explain its relatively good economic performance (compared to that of the private firms) in Barcelona.

9 Conclusion

In this paper we have evaluated the impact of firm ownership and competition on the total variable costs of a local bus transit service within a mixed delivery regime, paying particular attention to the presence of the transaction costs involved in private delivery. We have estimated a translog stochastic cost frontier in an attempt at shedding light on the effect of the delivery choices. The main results can be summarized as follows.

First, public delivery has been found to perform better than the private operators selected via competitive tendering. The public company provides bus services at a lower cost to that incurred by the private companies, and

[36] In this regard, we note that a limitation of our analysis is that we do not have access to detailed information on quality indicators such as punctuality, perceived quality, etc. Given that the regulator can impose stricter obligations on quality in private firms, this could bias the costs of private firms upward. We thank a referee for this insight.

differences in cost increase when we take into account transaction costs. This result is a clear indication that cost comparisons that ignore transaction costs may well be underestimating private costs and giving a misleading picture of the costs associated with public and private delivery.

Second, we find evidence that operating under tender procedures implies cost savings in the private segments of the service, because the tender process tends to affect variable costs. However, potential savings from tenders are not large enough to offset higher costs incurred by private firms when compared to public delivery. A possible explanation is the existence of relational contracting due to complex contracts and incompleteness. The regulator might prefer to establish strong relationships based on trust with specific private providers because it needs to make constant changes to the contract. In this regard, it is important to emphasize that maintaining public delivery helps the government to understand better the real costs of services, thereby mitigating information asymmetries.

The debate on the reform of public service delivery has most frequently focused on the dilemma between pure public and pure private delivery modes. Our analysis suggests that mixed delivery (as long as economic conditions for service fragmentation exist) creates a framework in which policy makers and regulators can employ different tools to pursue different objectives; for instance, using the more flexible management practices of private firms to adjust to non-contractual events or to unforeseen needs. Note, however, that this greater flexibility can result in higher costs being incurred by the regulator, which casts doubts on whether mixed delivery is used as an overall cost minimizing strategy.

We are aware that the generalization of our results is by no means straightforward: first, we analyze a specific service in a specific jurisdiction (that of Barcelona) and, second, while the number of mixed delivery systems is increasing, they are far from usual in the organization of the delivery of public services in local public transportation. This said, our findings should prove useful beyond the specific case of local bus services and be of relevance in other services where mixed delivery is frequent, in particular, those services, such as solid waste collection, where transportation is a major characteristic of the service. Clearly, future research needs to devote more attention to analyzing in greater depth the characteristics and effects of mixed and hybrid delivery choices. Likewise, further empirical studies of mixed delivery in other services would enhance our knowledge of its advantages and drawbacks.

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APPENDIX

Table A-1. Second order coefficients- Battese and Coelli specifications (standard errors in parentheses)

	Dep. variable without transaction costs		Dep. variable with transaction costs	
	(1)	(2)	(3)	(4)
$\beta_{Y'}$	-0.019**	-0.019**	0.042**	0.048***
	(0.009)	(0.008)	(0.018)	(0.016)
$\beta_{N'}$	0.006*	0.008**	0.000	0.004
	(0.003)	(0.003)	(0.049)	(0.0500)
$\beta_{SP'}$	0.132***	0.132***	0.110	0.104
	(0.005)	(0.005)	(0.083)	(0.101)
$\beta_{PL'}$	0.161***	0.162***	-0.019	-0.020
	(0.004)	(0.004)	(0.335)	(0.033)
$\beta_{K'}$	-0.078**	-0.071*	-0.086	-0.060
	(0.039)	(0.038)	(0.056)	(0.089)
β_{YN}	0.000	0.001	-0.164***	-0.164***
	(0.006)	(0.006)	(0.031)	(0.058)
β_{YSP}	-0.033**	-0.034***	0.074***	0.077
	(0.013)	(0.012)	(0.019)	(0.090)
β_{YPL}	0.001	-0.002	0.033**	0.029*
	(0.006)	(0.005)	(0.016)	(0.015)
β_{YK}	0.039**	0.037**	0.002	-0.010
	(0.019)	(0.018)	(0.026)	(0.031)
β_{NSP}	0.003	0.004	0.138**	0.135**
	(0.004)	(0.004)	(0.058)	(0.057)
β_{NPL}	0.001	0.000	-0.002	-0.004
	(0.003)	(0.003)	(0.021)	(0.027)
β_{NK}	-0.001	-0.005	0.273***	0.269**
	(0.012)	(0.011)	(0.059)	(0.111)
β_{SPK}	0.063**	0.064***	-0.200***	-0.204
	(0.026)	(0.024)	(0.047)	(0.171)
β_{SPPL}	-0.159***	-0.158***	-0.096***	-0.094**
	(0.006)	(0.005)	(0.035)	(0.042)
β_{PLK}	0.001	0.006	-0.107**	-0.101**
	(0.012)	(0.011)	(0.041)	(0.040)
Log likelihood	1728.33	1734.66	944.35	944.33
Observations	378	378	378	378

