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Essays on the Economics of Air Quality Control

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To Gilma Milena, Jorge Esteban, Juan Manuel,

my parents, my sister, my grandmother,

my mother-in-law , and

my whole family

"For as the rain and the snow come down from heaven and do not return there but water the earth, making it bring forth and sprout, giving seed to the sower and bread to the eater, so shall my word be that goes out from my mouth; it shall not return to me empty, but it shall accomplish that which I purpose, and shall succeed in the thing for which I sent it. (Isaiah 55:10-11)"

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As a child, I learnt that gratitude is more than being courteous. Although there are no accurate words to describe this great feeling, gratitude is a feeling that it is impossible not to express. It is like the joy when a new day begins. That day is a gift and it is enough reason for thanks. I have had the pleasure of meeting several people inside and outside of academia who have contributed to the achievement of my professional goals during this journey to reach the level of PhD. I want to manifest my gratitude to all of them.

I would like to start by expressing my infinite thanks to Jesus, the name that is above every name. Thanks my almighty God for your blessings and for hearing my prayers. I present this thesis to you and I would like you to receive it as a pleasing offering to your altar. My Lord, may all my work be to your glory. I am also grateful to my sweet Virgin Mary. Thanks, my heavenly mother, for your blessings, and please intercede for me and my family with the Lord.

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As in the Parable of the Talents, every person receives a talent, and it is our decision to invest and grow our talents. God gives the grace for using them at the service and for the good for others.

Jorge Alexander Bonilla Londoño
May, 2013
Gothenburg, Sweden

Abstracts

This thesis consists of five self-contained chapters:

Chapter 1: Effects of driving restrictions on air quality and car use in Bogota

Rationing car use at certain times of the day based on license plate numbers has become a popular policy to address traffic congestion and air pollution in several cities around the world. This paper analyzes the effects of moderate and drastic driving restrictions of the program *Pico y Placa* on air pollution and car use in Bogota. Because the program was implemented in phases, it was possible not only to study the impact of the program, but also to distinguish between the short- and long-run effects for each phase of restriction. Using hourly carbon monoxide data, monthly information on gasoline consumption, and vehicle registration and vehicle sales data, this paper shows differentiated effects of *Pico y Placa* on air quality and car use in the short- and long-run and between phases of the program. Although there was an initial improvement in air quality in both phases of the program, carbon monoxide concentrations, vehicle ownership, and total driving actually increased when drastic restrictions were implemented. Gasoline taxes, on the other hand, have tended to reduce gasoline usage in Bogota, suggesting that a price-based mechanism would be more effective in reducing driving.

JEL Classification: D62, R41, Q53, C54
Keywords: driving restrictions, air pollution, vehicle sales, policy evaluation.

Chapter 2: Air pollution dynamics and the need for temporally differentiated road pricing

Nowadays, road traffic is a major source of urban air pollution. In this paper we investigate the effects of the temporal variation of pollution dispersion, traffic flows and vehicular emissions on pollution concentration and illustrate the need for temporally differentiated road pricing through an application to the case of the congestion charge in Stockholm, Sweden. By accounting explicitly for the role of pollution dispersion on optimal road pricing, we allow for a more comprehensive view of the economy-ecology interactions at stake, showing that price differentiation is an optimal response to the physical

environment. Most congestion charges in place incorporate price bans to mitigate congestion. Our analysis indicates that, to ensure compliance with air quality standards, such price variations should also be a response to limited pollution dispersion.

JEL Classification: D62, R41, C54, Q53, Q57
Keywords: road pricing, congestion, air pollution, pollution dispersion.

Chapter 3: Synergies and trade-offs between climate and local air pollution policies in Sweden

In this paper, we explore the synergies and tradeoffs between abatement of global and local pollution. We build a unique dataset of Swedish combined heat and power plants with detailed boiler-level data 2001-2009 on not only production and inputs but also on emissions of CO_2 and NO_x. Both pollutants are regulated by strict policies in Sweden. CO_2 is subject to the European Union Emission Trading Scheme and Swedish carbon taxes; NO_x - as a precursor of acid rain and eutrophication - is regulated by a heavy fee. Using a quadratic directional output distance function, we characterize changes in technical efficiency as well as patterns of substitutability in response to the policies mentioned.

JEL Classification: H23, L51, L94, L98, Q48.
Keywords: Environmental policies, shadow pricing, directional distance function, climate change, local pollution, policy interactions.

Chapter 4: Diffusion of NO_x abatement technologies in Sweden

Though economists argue for the use of single instruments, we often observe the use of multiple instruments in actual regulations. These may include permit schemes, taxes, fees, subsidies and emission standards. In order to evaluate these combinations and to better understand their effects, we need more empirical analysis of how they interact. They might, for example, be either complements or substitutes; this might even vary between different types of instrument. As a case study we look at detailed data of NO_x emissions from large combustion plants in Sweden. These are regulated both by a refunded charge and at the same time plant level emission standards. We study the adoption of abatement technologies under the combined effect of these charges and standards. The results indicate that the net charge has an effect and one that is complementary to the standards, but only for end-of-pipe post-combustion technologies.

JEL Classification: H23, O33, O38, Q52

Keywords: Technology diffusion, NO_x abatement technologies, environmental regulations, refunded emission charge.

Chapter 5: Air quality combination forecasting with an application to Bogota

The bulk of existing work on the statistical forecasting of air quality is based on either neural networks or linear regressions. The present paper shows how forecast combination can be used to produce more accurate results. This is accomplished using both Monte Carlo simulation and an extensive application to air quality in Bogota, one of the largest and most polluted cities in Latin America.

JEL Classification: C45, C53, Q53

Keywords: Air quality forecasting, pollution, Bogota, forecast combination, neural networks.

Overview

The growth process and urbanization have brought with them the deterioration of the quality of air that people breathe in urban environments. Approximately 1.34 million people in the world die prematurely because of outdoor air pollution (WHO, 2011). Epidemiological studies have found that the risk of mortality increases under severe exposure to high concentrations (see Lippmann, 2003; Brunekreef and Holgate, 2002). Several of the health impacts are associated with chronic cardiovascular and respiratory diseases (see e.g., Kassomenos et al., 2008). The sources of air pollution are mainly emissions generated by the combustion of fossil fuels from industry facilities (point sources) and the transport sector (mobile sources).

Although environmentalists and scientists have gained some knowledge in the last decades about the causes of urban air pollution, and recently about its effects on human health, that knowledge is not very valuable if those findings are not translated into mechanisms to control air pollution and reduce human exposure. Because air quality is a public good, its socially optimal level of provision cannot be ensured through markets. Therefore, air pollution is considered an externality whose effects may cost-effectively be mitigated through the use of price-based instruments (see Sterner and Coria, 2012).

Developed countries seem to have improved in public awareness and reduction of emissions. The implementation of environmental standards, improved fuel quality, shifts away from heavy industry, and technological development tend to be the causal factors in that improvement. However, the situation in developing countries is not encouraging; their air quality is getting worse in most heavily populated cities. Nowadays the contribution of the transport sector to air pollution has become more evident and proportionately important because of increases in car ownership and driving.

Cities have applied different strategies to abate emissions and the effectiveness of those policies has been diverse (see Sterner and Coria, 2012). To understand those policy responses, more research is required in the field of air pollution control. This is not only because of general differences between policies, but also because of other specific factors: for instance, the sources of air pollution vary in composition and size, ambient air concentrations are governed by meteorological factors, and the regulated subjects may adapt their behavior over time as a consequence of the policy. This thesis seeks to analyze the effect of policies on air pollution control and study the design of empirical tools to prevent harmful health effects. The research here aims to contribute to the understanding and evaluation of air pollution control policies for some case studies.

The thesis is presented in three themes. The first theme is Mobile Sources. It contains two chapters associated with two policies regulating emissions from the transport sector: one in a developing country imposing restrictions on driving rights based on a command and control mechanism and another in a developed country employing price-based incentives to reduce vehicular traffic. The first chapter evaluates the effectiveness of the driving restriction program *Pico y Placa* implemented in Bogota, Colombia to reduce congestion and air pollution. Traffic congestion and air pollution have been major problems in the city for a long time. Because the restrictions were implemented in increasingly stringent phases, this chapter not only evaluates the effectiveness of the program but also distinguishes between the short- and long-run effects for each phase, and analyzes how drivers respond to a phased-in program. Using data on carbon monoxide (CO), gasoline consumption, and vehicle sales and registration, this study shows that CO concentrations and vehicle use did not decrease in the long run, despite an initial improvement in air quality for some periods of the day in both phases of the program. In fact, there is evidence of increased CO concentrations during the implementation of drastic restrictions. There is also evidence that *Pico y Placa* increased vehicle ownership and driving, more so in response to drastic restrictions, as drivers adapted to the program. In contrast, gasoline consumption in Bogota has responded negatively to increasing gasoline taxes, suggesting that market mechanisms may be more effective than driving restrictions.

Continuing with the theme of Mobile Sources, the second chapter analyzes how the congestion charge already implemented in Stockholm, Sweden could be used as an instrument to reduce vehicular traffic and satisfy air quality standards (AQS), accounting for temporal variations in the restricted dispersion of certain pollutants. The purpose of the chapter is to illustrate the need for temporally differentiated road pricing using Stockholm as a case study. By accounting explicitly for the role of pollution dispersion on optimal road pricing, this study allows for a more comprehensive view of the economy-ecology interactions at stake, showing that price differentiation is an optimal response to the physical environment. The study indicates that the achievement of AQSs in Stockholm would require the charge to be increased for all seasons and most hours of the day. In relative terms, a much larger increase is needed in the spring; the increment also should be larger in the morning to offset the negative effect of reduced assimilative capacity on pollution concentration. The basic principles and the methodology developed in this chapter could be easily adapted to other cities using information that is available in most countries.

The second theme is Point Sources. This theme also comprises two chapters, related to the implementation of environmental policies to reduce carbon and nitrogen oxides emissions from large combustion plants in Sweden. In particular, the third chapter analyzes the effects of the European Union Emissions Trading System (EU ETS), the Swedish carbon tax, and the refunded charge on nitrogen oxides (NO_x) on the relative performance of Swedish combined heat and power (CHP) plants with respect to carbon dioxide (CO_2) and NO_x emissions. This chapter attempts to study the interaction between multiple layers of regulation and the interaction between multiple pollutants. Hence, environmental policies aiming at reducing CO_2 emissions might affect emissions of other pollutants from firms adjusting their production processes in response to climate policy. Evaluating patterns of technical progress, substitution between CO_2 and NO_x and shadow prices of these pollutants between the periods 2001-2004 and 2006-2009, this study finds that CO_2 and NO_x are substitutes in the CHP sector and that the degree of substitution increased after the introduction of the EU ETS, as a response to technological development and regulatory changes that led to a reduced CO_2/NO_x relative price. The results also indicate that CO_2 is more sensitive to prices that NO_x. Therefore, if the regulator wants to encourage a large reduction in NO_x emissions, the charge must be increased to a much higher level than its current value.

The fourth chapter is about the interaction of the refunded nitrogen oxides charge and emission standards and their effect on the timing and decision to invest in abatement technologies. The chapter aims to explore whether these policy instruments are either complements or substitutes in encouraging the diffusion of NO_x reducing technologies. This study shows that the net NO_x charge does not seem to promote adoption of combustion or flue gas condensation technologies. The net NO_x charge only plays a role in stimulating adoption of the most expensive technologies: post-combustion installations. These types of technologies can be characterized as end-of-pipe solutions which allow firms to choose emissions independently from output to a much larger extent than the other technologies, possibly explaining why firms are more responsive to the charge. The results also point out that the emission standards and the charge tend to be complementary: a higher net charge promotes adoption among boilers in counties with more stringent emission standards.

The last theme is Forecasting. This consists of a single chapter that complements the previous analysis by proposing a forecasting method (forecast combination) of air pollutant concentrations as an alternative to more common statistical air quality forecasting approaches such as linear regressions (LR) and neural networks (NN). The method is applied to Bogota, the fifth most populated city in Latin America, with around 7.4 million inhabitants, where urban air pollutant concentrations have at times been well above the national air quality standards. The results show that forecast combination always performs better than using NN, the benchmark statistical approach. Likewise, the best performing individual forecast is generally dominated by the best performing forecast combination. Moreover, the combinations that perform relatively well in the Monte Carlo study also work well in forecasting pollution. Given the lack of forecasting models available to the environmental authority in the city, this tool can be used to inform in advance contingency plans that reduce the adverse impacts of air pollution on the population, as well as for other general policy purposes.

Overall this thesis attempts to answer policy-oriented questions related to air pollution. The implications are expected to be useful for policymakers.

References

Brunekreef, B., Holgate, S. T. (2002). Air pollution and health. Lancet 360, 1233–1242.

Kassomenos, P., Papaloukas, C. Petrakis, M., Karakitsios, S. (2008). Assessment and prediction of short term hospital admissions: the case of Athens, Greece. Atmospheric Environment 42, 7078–7086.

Lippmann, M. (2003). Air pollution and health – Studies in the Americas and Europe. In McGranahan, G., Murray, F. (Eds.), Air pollution and health in rapidly developing countries, Stockholm Environment Institute, 35–48.

Sterner, T., Coria, J. (2012). Policy instruments for environmental and natural resource management. Second Edition. New York, NY: RFF Press, Routledge.

WHO. (2011). Tackling the global clean air challenge. September 26. http://www.who.int/mediacentre/news/releases/2011/air_pollution_20110926/en/index.html (accessed September 5, 2012).

Chapter I

EFFECTS OF DRIVING RESTRICTIONS ON AIR QUALITY AND CAR USE IN BOGOTA*

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Abstract

Rationing car use at certain times of the day based on license plate numbers has become a popular policy to address traffic congestion and air pollution in several cities around the world. This paper analyzes the effects of moderate and drastic driving restrictions of the program *Pico y Placa* on air pollution and car use in Bogota. Because the program was implemented in phases, it was possible not only to study the impact of the program, but also to distinguish between the short- and long-run effects for each phase of restriction. Using hourly carbon monoxide data, monthly information on gasoline consumption, and vehicle registration and vehicle sales data, this paper shows differentiated effects of *Pico y Placa* on air quality and car use in the short- and long-run and between phases of the program. Although there was an initial improvement in air quality in both phases of the program, carbon monoxide concentrations, vehicle ownership, and total driving actually increased when drastic restrictions were implemented. Gasoline taxes, on the other hand, have tended to reduce gasoline usage in Bogota, suggesting that a price-based mechanism would be more effective in reducing driving.

JEL Classification: D62, R41, Q53, C54

Keywords: driving restrictions, air pollution, vehicle sales, policy evaluation.

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1 Introduction

This paper analyzes the effects of the *Pico y Placa* (PYP) driving restrictions on air pollution and car use in Bogota, Colombia. Because the restrictions were implemented in increasingly stringent phases, it was possible not only to evaluate the effectiveness of the program but also to distinguish between the short- and long-run effects for each phase. This paper is unique in analyzing how drivers respond to a phased-in program. Using data on carbon monoxide (CO), gasoline consumption, and vehicle sales and registrations, this study shows that CO concentrations and vehicle use did not decrease in the long run, despite an initial improvement in air quality for some periods of the day in both phases of the PYP program. In fact, there is evidence of increased CO concentrations during the implementation of drastic restrictions. There is also evidence that PYP increased vehicle ownership and driving, more so in response to drastic restrictions, as drivers adapted to the program. In contrast, gasoline consumption in Bogota has responded negatively to increasing gasoline taxes, suggesting that market mechanisms may be more effective than driving restrictions.

Traffic congestion and air pollution are major problems in heavily populated cities (Walsh, 2003; Han and Naeher, 2006). For instance, around 1.34 million premature deaths in the world are attributable to outdoor air pollution generated predominantly by motor transport (WHO, 2011). Moreover, approximately 1.44 million hours and 2.17 million gallons of fuel per year are wasted by drivers sitting in congested traffic on the busiest U.S. highway (Harbor Freeway/CA-110 NB in Los Angeles), implying an annual cost of US$ 95 million (Texas A&M Transportation Institute, 2011).

One way to mitigate these effects cost-effectively is through the use of market-based incentives such as fuel taxes and road pricing (see Sterner and Coria, 2012). An alternative approach is rationing car use at certain times of the day based on the last digit of the vehicle's license plate number. This approach has become very popular in several cities around the world. Examples of these programs are *Hoy No Circula* (HNC) in Mexico City and PYP in Bogota.[1] Policy makers generally justify driving restrictions in terms of welfare gains through the reduction of multiple externalities. Besides congestion and air pollution reduction, these programs might diminish crash risk and road and parking facility costs.

[1] Some other cities subject to driving restrictions are Santiago, Quito, Sao Paulo, Beijing, Athens, San Jose, and others in Asia and Latin America.

Another common argument is that such policies are easy to monitor and are considered fair because they target the rich and the poor equally. Moreover, they could potentially induce the use of public transportation, while decreasing the use of private transport (Eskeland and Feyzioglu, 1997a).

Assessments of the effects of driving restrictions on traffic congestion and air pollution in practice have yielded conflicting conclusions regarding their effectiveness. For example, Eskeland and Feyzioglu (1997) studied the effects of the HNC program on gasoline consumption and vehicle ownership and found that total driving increased because of additional car purchases under the regulation. Another assessment of the same program showed no evidence of improvements in air quality; restrictions rather led to an increase in vehicle registrations composed mainly of higher-polluting used vehicles (Davis, 2008). A recent theoretical and empirical study using a car-use and car-ownership model indicated that, although the HNC program has been ineffective in the long run, it reduced air pollution at peak hours by 7% during the first month of implementation (Gallego et al., 2011). That study also found that households responded by buying more vehicles, resulting in a rapid increase in the stock of vehicles in the first 11 months of implementation. In contrast to these studies, Viard and Fu's (2011) assessment of the driving restrictions implemented in preparation for the 2008 Beijing Olympics found that the program reached its initial goals: the every-other-day restrictions and one-day-a-week restrictions reduced total pollution by 19% and 8%, respectively.[2] These authors argue that high compliance and high vehicle ownership cost may explain the effectiveness of the program in Beijing.

The conflicting evidence emphasizes the importance of comparing the short- and long-run effects of a program so that policies can be revised before they are implemented in other cities. An additional important consideration omitted in previous studies is that driving restrictions may be introduced in stages, becoming stricter or increasing their coverage over time, which may potentially alter the extent of households' responsiveness to the program. The implementation in phases may be in response to political resistance and public opposition, which impede the immediate introduction of drastic restrictions.

[2] A recent evaluation of the driving restriction program in Quito provides similar conclusions, indicating that the ambient concentrations of carbon monoxide have decreased between 9 and 11% during peak traffic hours since the introduction of the program in May 2010 (Carrillo et al., 2013). The study suggests that the uncertainty about the program's permanency may have induced households not to buy a second car.

While one would expect that a stricter ban yields higher benefits in terms of a reduced number of vehicles circulating on the streets at a given time or on a given day, there may be several reasons why households may exhibit different behavior over time and that may affect the magnitude of their response. On the one hand, households may have more adaptation possibilities in the long run than in the short run; people might change habits and travel behavior over time, and become more familiar with new routes and travel times (see Björjesson et al., 2012). On the other hand, households might buy additional vehicles to circumvent the program, or the freed-up road space might stimulate new traffic (travelers with high value of time or travelers making more trips during the restricted hours). Indeed, the size of the short- and long-run effects may differ between phases of implementation of the program such that households with prior experience of driving restrictions might have a faster and larger adaptation response to changes in the policy than those who have not had such experience. Such responses based on experience might limit the effectiveness of the drastic phase of a phased-in program.

The present paper analyzes the effects of moderate and drastic driving restrictions on air pollution and car use in Bogota. Between August 1998 and February 2009, the program *Pico y Placa* introduced driving restrictions during peak hours (moderate) in Bogota to reduce congestion caused by privately-owned light vehicles. Then, the driving restriction became stricter, extending the program to 14 hours per day (drastic) to continue reducing congestion and to cut as well traffic emissions. The steps of implementation of PYP enable not only a study of the impact of the program, but also differentiation between the short- and long-run effects for each phase of restriction.

This study fills a gap in studying the effects of switching from moderate to drastic restrictions programs.[3] Although the effect of multiple changes in driving restrictions on pollution has been studied by Viard and Fu (2011), their analysis explores a complex mixture of changes within a very short period of time.[4] This makes it difficult to clearly separate levels

[3]Although a partial evaluation was conducted in San Jose for a driving restriction program that was implemented in similar phases (see Osakwe, 2010), that study analyzes only the effects of the program on national transport fuel sales. The results of that analysis show that, although San Jose's driving restrictions were successful in reducing gasoline sales from July 2008 to April 2009, i.e. the period when the policy was more stringent, low-income drivers were likely disproportionately affected by the program. That study applies an OLS approach instead of a regression discontinuity design, which may affect the reliability of the estimates.

[4]The driving restrictions in Beijing were initially introduced on July 20, 2008, but were lifted on September 20. The program was subsequently readjusted and reintroduced on October 11 and again readjusted on April

of stringency and estimate long-run effects. Unlike that study, the present paper also evaluates the effect of each phase of restriction on gasoline consumption and the impact of drastic restrictions on vehicle registrations and sales. The most closely related work in evaluating short- and long-run effects is Gallego et al. (2011). Although that study estimates the effects of the program on gasoline consumption and vehicle sales, this paper focuses on the effects of two different levels of stringency, develops a more detail analysis of the meteorological variables that may influence air quality, and models pollutant concentrations, gasoline consumption and vehicle sales, accounting for their dynamics using time series approach.

Carbon monoxide is used as a proxy for car use and air quality, because of its strong correlation with traffic activity. Using hourly CO data in an Autoregressive Distributive Lag (ARDL) model, the short- and long-run effects of moderate and drastic restrictions of PYP are estimated at different times of the day and week.[5] Using a similar ARDL model, the effects of PYP on gasoline consumption and vehicle registrations are examined. Then, the effects of the program on these variables are compared between phases.

The results indicate that, although there was an initial improvement in air quality for some periods of the day (evening peak or off-peak hours) in both phases of the PYP program, CO concentrations and vehicle use did not decrease in the long run. In fact, there is evidence of *increased* CO concentrations during the implementation of drastic restrictions. There is also evidence that PYP has increased the stock of private vehicles – an outcome that is consistent with no decrease in gasoline consumption. Overall, the results show an increase in vehicle ownership and total driving, suggesting that households were more responsive to drastic than moderate restrictions, i.e. drastic restrictions generated stronger counterproductive consequences.

This document is organized as follows. Section 2 presents a description of the PYP program. Section 3 describes the data, the econometric approach, and the effects of the initial

11, 2009. For details on the restrictions, see Viard and Fu (2011).

[5]During the development of this research, a study of driving restrictions in Bogota carried out by Lin et al. (2012) became available. Their analysis focuses on the effect of a mixture of driving restrictions for private and public transport on the concentrations of several pollutants. The authors do not evaluate the effects on car sales and registrations or gasoline consumption. Their method takes averages across all the monitoring stations to obtain a continuous data series of pollutants, including those stations with few valid observations. This approach may introduce biases in the average pollutant concentrations, which may explain why those authors find a reduction in CO after a minor adjustment of the restrictions in 2004. In particular, in 2002 and 2003, the monitoring network experienced maintenance problems, yielding the lowest number of valid observations in the entire history of the network, making it impossible to construct reliable continuous series for several pollutants.

and extended phases of PYP on CO. Section 4 presents the methodology and results regarding the effects of PYP on gasoline consumption, vehicle registrations and sales, and other variables. Section 5 brings all the previous analyses together in order to discuss the effects of switching from moderate to drastic restrictions of PYP. Finally, Section 6 concludes the paper.