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

Table A-2. Second order coefficients on True Random Effects specifications (standard errors in parentheses)

	Dep. variable without transaction costs		*Dep. variable with transaction costs*	
	(5)	(6)	(7)	(8)
$\beta_{Y'}$	0.058***	0.020**	0.094	0.118*
	(0.0126)	(0.0098)	(0.0687)	(0.0683)
$\beta_{N'}$	0.004	0.003	0.004	0.019
	(0.0063)	(0.0047)	(0.0257)	(0.0278)
$\beta_{SP'}$	0.113***	0.127***	0.130***	0.130**
	(0.0120)	(0.0072)	(0.049)	(0.0532)
$\beta_{PL'}$	0.173***	0.166***	-0.001	0.005
	(0.0083)	(0.0055)	(0.0319)	(0.0333)
$\beta_{K'}$	0.305***	0.110**	0.169	0.323
	(0.0523)	(0.043	(0.3121)	(0.3106)
β_{YN}	0.100***	0.032***	-0.122**	-0.096*
	(0.0104)	(0.0061)	(0.0502)	(0.0516)
β_{YSP}	-0.163***	-0.054***	0.046	0.0236
	(0.0233)	(0.0138)	(0.0855)	(0.0913)
β_{YPL}	-0.057***	-0.025***	0.000	-0.027
	(0.0115)	(0.0067)	(0.0467)	(0.0425)
β_{YK}	-0.134***	-0.048**	-0.118	-0.181
	(0.0254)	(0.0205)	(0.1453)	(0.1445)
β_{NSP}	0.003	0.006	0.128***	0.124***
	(0.0075)	(0.0060)	(0.0345)	(0.0365)
β_{NPL}	-0.027***	-0.007*	-0.012	-0.022
	(0.0050)	(0.004)	(0.0237)	(0.0245)
β_{NK}	-0.199***	-0.063***	0.198**	0.142
	(0.0197)	(0.0124)	(0.0973)	(0.1005)
β_{SPK}	0.325***	0.101***	-0.138	-0.092
	(0.0466)	(0.0279)	(0.172)	(0.1873)
β_{SPPL}	-0.137***	-0.160***	-0.098**	-0.101**
	(0.0095)	(0.0060)	(0.0391)	(0.0424)
β_{PLK}	0.124***	0.054***	-0.042	0.016
	(0.0231)	(0.0143)	(0.0967)	(0.0883)
$\lambda= \sigma_u/\sigma_v$	1.288	0.163	1.997***	2.879
Log likelihood	1532.50	1765.69	951.24	953.22
Observations	405	405	405	405

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

Table A-3. Second order coefficients - Battese and Coelli and True Random Effects model being dependent variable with transaction costs by passengers (standard errors in parentheses)

	Dependent variable with transaction costs by passengers			
	Battese and Coelli (1992)		True Random Effects model	
	(13)	(14)	(15)	(16)
$\beta_{Y'}$	0.024	-0.069	-0.015	0.046
	(0.0275)	(0.0441)	(0.0785)	(0.0553)
$\beta_{N'}$	-0.021	0.029	0.056***	0.033*
	(0.0175)	(0.0175)	(0.0209)	(0.0179)
$\beta_{SP'}$	0.137***	0.037	0.087	0.052
	(0.0225)	(0.0258)	(0.0566)	(0.0412)
$\beta_{PL'}$	0.146***	0.185***	0.202***	0.194***
	(0.0116)	(0.0195)	(0.0393)	(0.0309)
$\beta_{K'}$	0.220*	0.282	-0.066	0.190
	(0.1171)	(0.1926)	(0.2978)	(0.2114)
β_{YN}	0.035*	0.000	-0.006	0.009
	(0.0199)	(0.0292)	(0.0376)	(0.0269)
β_{YSP}	0.048	0.100	-0.055	0.031
	(0.0379)	(0.0633)	(0.1016)	(0.0824)
β_{YPL}	-0.003	0.019	0.009	0.022
	(0.0167)	(0.0279)	(0.0525)	(0.038)
β_{YK}	-0.082	-0.140	0.031	-0.093
	(0.0564)	(0.0916)	(0.1524)	(0.1077)
β_{NSP}	0.011	0.012	-0.068	-0.001
	(0.0135)	(0.0185)	(0.0286)	(0.0223)
β_{NPL}	-0.005	-0.026	-0.020	-0.032*
	(0.010)	(0.0138)	(0.0188)	(0.0164)
β_{NK}	-0.060	-0.012	-0.003	-0.028
	(0.0367)	(0.0569)	(0.0723)	(0.0520)
β_{SPK}	-0.095	-0.207	0.110	-0.063
	(0.0740)	(0.1257)	(0.2029)	(0.1629)
β_{SPPL}	-0.142***	-0.241***	-0.158***	-0.206***
	(0.0159)	(0.0269)	(0.0471)	(0.0418)
β_{PLK}	0.010	-0.006	0.002	-0.012
	(0.0338)	(0.0576)	(0.1061)	(0.0779)
$\lambda = \sigma_u/\sigma_v$	79.59	0.05	3.828***	2.015***
Log likelihood	1275.71	1127.25	1101.58	1142.51
Observations	378	378	405	405

Significance levels: * 10 per cent; ** 5 per cent; *** 1 per cent

CHAPTER 5

CONCLUSIONS, POLICY IMPLICATIONS, LIMITATIONS AND FUTURE RESEARCH

1 Conclusions and policy implications

The most important externalities caused by road transport are congestion, environmental damage, and accidents, among others. In the presence of externalities the market is incapable of achieving an efficient equilibrium. Since the market is incapable of reaching this equilibrium, a number of government interventions have been suggested. In chapters two and three we analyze those externalities related to environmental damage. While in chapter two we analyze CO_2 emissions from urban daily mobility, in chapter three we analyze the impact of speed limit policies on NO_x and PM_{10} pollutants. For reducing congestion, accidents and environmental damage, a sustainable public transport system for a city is to be desired, so chapter four proceeds to analyze and recommend policies that ensure value for money in public transportation.

From chapter 2, we obtain the Gini index on CO_2 emissions, which is 0.496. If we compare this index with the income-related Gini index for 2006 for the same area (recorded at 0.296), the inequality for mobility emissions is twenty points higher and, as such, is much more pronounced. Despite all cautions about calculating this index taking into account only one-day journeys, the difference, nevertheless, encourages the promotion of policies to reduce CO_2 transport externalities. Given that the top 10% of pollutants are responsible for 49% of total emissions, this implies that mobility policies should be addressed in the corresponding direction and with the purpose to cut CO_2 emissions from high-emitters. For the four mobility expenditures analyzed, three of them are designed in the right direction to reduce CO_2 externalities: fuel taxes, parking policies and public transportation. However, the non-existence of tolled corridors accessing the city of Barcelona, impacts in the wrong direction. Extending tolls to all motorways accessing the inner city should reduce the use of private vehicles and, thus, have the potential to reduce carbon emissions. Nowadays, there is a public debate in Catalonia about ending some motorway concessions. The regional government may promote more free motorways in the Barcelona metro area. Parallel to this policy, the regional government is interested in implementing a fixed tax rather than a vehicle/km travelled taxation (in line with the principles of the Euro vignette). Both policies are undesirable from the view of the social optimum, since the level of externalities will increase. So, our recommendation is not only to maintain tolled motorways around the city of Barcelona, but also to extend them.