2 The initial and extended phases of PYP

Bogota is the most important economic center in Colombia. With a population of 7.4 million, the pressure on roads and highways is very high. The urban roads are used predominantly by private vehicles – approximately 1,400,000 in number, which is roughly 75% of the total vehicle fleet in Bogota (Secretaria de Movilidad, 2009). Due to the growth of the vehicle fleet between 2000 and 2011, congestion has become a critical issue, giving rise to a progressive reduction in travel speeds (Secretaria de Movilidad, 2012). Moreover, private vehicles cause a deterioration of the air quality through the emission of 404,000 tons of CO, 18,200 tons of nitrogen oxides (NO_x), and 46,500 tons of hydrocarbons per year (SDA and Uniandes, 2009).[6]

To overcome traffic congestion, Bogota implemented PYP on August 18, 1998 (hereafter referred to as the initial phase).[7] The program banned the use of privately-owned light vehicles (automobiles, station wagons, and sport utility vehicles) in the urban area Monday through Friday during rush hours (07:00-09:00 and 17:30-19:30) according to a schedule based on the last digit of the license plate number. The aim was to reduce congestion by 160,000 vehicles during those hours of the day. The program applied to four different last digits of the license plates per day and annually assigned different days of the week for each group of digits.[8] Weekends and holidays were excluded from the regulation. Minor adjustments were made to these initial-phase rules for private vehicles in the subsequent years.[9]

[6]According to the guidelines for the prevention and control policy of air pollution in Colombia, the total terrestrial transportation sector contributes 86% of the air pollution in Bogota (CO, CO_2, and NO_x). See Conpes 3344 of 2005. Air pollution is also the environmental problem of major concern for Colombians (Lemoine, 2004).

[7]See Decree 626 of 1998.

[8]For instance, vehicles with a last license plate digit of 1, 2, 3, or 4 could not be operated on Mondays; 5, 6, 7, or 8 on Tuesdays; 9, 0, 1, or 2 on Wednesdays; 3, 4, 5, or 6 on Thursdays; and 7, 8, 9, or 0 on Fridays.

[9]Despite this program and other complementary actions, the district administration also decided to introduce the driving restriction program in 2001 for collective and individual public transport. The policy was implemented as a strategy to control the oversupply of public transport vehicles at a low level of occupancy,

In 2002, the restriction period was extended by 30 minutes in the morning (06:30-9:00) and shifted by 30 minutes in the evening (17:00-19:00). The restrictions were further increased in 2004 through the addition of 30 minutes in the morning (06:00-9:00) and one hour in the evening (16:00-19:00).

On February 6, 2009, the length of the restricted time per day was substantially modified (the period subsequent to this date (hereafter referred to as the extended phase), while other characteristics of the program remained unchanged.[10] In the extended phase, the use of private vehicles on weekdays was restricted from 06:00 to 20:00. This regulatory action aimed to reduce pollutant emissions and the number of accidents as well as congestion. Given that Bogota had already started a district plan of road building, the program was also justified as a mechanism to counteract future congestion. Because of the minor adjustments of PYP from 2002 to 2004, the analysis in the next section focuses only on the initial and extended phases of the program.

The PYP program has been enforced through the imposition of fines and vehicle immobilization.[11] If a driver is caught violating the restriction, then continues his trip and is caught in violation again, he must pay a new fine. According to reports by the Transport Agency of Bogota, drivers have generally complied with the restrictions. Since the initial phase of the program, the number of fines has decreased annually on average by 40%, and hence this infraction has moved from the second to fifth place in the list of the most frequent traffic fines imposed by the Metropolitan Transit Police (see Hernandez, 2003 and Secretaria de Movilidad, 2010b).

3 Effects of the initial and extended phases of PYP on carbon monoxide

Studies of the effects of PYP performed by the Environmental and Transport Agencies of Bogota have shown air quality improvements for some pollutants during certain periods of

which affected traffic congestion and environmental quality (Alcaldia Mayor de Bogota, 2001). Additional and similar regulations were applied in 2006 for public transport and load vehicles.

[10] Four different last digits of the license plates were also assigned per day, as in the initial phase. See Decree 033 of 2009.

[11] The Transport Agency of Bogota has also implemented educational periods without penalties, one week before the new rotation of license plates applies, in order to encourage observance of the norm and remind drivers the new restricted digits of the license plates. The current fine is approximately US$150.

the year, as well as traffic reductions for some areas in the city after the implementation of PYP (Secretaria de Movilidad, 2010a and 2010b). However, these reports do not show a clear trend over time. They are based on simple descriptive statistics that do not take into account the effects of factors such as meteorology, seasonality, and other unobserved time-varying components that may drive variations in traffic and pollution. Additional doubts regarding the effectiveness of the PYP program have also been raised in the media (El Tiempo, 2009). To this end, the present study provides more comprehensive insights into the effectiveness of the PYP program.

This next section explains the selection of carbon monoxide (CO) as the outcome variable in the policy evaluation, as well as the scientific fundamentals justifying the selection of meteorological variables and monitoring stations. It also describes the data and presents the econometric strategy and effects of the PYP on CO concentrations.

3.1 Selection of car use and pollution indicator

The identification and evaluation of policy effects requires selection of appropriate outcome variables. Traffic counts are possible variables for evaluating driving-restriction programs such as PYP since they serve as a direct measure of vehicle use at certain times of the day. However, such data is usually scarce. For example, traffic counts in Bogota are only available for the extended phase of PYP, cover only some days of the year and are only recorded at certain city locations. It is therefore not possible to construct a continuous time series before and after the implementation of PYP based on traffic counts.

Thus, instead of traffic counts, CO concentrations are used as a proxy for car use. Carbon monoxide has several advantages compared with other possible proxies, which makes it a useful variable in evaluating car use (Gallego et al., 2011). First, CO is mainly emitted by traffic (85-98% of total CO emissions in Bogota),[12] including mainly gasoline-powered vehicles (Derwent et al., 1995). In Bogota, almost the entire light vehicle fleet is gasoline operated (99% in 1998 and 96% in 2009[13]); privately-owned vehicles account for 90% of the CO emissions of this fleet (SDA and Uniandes, 2009).

[12] Contribution of total traffic to CO emissions in 2000 and 2007, respectively. See DAMA (2000) and Zarate et al. (2007).

[13] Own calculations using data from vehicle registration data provided by the Transport Agency of Bogota.

Second, CO is an inert tracer that reaches the highest levels during rush hours, when traffic demand is also the greatest (see Body et al., 2005; Comrie and Diem, 1999).[14] Third, carbon monoxide is also less chemically reactive in the atmosphere at short timescales (minutes, hours, or days) than other pollutants such as particulate matter (PM_{10}) and NO_x (see Body et al., 2005; Comrie and Diem, 1999). For example, PM_{10} and NO_x, unlike CO, are exposed to a chain of complex atmospheric reactions and are also emitted from more varied sources. Concentration levels or the rate at which these pollutants are accumulated in the atmosphere may be a result of multiple factors such as road dust or chemical and photochemical reactions among pollutants. These considerations prevent the use of other pollutant indicators to identify effects of policies like PYP. Therefore, the rapid response of CO levels to traffic emissions enables the monitoring of changes in traffic volume. Fourth, continuous series of hourly data are available for several city locations. Fifth, CO can be employed as a direct indicator to measure the effects of PYP on air quality. Although CO concentration levels tend to be, on average, below the air quality standard in the last five years (35 ppm per hour and 8.8 ppm for each 8 hours), compliance with the CO standard does not imply that other traffic pollutants are not generated. The vehicular activity identified through CO concentrations may be associated with emissions of other pollutants from the light vehicle fleet such as volatile organic compounds (VOC) and carbon dioxide (CO_2). The former is a precursor of ozone and the latter a greenhouse gas.

One factor that complicates the analysis of CO concentrations is meteorology. Meteorological variables such as wind speed, temperature, relative humidity, temperature inversion, and rainfall can alter CO concentrations (Aron and Aron, 1978; Huo et al., 2010; Maffeis, 1999). Increased wind speeds generally reduce CO levels. Stagnant air and low wind speeds, which are characteristics of high pressure systems, promote temperature inversions, impeding the dispersion and dilution of pollutants. In contrast, relative humidity is positively linked to CO concentrations – increases in relative humidity imply lower CO levels. Also, rainfall is expected to wash out gases, while wind direction may play an important role in transporting pollution between areas of the city. In general, Bogota's weather conditions suggest insufficient air mixing and dispersion, features that tend to be exacerbated during temperature inversion episodes, which hence alter the pollutant residence time in the ambi-

[14] Most of the monitoring equipment in Bogota is located 10-300m from the roads, and the manifolds take samples 3-25m above the ground level.

ent air. Nonlinearities in these factors are also exhibited in the Bogota's weather-CO profiles.

Of the meteorological variables described here, it is worth highlighting the relevance of temperature inversions and rainfall for policy evaluation and pollution modeling. Previous studies analyzing driving restrictions have not considered these factors.[15] Temperature inversion in Bogota is a particularly important factor influencing CO concentrations between 18:00 and 06:00, which overlaps with the end of the evening rush hour. Temperature inversions can also influence CO concentrations at the beginning of the morning rush hour. Under temperature inversion episodes, motor vehicle emissions are trapped near the ground level, thus potentially introducing a bias in the estimations of CO concentrations as a proxy for car use during those hours. Rainfall can either affect CO levels via a washing effect or function as a proxy for the willingness to substitute transportation mode at certain times of the day – individuals owning a car may opt to use it rather than public transport when discouraged by rain. The latter effect implies that individuals may drive more and increase traffic congestion during rainfall. An additional effect of rainfall is the increase in traffic congestion during rainy periods due to local flooding caused by the inadequate drainage system. These factors are important for evaluations of PYP in Bogota since maximum rainfall usually overlaps with the evening peak, when PYP applies. These possible confounding factors may be ruled out in the estimations by controlling for hourly precipitation.

3.2 Data

This study uses historical records of the monitored CO levels and meteorological variables to assess the effect of PYP on CO during two 2-year symmetrical time intervals centered around the start of the initial and extended phase, respectively (August 18, 1997 to August 17, 1999 and February 7, 2008 to February 5, 2010) – see Table 1 for descriptive statistics of these two time periods. Allowing the analysis to span two years for each phase of the program ensures accounting for seasonal variation and lessens the effect of possible confounding factors on CO concentrations (see Davis, 2008). Likewise, this time window is conservative enough to identify short- and long-run effects of PYP on CO since most of the adaptation responses to the program occur within the first year of implementation (see Gallego et al., 2011). The first

[15] The exception is Salas (2010), where precipitation is included in the model specification as a robustness check of the general estimations.

symmetrical time window is used to evaluate the effect of moderate restrictions (before PYP versus initial phase) and the second time interval to assess the effect of drastic restrictions (initial phase versus extended phase).

Measured data of hourly CO concentrations and meteorological variables were taken from the Air Quality Monitoring Network of Bogota (RMCAB) of the Environmental Agency of the District. The RMCAB is a system of ongoing and automatic monitoring of air quality dating back to August 1997. It transmits air quality via land phone and cellphone. At present, the RMCAB consists of 15 automatic point stations and a mobile station. Of the 15 stations, 13 measure pollutant concentrations while the remaining two stations record weather conditions. The monitored pollutants are mainly CO, NOx, nitrogen monoxide (NO), nitrogen dioxide (NO_2), PM_{10}, sulfur dioxide (SO_2), and ozone (O_3).[16] The meteorological variables monitored are wind speed, wind direction, relative humidity, superficial temperature, temperature at three heights, and rainfall. General reports about pollution trends and meteorological information in different places of the city are publicly available by the RMCAB through its website (http://www.ambientebogota.gov.co).

Monitoring stations with a total CO reporting of more than 75% of all possible hourly observations for the period of interest were selected.[17] In total, four monitoring stations were chosen during the implementation of the initial and the extended phases.[18] The percentage of hourly CO reporting of the selected stations ranges between 78% and 91% for the initial period and between 80% and 95% for the second period. Meteorological reporting of hourly data varies from 86% to 97% and from 92% to 100% for 1997-1999 and 2008-2009, respectively. This coverage is considered satisfactory in the field of environmental science because it is

[16]The equipment is in compliance with the regulations of the US-EPA referred to in the Quality Assurance Manual for measurement or monitoring systems for air pollution (Secretaria de Ambiente, 2010a). In the case of CO, the RMCAB equipment consists of infrared gas filter correlation analyzers (Secretaria de Ambiente, 2003).

[17]The selection of monitoring stations with valid reporting is required since some changes have been made in the RMCAB since its creation; new monitoring stations have been installed and others have been eliminated. Those changes cause some monitoring points to have low data representation and make it impossible to use the same monitoring stations for the initial and extended phases and for constructing a continuous data series for the period 1997-2010.

[18]These stations are Sagrado Corazon-MMA (St2), Carvajal-Sony (St3), Olaya (St4b), and Cazuca (St7) for the initial phase of PYP and the stations Parque Simon Bolivar-IDRD (St5), Las Ferias-Carrefour (St6), Puente Aranda (St13), and Fontibon (St14) for the extended phase. Out of all stations with an acceptable threshold (75%) of valid CO observations for the first time window, only one station was excluded (Suba Corpas - St11). The reason for its exclusion was the installation of a bus stop near the station where a significant fraction of vehicles operated with gasoline at the time when PYP was implemented. Although CO concentrations at this station are very low, increased CO levels were remarkably observable.

commonly accepted to use series with at least 75% valid observations (EPA, 2010).

To allow meaningful interpretations of wind circulation, wind direction was converted from azimuth bearings to a set of dummy variables corresponding to the 8-point compass international convention. Temperature at different heights was used to define temperature inversion. A temperature inversion episode occurs when the temperature gradient between two different heights is positive, i.e. temperature increases with altitude the opposite of normal conditions. Thus, temperature inversion is an indicator variable that takes the value of one under those episodes, and zero otherwise. The height interval 20m-2m was chosen to compute the temperature gradient because temperature inversions are severe near ground level (Comrie and Diem, 1999).

3.3 Econometric approach

Unlike other studies that use hourly data to analyze the effect of transport policies on pollutants, this study uses an approach that treats CO concentrations dynamically by taking into account the persistence of CO in the atmosphere (Dennis et al., 1996; EPA, 2010) through the addition of lags in the specification (Aron and Aron, 1978). These lags also account for the inertia of the time series when congestion is building up to a peak or falling off from the peak, and control for any other transitory shock on CO other than PYP, for instance, transitory road construction. Thus, the effect of PYP on CO concentrations is analyzed using the following general Autoregressive Distributed Lag (ARDL) model for $t = 1, 2, ..., T$:

$$y_t = \alpha + \beta PYP_t + \sum_{i=1}^{m} \gamma_i t_t^i + \sum_{j=1}^{p} \delta_j y_{t-j} + \sum_{s}^{S} \sum_{k}^{K} \sum_{r=0}^{R} \omega_{skr} x_{s,t-r}^k + \mathbf{Z}_t' \theta + \mathbf{D}_t' \eta + \varepsilon_t \ , \quad (1)$$

where y_t is the CO concentration in logs at period t, PYP is an indicator variable equal to one from August 1998 to January 2009 and zero otherwise, i.e., the initial phase of PYP, or equal to one after February 2009 for the extended phase and zero otherwise. $x_{s,t-r}^k$ represents any of the s meteorological variables described in Section 3.1 in a polynomial form of order k at time t and their corresponding r distributive lags, except temperature inversion and wind direction, which are dummies and are included in $\mathbf{Z}_t$. $\mathbf{D}_t$ denotes seasonal dummies for month of the year, day of the week, hour of the day, and interactions between weekends and hour of the day. p is the autoregression order, m is the polynomial order of the time trend, and ε_t is the error term at time t. CO concentrations and meteorological variables are included in the model at the mean city level.

There are two major advantages of using ARDL for evaluating the effects of PYP. First, ARDL captures the short- and long-run dynamics of the time series. Taking into consideration the appropriate number of lags, the ARDL analysis therefore entails a general-to-specific modeling framework. Second, ARDL enables the analysis of structural breaks of the time series, for instance those induced by policy effects. Hence, the coefficient β in equation (1) reflects the short-run dynamics of the average effect of PYP on the mean CO concentrations, whereas $\beta/(1-\sum_{j=1}^{p}\delta_j)$ accounts for the long-run effects.[19]

The estimation of the ARDL model is conducted within the framework of Regression Discontinuity Design (RDD), allowing for a polynomial time trend to control for unobserved time-varying factors that may influence the CO levels and make the estimates of the PYP effect less informative. The underlying assumption of RDD is that unobserved factors influencing CO levels change smoothly at the policy date when PYP was implemented (see Hahn, Todd, and Van der Klaauw, 2001). The time trend may capture unobservable time-varying variables such as changes in vehicle age, vehicle technology to control emissions (three-way catalysts), size of the engine, adjustments in the vehicle fleet composition, and other possible economic trends. Therefore, the information in the time series before the policy serves as a counterfactual of the observations after the PYP program.[20]

Equation (1) is estimated for the two symmetrical time windows considered in this study (1997-1999 and 2008-2010). First, the impact of PYP on CO is analyzed for the series containing hours of the day (hereinafter all hours) when traffic is most active (05:00-21:00). Although, in the initial phase of PYP, restrictions applied only during peak hours, this overall model provides insights regarding whether the program has an effect in the average CO concentrations during the day. Second, to evaluate the effects of the program and identify intertemporal substitution among restricted and non-restricted periods, equation (1) is esti-

[19] It assumes stability; i.e., the process is stationary in the autoregressive components.

[20] An alternative method to evaluate the effectiveness of the program is difference-in-difference approach. In the context of driving restrictions, it would require the use of information from another location (or city) with a similar development trend and vehicle fleet composition, where the program was not implemented, as a counterfactual. As Davis (2008) points out, these characteristics are unlikely to be met by another city, which prevents the use of that approach. PYP was implemented at the same time in the whole urban area of the city. Even though during the initial phase this was restricted to only peak hours, driving during other untreated times might have been affected by the program.

mated for time-subsamples at different periods: morning peak, evening peak, off-peak, and weekends.[21]

Differentiated short-run effects of the program over time are estimated using the ARDL model; i.e., it is of interest to evaluate how the effects of the program evolve progressively from the beginning of the policy date. The implemented approach consists of estimating the model recursively by fixing the sample for the period before the PYP program and then extending it forward by two months at a time. This allows the β coefficient to change. Hence, if the PYP program were effective at time τ after the policy date, one would expect the estimate of $\beta_\tau/(1-\sum_{j=1}^{p}\delta_{j,\tau})$ to be negative and statistically significant. Note that Pesaran and Smith's (2012) approach is similar since the counterfactual is computed recursively from the policy date onward. The estimations are carried out for all hours of the day, morning peak, evening peak, off-peak, and weekends.

Before estimating equation (1), the temporal variation of each series is analyzed. Standard time series techniques assume that the process underlying the observations is weakly stationary. The Augmented Dickey Fuller (ADF) test and the Phillips-Perron (PP) test are used to examine whether the series are stationary. CO and meteorological variables were found to be stationary across all the tests. Hence, the ARDL model was estimated with the variables in levels. The maximum polynomial orders m and k, the lag orders p, and the number of lags for the weather variables were selected based on the Bayesian Information Criterion (BIC). In the case of the meteorological variables, the selection criteria were also supplemented with visual inspection of scatter plots ($k = 2$ is used in all models)[22]. Moreover, $m = 4$ was implemented for the model of all hours of the day, whereas $m = 3$ was applied for the morning peak, evening peak, off-peak, and weekend subsamples.[23] Finally,

[21] Restricting the sample to time intervals was initially suggested by Davis (2008) (see also Gallego et al., 2011). Indicator variables representing the interactions between weekends and hour of the day are excluded in the subsample models. For the initial phase, the sample is restricted to time slots such as 07:00-09:00 (morning peak), 17:00-20:00 (evening peak), and 10:00-17:00 (off-peak) during working days. Similarly, for the extended phase analysis, the sample is restricted to time intervals such as 06:00-09:00 (morning peak), 16:00-19:00 (evening peak), and 10:00-16:00 (off-peak) during working days. For weekends, the sample includes the aggregated time slot 07:00-09:00 and 17:00-20:00 for the initial phase, and the aggregated time intervals 06:00-09:00 and 16:00-19:00 for the extended phase.

[22] This rule was used when the value of BIC monotonically decreases for large orders of the polynomial. In this case, the scatter plots provide information about the degree of nonlinearity of the relationship between meteorological variables and CO, and are useful to construct a parsimonious specification.

[23] The value of BIC monotonically decreases in some time-subsamples for the extended phase analysis, indicating a very large number of m. Hence, the same orders of m employed for the initial phase of PYP are used

$p = 1$ was employed during the initial phase and $p = 2$ for the extended phase.

3.4 Results

The effects of PYP on CO concentrations for the all hours model using equation (1) are summarized in Table 2, where columns 1 and 2 contain the estimates for the initial and the extended phase, respectively. The results show no evidence of a decrease in the average CO concentrations due to PYP. For instance, the initial phase β coefficient is negative (-0.006) and statistically insignificant at conventional levels, whereas the extended phase β estimate is positive (0.0274) and statistically different from zero at the 10% significance level. As regards the Autoregressive (AR) components in the ARDL models, the coefficients of the first lag of CO concentrations for the periods 1997-1999 and 2008-2010 are remarkably high (0.68 and 0.80) and statistically significant. This suggests a high persistence of pollution and supports the notion of accounting for CO dynamics when modeling CO concentrations.

The results in Table 2 also show that the relationship between the meteorological variables and pollution is adequately taken into account in the model, as most of the meteorological variables are statistically significant and all their aggregated coefficients have the expected signs. CO exhibits an inverted-U relationship with rainfall for the period 1997-1999. As previously explained, the reason for this is that, during the rainy episodes, traffic congestion and hence CO concentrations rise, but then increased precipitation cleanses the air, reducing CO levels. For 2008-2010, CO concentrations also increase when rainfall increases, yet the washing effect does not seem to be present. This might be because rainfall was, on average, slightly lower than in the initial phase period. An analysis of the temperature-inversion coefficients also indicates that a positive temperature gradient results in high CO levels. Thus, during temperature inversion episodes, CO concentrations were 17% and 12% higher for the initial and extended phases, respectively, than in normal conditions.[24]

The estimates of the effects of PYP on CO concentrations for different time-interval subsamples (morning peak, evening peak, off-peak, and weekend) during the initial and extended phases of the PYP program are presented in Tables 3 and 4, respectively. Panel A corresponds to estimates using equation (1), while Panel B refers to the estimates of the re-

here.

[24] This effect takes into account the adjustment due to the AR components.

cursive short-run effects of PYP. In Tables 3 and 4, the last row shows the computed long-run effects of the program. In all models, the R^2 was found to be higher than 0.70, which represents a good fit of data. Likewise, all AR components were very high, confirming the high degree of persistence of CO across time-interval subsamples.

The results indicate that the PYP program did not lead to any long-run reductions in CO concentrations during the initial phase for any of the subsamples (Panel A, Table 3). Although some of the PYP coefficients are negative, they are statistically insignificant. Panel B shows that PYP generated a reduction of 21% and 14% of CO concentrations during the evening peak hours and all hours time interval, respectively, just two months after the policy was implemented. The decline during the evening rush hours was in line with the purposes of the restriction. The program restricted 40% of the vehicles, but due to possible reallocation of trips, the Transport Agency estimated the net reduction in vehicular flows to 31% (Acevedo, 1998). However, the coefficient for the morning peak hours during these two months was not statistically significant. The short-run PYP estimates for four, six, eight, and ten months after the policy implementation indicate no reduction in CO levels for all periods of the day. In fact, there is even evidence of increased CO concentrations during the off-peak hours eight months after the implementation of the program. Figure 1 shows the evolution of the short-run PYP coefficients over time with their corresponding confidence intervals at 95%. As can be seen, overall these results suggest that the effect of the initial phase of PYP tended to vanish in the long run.

Considering that the policy did indeed have an effect on the evening peak period, one can ask why the CO levels were not correspondingly reduced during the morning peak. One possible explanation could be that, during morning hours, some drivers might have changed routes and travel times in ways that reduced the effect of the program. For example, some drivers who usually started their trips at the end of the morning rush to avoid congestion before the policy date might have begun traveling during the morning peak as they expected lower congestion in these hours after the program was put in place. Additionally, it seems to have been very common to pick up friends along the way during the restricted times. In the latter case, vehicles might have been driven longer distances, counteracting the effect of the program. These coordination activities tend to be more difficult in the evening than in the morning because of more dispersed departure times after work.

Nor were there any PYP-induced long-run reductions in CO concentrations in the extended phase across the subsamples. In fact, CO levels increased by 26% and 36% in the long run during the morning and evening rush hours, respectively, after the launching of the program. PYP also led to higher concentrations in the all hours model (10%) compared with the period before the policy implementation. Although the program was intended to reduce vehicular traffic during off-peak hours, the estimates indicate that this objective was not achieved in the long run.

All the recursive short-run coefficients during the morning peak are positive, and several of them are statistically significant. Interestingly, two months after the program was implemented, the effect of PYP on CO concentrations during the off-peak time interval was negative and statistically significant at the 10% level. The magnitude of this reduction (15%) was much lower than the expected decline in traffic (40%). During these two months, there was no observed effect on CO levels at other times of the day or during weekends. Progressively in the next several months, the effect of PYP in the off-peak hours increased significantly, reaching the highest reduction (36%) eight months after the policy introduction, which was much closer to the magnitude of the effect planned by the policymakers. However, this effect decreases rapidly and disappears in the long run. An opposite trend is found for the PYP short-run effects in the morning and evening rush hours (Figure 2 depicts the confidence intervals at 95% of the short-run coefficients for the extended phase). Initially, PYP did not affect CO concentrations during the morning and evening peak hours, but later it systematically led to increased CO levels. Possible explanations for the unexpected consequences of the extended phase during peak hours is that PYP may have induced new traffic due to the freed-up road space, may have led to an increase in the number of trips, or households simply found alternatives to substitute for the restricted trips, such as buying a second car with a different license plate number.

3.5 Robustness checks

To analyze the stability of the results, additional robustness checks were performed. Each ARDL model in the following analyses includes the same autoregressive and polynomial orders, meteorological covariates, and seasonal controls employed to estimate equation (1) in section 3.4. The long-run coefficients of the initial and extended PYP phases for the various time-interval periods are shown in Tables 4 and 5, respectively.

A potential concern when estimating equation (1) is that the model does not explicitly include explanatory variables that may be strongly related to car use. To address this issue, two specifications that account for gasoline price and the real exchange rate were included. Furthermore, two additional specifications account for the possible confounding effects related to industrial emissions; one pertains to the industrial production index, the other specification adds SO_2 as a covariate. [25] Also, estimations control for a few environmental regulations. A regulation in 1998 established emission standards for new vehicles. Thus, a regression includes an indicator variable equal to one from January 1998 and zero otherwise. A similar regulation of emissions standards for new vehicles has applied since 2009; hence, a specification adds an indicator variable equal to one from January 2009 and zero otherwise.[26] Because roadwork had been intensified during the extended phase, a regression adding a variable with the annual total roadwork investments is included. The sensitiveness of the estimates to the time trend polynomial order is also evaluated. Two different polynomial orders are tested: 4th and 5th. Due to the concern of collinearity among meteorological variables, which may make the estimates less precise, one specification is estimated replacing the meteorological covariates with their corresponding principal components. The selected components are those that yield eigenvalues larger than one.

The results for the initial phase of PYP across all specifications remain unchanged. In the case of the extended phase, most of the models show that the coefficients are stable to the inclusion of controls during the morning and evening peak subsamples. Out of nine specifications during the off-peak, only one indicates a reduction in CO concentrations as a result of the extended phase of PYP, where the coefficient is significant at the 5% level.

In conclusion, the results found in the preceding section and the robustness checks indicate that the PYP program has not led to long-term improvements in air quality but has in fact increased car use. Nevertheless, there is evidence that the program may discourage driving in short-time scales. Drivers seem to respond faster during peak than off-peak hours, while weekend traffic behavior, on average, remains unchanged. The evening peak tends to

[25]Gasoline price, real exchange rate, and industrial production index are in the form of first monthly differences because these variables are nonstationary. Moreover, SO2 is included in lagged form because the use of lags tends to lessen the effect of possible endogeneity.

[26]These technology dummies are not included in the original model due to uncertainties regarding whether these regulations were enforced.

be more elastic to changes in the policy than the morning rush hours. Likewise, households' responses over time vary depending of the PYP program scheme. A drastic program, as in the extended phase of PYP, induced a faster, more prolonged, and unintended response to the policy and seems to have promoted more driving than a less stringent phase. Some households' responses that support the general results are discussed in Sections 4 and 5.

4 Additional evidence

In addition to the analysis of the effects of PYP on CO levels, this study also evaluates the effect of PYP on other outcome variables that are also associated with car use and may play an important role as behavioral indicators of how households responded to the program. The variables considered were gasoline consumption, vehicle registrations, and vehicle sales. The analysis of these variables aims to provide additional clues as to why the PYP program did not result in any CO reductions in the long run. Moreover, because of the availability of information on gasoline consumption and vehicle registration, the effect of the moderate and drastic driving restrictions can be analyzed within the same regression, allowing for comparison of the size of the PYP estimates between the initial and the extended phases. With respect to gasoline consumption, data on gasoline taxes and price elasticity are used to compare the relative effectiveness of driving restrictions and gasoline prices in reducing driving. In each case, the empirical strategy and corresponding findings are presented. A brief discussion is also presented regarding the effect of PYP on other variables.

4.1 The effect of PYP on gasoline consumption

When analyzing gasoline consumption, it is important to distinguish the effect of PYP from the effect of gasoline prices, as this will help policy makers evaluate the relative effectiveness of market and non-market policies. This analysis utilizes data on total monthly consumption of gasoline (regular plus premium) obtained from the Finance Agency of Bogota, gasoline price data from Ecopetrol[27] (the largest hydrocarbon company in Colombia), and GDP and

[27] Monthly prices from 1999 to 2010 were obtained from http://www.ecopetrol.com.co/precios.htm. For some months between 1995 and 1999, written and online reports from Ecopetrol and information in the archive of digital news of El Tiempo (www.eltiempo.com) were accessed in order to fill in the gaps in series. The prices taken from the news corresponded to official prices and matched perfectly with those prices for which information of Ecopetrol was available.

total population figures January 1996 to September 2010, provided by the National Department of Statistics (DANE). The consumption equation is as follows:

$$C_t = \alpha + \pi_1 PYP1_t + \pi_2 PYP2_t + \sum_{i=1}^{m} \gamma_i t_t^i + \lambda_1 P_t + \lambda_2 M_t + \mathbf{D}_t' \eta + \varepsilon_t \ , \quad t = 1, ..., T \qquad (2)$$

where C_t is the total gasoline consumption per capita in logs at time t, PYP1 is an indicator variable equal to one from August 1998 to January 2009 and zero otherwise, and PYP2 is an indicator variable equal to one after February 2009 and zero otherwise. P is the gasoline price and M is the GDP per capita, both in logs at time t. Furthermore, m is the polynomial order of the time trend, $\mathbf{D}_t$ is a group of monthly dummies, and ε_t is the error term at time t. π_1 and π_2 measure the effect of PYP on consumption. This specification is closely related to the work by Eskeland and Feyzioglu (1997a) for the case of HNC, although here the estimation includes a polynomial time trend to control for unobservables over time.

Unit root tests such as ADF, PP, and the Generalized Least Squared detrended ADF tests (ADF-GLS) indicate that consumption, prices, and GDP are $I(1)$. Information criteria (BIC) indicate that it is sufficient to consider a linear polynomial ($m = 1$) to adequately describe the underlying trend. The Engle and Granger test (1987) was applied to the residuals in equation (2). The test suggests that the series are cointegrated, which rules out the possibility of spurious relationships among the variables.[28] Cointegration indicates that there is no need to include lags of consumption; hence, equation (2) represents the long-run relationship. Furthermore, an Error Correction Model (ECM) is estimated in order to explore the short-run relationship.

The estimations show no evidence of a decrease in gasoline consumption as a result of PYP during the initial phase. In contrast, estimates of the effect of the extended phase were shown to be positive and statistically significantly different from zero (see Table 7A). The long-run PYP elasticity of gasoline consumption during the extended phase was around 10%. This implies that an individual who consumed on average 2.6 gallons per month of gasoline prior to the policy implementation date was stimulated to increase his/her consumption to 2.9 gallons per month in the long-run as a result of PYP. This finding is in line with Eskeland and Feyzioglu (1997a) who found that the HNC program increased vehicle use and is consistent with the increased CO levels during the extended phase of PYP.

[28]Johansen's test was also conducted. It indicated the presence of only one cointegration relationship.

Three specifications were conducted to evaluate the robustness of this result. Equation (2) is estimated without controlling for seasonality (column 1), adding the monthly dummies (column 2), and by Dynamic Ordinary Least Squares (DOLS) (columns 3)[29]. DOLS was estimated since this estimator has been shown to have better small sample properties than other alternative estimators when estimating cointegration relationships (Stock and Watson, 1993) such as demand equations.[30] The conclusions about the effect of the program on gasoline consumption remain unaltered across these specifications.

Estimations also indicate that gasoline price and GDP have the expected signs in all specifications. The long-run price elasticities oscillate between -0.15 and -0.18, and the long-run income elasticities range between 0.63 and 0.73. Estimates of the price and income elasticities are statistically significantly different from zero in most of the specifications. Dahl and Sterner (1991a) surveyed around hundred studies that may vary significantly in scope and methodology around the world and provided average price and income elasticities that seem to be similar to those reported here. For lagged consumption models with monthly periodicity, they found mean long-run price and income elasticities of -0.23 and 0.85, respectively. In the case of static models, they showed mean values of -0.29 and 0.52 for price and income elasticities (see also Dahl and Sterner, 1991b).[31]

Besides the possible developments in vehicle technology[32], gasoline price seems to play an important role in the decreasing trend of gasoline consumption in Bogota during 1996-2008. At the beginning of 1996, a gasoline tax of 13% was introduced that shifted the patterns of gasoline consumption in the earlier years. The tax progressively increased to 14% in 1997, 20% in 1999, and 25% in 2003. Those changes were implemented to finance road infrastructure and were accompanied by the elimination of a set of domestic subsidies to allow price fluctuations according to international prices. Given that monthly gasoline consumption in

[29] Eskeland and Feyzioglu (1997) also consider that the program may cause a change in the price and income slopes. Interaction terms of PYP with price and income were also included in the gasoline consumption equation. Those terms were not statistically significant different from zero at the 1% significance level.

[30] This estimator also deals with the concern of potential simultaneity bias among regressors. The procedure consists of adding lagged and lead values of the first differences of the regressors and the first differences of the original regressors.

[31] Dahl and Sterner (1991a, 1991b) classify static models as those where gasoline demand is a function of the real gasoline price and income. In addition to these regressors, lagged consumption models include lags of the dependent variable.

[32] There has been technological progress in vehicle fuel economy over time. For example, in 1996 a light vehicle in Bogota could travel 25 Km per gallon. Currently the fuel economy of a car is roughly 50 Km/gallon. The time trend included in equation (2) attempts to control for this technological development over time.

Bogota decreased from a total of 42 million gallons at the beginning of 1996 to around 24 gallons in 2008, the negative long-run price elasticity implies that the gasoline price was responsible for a non-negligible portion of the volume decline. This suggests that a price-based mechanism would be more effective in discouraging driving than PYP.

In contrast to the effect of the gasoline price, the estimates of the ECM analysis show that neither the initial phase nor the extended phases of the PYP program led to short-run reductions in gasoline consumption (Table 7B). This result is demonstrated by the lack of evidence of reduced CO during several time-interval subsamples during the first months of implementation of the program and the relatively quick monthly rate of adjustment of gasoline consumption towards the long-run equilibrium – for example, between 27% and 42% of the adjustments happen in the first month. Short-run elasticities were statistically significant and varied from -0.43 to -0.62 for price and from 0.27 to 0.42 for GDP. The estimates of the price elasticities also show a higher response of the gasoline demand to pricing in the short-run than in the long run. Eskeland and Feyzioglu (1997b) studied the gasoline demand in Mexico and found results in the same direction.

4.2 The effect of PYP on vehicle registrations and new vehicle sales

That PYP did not lead to reductions in CO concentrations in the long run and in fact led to increases in CO during the extended phase can be explained by households having responded to the program by increasing their vehicle stock or by increasing the number of trips. One approach to analyzing this type of behavioral response is to evaluate possible increases in the number of registered vehicles and in the sales of new cars after the implementation of the program. Data on monthly vehicle registrations between January 1997 and November 2010 from the Transport Agency of Bogota and on new vehicle sales between July 2000 and September 2011 from Econometria S.A is used to evaluate this effect.

The effect of PYP on vehicle registrations was explored using also an ARDL specification. The model can be represented as:

$$V_t = \alpha + \omega_1 PYP1_t + \omega_2 PYP2_t + \sum_{i=1}^{m} \gamma_i t_t^i + \sum_{j=1}^{p} \phi_j V_{t-j} + \mathbf{D}_t' \eta + u_t \ , \quad t = 1, ..., T \tag{3}$$

where V_t is the registration of vehicles in logs at period t, PYP1 and PYP2 are as described above, u_t is the error term at time t, and p and m are the lags of registrations and the polynomial order, respectively. ω_1 and ω_2 measure the short-run effect of the initial and extended phases of PYP on vehicle registrations, respectively. Fees for vehicle registration are not included in the model because they were not available. Equation (3) is estimated for registrations of all private vehicles and automobiles, controlling for seasonal variation. Standard unit root tests (ADF, PP, ADF-GLS) indicate nonstationarity of the registered automobiles and private vehicles. Hence the dependent variable is the first difference of monthly vehicle registrations. Using BIC, the identified model includes the first, second and twelfth autoregressive lags, and a quadratic time trend.

Results are presented in Table 8 – see column (1) for automobiles and column (2) for private vehicles. In both specifications, the coefficients of the initial and extended phases of the program are positive. However, the estimates are statistically significant at conventional levels only for the extended phase, indicating that increases in vehicle registrations during this period are associated with PYP. Given the size of the long-run estimates for the extended phase (10%), drastic restrictions appear to have stronger impacts than the initial program. This result is certainly consistent with increases in the CO levels during the extended phase in the long run and supports the idea that households adapted to the policy, making it less effective. The lack of evidence of increased registration in the initial phase indicates that other mechanisms may have influenced the net effect on CO concentrations. As a robustness check, the number of registrations of load vehicles (trucks) is tested for possible effects of the program, using a similar ARDL model. These types of vehicles were not restricted by the program and one might expect that PYP did not affect load vehicle registrations (see column 3 in Table 8). Indeed, neither the initial phase nor the extended phase of the program influenced the number of truck registrations.

Because the available data on sales of new cars are for the period after the initial phase of PYP, the effect of the program can only be estimated for the extended phase. Considering that vehicle sales may be affected by the performance of the economy, the GDP growth rate is also used as a control in the estimations. The ARDL model implemented to estimate the effect of PYP on new sales is represented by the following equation:

$$S_t = \alpha + \rho PYP2_t + \sum_{i=1}^{m} \gamma_i t_t^i + \sum_{j=1}^{p} \mu_j S_{t-j} + \lambda R_t + \mathbf{D}_t' \eta + \varepsilon_t \ , \quad t = 1, ..., T \tag{4}$$

where S_t is new automobile sales in logs at period t, PYP2 is as described above, ε_t is the error term at time t, and R_t is the GDP growth rate in logs at time t. ρ is the short-run effect of the extended phase on new automobiles sales. The GDP growth rate is obtained from DANE. The series exhibit nonstationarity and seasonality. Although the variables are $I(1)$, they appear not to be cointegrated under the Engel and Granger test. Therefore, the variables are in first differences. BIC indicates that it is sufficient to include two autoregressive terms and a quadratic time trend.

Estimates are shown in Table 9. Column (1) depicts the estimates of equation (4) for automobile sales. The short- and long-run coefficients of PYP2 are positive and statistically significant at the 10% level. The long-run coefficient of PYP2 implies an increase of around 5.5% in the growth rate of automobile sales associated with the program intervention. This result is consistent with the increased vehicle registrations for that period. Although vehicle prices were not available, a price index was created using the ratio of the total monetary value of sales in the country to the total number of new automobile sales[33]. The effect of the program on automobile registrations in this case is 8%, which is higher than the value obtained when controlling by GDP growth (see column 2 in Table 9). A possible concern when constructing the price index is that it might be endogenous because the vehicle sales in Bogota represent almost 50% of the total sales in Colombia. To address this issue, Two Stage Least Squares (2SLS) is estimated by instrumenting the price index with the real exchange rate. The results imply a slightly higher effect on automobile registrations than the estimate in the two previous specifications.

As a robustness check, the ARDL model is run for automobile sales of other cities in the country. Because some cities have already implemented driving restrictions similar to PYP, it is likely that vehicle sales in those cities also increased as a result of the restrictions, and hence this simultaneous effect of other restrictions may yield misleading estimates. Therefore, the model is run for the aggregated automobile sales excluding those cities. Columns (4), (5) and (6) of Table 9 presents the results. The short- and long-run coefficients for PYP in those estimations are statistically insignificant, supporting the fact that the changes occurred in Bogota at the time the program was implemented. Overall, these results show that PYP spurred households to acquire more new automobiles.

[33]Information on these variables was obtained from DANE.

4.3 The effect of PYP on other variables

The evidence in the preceding subsection explored the effect of the policy on new vehicle sales. However, this might not be the only channel through which households adapted to the policy. Analysis of used vehicle sales would be an alternative way to shed light on changes in car ownership. However, statistics on used vehicle sales are scarce, impeding the development of an econometric model. In 2010, the Ministry of Transport and the Unified National Transit Registration System (RUNT) presented a broad figure concerning the total transfers of used private vehicles. It reported that around 104,061 used automobiles were sold in the city, corresponding to 33.3% of the total market in the country. At that time, the sales of new automobiles in Bogota reached 47,491, implying a ratio of roughly two used cars per each new car. Basic statistics on transfers of used automobiles indicate that sales increased from 5,784 to 8,556 cars from January to February 2009, just one month after the implementation of the extended phase of PYP. Including the station wagon and sport utility vehicles in the statistics it results in an increase during the same period from 7,815 to 11,832 vehicles. This represents an increase of about 48-50% in the sales of used vehicles. A comparison of the sales between January and February one year before yields a lower increase of 11-14%. Therefore, the transfers seem to increase four times more during the first month after the implementation of the extended phase than during the same month in the previous year.

The behavior of traffic counts are also analyzed using descriptive information for the period before and after the implementation of the extended phase of PYP. Traffic data during the application of the initial phase were not available. Figure 3 depicts the diurnal profile of traffic counts for light vehicles across several measurement stations in the city during the first quarter of 2008, 2009, and 2010. Simple observation indicates that the number of vehicles in circulation clearly increased between the first quarters of 2008 and 2010 during the peak hours: by approximately 13% in the morning and 17% in the evening. Interestingly, the changes in traffic during the off-peak hours between the first quarter of 2008 and 2009 and the first quarter of 2009 and 2010 followed the same trend as the short-run coefficients of the PYP program on CO concentrations shown in Figure 2. Initially traffic declined during the first quarter of 2009, but later it returned to similar levels as in the first quarter of 2008. These statistics suggest that the driving restriction program had a limited effect on traffic congestion in the long run.

However, caution is advised when interpreting this type of statistics. For instance, the increase in vehicle flow may be attributed to endogenous changes in the pattern of the traffic. That is, if a fraction of cars are not used because of the driving restriction, the roads tend to be less congested and consequently the number of vehicles per hour crossing the traffic count monitoring point may increase. It seems that this might not have been the case in Bogota. The timespan corresponding to the morning and evening rush hours had already been constrained several years before the extended phase of PYP was implemented, so PYP was not expected to strongly affect the average traffic flow at those times. Therefore, it would be expected that there would be roughly a similar number of vehicles in circulation. This would rule out the argument that, with the same vehicle fleet, more vehicles are observed at the monitoring points due to the benefits of less congested streets after the implementation of the program. To the contrary, if traffic counts increased during the morning and evening rush hours, this may be associated with increases in the number of trips, and sales of new or used vehicles that might have been stimulated by the reduction in the relative driving costs of the morning peak/off-peak, particularly for drivers with high value of time, which is also consistent with the increases in CO concentrations.

5 Compiling the evidence: the more stringent, the better?

In this section, the results are compiled between periods to analyze whether a drastic ban has stronger effects than the moderate restrictions. The evidence provided in the preceding sections points out that neither phase of the policy achieved the expected objectives in the long run.

A decrease in CO concentrations in the short run (21%) during the initial phase of PYP suggests that improvements in air quality and therefore reductions in car use can be achieved, yet the social benefits are limited. Most of the effects of the policy during the initial phase were expected during the morning and evening peak, but rather these effects wear off over time. Increases in the vehicle fleet seem not to explain the ineffectiveness of the program in the long run. On the one hand, this phase consisted of moderate restrictions allowing driving during off-peak hours, which may have had less influence on the vehicle ownership. On the other hand, households had not had prior experience with the program, and hence a precautionary attitude might have been to wait until the Mayor would announce the PYP program as a permanent policy.

Other channels such as displacement of trips and increase mileage might have been the causes of the limited effectiveness of the program. An ex-ante study of the initial phase of the PYP program mentions that the trips affected by the restriction might be displaced to other hours of the day, which may shift earlier or later than the restriction periods (Acevedo, 1998). Nevertheless, new traffic during peak hours might also be indirectly stimulated by the program since drivers would expect to find less congested roads.

Although enforcement of the program has been considered in general satisfactory, there are some facts that during the initial phase of the program might question the full effectiveness of the program. Acevedo (1998) describes that the procedure control was in charge of a few number of police officers (only 150 motorized agents and 900 officers) who might not have carried out a full coverage of the city to enforce the driving restriction. Infringers may have found routes where the probability of being caught in violation was very low. Indeed, three months after the implementation of the program a newspaper reported that drivers took secondary roads to avoid the police controls and informed that some of the police officers had to execute other tasks reducing the coverage in other areas of the city (El Tiempo, 1998b). Moreover, not suppressing trips but rather shifting them in time caused also bottlenecks in some of the main routes, particularly at the beginning of the morning peak (El Tiempo, 1998a). Overall these factors may have contributed to counteract the effect of the PYP program during the initial phase.