NO_x and PM_{10} can negatively impact on the human respiratory system, reduce lung function and increase premature mortality. In chapter 3 we analyze the effects of reducing maximum speed limit to 80 km/h and implementing variable speed systems, in order to reduce both pollutants. For nitrogen dioxides, the 80 km/h speed limit has not shown an air quality improvement, while for PM_{10} there is a 5.6% deterioration. On the other hand, the policy of implementing variable speed controls has positive environmental effects in reducing both pollutants. Congestion reduction in peak hours due to a variable speed system can explain this positive effect in comparison to the 80 km/h speed limit. The important effect of ensuring a free flow of traffic and avoiding stop-and-starts might be achieved under the variable speed limit system rather than by decreasing the maximum speed limit from 120/100 km/h to 80 km/h. Therefore, we recommend extending the variable speed limit, which has been expanded in the Barcelona metro area since 2010. Furthermore, in other cities that face pollution problems related to NO_x and PM_{10} from traffic, such as Madrid, implementing a variable speed limit system should reduce this pollution. Thanks to the use of quantiles, we can see that the recommendation to activate an 80 km/h speed limit only on days when there are high levels of both pollutants, makes no sense if the objective is to reduce the pollution levels on those days.

In chapter 4 we analyze a mixed delivery scheme for the Barcelona urban bus system. Ensuring a viable public transport system, should not increase costs significantly. So we have evaluated the impact of firm ownership and competition on the total variable costs of the Barcelona local bus service. We find that public delivery costs are lower than private costs, and competition, through the tender process, contributes to a decrease in private delivery costs. Furthermore, our results show the importance of considering transaction costs, as they reflect the costs paid by the citizens to maintain the regulator, and they ensure a better comparison of the total costs paid by bus users and taxpayers. The failure to consider transaction costs in previous empirical studies may well have biased the results of the cost comparisons undertaken, to the detriment of public firms. Private production within a mixed delivery system may also be used as the basis for a credible threat to privatize some of the publicly delivered services. Indeed, this threat has been used in Barcelona, at the time of labor conflicts within the TMB.

To summarize, the results obtained in this dissertation provide a better understanding of the evaluation of mobility policies. Although the three chapters are framed specifically around the Metropolitan Area of Barcelona, the contribution to the methodology for evaluating public policies can be equally well adapted for other urban areas where policy-makers may have an interest. After all, this thesis was not intended to be an analysis of a case study of the conurbation of Barcelona, but to design an evaluation of public policies on mobility, applicable to other cities. This thesis presents a contribution to the debate on evaluating mobility public policies in Spain. Since, while most of the European Union member states have a high degree of evaluation culture maturity, in Spain in general this is still pending, and specifically, in mobility and infrastructures.

2 Limitations and future research

In chapter two, the household travel survey data used have several shortcomings. First, there is insufficient information to compute the CO_2 emissions of commuters that use more than one travel mode. Future surveys would be greatly improved if respondents were asked to indicate where they changed modes of transport. Second, the sample bias with regard to mobility expenditure means high-emitters are over- represented. However, it is our belief that these shortcomings have not seriously affected our main findings and conclusions. A possible emissions-per-capita bias might be present because we only have one-day emissions calculations, and none for a longer period of time. The Gini Index should be taken with caution. Most of the CO_2 emissions related to mobility are attributable to factors and attitudes intrinsic to each person, which means an individual's socio-economic characteristics only account for a part of this variability. Mobility patterns differ from one city to another as do the socio-economic characteristics of their respective citizens. We recommend expanding future research in other cities using the quantiles technique.

In chapter three, the pollutant measures were daily rather hourly. In this case, congestion from peak hours might be difficult to identify and some effects may not be taken into account. We recommend expanding difference-in-differences analysis on these policies with a more precise time period.

In chapter four, future research on the provider-awarding mechanism in the bus sector should be undertaken. Governments and regulators have to fulfill the requirements of competitive tendering that emanate from higher legal frameworks, promoting contests in which there is no common carrier providing the service. Besides, they are forced to seek more flexible agreements to guarantee a workable relationship based on trust with the private operator. And, competitive tendering involves greater rigidities than a negotiated contract, since it reduces government opportunities for modifying the delivery conditions. We propose that further research should focus on comparing competitive tendering and negotiated contracts, especially in small municipalities, where transaction costs might be higher.

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