In the case of the extended phase, PYP pursued a decrease in vehicular traffic during the off-peak hours. Rush hours had already been restricted since some years earlier. Although CO reductions were observed in the short run during off-peak hours, this effect disappeared after one year. Instead, CO levels increased in the long-run for the morning (26%) and evening peak (36%). The explanation to this failure is that behavioral responses may arise to circumvent the policy, increasing car use and CO levels. One way to get around the program is to buy new vehicles. There is evidence in this direction, indicating that households responded by increasing their vehicle stock. Given that increases in the vehicles registrations during the initial phase were not present, the estimated program effect on the growth rate of vehicle registrations during the extended phase (10%) indicates that households may learn from prior experiences, generating a faster response. Household were exposed to moderate driving restrictions for more than one decade, thus the shift towards drastic restrictions generated more incentives to buy new vehicles to circumvent the program. Gasoline consumption and CO concentrations during the peak hours have increased during the extended phase of the program to a higher extent than the initial phase. Overall, these results indicate that drastic restrictions generated stronger counterproductive consequences.

So who acquired new cars? Official statistics in Bogota for 2008 indicate that 21.7% of the households owned a car. The figure for 2010 increased slightly to 21.8%.[34] Therefore, if the new vehicle sales increased while the ownership fraction remained almost constant during that period, the households that acquired a new vehicle were those that already owned a car. This is consistent with what other studies mention about strategies to skip the driving restriction, i.e., households acquire a second or third car (see Eskeland and Feyzioglu, 1997; Davis, 2008; Gallego et al., 2011).

Another question that arises is how in practice a household can avoid the restriction by buying a new car. Not surprisingly, the transaction costs of choosing the license plate number seem to be very low. Car dealers buy a group of license plate numbers, which are pre-allocated by the Transport Agency of Bogota through the Integral Services of Mobility (SIM). These license plate numbers are allocated in certain combinations of digits from 0 to 9. Car dealers are not allowed to choose the numbers of the plates systematically; that is, they cannot, for example, purchase 10 plates ending with 1, and 10 plates ending with 2, or any other similar arrangement. Those who can choose the plate numbers are the customers. This implies an opportunity to avoid PYP. Information in the media supports the idea that PYP is a relevant factor in the bargaining process in the market for new and used cars.

Wealthier households are the ones who might afford to buy an additional car. The demand for private vehicles in this group is higher than for middle-income or low-income households. For example, while on average a high-income household in Bogota owns more than one car, there is one car per each two middle-income households and one car for each four low-income households.[35] For households where the likelihood of increasing the vehi-

[34]Data accessed from DANE. Encuesta Nacional de Calidad de Vida 2008 y 2010.

[35]Figures are taken from the Mobility Survey 2011.

cle stock seems to be more limited, one behavioral response to the program may have been to substitute trips at other times of the day. This is inequitable because the trips of these groups are affected disproportionately due to the lower values of time and higher value of money (Verhoef, 1995). Evidence is found in this study to support this adaptation mechanism. For instance, estimates in a middle-income area of the city during the initial phase of PYP provide some indication of intertemporal substitution of trips in the hour before the restricted morning peak. The long-run program effect coefficient indicates an increase in CO concentrations of 18% during the pre-morning peak after the implementation of the initial phase of program (see Table 10). This fact is also consistent with media reports indicating that households in those areas started getting up early in response to the restriction.[36]

Although the effects of the extended phase of PYP on vehicle sales have contributed to the ineffectiveness of the program, this increase alone might not explain all the increase in either gasoline consumption or CO concentrations during the peak hours. The increase of 9% in vehicle sales corresponds to an increase of only 4800 vehicles after one year of implementation. This may be few vehicles to account for all the effects on increased concentrations. Furthermore, new cars must satisfy environmental standards, including the three-way catalytic converter (TWC), which removes a large fraction of CO emissions. Therefore, the effect of the new vehicles on CO might be limited, unless it has been accompanied by an increase in the number of trips.

Several facts here suggest that the extended phase of PYP caused an increase in total driving. On the one hand, gasoline consumption increased by 10% (2 million gallons per month) indicating that households (either with new or used vehicles) have increased vehicle use. A second car usually brings additional vehicle use because trips in the new car not only substitute for the trips restricted by the program but also expand the possibility of using both the new and used car for other purposes. On the other hand, used cars tend to be less fuel-efficient than new ones; moreover, the lifetime of the TWC for some used cars in Bogota has already expired (more than 10 years of use). Furthermore, more than 50% of the automobiles in the city do not have TWC. These factors explain the increased CO concentrations during the extended phase.

[36]Another study carried out by Medina (2009) reveals that households with very low, low, and middle income may have lost 14% of their income due to PYP.

Households have also responded to increases in gasoline price. Based on back-of-the-envelope estimates, it is possible to calculate the price increase that would be necessary to achieve a reduction of 21% in CO concentrations, as happened in the first two months of implementation of the initial phase of PYP. To investigate the sensitiveness of carbon monoxide due to changes in prices, the relationship between average weekly CO concentrations and gasoline price during 1997-1999 was estimated. Weekly CO levels were chosen because it is plausible that households fill up the gas tank on a weekly basis. CO concentrations and gasoline price are cointegrated, enabling the estimation of the long-run equation. The price elasticity estimate was -1.53, suggesting that, to generate a decline of 21% in CO levels, the gasoline price should have increased 13.7%. Interestingly, this price increase was of the same order of magnitude as the gasoline tax introduced in 1996. Likewise, if a reduction of 40% in traffic would imply a decline of 40% in CO concentrations (keeping constant other factors such as meteorology), the gasoline price associated with that reduction of traffic should have increased 26.1%, a similar value to the current gasoline tax.[37] Given that driving also is associated with other pollutants such as greenhouse gases, the decrease in vehicular traffic would induce a reduction in carbon emissions.

It is worth pointing out that there are other effects not considered in the present analysis, such as impacts on commerce and labor (see Echeverry et al., 2009) or on other pollutants and health. These issues are outside the scope of this study but may increase the overall costs of the program, given the long-run ineffectiveness of PYP.

6 Summary and conclusions

Driving restrictions have been used in several cities around the world to overcome traffic congestion and air pollution. At present, seven cities in Colombia are subject to driving restrictions. This study contrasts previous studies evaluating programs implemented in a drastic fashion by assessing the effects on air quality and car use of shifting the regulation from moderate to drastic restrictions. Rather than reducing congestion and air pollution, the most stringent phase of the program generated stronger counterproductive effects than the least stringent one. The extended phased of the program increased CO concentrations,

[37]Eskeland (1994) also suggests that gasoline taxes can be used to complement other abatement measures to reduce vehicle emissions.

vehicle ownership, and total driving, indicating that households were quite responsive to drastic restrictions.

Although driving restriction programs were associated with an immediate reduction in air pollution or car use, the social benefits are limited. These effects wear off over time because households find alternative ways to avoid the restrictions. These programs, besides being ineffective in the long-run, were inefficient in that they affected many households by increasing their commuting costs. The findings of this study question the rationale for extending the program to other cities or making it more stringent, as well as whether it should continue to be applied where already implemented.

Gasoline pricing may be a more effective tool to discourage driving than PYP. An additional tax equivalent to the current gasoline tax might significantly reduce CO concentrations and congestion. As Sterner (2007) argues, although in some cases the purpose of gasoline taxes is to generate revenue or for other reasons unrelated to reduce pollution, gasoline taxes do substantially affect carbon emissions.

Other policy alternatives such as road tolls might also be explored to ration the scarce resource of road infrastructure. In Stockholm and London, for example, road tolls have been effective in dealing with congestion and pollution externalities (see Björjesson et al., 2012). The fees used in these programs depend on the time of day and increase during congestion-prone times. Interestingly, the evidence in Stockholm indicates that the fee elasticity of car use is greater in the long-run, implying that the effect of the fee does not vanish over time. Bogota might benefit from the lessons learned from the programs used in these cities. Simulation studies of a congestion charge in Bogota would be useful to quantify its impact on traffic and air quality.

References

Acevedo, J. E. (1998). Evaluacion de la restriccion de transito vehicular por numero de placa: programa pico y placa. Camara de Comercio de Bogota.

Alcaldia Mayor de Bogota. (2009). Decreto 033 de 2009. ”Por el cual se dictan disposiciones para el mejor ordenamiento del transito de personas y vehiculos por las vias publicas”. Febrero 5. Bogota D.C.

Aron, R., Aron, I-Ming. (1978). Statistical forecasting models: I. Carbon monoxide concentrations in the Los Angeles basin. *Journal of the Air Pollution Control Association* **28**(7), 681–681.

Body, J.W.D, Smalley, R.J., Dixon, N.S., Tate, J.E., Tomlin, A.S. (2005). The spatial variability in concentrations of traffic-related pollutant in two street canyons in York, UK - Part I: The influence of background winds. *Atmospheric Environment* **39**, 3147–3161.

Börjesson, M., J. Eliasson, M. B. Hugosson, K. Brundell-Freij. (2012). The Stockholm congestion charges5 years on. Effects, acceptability and lessons learnt. *Transport Policy* **20**(1), 1–12.

Carrillo, P. E., Malik, Arun S., Yoo, Y. 2013. Driving restrictions that work? Quito’s Pico y Placa program. Department of Economics, George Washington University.

Comrie, A.C., Diem, J.E. (1999). Climatology and forecast modeling of ambient carbon monoxide in Phoenix, Arizona. *Atmospheric Environment* **33**, 5023–5036.

Dahl C., Sterner, T. 1991a. A survey of econometric gasoline demand elasticities. *International Journal of Energy Systems* **11** (2), 53–76.

Dahl C., Sterner, T. 1991b. Analysing gasoline demand elasticities: a survey. *Energy Economics* **13** (3), 203–210.

Davis, L., (2008). The Effect of Driving Restrictions on Air Quality in Mexico City. *Journal of Political Economy* **116**(1), 38–81.

Dennis, R.L., Byun D.W., Novak, J.H. Gallupi, K.J., Coats, C.J., Vouk, M.A. (1996). The next generation of integrated air quality modelling: EPA’s Models-3. *Atmospheric Environment* **30**, 1925–1938.

Derwent, R.G., Middleton, D.R., Field, R.A., Goldstone, M.E., Lester, J.N., Perry, R. (1995). Analysis and interpretation of air quality data from an urban roadside location in central London over the period from July 1991 to July 1992. *Atmospheric Environment* **29**, 923–946.

Echeverry, J.C., Navas, V., Gomez, M.P. (2009). Evaluacion del impacto del pico y placa sobre el comercio de Bogota. Econcept.

El Tiempo. (1998a). La congestion vial le madrugo a Bogota. August 19,1998. http://www.eltiempo.com (accessed October 19, 2010).

El Tiempo. (1998b). La restriccion, sin control. November 12, 1998. http://www.eltiempo.com (accessed October 19, 2010).

El Tiempo. (2009). 14 por ciento de su ingreso pierden hogares de estratos 1,2 y 3 por efecto del nuevo Pico y placa. September 19, 2009. http://www.eltiempo.com (accessed September 22, 2009).

EPA. (2010). Integrated science assessment for carbon monoxide. EPA/600/R-09/019F.

Eskeland, G. (1994). A presumptive Pigouvian tax: complementing regulation to mimic an emission fee. *The World Bank Economic Review* **8**(3), 373–394.

Eskeland, G. and Feyzioglu T. (1997a). Rationing can backfire: The "Day without a Car" in Mexico City. *The World Bank Economic Review* **11**(3), 383-408.

Eskeland, G. and Feyzioglu T. (1997b). Is demand for polluting goods manageable? An econometric study of car ownership and use in Mexico. *Journal of Development Economics* **53**, 423–445.

Gallego, F., Montero, J.M., Salas, C. (2011). The effect of transport policies on car use: Theory and evidence from Latin American cities. Documento de Trabajo N 407. Pontificia Universidad Catolica de Chile Instituto de Economia.

Hahn, J., P. Todd, and W. Van der Klaauw. (2001). Identification and estimation of treatment effects with a Regression-Discontinuity Design. *Econometrica* **69**(1), 201–209.

Han X., Naeher, L.P. (2006). A review of traffic-related air pollution exposure assessment studies in the developing world. *Environmental International* **32**, 106–120.

Huo, M., Sun, Q., Bai, Y., Xie, P., Liu, Z., Li, J., Wang, X., Lu, S. (2010). Chemical character of precipitation and related particles and trace gases in the North and South of China. *Journal of Atmospheric Chemistry* **67**, 29-43.

Lin, C., Zhang, W., Umanskaya, V. (2012). The effects of driving restrictions on air quality: evidence from Bogota, Sao Paulo, Beijing and Tianjin. Working paper.

Maffeis, G. (1999). Prediction of carbon monoxide acute air pollution episodes. Model formulation and first application in Lombardy. *Atmospheric Environment* **33**, 3859–3872.

Medina, C. (2009). Aglomeracion economica, congestion y restriccion al trafico vehicular: Es aconsejable un dia sin auto por Semana?. Banco de la Republica, Sucursal Medellin.

Osakwe, R. (2010). Political Economy of Private Transport Regulation in Costa Rica. Magister Scientiae in Environmental Socioeconomics.

Pesaran, M.H. and Smith, R. (2012). Counterfactual analysis in macroeconometrics: An empirical investigation into the effects of quantitative easing. IZA Discussion Paper No. 6618.

Salas, C., (2010). Evaluating Public Policies with High Frequency Data: Evidence for Driving Restrictions in Mexico City Revisited. Documento de Trabajo N 374. Pontificia Universidad Catolica de Chile.

SDA and Uniandes. (2009). Parte 2: Inventario de emisiones provenientes de fuentes fijas y moviles. Elementos tecnicos del plan decenal de descontaminacion de Bogota.

Secretaria de Ambiente. (2003). Red de Monitoreo de Calidad del Aire de Bogota. Informe Anual, 2002. Bogota.

Secretaria de Ambiente. (2010a). Red de Calidad del Aire de Bogota. http: //secretariadeambiente.gov.co/sda/libreria/ (accessed September 15, 2010).

Secretaria de Movilidad. (2009). Bogota en vehiculos particulares. Bogota D.C. http: //www.movilidadbogota.gov.co (accessed August 29, 2009).

Secretaria de Movilidad. (2010a). Informe trimestral. Medida de Pico y Placa Extendida – Decreto 033 de 2009. Primer trimestre de 2010.

Secretaria de Movilidad. (2010b). Informe anual. Medida de Pico y Placa expandida – Decreto 033 de 2009.

Secretaria de Movilidad. (2012). Evaluacion de alternativas de modificacion a la medida de restriccion del transito de vehiculos particulares "Pico y Placa". Revision de Decreto Distrital 025 de 2012.

Sterner, T. 2007. Fuel taxes: An important instrument for climate policy. *Energy Policy* **35**, 3194–3202.

Sterner, T.; Coria, J. (2012). Policy instruments for environmental and natural resource management. Second Edition. New York, NY: RFF Press, Routledge.

Stock, J.; Watson, M. (1993). A simple estimator of cointegrating vectors in higher order integrated systems.*Econometrica* **61**(4), 783–820.

Texas A&M Transportation Institute. (2011). Urban Mobility information. Table 2. Congestion Leaders (Top 40). http://mobility.tamu.edu/corridors/summary-tables/ (accessed September 5, 2012).

Verhoef, E. (1995). The demand curve under road pricing and the problem of political feasibility: A comment. *Transportation Research Part A: Policy and Practice* **29**(6), 459–465.

Viard, V., Fu; S. (2011). The effect of Beijing's driving restrictions on pollution and economic activity. SSRN working paper.

Walsh, M. (2003). Vehicle emissions and health in developing countries. In G. McGranahan and F. Murray, (Eds.), Air pollution and health in rapidly developing countries. Stockholm: Earthscan Publications Ltd. pp 146–175.

WHO. (2011). Tackling the global clean air challenge. September 26, 2011. http://www.who.int/mediacentre/news/releases/2011/ (accessed September 5, 2012).

Zarate, E., Belalcazar, L.C., Clappier, A., Manzi, V. (2007). Air quality modelling over Bogota, Colombia: Combined techniques to estimate and evaluate emissions inventories. *Atmospheric Environment* **42**, 6302–6318.

Table 1: Descriptive Statistics

Variable	Observations	Mean	Std. Deviation	Minimum	Maximum
Initial phase: August 18, 1997- August 17, 1999					
Carbon monoxide (ppm)	17027	3.11	1.53	0.1	11.7
Wind speed (m/s)	17037	1.16	0.76	0	4.4
Temperature(C)	17013	12.88	3.73	2.4	23.4
Relative humidity (%)	15131	72.69	12.78	16	97
Rain (mm)	17014	0.11	0.61	0	19
Wind direction (degrees)	17037	171.18	44.51	20	357
Temperature inversion	15127	0.24	0.43	0	1
Extended phase: February 7, 2008-February 5, 2010					
Carbon monoxide (ppm)	17503	0.84	0.47	0.1	6.1
Wind speed (m/s)	17511	2.33	1.15	0.4	6.5
Temperature(C)	17511	14.30	2.86	4.3	23.1
Relative humidity (%)	17473	67.46	13.74	13.0	90.9
Rain (mm)	17511	0.10	0.55	0	19.9
Wind direction (degrees)	17511	189.65	78.19	6.5	356.5
Temperature inversion	16846	0.24	0.43	0	1

Table 2: Effect of initial and extended phases of PYP on CO concentration for all hours of the day

Variable	(1) CO 1997-1999	(2) CO 2008-2010	Variable	(1) CO 1997-1999	(2) CO 2008-2010
L.CO	0.6825***	0.8035***	L.RH2	0.00005***	0.0001***
	(0.0120)	(0.0126)		(0.00002)	(0.00002)
L2.CO		-0.0833***	RAIN	0.0151***	-0.0123**
		(0.0109)		(0.0052)	(0.0050)
PYP	-0.0056	0.0274*	L.RAIN	0.0171***	0.0155***
	(0.0116)	(0.0159)		(0.0060)	(0.0052)
WS	-0.1182***	-0.1685***	RAIN2	-0.0010***	0.0004
	(0.0117)	(0.0124)		(0.0003)	(0.0005)
L.WS	-0.0167	-0.0494***	L.RAIN2	-0.0010**	0.0004
	(0.0118)	(0.0118)		(0.0004)	(0.0005)
WS2	0.0040	0.0143***	WINDD2	0.0962***	-0.0148
	(0.0026)	(0.0019)		(0.0290)	(0.0224)
L.WS2	0.0140***	0.0082***	WINDD3	0.0472*	-0.0500**
	(0.0026)	(0.0018)		(0.0274)	(0.0215)
TMP	0.0232**	0.0490**	WINDD4	0.0063	-0.0257
	(0.0094)	(0.0199)		(0.0260)	(0.0211)
L.TMP	-0.0573***	-0.0978***	WINDD5	-0.0049	-0.0040
	(0.0084)	(0.0185)		(0.0260)	(0.0212)
TMP2	-0.0012***	-0.0019***	WINDD6	-0.0180	0.0311
	(0.0003)	(0.0006)		(0.0261)	(0.0212)
L.TMP2	0.0021***	0.0030***	WINDD7	-0.0340	0.0170
	(0.0003)	(0.0006)		(0.0264)	(0.0211)
RH	0.0094***	0.0359***	WINDD8	-0.0761***	0.0148
	(0.0025)	(0.0034)		(0.0265)	(0.0214)
L.RH	-0.0077***	-0.0314***	TI	0.0525***	0.0334***
	(0.0024)	(0.0033)		(0.0072)	(0.0053)
RH2	-0.0001***	-0.0002***			
	(0.00002)	(0.00003)			
N	9999	10869	R^2	0.8307	0.8862
F(weather)	55.41	124.69	P-value	0.0000	0.0000
Sum of CO lags	0.6825	0.7202	AR stability	Yes	Yes

Notes: This table shows estimates from two regressions: (1) 1997-1999 and (2) 2008-2010 for all hours models (05:00-21:00). The dependent variable is carbon monoxide (CO) in logs. L.CO and L2.CO are the first and second lags of CO. PYP is an indicator variable equal to one after August 18, 1998 and zero otherwise in Column (1) and equal to one after February 6, 2009 and zero otherwise in Column (2), respectively. Meteorological variables are wind speed (WS), superficial temperature (TMP), relative humidity (RH), and rainfall (RAIN) in quadratic form and their corresponding first lags, wind direction for the 8-point compass (WINDD2- WINDD8), and temperature inversion (TI). Regressions also include a polynomial time trend of degree four and indicator variables for month of the year, day of the week, hour of the day, and interactions between weekends and hour of the day. Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 3: Effect of initial phase of PYP on CO concentration at different times of the day and week

	(1) CO All hours	(2) CO Morning peak	(3) CO Evening peak	(4) CO Off-peak	(5) CO Weekends
	Panel A. General model				
L.CO	0.6825***	0.6326***	0.7287***	0.6446***	0.7558***
	(0.0120)	(0.0284)	(0.0254)	(0.0323)	(0.0252)
PYP	-0.0056	-0.0573	0.0000	-0.0084	0.0175
	(0.0116)	(0.0451)	(0.0307)	(0.0219)	(0.0357)
N	9999	825	1249	2875	884
R^2	0.8307	0.7853	0.8280	0.7067	0.8530
AR stability	Yes	Yes	Yes	Yes	Yes
	Panel B. Model with short-run coefficients				
Two months later	-0.1397***	-0.1195	-0.2107*	-0.0585	-0.1005
	(0.0368)	(0.1055)	(0.1138)	(0.0544)	(0.1584)
Four months later	-0.0215	-0.0403	-0.0257	0.0790	0.0501
	(0.0402)	(0.1103)	(0.1358)	(0.0576)	(0.1511)
Six months later	-0.0191	-0.0607	-0.0412	0.0780	0.0600
	(0.0388)	(0.1068)	(0.1297)	(0.0577)	(0.1551)
Eight months later	0.0566	0.0346	0.0572	0.1379**	0.1572
	(0.0380)	(0.1175)	(0.1369)	(0.0589)	(0.1681)
Ten months later	0.0408	-0.0089	0.0135	0.0130	0.0952
	(0.0382)	(0.1164)	(0.1284)	(0.0599)	(0.1482)
Long-run PYP	-0.0176	-0.1558	0.0001	-0.0235	0.0717
	(0.0366)	(0.1241)	(0.1131)	(0.0623)	(0.1463)

Notes: This table shows estimates from 30 regressions for the period 1997-1999. The dependent variable is carbon monoxide (CO) in logs. L.CO is the first lag of CO. All hours includes observations for 05:00-21:00 all days of the week; morning peak, evening peak, and off-peak restrict the sample to 07:00-09:00, 17:00-20:00, and 10:00-17:00, respectively, during weekdays. Weekend corresponds to the aggregated time slot 07:00-09:00 and 17:00-20:00 during weekends. PYP is an indicator variable equal to one after August 18, 1998 and zero otherwise. Panel B shows short-run coefficients and standard errors of PYP. All specifications include meteorological variables such as wind speed, superficial temperature, relative humidity, and rainfall in quadratic form and their corresponding first lags, wind direction using the 8-point compass, and temperature inversion. Regressions are also fitted along a polynomial time trend of degree four for the all hours model and degree three for the other models, and include indicator variables for month of the year, day of the week, and hour of the day. Interactions between weekends and hour of the day are added only in the all hours model. Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 4: Effect of the extended phase of PYP on CO concentration at different times of the day and week

	(1) CO All hours	(2) CO Morning peak	(3) CO Evening peak	(4) CO Off-peak	(5) CO Weekends
Panel A. General model					
L.CO	0.8035***	0.8226***	0.7499***	0.7218***	0.8079***
	(0.0126)	(0.0338)	(0.0279)	(0.0284)	(0.0348)
L2.CO	-0.0833***	-0.1487***	0.0175	-0.0090	-0.0778**
	(0.0109)	(0.0286)	(0.0247)	(0.0232)	(0.0307)
PYP	0.0274*	0.0861*	0.0836***	-0.0099	-0.0269
	(0.0159)	(0.0452)	(0.0297)	(0.0275)	(0.0607)
N	10869	1418	1404	2803	1170
R^2	0.8862	0.8087	0.8908	0.8283	0.8929
AR stability	Yes	Yes	Yes	Yes	Yes
Panel B. Model with short-run coefficients					
Two months later	-0.0155	0.1962	0.2178	-0.1510*	-0.0458
	(0.0556)	(0.1281)	(0.1417)	(0.0789)	(0.2163)
Four months later	-0.0886	0.2604*	0.2776*	-0.2271**	-0.1159
	(0.0614)	(0.1421)	(0.1595)	(0.0884)	(0.2315)
Six months later	-0.0698	0.2948**	0.2698*	-0.2501***	-0.1866
	(0.0620)	(0.1476)	(0.1530)	(0.0966)	(0.2294)
Eight months later	-0.0878	0.2717*	0.1891	-0.3609***	-0.1900
	(0.0634)	(0.1467)	(0.1438)	(0.1030)	(0.2257)
Ten months later	-0.0830	0.2234	0.1952	-0.2585***	-0.1357
	(0.0635)	(0.1462)	(0.1342)	(0.0958)	(0.2267)
Long-run PYP	0.0979*	0.2642*	0.3594***	-0.0344	-0.0998
	(0.0566)	(0.1395)	(0.1298)	(0.0959)	(0.2249)

Notes: This table shows estimates from 30 regressions for the period 2008-2010. The dependent variable is carbon monoxide (CO) in logs. L.CO and L2.CO are the first and second lags of CO. All hours includes observations for 05:00-21:00 all days of the week; morning peak, evening peak, and off-peak restrict the sample to 06:00-09:00, 16:00-19:00, and 10:00-16:00, respectively, during weekdays. Weekend corresponds to the aggregated time slot 06:00-09:00 and 16:00-19:00 during weekends. PYP is an indicator variable equal to one after February 6, 2009 and zero otherwise. Panel B shows short-run coefficients and standard errors of PYP. All specifications include meteorological variables such as wind speed, superficial temperature, relative humidity, and rainfall in quadratic form and their corresponding first lags, wind direction using the 8-point compass, and temperature inversion. Regressions are also fitted along a polynomial time trend of degree four for the all hours model and degree three for the other models, and include indicator variables for month of the year, day of the week, and hour of the day. Interactions between weekends and hour of the day are added only in the all hours model. Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 5: Effect of initial phase of PYP on CO concentration: Robustness checks

	(1) CO All hours	(2) CO Morning peak	(3) CO Evening peak	(4) CO Off-peak	(5) CO Weekends
Panel A. Specifications with economic variables					
With gasoline prices	-0.0153	-0.1443	0.0052	-0.0242	0.0475
	(0.0356)	(0.1203)	(0.1121)	(0.0622)	(0.1450)
With real exchange rate	-0.0439	-0.0247	-0.0127	-0.0850	0.0419
	(0.0383)	(0.1342)	(0.1295)	(0.0626)	(0.1581)
Panel B. Specifications with other controls					
With industrial production index	-0.0419	-0.1817	0.0144	-0.0644	0.0049
	(0.0367)	(0.1280)	(0.1196)	(0.0618)	(0.1521)
With SO2	-0.0260	-0.1560	-0.0070	-0.0343	0.0651
	(0.0362)	(0.1245)	(0.1130)	(0.0622)	(0.1454)
With technology regulation	-0.0105	-0.1698	-0.0015	0.0054	0.0780
	(0.0363)	(0.1242)	(0.1135)	(0.0602)	(0.1484)
Panel C. Alternative polynomial orders of the time trend					
4th- order polynomial	-0.0176	-0.1505	-0.0042	-0.0355	0.0571
	(0.0366)	(0.1227)	(0.1125)	(0.0613)	(0.1482)
5th- order polynomial	-0.0121	-0.1365	-0.0402	0.0010	0.0686
	(0.0361)	(0.1237)	(0.1169)	(0.0572)	(0.1508)
Panel D. Specifications with orthogonal regressors					
With principal components	0.0374	-0.0747	0.0381	0.0179	0.1147
	(0.0377)	(0.1267)	(0.1110)	(0.0609)	(0.1442)

Notes: This table shows estimates from 40 regressions for the period 1997-1999. PYP is an indicator variable equal to one after August 18, 1998 and zero otherwise. Each estimate reports the PYP long-run coefficient of an alternative specification. The dependent variable is carbon monoxide (CO) in logs. All hours includes observations for 05:00-21:00 all the days of the week; morning peak, evening peak, and off-peak restrict the sample to 07:00-09:00, 17:00-20:00, and 10:00-17:00, respectively, during weekdays. Weekend corresponds to the aggregated time slot 07:00-09:00 and 17:00-20:00 during weekends. All specifications include meteorological variables. Regressions are also fitted along a polynomial time trend of degree four for all hours model and degree three for the other models, and include indicator variables for month of the year, day of the week, and hour of the day. Interactions between weekends and hour of the day are added only in the all hours model. Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 6: Effect of extended phase of PYP on CO concentration: Robustness checks

	(1) CO All hours	(2) CO Morning peak	(3) CO Evening peak	(4) CO Off-peak	(5) CO Weekends
Panel A. Specifications with economic variables					
With gasoline prices	0.0485	0.1678	0.2801**	-0.0232	-0.1046
	(0.0583)	(0.1432)	(0.1350)	(0.0994)	(0.2252)
With real exchange rate	0.0079	0.3364**	0.3078**	-0.1881**	-0.0985
	(0.0563)	(0.1429)	(0.1271)	(0.0893)	(0.2247)
Panel B. Specifications with other controls					
With industrial production index	0.0912	0.2555*	0.3449***	-0.0451	-0.1160
	(0.0565)	(0.1394)	(0.1286)	(0.0952)	(0.2223)
With SO2	0.0999*	0.2024	0.3652***	-0.0374	-0.1147
	(0.0561)	(0.1309)	(0.1281)	(0.0967)	(0.2232)
With technology regulation	0.1061*	0.3160**	0.3616***	-0.0509	-0.0963
	(0.0565)	(0.1397)	(0.1276)	(0.0938)	(0.2236)
With roadwork investment	0.1041*	0.3162**	0.3581***	-0.0524	-0.1074
	(0.0565)	(0.1398)	(0.1273)	(0.0937)	(0.2243)
Panel C. Alternative polynomial orders of the time trend					
4th- order polynomial	0.0979*	0.2733**	0.3528***	-0.0459	-0.1440
	(0.0566)	(0.1393)	(0.1271)	(0.0937)	(0.2240)
5th- order polynomial	0.1527***	0.3677***	0.4022***	0.0013	-0.1314
	(0.0577)	(0.1399)	(0.1292)	(0.0948)	(0.2236)
Panel D. Specifications with orthogonal regressors					
With principal components	0.1235**	0.2129	0.3271***	0.0842	0.0351
	(0.0591)	(0.1403)	(0.1266)	(0.0917)	(0.2454)

Notes: This table shows estimates from 45 regressions for the period 2008-2010. PYP is an indicator variable equal to one after February 6, 2009 and zero otherwise. Each estimate reports the PYP long-run coefficient of an alternative specification. The dependent variable is carbon monoxide (CO) in logs. All hours includes observations for 05:00-21:00 all days of the week; morning peak, evening peak, and off-peak restrict the sample to 06:00-09:00, 16:00-19:00, and 10:00-16:00, respectively, during weekdays. Weekend corresponds to the aggregated time slot 06:00-09:00 and 16:00-19:00 during weekends. All specifications include meteorological variables. Regressions are also fitted along a polynomial time trend of degree four for all hours model and degree three for the other models, and include indicator variables for month of the year, day of the week, and hour of the day. Interactions between weekends and hour of the day are added only in the all hours model. Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 7: Effect of initial and extended phases of PYP on gasoline consumption. Long-run equation

	(1) Gasoline consumption (CONS)	(2) Gasoline consumption (CONS)	(3) Gasoline consumption (CONS)
PYP1	0.0054	0.0082	0.0102
	(0.0210)	(0.0258)	(0.0469)
PYP2	0.1012***	0.0932**	0.1079*
	(0.0325)	(0.0363)	(0.0651)
PRICE	-0.1527*	-0.1784**	-0.1217
	(0.0831)	(0.0768)	(0.1402)
GDP per capita	0.6335***	0.6495***	0.7340**
	(0.0752)	(0.1572)	(0.2857)
Monthly dummies	No	Yes	Yes
Lags and leads of D.PRICE and D.GDP	No	No	Yes
N	177	177	176
R^2	0.9585	0.9725	0.9766
F	790.32	353.63	267.54
P-value	0.0000	0.0000	0.0000

Notes: This table shows estimates from four regressions for the period January 1996-September 2010. The dependent variable is gasoline consumption in logs (CONS). Gasoline price (PRICE) and GDP per capita are also in logs. D.PRICE and D.GDP are first differences of PRICE and GDP. PYP1 is an indicator variable equal to one from August 1998 to January 2009 and zero otherwise. PYP2 is an indicator variable equal to one after February, 2009 and zero otherwise. Specification (3) corresponds to the DOLS model. Regressions are fitted along a linear time trend. Conventional (Columns 1 and 2) and Newey-West 4 lags standard errors (Column 3) are reported in parentheses. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 8: Effect of initial and extended phases of PYP on gasoline consumption. Error correction model

	(1) Gasoline consumption (D.CONS)	(2) Gasoline consumption (D.CONS)	(3) Gasoline consumption (D.CONS)
PYP1	0.0031	0.0036	0.0048
	(0.0150)	(0.0114)	(0.0114)
PYP2	0.0116	0.0076	0.0091
	(0.0250)	(0.0190)	(0.0191)
D.PRICE	-0.1554	-0.4317***	-0.6219***
	(0.1707)	(0.1343)	(0.1283)
D.GDP per capita	0.8265***	0.5048*	0.4805*
	(0.0850)	(0.2625)	(0.2630)
L.ε	-0.4242***	-0.2778***	-0.2737***
	(0.0939)	(0.0727)	(0.0751)
N	176	176	176
R^2	0.6523	0.8128	0.8113
F	39.16	37.86	37.51
P-value	0.0000	0.0000	0.0000

Notes: This table shows estimates from three regressions for the period January 1996-September 2010. The dependent variable is the first difference of monthly gasoline consumption in logs (D.CONS). Gasoline price and GDP per capita are also in logs and first differences (D.PRICE and D.GDP per capita). Specifications (1), (2), and (3) are the ECM for models in columns (1), (2), and (3) of Table 7A, respectively. PYP1 is an indicator variable equal to one from August 1998 to January 2009 and zero otherwise. PYP2 is an indicator variable equal to one after February, 2009 and zero otherwise. L. is the first lag of the residual estimated from the long-run relationship among variables. Regressions are fitted along a linear time trend and including first lagged differences of CONS. Standard errors are reported in parentheses. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 9: Effect of initial and extended phases of PYP on vehicle registrations

	(1) Automobiles	(2) Private vehicles	(3) Load vehicles
PYP1	0.0566	0.0610	-0.2616
	(0.0804)	(0.0748)	(0.2923)
PYP2	0.1844**	0.1681*	0.0652
	(0.0919)	(0.0855)	(0.3298)
L1.	-0.4665***	-0.4441***	-0.6116***
	(0.0836)	(0.0837)	(0.0838)
L2.	-0.2643***	-0.2695***	-0.2460***
	(0.0825)	(0.0827)	(0.0835)
L12.	0.0662	0.0323	-0.0256
	(0.0776)	(0.0779)	(0.0718)
Long-run PYP2	0.0340	0.0363	-0.1389
	(0.0484)	(0.0447)	(0.1543)
Long-run PYP2	0.1108**	0.1000*	0.0346
	(0.0554)	(0.0511)	(0.1752)
N	154	154	154
R^2	0.73	0.77	0.37
F	19.82	24.72	4.41
P-value	0.0000	0.0000	0.0000

Notes: This table shows estimates from two regressions for the period 1997-2010. The dependent variable is the first difference of monthly vehicle registrations in logs. Automobiles corresponds to the sample only for automobiles; Private vehicles, in addition to automobiles, includes station wagons and sport utility vehicles; and Load vehicles corresponds to the sample of trucks. L1., L2., and L12. are the first, second, and twelfth autoregressive components. PYP1 is an indicator variable equal to one from August 1998 to January 2009 and zero otherwise. PYP2 is an indicator variable equal to one after February, 2009 and zero otherwise. Regressions are fitted along a quadratic time trend. Standard errors are in parentheses. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 10: Effect of extended phase of PYP on vehicle sales: Automobiles

	(1) Bogota OLS	(2) Bogota OLS	(3) Bogota 2SLS	(4) Other cities OLS	(5) Other cities OLS	(6) Other cities 2SLS
PYP2	0.0963*	0.1434***	0.1655*	0.0292	0.0249	0.0914
	(0.0499)	(0.0509)	(0.0942)	(0.0605)	(0.0638)	(0.1226)
GDP growth rate	0.0002			0.0088		
	(0.0060)			(0.0073)		
Price index		-0.7260**	-1.0684		0.3113	-0.7518
		(0.2946)	(1.2934)		(0.3709)	(1.7345)
L.D.	-0.5045***	-0.5221***	-0.5300***	-0.3424***	-0.3352***	-0.3224***
	(0.0916)	(0.0884)	(0.0877)	(0.0935)	(0.0935)	(0.0923)
L2.D.	-0.2415***	-0.2463***	-0.2485***	0.0010	0.0097	0.0035
	(0.0897)	(0.0873)	(0.0820)	(0.0923)	(0.0925)	(0.0895)
Long-run PYP2	0.0551*	0.0811***	0.0930*	0.0218	0.0188	0.0693
	(0.0284)	(0.0289)	(0.0514)	(0.0450)	(0.0480)	(0.0935)
N	132	132	132	132	132	132
R^2	0.50	0.53	0.52	0.42	0.41	0.37
F	6.79	7.51	7.10	4.83	4.75	4.40
P-value	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

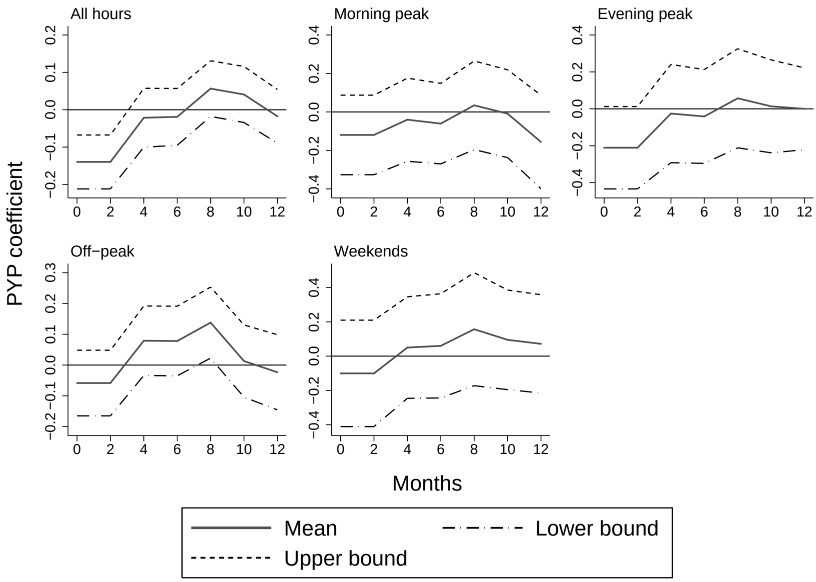
Notes: This table shows estimates from six regressions for the period July 2000-September 2011. The dependent variable is the first difference of monthly vehicle sales (automobiles) in logs. L.D. and L2.D. are the first and second lags of the dependent variable. PYP2 is an indicator variable equal to one after February 6, 2009 and zero otherwise. Regressions are fitted along a quadratic time trend. Standard errors are in parentheses. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 11: Pre-morning peak effect

Model with anticipated response: pre-morning peak of CO	
Variable	Coefficient
L.CO	0.3369***
	(0.0248)
PYP	0.1218*
	(0.0703)
Long-run PYP	0.1837*
	(0.1051)
N	373
R^2	0.84
AR stability	Yes

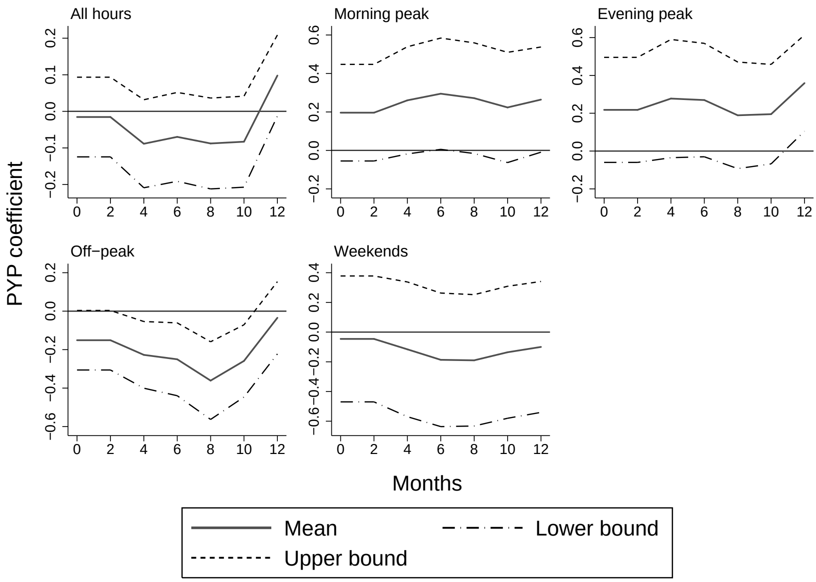
Notes: This table shows estimates from one regression for the period 1997-1999. The dependent variable is carbon monoxide in logs (CO). L.CO is the first lag of CO. PYP is an indicator variable equal to one after August 18, 1998 and zero otherwise. The specification includes weather variables such as wind speed, superficial temperature, relative humidity, and rainfall in quadratic form and their corresponding first lags, wind direction using the 8-point compass, and temperature inversion. Regressions are also fitted along a polynomial time trend of degree three and include indicator variables for month of the year and day of the week. Estimates correspond to Station 3 in the Southwest (one of the farthest Stations from the city center located in a middle income area). Standard errors, in parentheses, are robust to heteroskedasticity. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Figure 1: Short-run PYP effect during the initial phase of PYP



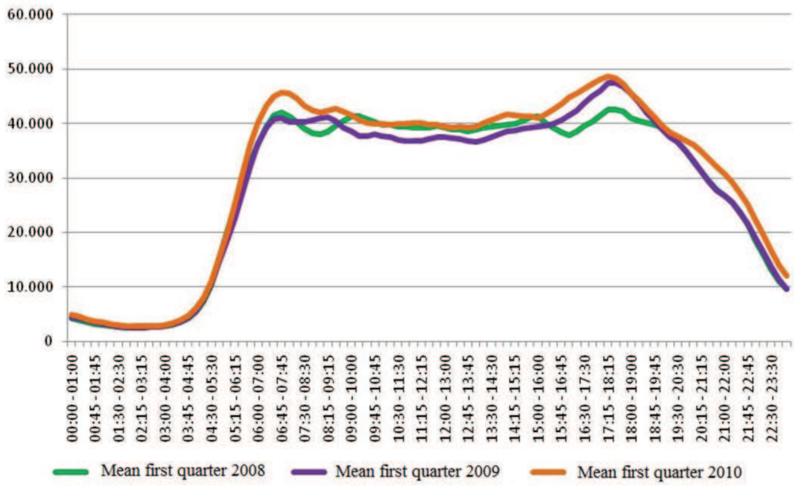
Notes: Dotted lines in the figure are the lower and upper bounds of the confidence interval at 95%.

Figure 2: Short-run PYP effect during the extended phase of PYP



Notes: Dotted lines in the figure are the lower and upper bounds of the confidence interval at 95%.

Figure 3: Mean traffic counts across several measurement stations



Source: Consorcio Monitoreo Transito y Transporte Urbano 2009 – SDM. April, 2010. See Secretaria de Movilidad (2010a).

Chapter II

Air Pollution Dynamics and the Need for Temporally Differentiated Road Pricing

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Abstract

Nowadays, road traffic is a major source of urban air pollution. In this paper we investigate the effects of the temporal variation of pollution dispersion, traffic flows and vehicular emissions on pollution concentration and illustrate the need for temporally differentiated road pricing through an application to the case of the congestion charge in Stockholm, Sweden. By accounting explicitly for the role of pollution dispersion on optimal road pricing, we allow for a more comprehensive view of the economy-ecology interactions at stake, showing that price differentiation is an optimal response to the physical environment. Most congestion charges in place incorporate price bans to mitigate congestion. Our analysis indicates that, to ensure compliance with air quality standards, such price variations should also be a response to limited pollution dispersion.

JEL Classification: D62, R41, C54, Q53, Q57

Keywords: road pricing, congestion, air pollution, pollution dispersion.

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1. Introduction

Despite the efforts to reduce urban air pollution in many places around the world, the magnitude of the impacts of air pollution on human health remains substantial. For instance, it is estimated that, just in Europe, exposure to urban pollution will cause a loss of 210 million life-years and 18 000 premature deaths (1, 2) in 2030. In the past, industrial activities and domestic heating were major drivers of poor air quality. Nowadays the major source of urban air pollutants is road traffic (3).

Facing the need for air quality improvement, policymakers must decide on air quality standards (hereinafter AQSs) and the regulatory instruments to meet these. In some cases, the damages caused by emissions of pollutants are more or less directly proportional to emission rates and the number of persons being exposed. However, under other circumstances, the possibility of reaching environmental targets depends not only on the emission rates of pollutants, but also on the assimilative capacity of the environment. This is defined as the capacity of an environment to cleanse itself after receiving a determined level of emissions, by degrading or dispersing the emissions and converting them into substances that are harmless to humans or ecosystems. In the case of urban air pollution, the assimilative capacity is mainly driven by the meteorological factors that govern air mixing and thus dispersion of the pollutants. Due to the large temporal variation of these meteorological factors, there is a strong day-to-day and diurnal variation in the assimilative capacity, in addition to the variation in hourly traffic flows and vehicular emissions (4, 5, 6, 7, 8).

In this paper we investigate the effects of the temporal variation of the assimilative capacity on pollution concentration and illustrate the need for temporally differentiated road pricing through an application to the case of the congestion charge in Stockholm, Sweden.

To the best of our knowledge, this is the first study analyzing how the variation in air mixing and pollution dispersion should be accounted for in road pricing. Most previous studies of urban road pricing have focused on its role in relieving congestion. Only recently, attention has turned toward using road pricing as an instrument to improve air quality (9, 10, 11, 12). By accounting explicitly for the role of the assimilative capacity on optimal road pricing, we allow for a more comprehensive view of the economy-ecology interactions at stake, showing that price differentiation is an optimal response to the physical environment; a higher charge at certain times of the day are optimal because they discourage traffic when the assimilative capacity is constrained. Furthermore, our results are of practical relevance, since most congestion charges in place resemble variable price schemes as they incorporate price bans to mitigate congestion.[1] Our analysis indicates that such price variations should also be a response to limited air mixing resulting in reduced assimilative capacity.

This paper is organized as follows. Section 2 describes general patterns of air pollution in some cities around the world. Section 3 introduces a congestion charge that takes into account the role and dynamics of the assimilative capacity in Stockholm (Sweden), where a congestion charge was introduced as a trial in 2006 and permanently in 2007. We start out from the scheme currently in place and look for modifications that would be consistent with the AQSs for nitrogen dioxide (NO_2) and particulate matter (PM_{10}) concentrations (hereinafter denoted as [NO_2] and [PM_{10}]) and the meteorological factors that govern air mixing and dispersion of air pollutants through the day. Finally, Section 4 discusses the main results and concludes the paper.

2. Temporal variation of air pollution and assimilative capacity

Understanding urban air pollution due to road transportation is complex because several factors affect pollutants' dynamics and air quality. Air pollution levels from road transportation depend on polluters' type and number, meteorology, topography including the arrangement of houses along streets and traffic routes, driving patterns, and sectorial policies in place. Because the levels and particular causes of pollution vary from one city to another, we present a sample of cities around the world to highlight the main patterns of diurnal variation of air pollution. Our choice is based on availability of data and the importance of traffic-related air pollution: it includes heavily polluted as well as cleaner cities. As shown in Figure 1, we display the diurnal patterns of [NO_2] and [PM_{10}], and traffic flows in Stockholm (Sweden), London (United Kingdom), and Santiago (Chile).

Regarding our choice of cities, Stockholm is ranked as one of the cleanest large cities in Europe (15). Nevertheless, air pollution due to road transportation continues to pose a threat to human health; long-term exposure to local traffic-related air pollution has been shown to contribute to lung cancer, respiratory and heart diseases, and allergies (16, 17). The significant contribution of road transport to air pollution in London is widely acknowledged. Official government figures revealed, for instance, that in 2011 traffic was responsible for approximately 60% of the total emissions of nitrogen oxide (NO_x), PM_{10} and $PM_{2.5}$ within the City of London (18). Finally, Santiago is one of the Latin American capitals with the worst air quality, and the transportation sector is estimated to account for over 86% and 75% of PM_{10} and NO_2 emissions, respectively (19).

[1] Congestion charges can be classified as uniform (charge is constant over the entire application period), quasi-uniform (charge is constant over a specific time period and zero otherwise) and variable (charge is time-varying). See (13) and (14) for further discussion.

For all cities, we used hourly data on pollutant concentrations from the national Air Quality Networks to describe the average daily profiles of concentrations. In each case, we selected monitoring stations located in central areas that are most affected by vehicular traffic (see supplementary material for further details). To explore the relationship between traffic and pollution, we plotted the flow of vehicles per hour in the traffic monitoring station closest to the pollution monitoring station (Figure 1).

Some interesting patters appear:

Hourly [NO_2] and [PM_{10}] vary significantly during the day, as does traffic flows. In our sampled cities, traffic flows exhibit two clear peaks (one in the morning and one in the afternoon). Whether pollution peaks occur in hours of peak traffic depends on the pollutant under analysis. For example, [NO_2] is clearly related to traffic flows, while [PM_{10}] tends to peak after the first traffic peak has occurred.

Nevertheless, despite [NO_2] being strongly influenced by traffic, it is not perfectly correlated to traffic peaks. For instance, [NO_2] responds quickly to changes in the vehicle fleet in the morning (i.e. the concentration peak is reached within one hour after the traffic peak) while it responds more slowly and to a smaller extent in the afternoon. Interestingly, the traffic peak in the afternoon (in most cases larger than the morning traffic peak) is associated with a much lower level of [NO_2]. The ratio [NO_2] morning/afternoon is approximately 1.5 in Santiago, 1.12 in London and 1.25 in Stockholm.[2]

The correlation between traffic flows and [PM_{10}] is less clear than in the case of [NO_2]. This might be explained by the fact that a fraction of PM_{10} emissions comes from other sources, e.g. road dust, construction work, pollen material, spores and sea spray. These sources are subject to seasonal variation. Suspended pollen material contributes mainly during the growing season. Particles originating from road dust increase during dry conditions and high wind speeds. In Sweden, this is especially important during spring, when roads dry up and big deposits of grit from the winter remain on the roads (24). Moreover, [PM_{10}] might be affected by meteorological conditions that do not exhibit a clear pattern during the day, such as precipitation that reduces airborne particles and flushes away PM_{10} emissions. Nevertheless, from the figures, we can still observe that, for the cities included, [PM_{10}] closely follows the pattern of traffic when the traffic is building to the peak. In addition, morning traffic peaks coincide with peaks of [PM_{10}], but the afternoon traffic peak shows a comparatively lower level of [PM_{10}]. This could be explained by the diurnal wind pattern. Higher wind speeds in the afternoon tend to disperse the emitted particles better, resulting in a lower [PM_{10}].

[2] Similar evidence is found by (20) for Los Angeles (United States), by (21) for Beijing (China), by (22) for Toronto (Canada), and by (23) for Seoul. Higher relative concentrations of pollutants occur in the morning than in the afternoon, reflecting the diurnal variations of mixing depths and wind speeds.

Alternatively, the contribution of non-traffic related sources is larger during off-peak traffic hours, which would explain for, example, the high [PM_{10}] in Stockholm at midday when traffic is at a dip.

The imperfect correlation between pollution concentrations and traffic flows observed for our cities is consistent with scientific literature reporting that the effect of road transportation emissions on concentrations is critically affected by changes in the assimilative capacity, which is heavily influenced by meteorological conditions (25, 26, 27). For instance, anticyclonic weather and temperature inversions near the ground favor high pollutant concentrations. During these conditions, a stagnant layer of air traps air pollutants near the ground, impeding pollutant dispersion (28).

What are the implications of the strong variation in dispersion/assimilative capacity for optimal road pricing? The optimal management of a pollutant requires policies that restrict emissions when they cause the most damage, while inducing behavioral adjustments leading to relatively higher emissions when the assimilative capacity of the environment is larger. In the following section, we illustrate the dynamics of a congestion charge consistent with the variation in assimilative capacity in the case of Stockholm, Sweden. We focus on [NO_2] and [PM_{10}] with current AQSs at 90 $\mu g/m^3$ per hour and 50 $\mu g/m^3$ per day, respectively. The [NO_2] standard should not be violated more than 175 times per year (2% of the time). In the case of [PM_{10}], the standard can be exceeded no more than 35 times per year (10% of the time). Despite the implementation of the congestion charge, NO_2 and PM_{10} are the urban air pollutants for which concentrations allowed under AQSs are regularly exceeded along densely trafficked streets of Stockholm (30).

3. Differentiated Road Pricing: An Application to the Case of Stockholm

Our methodology is based on the following steps. Firstly, using data for Stockholm during the period 2002-2010, we estimate the empirical relationship between [NO_2], [PM_{10}], traffic flows and assimilative capacity per hour and season. In line with previous studies, we use wind speed as a proxy of assimilative capacity (1, 30, 31). Secondly, we use the estimates above to forecast the traffic flow per hour (and season) consistent with compliance of the Swedish AQSs for [NO_2] and [PM_{10}]. Finally, we estimate the responsiveness of the traffic flow across the Stockholm's cordon with respect to the congestion charge (i.e., price elasticity) to calculate the variation in the charge needed to bring the actual level of traffic flow to the level of traffic flow needed to comply with the AQSs calculated in the second step. Thus, we used predicted and observed traffic flows in the cordon, and the elasticity of traffic to the congestion charge as inputs to obtain the time-varying congestion charge consistent with the AQSs.

From a theoretical point of view, it would be optimal to implement a time-varying congestion charge that takes account not only of variation of the assimilative capacity throughout the day but also across days. However, from a practical point of view, the main purpose of the congestion charge is to ensure that in deciding when or whether to travel or by which route, travellers take account of the costs that their travel choice will impose on others. If the congestion charge is to fulfill this price signaling function, it is essential that it is known accurately by each traveller before they make their travel choices. For this reason, and to keep the administrative costs low, it may be desirable to maintain the daily time-varying pattern for a certain part of the year.

We chose to develop our estimations per season because it allows us to characterize major variations in the assimilative capacity due to seasonal variation in meteorological conditions, while maintaining the potential for policy implementation of our results.

This section is organized as follows. Firstly, we briefly describe the main features of the congestion charge in place and its effect on reducing congestion and pollution to date. Secondly, we present the model employed to characterize the dynamics of assimilative capacity during the day and its effects on pollution concentration. Thirdly, we compute our estimates of the time-varying charge that account for variations in the assimilative capacity.

3.1 Stockholm's Congestion Charge

The congestion charge in Stockholm is a cordon toll system that surrounds the entire city center, with a total area of approximately 35.5 km^2. It was implemented on 1 August 2007 with the purpose of reducing both traffic congestion and emissions. It is incurred both at entry into and exit from any of the 18 entry and exit points between 6.30 and 18.30. It has three price bands, depending on time of the day, which makes it a variable price scheme. The charge is higher when congestion is expected to build up to a peak and lower at other times of the day. Thus, the cost of passing the cordon on week days is SEK 20 (approx. € 2) during peak hours (7:30–8:30, 16:00–17:30), SEK 15 during the shoulders of the peaks (30 minutes before and after peak periods: 7:00-7:30, 8:30-9:00, 15:30-16:00 and 17:30-18:00) and SEK 10 during the rest of the period (6:30-7:00 and 9:00-15:30). The charge is levied in both directions, and the maximum total charge per day is SEK 60. There is no charge in the evening or at night; on Saturdays, Sundays, public holidays or the day before such a holiday; or during the month of July.

Vehicles are registered and identified automatically by taking a photograph of their number plate, which is analyzed using Optical Character Recognition technology. Payment is made monthly after an invoice is sent to the registered owner of the vehicle.

The charge has been in place for more than five years, and measurement of the effects has been extensive (32, 33, 34, 35, 36, and 37). Studies have shown that traffic over the taxed cordon was reduced significantly after the permanent implementation of the charge (approximately 18.75% over the period 2008-2011). The congestion charge also proved to have a significant effect in reducing journey times: queuing times were reduced to half during morning peak in the inner road system subject to taxation, whereas there was no increase over the 2007 and 2008 traffic levels in the inner city, suggesting that there was no tendency to increase use of road space free from congestion charge. Moreover, emissions from traffic were also reduced (the estimated reductions of air-borne pollutants inside the cordon varied from 10% to 14%). The number of traffic accidents was reduced as well, contributing to an increase in traffic safety, both in terms of fatal accidents and severe injury accidents (34, 35). [3]

3.2 The Dynamics of the Assimilative Capacity

Using data for Stockholm, we estimate separately the relationship between assimilative capacity per season and $[NO_2]$ and $[PM_{10}]$ as:

(6) $$C_t = k_1 + \frac{k_2 F_t}{A_t^{k_{3t}}}.$$

where C_t, represents $[NO_2]$ or $[PM_{10}]$; F_t and A_t are traffic flow and wind speed, respectively, at time t; k_1 and k_2 are two time-invariant constants; and k_3 is a time-variant (by hour of the day) constant that accounts for the role of wind speed dispersing pollution at different rates throughout the day. k_1 represents the background concentration, which is not affected by local traffic. k_2 is a proportionality constant which assumes a linear relationship between traffic flow and the contribution to air pollution concentration by local traffic. A_t^{k3t} represents the assimilative capacity. It can be shown that urban air pollution concentration above a background level (represented by k_1) is strongly and negatively related to wind speed (26, 31) and that the relationship is strongly non-linear. Rising wind speed (A_t) to the power of k_{3t} allows for this non-linearity. Moreover, the probability that concentrations will exceed the AQSs depends strongly on both traffic flows and wind speed. As seen in Figure 2, for the highest wind speed interval (i.e., wind speed larger than 8 m/s), the probability of exceeding the concentration allowed under the AQS for $[NO_2]$ is almost zero, even for large traffic flows. For $[PM_{10}]$, the

[3] There is some evidence of the morning peak of traffic spreading toward a later departure time due to the charge (36). Changes in departure time are important in the context of equity and redistribution, as the poor may be less flexible in departure time than the rich. This aspect is, however, beyond the scope of our study.

relationship is less clear, since, at low wind speeds, the small combustion particles are likely to be predominant, while the fraction of dust particles will increase at higher wind speeds.

[PM_{10}] is also sensitive to precipitation, which the model does not take into consideration (25, 26).[4] Nevertheless, a statistical analysis of the daily profile of precipitation in Stockholm indicates a random pattern over the day within each season, with most of the variation occurring across seasons. Thus, though precipitation might affect the level of pollution concentration (which is captured through k_1), it does not explain its variation throughout the day.

We plot the observed and predicted hourly means of [NO_2] and [PM_{10}] along the traffic flows and wind speed for each season in Figure 3 and 4. As can be seen from the figures, despite its simplicity, our model succeeds in replicating the level and rate of variation of pollution concentration throughout the day in Stockholm.

3.3 Assimilative Capacity and Time-Varying Charges in Stockholm

By using equation (6), we solve for the traffic flows consistent with the air quality standards of [NO_2] and [PM_{10}] as follows[5]:

$$(7) \qquad F_t = \frac{A_t^{k_3 t}\left[C_t^{AQS} - k_1\right]}{k_2}.$$

Note that in order to forecast the traffic flow, we need to feed our formula with the parameters estimated in the previous section, and the mean values of wind speed for each hour of the day per season. Because the prediction is based on the mean hourly value of wind speed, this method does not take account of the variability of wind speed between days and its effects on pollution concentration. To ensure that the traffic flows are consistent with concentrations that do not exceed permissible levels under AQSs, we estimate the empirical relationship between the pollutants' concentrations predicted by our model and the actual fraction of hours with [NO_2] larger than 90 $\mu g/m^3$ and [PM_{10}] larger than

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[4] Precipitation and mixing layer height seem to be the meteorological variables influencing near-surface [PM_{10}] most significantly within cities (26, 31); the absence of precipitation and low values of the mixing layer height lead to comparatively high PM_{10} levels. Because wind speed normally varies with height, in this study we use the average wind speed through the mixing depth as a convenient representation of the horizontal transport of air within the mixing layer.

[5] Note that our analysis is performed on an hourly basis. In the case of [NO_2], this is consistent with the time frame of the AQS, which is defined as a maximum concentration per hour. In the case of [PM_{10}], however, the standard is defined as a 24 hour average. There are many profiles of concentrations per hour consistent with such a standard. To keep the analysis simple, we only focus on the case where the hourly concentration of PM_{10} does not violate the standard at any hour of the day. This precautionary approach is nevertheless consistent with evidence from toxicological and clinical studies showing that peak exposures of short duration to PM_{10} (ranging from less than an hour up to a few hours) lead to immediate physiological changes (2).

50 μg/m^3. These empirical distributions are presented in Figure 5. They indicate that to ensure that [NO_2] and [PM_{10}] AQSs are exceeded no more than 2% and 10% of the time, respectively, one has to target a much lower average hourly concentration than the concentration imposed by the AQSs. Notably, the mean [NO_2] and [PM_{10}] should be equal to 55 μg/m^3 and 35 μg/m^3 to avoid exceeding of AQSs (Figure 5). This is to say that in our estimates C_t^{AQS} takes a value equal to 55 μg/m^3 and 35 μg/m^3 for [NO_2] and [PM_{10}] respectively.

Our next step is to calculate the difference between the actual traffic flow (F_t) and traffic flow per hour consistent with the AQS (F_t^{AQS}) by using historical records and the estimates computed through equation (7). For each pollutant, this difference is defined as $\Delta F_t = \frac{F_t - F_t^{AQS}}{F_t}$.

For those hours where $\Delta F_t > 0$, the actual traffic flow should be reduced through an increased congestion charge, while the reverse holds when $\Delta F_t < 0$. To what extent should the charge be increased or reduced? In order to provide an answer to this question, we estimate the price elasticity of traffic flows to the congestion charge η_t. We estimate η_t for three time windows: morning (06:30-09:00), off-peak period (09:00-15:30) and afternoon (15:30-18:30) to take account of the fact that the responsiveness to the charge might vary throughout the day. This yields elasticity estimates of -0.58, -1.14 and -0.89, respectively.

By using these estimates, for each pollutant we compute the difference between the actual congestion charge (P_t) and the congestion charge representing no concentrations exceeding permissible levels (P_t^{AQS}) as $\Delta P_t = \Delta F_t / \eta_t$, where $\Delta P_t = \frac{[P_t - P_t^{AQS}]}{P_t}$.

Finally, note that the daily profiles for [NO_2] and [PM_{10}] differ considerably, leading to a very different time varying congestion charge. To account for this, we take a precautionary approach and compute the (final) charge as the value needed to comply with both AQSs, i.e. $P_t^{AQS} = max\left[P_t^{AQS_{NO_2}}, P_t^{AQS_{PM_{10}}}\right]$. The results are presented in Figure 6.

Figure 6 displays the combined effect of different traffic flows per hour and season, hourly and seasonal dynamics of [NO_2] and [PM_{10}], and different price elasticities per time of the day. It indicates that, with regard to the current time-varying schedule, the achievement of AQSs in Stockholm would require the charge to be increased for all seasons and most hours of the day. In relative terms, a much larger increase is needed in spring; the increment should be also larger in the morning to offset the negative effect of reduced assimilative capacity on pollution concentration. The increment in the charge should be smaller for the remaining seasons due to increased pollution dispersion resulting from more favorable meteorological conditions and, in the case of summer, much reduced traffic flows. Moreover, the charge should be of equivalent magnitude during morning and afternoon traffic peaks. Despite the increased assimilative capacity in the afternoon, a large charge is needed to reduce larger traffic flows at this time of the day.

The seasonal dynamics of [PM_{10}] is the main driver behind the need for an increased charge in spring, while [NO_2] explains the need for an increased charge for the remaining seasons. The need for a larger time-varying congestion charge in spring coincides with an increased contribution of PM_{10} emissions by other sources, such as suspended road dust and pollen fragments. Our results indicate that, in order to achieve the same AQS for [PM_{10}], a much larger contribution to reductions from road transportation is required to compensate for the increased emissions coming from other sources.

4. Discussion

Considering the urgency of improving air quality in many countries around the world, it is important to use environmental policy instruments that restrict emissions when they cause the most damage. In this paper, we suggest a congestion charge that takes account of the economy-ecology interactions at stake in Stockholm, Sweden. A distinguishing feature of our analysis is that the time-varying charge depends on the rate at which pollution is dispersed. Our results indicate that, in order to achieve the AQSs, the congestion charge should be increased for all seasons and most hours of the day. Moreover, there is important variation in the level of the time-varying charge per season; the level should be larger in spring due to less favorable meteorological conditions for pollution dispersion and increased contribution from pollution sources other than vehicle exhaust. Regarding the daily profile, during spring a larger charge is also needed in the mornings to offset the negative effects of the limited assimilative capacity. However, the diurnal profile of the suggested congestion charge resembles the existing charge to a large extent during the remaining seasons.

In our analysis, we have modeled the dynamics of the relationship between pollution concentration and traffic flows in the specific case of Stockholm. Nevertheless, we believe that the basic principles and the methodology developed in this paper could be easily adapted to other cities. In some cases, this might require inclusion of further meteorological variables in the analysis, for example precipitation in the case of [PM_{10}] where wind speed has a strong diurnal or seasonal pattern. Even if this might impose additional challenges, such information is available for most countries.

Materials and Methods

Information on hourly pollutant concentrations, wind speed and traffic flow was obtained from the SLB-analysis unit of Stockholm City's Environment and Health Administration. [NO_2] and [PM_{10}] are obtained from the monitoring site Hornsgatan located at ground level on the south side of the street in a typical inner city environment that is highly influenced by traffic, with approximately 30 000 vehicles per workday. Wind speed measurements are collected at the monitoring station Torkel Knutssonsgatan, located around 100 m south of the street Hornsgatan. Traffic counts correspond to the total traffic for four lanes, and are measured on Hornsgatan adjacent to the Hornsgatan air quality station between the streets Ringvägen and Anskariegatan.

We estimate equation (6) using Nonlinear Least Squares (NLS) for each pollutant and season of the year during workdays. Our estimates are reported with heteroscedasticity and autocorrelation consistent Newey-West standard errors for 24 lags of the diurnal profile for concentrations. As expected, the coefficients for background concentrations and traffic volume are positive and statistically significant (see Appendix B for estimates of the parameters for the regressions for [NO_2] and [PM_{10}] respectively). Moreover, for both pollutants, several of the time-variant coefficients associated with wind speed are statistically significant, which happens at times when the highest concentrations or concentrations exceeding permissible levels occur. The variation in size of these coefficients also supports the presence of different effects of wind dispersing pollutants throughout the day.

The overall performance of the model was also statistically evaluated using the Likelihood Ratio Test (LR). We also performed LR tests of the null hypothesis that k_3 is a time-invariant coefficient versus the alternative hypothesis that k_3 varies over time as in specification (6). The tests reject the hypothesis of no temporal variation in the influence of wind speed on assimilative capacity during day, at the 1% significance level. Moreover, plots of the predicted and observed hourly means of [NO_2] and [PM_{10}] show a very high goodness of fit with an R-squared of around 99%, pointing out that the model satisfactorily captures and reproduces the diurnal pattern of concentrations.

Elasticity to the Congestion Charge: In calculating the elasticity, we follow the methodology by (32), who computes the elasticity in response to the congestion charge as $\eta_i = \frac{\ln(F^i_{2010}/F^i_{2005})}{\ln(P^i_{2010}/P^i_{2005})}$, where ($F^i_{2010}, P^i_{2010}$) and ($F^i_{2005}, P^i_{2005}$) are the traffic flows of non-exempt traffic[6] and total travel costs in

[6] In 2007, the charge became tax deductible for those commuters travelling more than 5 km and who could save at least one hour compared to public transportation one way; this applied to approximately 8% of all car trips across the cordon. They were granted a 60% charge reduction. Moreover, ‘‘company cars’’ (i.e., company-owned cars used by an employee for both work-related and private purposes), which account for 23% of all

real terms in years 2010 and 2005, respectively, at time period i, i.e., morning (6:30-9:00), off-peak (9:00-15:30) and afternoon (15:30-18:30). The overall average traffic reductions of non-exempt traffic during the morning, off-peak and afternoon between the years 2010 and 2005 were equivalent to 25.2%, 32.5% and 37.2%, respectively. Following (31), we assume that the mean length of trips crossing the cordon was 13 km, as indicated by surveys carried out before and after the implementation of the congestion charge. The average total trip costs in real terms for 2010 corresponded to 32.3, 27.5 and 32.8 SEK, respectively, in contrast to a value of 19.5 SEK in 2005.

vehicles crossing the charging cordon, receive at least a 60% reduction of the congestion charge, and about 20% of all company cars pay no charge at all (32).

References

(1) International Institute for Applied Systems Analysis IIASA. 2012. TSAP-2012 Baseline: Health and Environmental Impacts. Report available at: http://www.iiasa.ac.at/web/home/research/researchPrograms/MitigationofAirPollutionandGreenhousegases/TSAP-_IMPACTS-20121126.pdf

(2) World Health Organization - European Centre for Environment and Health. 2013. Review of evidence on health aspects of air pollution – REVIHAAP". Report available at: http://www.euro.who.int/__data/assets/pdf_file/0020/182432/e96762-final.pdf

(3) Goyal, P., S. Anand, and B.S. Gera. 2006. "Assimilative capacity and pollutant dispersion studies for Gangtok city." *Atmospheric Environment* 40: 1671-1682.

(4) Hayas, A., C.F. Gonzalez, G. Pardo, and M.C. Martinez. 1981. "Diurnal variation of particulate pollution of the atmosphere in an urban area." *Atmospheric Environment* 15(3): 411–413.

(5) Viana, M., C. Pérez, X. Querol, A. Alastuey, S. Nickovic, and J.M. Baldasano. 2005. "Spatial and temporal variability of PM levels and composition in a complex summer atmospheric scenario in Barcelona (NE Spain)." *Atmospheric Environment* 39: 5343–5361

(6) Mikhailuta, S.V., O.V. Taseiko, A. Pitt, A.A. Lezhenin, and Y.V. Zakharov. 2009. "Seasonal variations of air pollutant concentrations within Krasnoyarsk City." *Environ Monitoring Assessment* 149(1-4): 329-341.

(7) Toth, I., P. Hercog, R. Peternel, and B. Vojniković. 2011. Seasonal, daily and intradiurnal variation of PM10, NO2, NO and O3 in residential part of Zagreb, Croatia. Collegium antropologicum 35(2):25-30.

(8) Kim, W., H. Lee, J. Kim, U. Jeong, and J. Kweon. 2012. "Estimation of seasonal diurnal variations in primary and secondary organic carbon concentrations in the urban atmosphere: EC tracer and multiple regression approaches." *Atmospheric Environment* 56: 101–108.

(9) Verhoef, E. 2000. "The implementation of marginal external cost pricing in road transport. Long run vs. short run and first-best vs. second-best." Papers in Regional Sciences 79(3): 307-332.

(10) Tsekeris, T. and S. Vob. 2009. "Design and evaluation of road pricing: state-of-the-art and methodological advances." *Netnomics* 10:5-52.

(11) Anas, A. and R. Lindsey. 2011. "Reducing Urban Road Transportation Externalities: Road Pricing in Theory and in Practice." Review of Environmental Economic Policy 5 (1): 66-88.

(12) Johansson-Stenman, O. 2006. "Optimal environmental road pricing." *Economic Letters* 90(2):225-229.

(13) Wie, B. and R.L. Tobin. 1998. Dynamic Congestion Pricing Models for General Traffic Networks. Transportation Record B 32(5): 313-327.

(14) Joksimovic, D., M.C. Bliemer, and P.H.L. Bovy. 2005. “Optimal toll design problems in dynamic traffic networks with joint route and departure time choice.” Transportation Research Record 1923, 61-72.

(15) Aphekom. 2011. Summary report of the Aphekom project 2008-2011. Improving Knowledge and Communication for Decision Making on Air Pollution and Health in Europe.
Report downloadable from: www.aphekom.org/c/document_library/get_file?uuid=5532fafa-921f-4ab1-9ed9-c0148f7da36a&groupId=10347

(16) Nyberg, F., P. Gustavsson, L. Järup, T. Bellander, N. Berglind, R. Jakobsson, and G. Pershagen. 2000. “Urban Air Pollution and Lung Cancer in Stockholm.” *Epidemiology* 11(5): 487-495.

(17) Forsberg, B. and K. Sjöberg. 2005. Quantification of deaths attributed to air pollution in Sweden using estimated population exposure to nitrogen dioxide as an indicator. Report by the Swedish Environmental Research Institute available at:
http://www.imm.ki.se/Datavard/PDF/B1648_IVL_Ume%C3%A5_NO2%20och%20antal%20d%C3%B6da.pdf

(18) CERC (Cambridge Environmental Research Consultants). 2011. Source Apportionment for the City of London Corporation. Report available at:
http://www.cityoflondon.gov.uk/business/environmental-health/environmental-protection/air-quality/air-quality-reports/Documents/source%20apportionment%2015Jun11.pdf

(19) Bell, M. L., D.L. Davis, N. Gouveia, V.H. Borja Aburto, and L. Cifuentes. 2006. “The avoidable health effects of air pollution in three Latin American cities Santiago Sao Paulo and Mexico City.” *Environmental Research* 1: 431-44.

(20) Holzworth, G.C. 1967. “Mixing Depths, Wind Speeds and Air Pollution Potential for Selected Locations in United States.” *Journal of Applied Meteorology* 6: 1039-1044.

(21) Wang, T. and S. Xie. 2009. “Assessment of traffic-related air pollution in the urban streets before and during the 2008 Beijing Olympic Games traffic control period.” *Atmospheric Environment* 43(35):5682-5690.

(22) Arain, M.A., R. Blair, N. Finkelstein, J.R. Brook, T. Sahsuvaroglu, B. Beckerman, L. Zhang, and M. Jerrett. 2007. “The use of wind fields in a land use regression model to predict air pollution concentrations for health exposure studies.” *Atmospheric Environment* 41(16): 3453–3464.

(23) Kim, W., H. Lee, J. Kim, U. Jeong, and J. Kweon. 2012. “Estimation of seasonal diurnal variations in primary and secondary organic carbon concentrations in the urban atmosphere: EC tracer and multiple regression approaches.” *Atmospheric Environment* 56(1): 101-108.

(24) Johansson, C., M. Norman, and L. Gidhagen. 2007. “Spatial & temporal variations of PM_{10} and particle number concentrations in urban air.” *Environmental Monitoring and Assessment* 127(1-3): 477-487.

(25) Holst, J., H. Mayer, and T. Holst. 2008. “Effect of meteorological exchange conditions on PM10 concentration.” *Meteorologische Zeitschrift* 17(3): 273-282.

(26) Rost, J., T. Holst, E. Sähn, M. Klingner, K. Anke, D. Ahrens, and H. Mayer. 2009. “Variability of PM10 concentrations dependent on meteorological conditions.” *International Journal of Environment & Pollution* 36(1-3): 3 – 18.

(27) Jones, A.M., R.M. Harrison, and J. Baker. 2010. “The wind speed dependence of the concentrations of airborne particulate matter and NO_x.” *Atmospheric Environment* 44: 1682-1690.

(28) Grundström, M., H.W. Linderholm, J. Klingberg, and H. Pleijel. 2011. “Urban NO2 and NO pollution in relation to the North Atlantic Oscillation NAO.” *Atmospheric Environment* 45(4): 883-888.

(29) Johansson, C., L. Burman, and B. Forsberg. 2009. “The effects of congestions tax on air quality and health.” *Atmospheric Environment* 4: 4843-4854.

(30) Goyal, P. and C.V. Chalapi Rao. 2007. “Assessment of atmospheric assimilation potential for industrial development in an urban environment: Kochi (India).” *Science of the Total Environment* 376: 27–39.

(31) Jones, A. M., R. M. Harrison, and J. Baker. 2010. “The wind speed dependence of the concentrations of airborne particulate matter and NOx.” *Atmospheric Environment* 44: 1682-1690

(32) Börjesson, M., J. Eliasson, M. B. Hugosson, and K. Brundell-Freij. 2012. “The Stockholm congestion charges—5 years on. Effects, acceptability and lessons learnt.” *Transport Policy* 20(1): 1-12.

(33) Eliasson, J. 2008. “Lessons from the Stockholm Congestion Trial.” *Transport Policy* 15(6): 395-404.

(34) Eliasson, J. 2009. A cost-benefit analysis of the Stockholm congestion charging system. Transportation Research Part A: Policy and Practice 43 (4): 468-480.

(35) Eliasson, J., L. Hultkrantz, L. Nerhagen, and L. Smidfelt Rosqvist. 2009. The Stockholm congestion - charging trial 2006: Overview of effects. Transportation Research Part A: Policy and Practice 43(3): 240-250.

(36) Karlström, A. and J.P. Franklin. 2009. Behavioral adjustments and equity effects of congestion pricing: Analysis of morning commutes during the Stockholm Trial. Transportation Research Part A 43: 283-296.

(37)Kristofferson, I. 2011. Impacts of time-varying cordon pricing: Validation and application of mesoscopic model for Stockholm. Forthcoming in Transport Policy.

Figure 1: Hourly patterns of air pollution in Stockholm, London and Santiago.

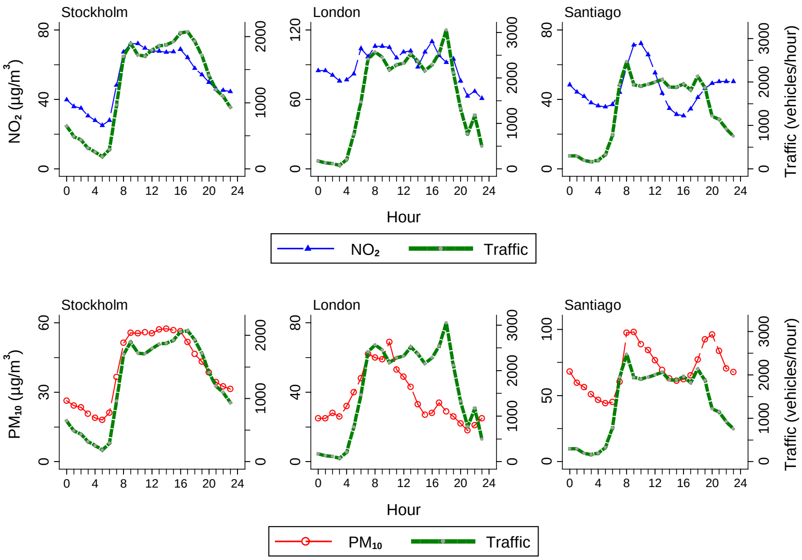


Figure 2: Traffic Flows, Wind Speed and Probability of Exceeding AQS for $[NO_2]$ and $[PM_{10}]$ AQSs

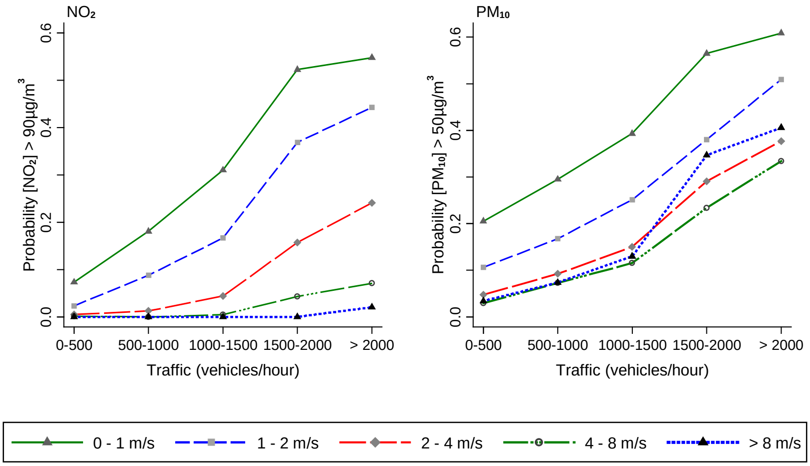


Figure 3: Diurnal profile of observed and fitted [NO_2], traffic flows and wind speed by season

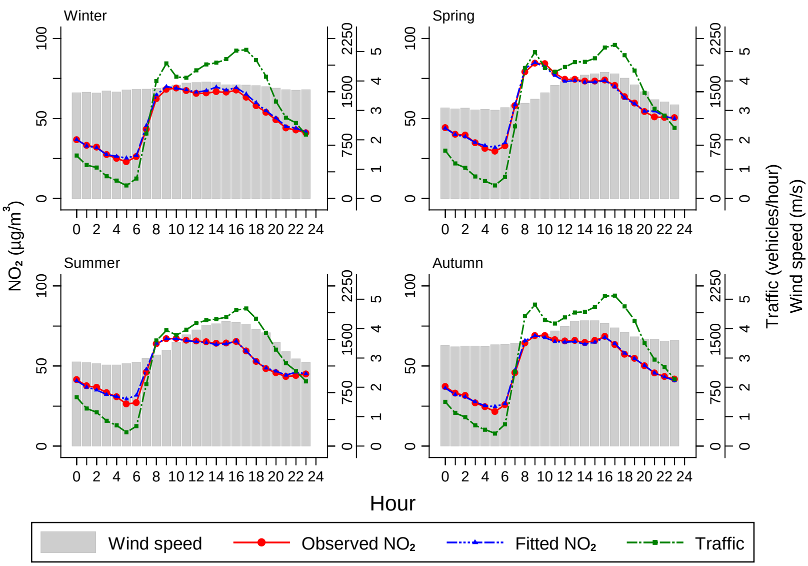


Figure 4: Diurnal profile of observed and fitted [PM_{10}], traffic flows and wind speed by season

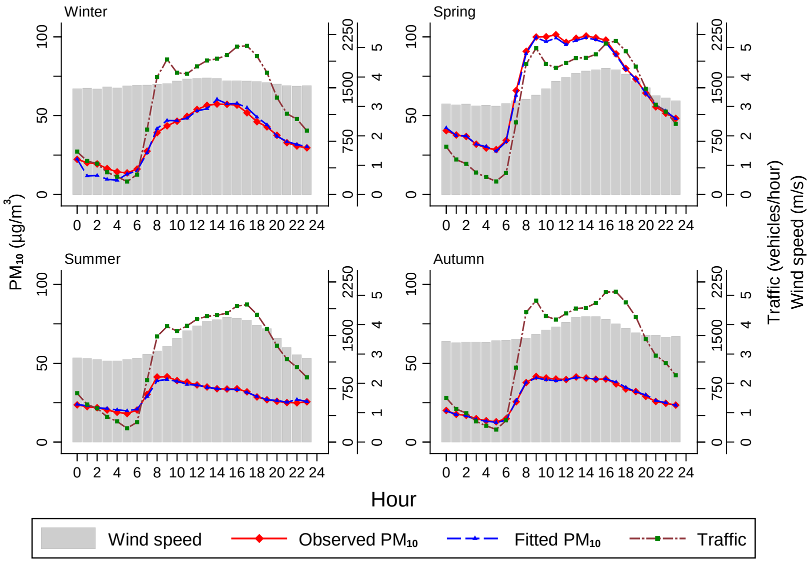
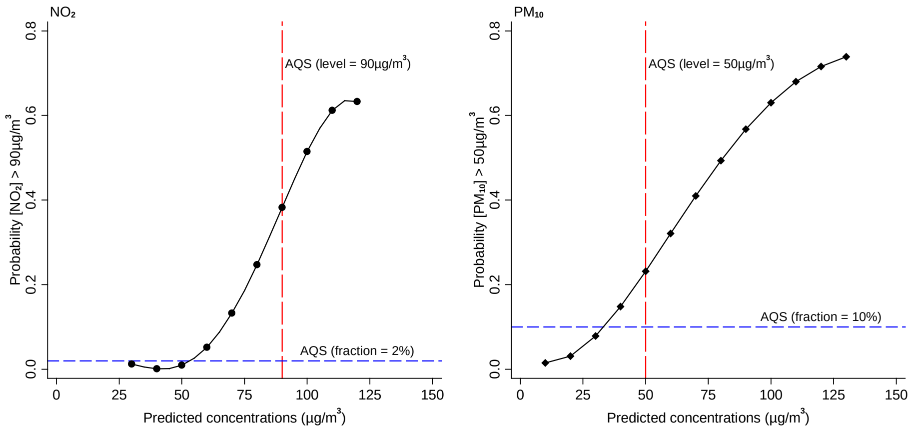
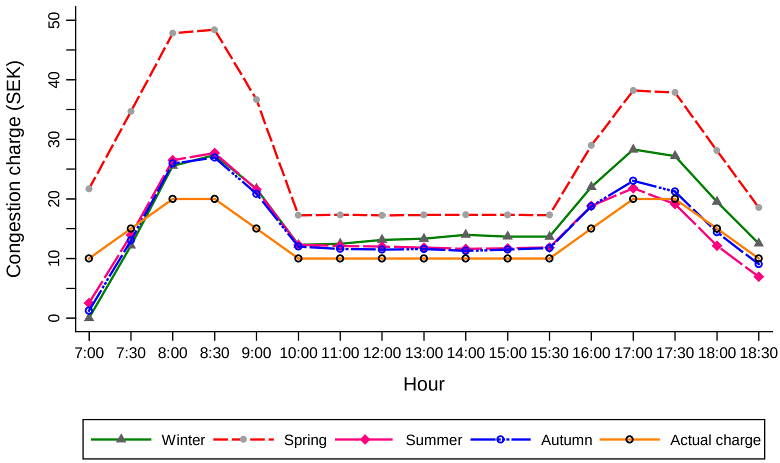


Figure 5: Empirical Distribution of Exceeding AQS and Predicted [NO_2] and [PM_{10}][7]



[7] The graphs of the empirical distribution of the probability of exceeding AQS and the predicted [NO_2] and [PM_{10}] were fitted with a polynomial of order four. The empirical and the fitted distribution showed a high goodness of fit ($R^2 = 0.99$).

Figure 6: Actual vs. Time-Varying Charge Consistent with AQS per Season



Appendix

Table 1. [NO_2] by season

	[NO_2] Winter	[NO_2] Spring	[NO_2] Summer	[NO_2] Autumn
k1	23.5275***	29.6248***	25.7822***	22.8315***
	(1.2588)	(1.4809)	(1.6034)	(1.1994)
k2	0.0472***	0.0459***	0.0415***	0.0382***
	(0.0021)	(0.0020)	(0.0018)	(0.0015)
g0	0.6592***	0.8198***	0.7115***	0.5549***
	(0.0732)	(0.0784)	(0.0948)	(0.0628)
g1	0.1317	0.0506	0.0891	0.0757
	(0.1001)	(0.1459)	(0.0848)	(0.0674)
g2	0.1768	0.0256	0.1951*	0.1038
	(0.1370)	(0.0961)	(0.1169)	(0.0897)
g3	0.5736	0.2796*	0.3747**	0.3295**
	(0.3547)	(0.1678)	(0.1533)	(0.1351)
g4	0.7179	0.6461	0.4520*	0.9251**
	(0.5628)	(0.5180)	(0.2699)	(0.4489)
g5	0.9547***	0.8653**	0.4309*	0.9409***
	(0.2568)	(0.4116)	(0.2560)	(0.1936)
g6	0.7060**	0.3330**	0.1776	0.7441***
	(0.2841)	(0.1450)	(0.1764)	(0.2204)
g7	-0.0645	-0.3443***	-0.1898**	-0.0586
	(0.0549)	(0.0704)	(0.0785)	(0.0535)
g8	-0.1048*	-0.3387***	-0.2372***	-0.0921
	(0.0618)	(0.0785)	(0.0894)	(0.0603)
g9	-0.0965	-0.3161***	-0.2505***	-0.0904
	(0.0625)	(0.0794)	(0.0912)	(0.0612)
g10	-0.1894***	-0.4117***	-0.3203***	-0.1948***
	(0.0622)	(0.0786)	(0.0905)	(0.0609)
g11	-0.1895***	-0.3794***	-0.2876***	-0.1814***
	(0.0615)	(0.0770)	(0.0903)	(0.0598)
g12	-0.1126*	-0.2996***	-0.2480***	-0.1465**
	(0.0615)	(0.0748)	(0.0902)	(0.0594)
g13	-0.0941	-0.2958***	-0.2278**	-0.1408**
	(0.0614)	(0.0752)	(0.0891)	(0.0592)
g14	-0.1276*	-0.2887***	-0.2049**	-0.1100*
	(0.0674)	(0.0747)	(0.0888)	(0.0591)
g15	-0.0644	-0.2762***	-0.2091**	-0.1050*
	(0.0612)	(0.0747)	(0.0891)	(0.0588)
g16	-0.0417	-0.2429***	-0.1981**	-0.0914
	(0.0607)	(0.0753)	(0.0899)	(0.0602)
g17	0.0539	-0.1587**	-0.0621	0.0001
	(0.0609)	(0.0730)	(0.0864)	(0.0588)
g18	0.1160**	-0.0546	0.0771	0.0943*
	(0.0589)	(0.0722)	(0.0804)	(0.0554)
g19	0.1405***	-0.0020	0.1367*	0.1080**
	(0.0539)	(0.0686)	(0.0749)	(0.0531)
g20	0.0991**	0.0433	0.1543**	0.0898*
	(0.0493)	(0.0642)	(0.0709)	(0.0507)
g21	0.1551***	-0.0408	0.2095***	0.1031**

	(0.0472)	(0.1114)	(0.0662)	(0.0484)
g22	0.1448***	0.0474	0.0790	0.1316***
	(0.0473)	(0.0605)	(0.0922)	(0.0438)
g23	0.0990**	-0.0150	0.0016	0.0765*
	(0.0441)	(0.0550)	(0.0599)	(0.0410)
N	9375	10291	10401	10517
R^2	0.5354	0.4684	0.4072	0.5139
Log-likelihood	-40834.13	-46311.06	-45952.76	-45339.48
LR test chi2(23)	466.19	660.98	647.02	507.90
P-value	0.0000	0.0000	0.0000	0.0000
AIC	81720.27	92674.12	91957.52	90730.97
BIC	81906.06	92862.34	92146.01	90919.75

Table reports Non Linear Square estimates of four separate regressions. The dependent variable is pollutant [NO_2] ($\mu g/m^3$). All the specifications correspond to the equation $[NO2]_t = k_1 + k_2\, traffic_t / ws_t^{\sum_{i=0}^{23} g_i h_i} + \varepsilon_t$. *traffic* is the total traffic in four lanes, *ws* is wind speed and h_i is a variable indicator for the hour of the day *i*. Standard errors, in parentheses, are heteroskedasticity and autocorrelation-consistent (HAC) standard errors using Newey-West's estimator for 24 lags. In the LR test, each specification is compared with the nested model $[NO2]_t = k_1 + k_2\, traffic_t / ws_t^{k_3} + \varepsilon_t$ under the null hypothesis. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 2. [PM_{10}] by season

	[PM_{10}] Winter	[PM_{10}] Spring	[PM_{10}] Summer	[PM_{10}]Autumn
k1	7.8883***	23.0488***	18.1928***	10.5288***
	(2.1699)	(2.8694)	(0.8098)	(1.5412)
k2	0.0497***	0.0632***	0.0196***	0.0270***
	(0.0158)	(0.0084)	(0.0020)	(0.0026)
g0	0.6211***	0.8588***	1.0229***	0.5986***
	(0.1107)	(0.1118)	(0.1557)	(0.1035)
g1	1.9953***	0.0085	-0.1161	-0.0034
	(0.3509)	(0.1647)	(0.1445)	(0.0938)
g2	1.4370***	-0.1551	0.2593*	-0.0343
	(0.3033)	(0.1149)	(0.1389)	(0.1322)
g3	2.5280***	0.0066	0.1052	0.1828
	(0.6695)	(0.1863)	(0.1947)	(0.1606)
g4	2.9688**	-0.0389	0.2787	0.2534
	(1.4304)	(0.3206)	(0.3297)	(0.3971)
g5	-0.1439	0.2630	0.1446	0.3260
	(0.1843)	(0.5117)	(0.3020)	(0.4588)
g6	-0.0383	-0.1919	-0.0818	0.3531
	(0.1518)	(0.1615)	(0.1796)	(0.2730)
g7	0.1996	-0.3753***	-0.5269***	0.0034
	(0.2094)	(0.1196)	(0.1404)	(0.0773)
g8	0.1570	-0.2990**	-0.6811***	-0.0268
	(0.1635)	(0.1325)	(0.1542)	(0.0905)
g9	0.1418	-0.3588***	-0.6626***	-0.0523
	(0.1550)	(0.1359)	(0.1557)	(0.0944)
g10	0.0273	-0.4696***	-0.6734***	-0.1534*
	(0.1565)	(0.1315)	(0.1551)	(0.0931)
g11	-0.0379	-0.5523***	-0.5785***	-0.1758*
	(0.1654)	(0.1247)	(0.1571)	(0.0934)
g12	-0.0772	-0.4891***	-0.5340***	-0.1620*
	(0.1614)	(0.1241)	(0.1497)	(0.0939)
g13	-0.0694	-0.4984***	-0.4633***	-0.1919**
	(0.1870)	(0.1238)	(0.1487)	(0.0960)
g14	-0.1563	-0.5217***	-0.4177***	-0.1842**
	(0.1454)	(0.1242)	(0.1468)	(0.0938)
g15	-0.0829	-0.4938***	-0.4025***	-0.1357
	(0.1803)	(0.1225)	(0.1474)	(0.0932)
g16	-0.0324	-0.4261***	-0.3417**	-0.0677
	(0.1730)	(0.1217)	(0.1469)	(0.0944)
g17	0.0304	-0.3306***	-0.2392*	0.0066
	(0.1662)	(0.1188)	(0.1444)	(0.0924)
g18	0.0891	-0.2559**	-0.0916	0.0797
	(0.1519)	(0.1152)	(0.1385)	(0.0882)
g19	0.0874	-0.2222**	0.0146	0.1084
	(0.1436)	(0.1133)	(0.1320)	(0.0860)
g20	0.1245	-0.1650	0.0404	0.0467
	(0.2386)	(0.1069)	(0.1287)	(0.0746)
g21	0.0740	-0.0622	0.1178	0.0718
	(0.1362)	(0.0965)	(0.1236)	(0.0651)
g22	0.0894	0.0017	-0.1331	0.0813

	(0.1039)	(0.0897)	(0.1852)	(0.0548)
g23	-0.0164	0.0840	-0.0579	0.0315
	(0.1064)	(0.0763)	(0.1121)	(0.0456)
N	9522	9860	10351	10467
R^2	0.1183	0.1959	0.2092	0.2650
Log-likelihood	-52631	-54755.74	-43058.89	-46627.94
LR test chi2(23)	82.42	192.44	477.45	154.32
P-value	0.0000	0.0000	0.0000	0.0000
AIC	105314	109563.5	86169.78	93307.89
BIC	105500.2	109750.6	86358.15	93496.54

Table reports Non Linear Square estimates of four separate regressions. The dependent variable is $[PM_{10}]$ ($\mu g/m^3$). All the specifications correspond to the equation $[\mathrm{PM10}]_t = k_1 + k_2\, traffic_t / ws_t^{\sum_{i=0}^{23} g_i h_i} + \varepsilon_t$. *traffic* is the total traffic in four lanes, *ws* is wind speed and h_i is a variable indicator for the hour of the day *i*. Standard errors, in parentheses, are heteroskedasticity and autocorrelation-consistent (HAC) standard errors using Newey-West's estimator for 24 lags. In the LR test, each specification is compared with the nested model $[\mathrm{PM10}]_t = k_1 + k_2\, traffic_t / ws_t^{k_3} + \varepsilon_t$ under the null hypothesis. Estimates marked * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Supplementary Material

Table 3. Description of traffic and air quality monitoring points

City	Traffic data			Air quality station	Information	Source
	Count points	Location	District			
London	A3211: 75401	Arthur St - Fish St Hill	City of London	Top of Form Upper Thames Street and Walbrook Wharf	October 6th and 16th, 2009. Traffic at 0-6h and 19-23h is projected using the traffic growth rate of M25 road	DfT and LAQN
	A3211: 75400	Queen St Place - Arthur St	City of London			
Santiago	Alameda: E036O0	Jose Victorino Lastarria street- Portugal street	Santiago	Parque O'Higgins	January 2010-December 2010	UOCT and SINCA
	Cardenal Jose Maria Caro: E073O0	Jose Miguel De La Barra street - Purisima Street	Santiago			
Stockholm	Hornsgatan	Ringvägen - Anskariegatan	Central	Hornsgatan	Setpember 2002- December 2010	SLB

Chapter III

Synergies and Trade-offs between Climate and Local Air Pollution Policies in Sweden

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Abstract

In this paper, we explore the synergies and tradeoffs between abatement of global and local pollution. We build a unique dataset of Swedish combined heat and power plants with detailed boiler-level data 2001-2009 on not only production and inputs but also on emissions of CO_2 and NO_x. Both pollutants are regulated by strict policies in Sweden. CO_2 is subject to the European Union Emission Trading Scheme and Swedish carbon taxes; NO_x - as a precursor of acid rain and eutrophication - is regulated by a heavy fee. Using a quadratic directional output distance function, we characterize changes in technical efficiency as well as patterns of substitutability in response to the policies mentioned.

JEL Classification: H23, L51, L94, L98, Q48.

Keywords: Environmental policies, shadow pricing, directional distance function, climate change, local pollution, policy interactions.

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1. Introduction

This paper deals with both the interaction between multiple layers of regulation and the interaction between multiple pollutants. Climate change policy is affected by multiple decision makers at local, national, and international levels. Usually these decision makers are not fully coordinated (neither with respect to goals nor methods) and, the existence of several layers of governance may encourage strategic behavior from powerful local actors trying to enhance their own positions (Caillaud et al. 1996). Multi-level climate change governance is also related to governance of local air pollutants since production processes often involve emitting several air pollutants simultaneously. Environmental policies aiming at reducing CO_2 emissions might therefore create spillovers: decreases or increases in emissions of other pollutants from firms changing or modifying their production processes in response to climate policy. For example, a common strategy to reduce CO_2 emissions is switching the fuel mix from fuel oil towards biofuels which are counted as having zero carbon. While net CO_2 emissions do fall dramatically, biofuels often mean an increase in nitrogen oxides (NO_x), particulate matter (PM), carbon monoxide (CO) and volatile organic compound (VOC) emissions (Brännlund and Kriström 2001, Burtraw et al. 2003).

The aim of this paper is to study the effects of the interaction between the European Union Emissions Trading System (EU ETS) and the Swedish CO_2 tax and refundable charge on NO_x on the relative performance of Swedish combined heat and power plants with respect to CO_2 and NO_x emissions. For this purpose, we built a unique dataset of Swedish combined heat and power plants for the period 2001-2009 that has detailed boiler-level data on not only production and inputs but also on two pollutants CO_2 and NO_x. We use a quadratic directional output distance function to study and compare patterns of technical progress, substitution between CO_2 and NO_x and shadow prices of these pollutants between the periods 2001-2004 and 2006-2009.

Our choice of country and pollutants is motivated by the availability and stringency of environmental policies. The Swedish government has been at the forefront of greenhouse gases' emissions reduction for a long time now. Sweden is one of the few countries that present emissions below the level recorded in 1990, which is mostly explained by a number of initiatives taken in order to reduce CO_2 emissions. In 1991, for example, Sweden was the first country in the world to introduce a carbon tax based on the carbon content of various fuels. Since 2005, most installations within the heat and power sector are part of the EU ETS - cornerstone of the European Union's policy to combat climate change.

The most obvious effect of the carbon tax and the EU ETS has been the expansion of biofuel use with the potential negative side effect of increased NO_x emissions. Since Sweden has ecosystems that are naturally very sensitive to acidification, the country has a very aggressive policy on the precursors, notably NO_x. Indeed, to reduce the emissions of this pollutant a heavy refundable charge has been in place since 1992. Though significant reduction in NO_x emission was noted after the introduction of the NO_x charge reductions appear to have leveled off during the last decade, and hence, the charge level was raised in 2008 to encourage further NO_x reductions.

To the best of our knowledge, this is the first study analyzing and quantifying the effects of the multi-governance of climate change policy and its interaction with the other national policy instruments aimed to reduce local pollutants. However, some previous studies have employed directional output distance functions to analyze the technological non-separability and substitutability among air and water pollutants (see for instance, Murty et al. 2007, Kumar and Managi 2011 and Färe et al. 2012). In this regard, perhaps the most closely related work is Färe et al. (2005), who use a quadratic directional output distance function to estimate the shadow price of SO_2 and the output substitution between electricity and SO_2 before and after the implementation of Phase I of the Acid Rain Program in United States. Unlike Färe et al. (2005), our focus is on policy-induced substitutability *across* pollutants and the changes in emissions' relative prices introduced by environmental multi-level governance.

To what extent could an increased relative CO_2/NO_x price have contributed to increased/decreased emissions of NO_x? Theory shows that NO_x emissions may either increase or decrease depending on whether NO_x is a substitute or a complement to CO_2, respectively (see Ambec and Coria 2013). Technological progress does, however, also play an important role as it might lead to reduced emissions of one or of all pollutants. Hence, even in the case that pollutants are substitutes, the final effect on emissions would depend on the direction of technological progress (i.e., CO_2 and/or NO_x saving) and whether technological progress outweighs the substitution effect.

This paper is organized as follows. Section 2 briefly describes the climate and NO_x policy in place in Sweden and the changes on the relative price CO_2/NO_x over the period 2001-2009. Section 3 presents the theoretical and empirical framework of the joint production of heat and power, and CO_2 and NO_x emissions. Section 4 discusses the data and empirical results. Finally, Section 5 concludes the paper.

2. Climate and NO_x Policy in Sweden: Carbon Tax, EU ETS, and the Refundable NO_x Charge

As much focus of the climate challenge discussion during the 1980s had been on oil substitution, Sweden reformed its taxation system in 1991, introducing a carbon tax specifically designed to discourage oil use. Thus, it was directly connected to the carbon content of the fuel (see Figure 1), though still allowing for some differentiation among sectors.[1] Initially, the general tax was equivalent to 25 €/ ton of CO_2. However, it has increased steadily during the last decade, and at present, it corresponds to 105 €/ ton. This is by all accounts a very high – some would say extreme - carbon tax. To put it in context, the carbon dioxide permits on United States markets such as RGGI and Chicago are trading around 4 €/ ton; the EU ETS has varied around a mean of

[1] For example, there is no carbon tax on electricity production but non-industrial consumers have to pay a tax on electricity consumption.

15-20 €/ ton; and France tried to introduce a carbon tax of 17 €/ ton – but failed because of fears that such a level would be detrimental to the economy.

Since the tax is very large and Sweden is a small open economy, there was quite some concern for the competitiveness of some energy intensive industries. Thus, a number of deductions and exemptions were created in those sectors that are open to competition, and a series of reduced rates applied. In the case of the heat and power sector, the carbon tax varies according to the type of generation. From 2005 until July 2008 the carbon tax and the EU ETS overlapped. Since that date the tax was in principle replaced by the EU ETS (see Figure 1). Combined heat and power plants (hereinafter CHP) were granted a tax reduction of 85%. Since the level of the permit price is much lower than the Swedish tax level, this harmonization with the EU actually implied a sizeable fall in the price of carbon emissions for most CHP plants.

Indeed, regarding the prices of CO_2 allowances in the EU ETS, they have faced a great deal of variation since the launch of the first phase. They started around 5 €/ton but quickly increased to a range of 20-30 €/ton, peaking at over 30 €/ton early in 2006. However, the prices fell dramatically in 2006— as it became clear that the market was long on allowances. Since very low allowances prices in Phase I (2005–2007) could jeopardize the credibility of the trading scheme, the European Commission tightened the cap for the second trading period. Remaining at levels of €/ton 22-23 during the early 2008, the CO_2 price dropped during the market crashes. Though fundamentals of the carbon market in the road ahead to Phase III are yet unchanged, the CO_2 price is somewhat volatile, depending in the short run on both fuel prices and international climate negotiations. In 2011, for instance, the CO_2 price peaked in April after the German decision to phase out nukes; it decreased after June when the European Commissions proposed a mandatory target for energy efficiency improvement and it jumped again with increasing gas and power prices in August.

The tax reform of 1991 did not only introduce carbon taxes but also other taxes including for instance a high fee on NO_x. The fee was initially confined to all NO_x emissions from electricity and heat-producing boilers, stationary combustion engines and gas turbines with a useful energy production of at least 50 gigawatt hours (GWh) per year (approx. 182 boilers). Nevertheless, because of its effectiveness in emission reduction and simultaneously falling monitoring costs, in 1996 the charge system was extended to include all boilers producing at least 40 GWh and in 1997 the limit was lowered to 25 GWh of useful energy per year.

The total fees are returned to the participating plants[2] in proportion to their production of useful energy. Hence, the system encourages plants to reduce NO_x emissions per unit of energy to the largest possible extent, since plants with lower emissions relative to energy output are net receivers of the refund. The fee was originally set at 4.3 €/kg - which again is an extremely high level compared to other countries. It corresponds to 4300 €/ton which can be compared to the Taxe Parafiscale in France of 40.85 €/ton, the Norwegian fee of 525 €/ton, or the Summer Seasonal NO_x emissions trading program' price in United States of approximately 600 €/ton.[3] The refund varies from year to year, but in recent years it has been around 0.9 € /megawatt hour (MWh) of useful energy while the average NO_x emissions per unit of energy has been 0.23 kilograms of NO_x per MWh.

The Swedish NO_x charge has been evaluated extensively (see for instance, Höglund 2005; Sterner and Höglund 2006; and Sterner and Turnheim 2009). It has been shown to be very effective lowering emissions. Indeed, the empirical findings suggest that extensive emission reductions have taken place due to learning and technological development in abatement. NO_x is produced largely from an unintended chemical reaction between nitrogen and oxygen in the combustion chamber. The process is quite non-linear in temperature and other parameters of the combustion process, which implies that there is a large scope for NO_x reduction through various technical measures

[2] With the exception of 0.7 percent that is kept for administration costs.
[3] Price in 2008. NO_x prices in the Summer Seasonal NO_x emissions trading program' price in United States have dropped dramatically since then to approx. 12 €/ton by 2011.

(including changing the shape, temperature or oxygen and moisture content of the combustion chamber), fuel switching, and through other abatement strategies involving the addition of ammonia or passing exhausts through catalytic converters (Millock and Sterner 2004). Adoption of NO_x reduction technologies has been a major driver of emissions reduction. Nevertheless, emissions fell mostly in the early years. The decrease has continued since then, but at a reduced pace. Hence, as the impact of the charge seemed to be lower than before, in 2008 the Swedish government decided to raise the tax to 5.3 €/ kg to foster further adoption of more effective treatment techniques (SEPA 2003, 2007).

How have all the regulations described above affected the relative cost of CO_2-NO_x emissions? To assess the overall picture is not altogether easy. Although large industrial plants in the energy sector can to some extent adjust their technology in response to short run price variations (for instance through fuel switching), many features of their design take a decade to build and are adapted to expected price trends over a longer time horizon (though clearly, the ability to switch fuels may well be one such feature). Furthermore, the carbon tax paid and allowances used depend on the type of fuel being burned, which is endogenous to the stringency of CO_2 policies in previous years. Due to the large fraction of boilers in the heat and power sector fully relying on biofuels, the actual payment of CO_2 regulations is close to zero. Hence, the actual cost of this regulation to generating units would provide a misleading estimate of the real opportunity cost.

Nevertheless, in order to provide an indication of the relative stringency of CO_2 and NO_x policies, we compute the relative opportunity cost of CO_2 and NO_x emissions per unit of output for an average CHP plant. As shown in Figure 2, it seems clear that policy signals in Sweden have told power companies to avoid fossil fuels. The opportunity cost of CO_2 emissions is much larger than the opportunity cost of emitting NO_x.[4] For example, in 2003, an average CHP plant emitted 0.082

[4] Note that the Swedish NO_x charge is refunded to firms in proportion to energy produced, implying that some firms receive a net refund while others are net payers. Thus, refunding reduces the total cost of the

tons of CO_2 and 0.248 kilograms of NO_x to produce 1 Mwh of useful energy. Given the magnitude of the carbon tax and NO_x fee at that time, this implied a cost of 6.116 and 0.936 €/Mwh, respectively. This is to say that the opportunity cost of CO_2 emissions per unit of output was over six times the cost of NO_x emissions.

Moreover, the variation observed in Figure 2 suggests us that CO_2 policy, on average, did become less stringent (relative to NO_x) due to the carbon tax phase-out. Indeed, the reduction started to take place already in 2004 when CHP plants were granted a significant carbon tax reduction. Moreover, Sweden increased the NO_x charge for all regulated boilers in 2008, adding to the effect of the reduced carbon tax on the relative opportunity cost CO_2/NO_x.

It seems clear that most abatement efforts should be directed to reducing CO_2 emissions as the economic effect of CO_2 regulations on firms' profitability is much larger than NO_x's regulation. However, the variations in the opportunity cost of CO_2-NO_x emissions should induce some variations on the relative CO_2-NO_x abatement efforts if generating units want to minimize the cost of compliance with environmental regulations. The magnitude and direction of the changes on the optimal mix of CO_2-NO_x emissions would depend, however, on a series of factors such as technological development and whether CO_2 and NO_x are substitutes or complements in abatement. For instance, in the absence of technological development, one would expect emissions of NO_x from CHPs to decrease to a relatively larger extend during the period 2006-2009 than to the period 2001-2004 if pollutants are substitutes, while the reverse holds if pollutants are complements.

On the other hand, the high relative opportunity cost of CO_2 emissions should also trigger technological fixes and fuel switching aiming to reduce them. For instance, CO_2 emissions can be reduced by raising combustion temperature since this increases thermal efficiency – but it also increases NO_x emissions. Alternatively, NO_x emissions can be reduced through post combustion measures but they usually require energy and thus will increase carbon emissions (at least relative

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regulation; however, it does not affect the opportunity costs of releasing further NO_x emissions, which is given by the NO_x charge.

to output). Hence, given the relatively stringency of CO_2-NO_x regulations, we would expect technological efforts to be overall biased towards CO_2 emissions reductions. In the following sections we use a quadratic directional output distance function to derive the relative shadow prices of emissions for each generating unit and analyze the changes on technical efficiency and abatement efforts induced by the regulatory changes, but we start first by describing the estimation strategy.

3. Estimation Strategy and Data

In our approach we employ the directional output distance function to derive estimates of technical efficiencies, elasticities of substitution output/pollutants and between pollutants, and relative shadow prices CO_2/NO_x. In the past, researchers first estimated a frontier production function and then the distances of individual plants from the frontier. The function used here seeks the simultaneous expansion of good outputs and contraction of bad outputs, which is very suitable to our case, i.e., modeling the technology in this manner allows for the adoption of abatement measures in order reduce the bad outputs (emissions) and still increase, or hold constant, the production of heat and power.

3.1 Theoretical Background

Following Färe et al. (2005), we treat emissions as bad or undesirable outputs generated in the boilers combustion process and model jointly the production of heat and power and emissions. Let the set of output possibilities $P(x) = \{(y, b): x \text{ can produce } (y, b)\}$ represent all the feasible input-output possibilities of boilers that jointly generate heat and/or power (y) and emissions of CO_2 and NO_x (denoted by $b = (b_1, b_2)$, respectively) using an input vector $x = (x_1, x_2, x_3)$ containing installed capacity, fuel consumption (as input energy) and labor, respectively. We assume that inputs are strongly disposable, which implies that the output set is not shrinking if the inputs are expanding. Furthermore, we assume null-jointness, implying that no good output is produced without bad outputs. Moreover, all outputs are assumed to be jointly weakly disposable. That is, for a given input vector, emissions can be reduced to the expense of reduced output. Finally, we consider that the good output is strongly disposable, i.e. it is always possible to reduce the good output without reductions in undesirable outputs for a feasible good and bad output vector. The directional output distance function is characterized as follows:

$$\vec{D}_o(x, y, b; g) = max\{\beta: (y + \beta g_y, b - \beta g_b) \in P(x)\} \qquad (1)$$

Where $g = (g_y, g_b)$ is the directional vector and $\beta^* = \vec{D}_o(x, y, b; g)$. Equation (1) is a functional representation of the technology that is consistent with $P(x)$ and its associated properties. For a given g>0, if a generating unit that produces a particular combination of b and y outputs were to work efficiently, it could expand the good output and contract the bad outputs along the g direction until reaching the boundary of $P(x)$ at $(b - \beta^* g_b\, , y + \beta^* g_y)$.

The directional output distance function has several properties. (i) It is non-negative for feasible output vectors. Thus, the function takes the value of zero for generating units with observed output vectors operating with maximum technical efficiency on the frontier of $P(x)$, or takes positive values for generating units operating with inefficient output vectors in the interior of $P(x)$. So the function provides scores of technical and environmental efficiency where higher values of $\vec{D}_o(x, y, b; g)$ indicate higher inefficiency. This also satisfies monotonicity: (ii) it is non-decreasing in undesirable outputs, (iii) non-increasing in good output, and (iv) non-decreasing in inputs. Moreover, it is (v) concave in (y, b), and fulfills (vi) weak disposability of good output and bad outputs and (vii) null-jointness. (viii) Additionally, the directional output distance function satisfies by its definition the translation property, expressed as $\vec{D}_o(x, y + \alpha g_y, b - \alpha g_b; g) = \vec{D}_o(x, y, b; g) - \alpha\, , \alpha \in \Re$.[5]

[5] The translation property is equivalent to homogeneity in outputs of the Shepard's (1970) output distance function (see Färe et al. 2006). For a detailed description of the directional output distance function's properties see Färe et al. (2005).

The directional output distance function approach allows us not only to account for technical efficiency but also calculate the elasticity of substitution between CO_2 and NO_x emissions and the relative shadow prices of these pollutants. Regarding the first, the indirect Morishima elasticity of substitution provides us with information of how much the relative shadow prices of outputs will change in response to changes in emission intensities (see Blackorby and Russell 1981, 1989). In the context of the technology described by the directional output distance function in (1) the indirect Morishima elasticity of substitution between pollutants b_1 and b_2 can be expressed as:

$$M_{b_1 b_2} = \frac{\partial \ln(q_1/q_2)}{\partial \ln(b_2/b_1)} = b_2{}^{*}.\left(\frac{D_{b_1 b_2}}{D_{b_1}} - \frac{D_{b_2 b_2}}{D_{b_2}}\right) \qquad (2)$$

and between pollutant b_i (with *i*=1,2) and good output is:

$$M_{b_i y} = \frac{\partial \ln(q_i/p)}{\partial \ln(y/b_i)} = y^{*}.\left(\frac{D_{b_i y}}{D_{b_i}} - \frac{D_{yy}}{D_y}\right) \qquad (3)$$

Where $q = (q_1, q_2)$ denotes the emissions price vector containing CO_2 and NO_x prices, respectively, and p denotes the output price. D_{b_i} and D_y are the first order derivatives of $\vec{D}_o(x,y,b;g)$ with respect to pollutant b_i and good output, respectively; $D_{b_1 b_2}$, $D_{b_2 b_2}$, $D_{b_i y}$ and D_{yy} are second order derivatives of the directional output distance function; $y^* = y + \vec{D}_o(x,y,b;g)$ and $b_i^* = b_i - \vec{D}_o(x,y,b;g)$. If $M_{b_1 b_2} > 0$, b_1 and b_2 are Morishima substitutes. That is, pollutants are substitutes if the emission intensity (b_2/b_1) increases when the relative marginal cost of abatement between these pollutants (q_2/q_1) decreases; in other words, emissions reductions in b_1 are accompanied by increased emissions in b_2. Conversely, b_1 and b_2 are complements when $M_{b_1 b_2} < 0$. The Morishima elasticities are inherently asymmetric ($M_{b_1 b_2} \neq M_{b_2 b_1}$). In terms of the analysis, the asymmetric substitutability tells us which pollutant is easier to substitute for another pollutant for a fixed amount of output (Stiroh 1999). Note that a lower value of the indirect Morishima elasticities of substitution indicates larger substitution possibilities between pollutants. The intuition behind this is that to generate the same change in the emission intensity (b_2/b_1), a smaller change in the prices (q_1/q_2) is required when the pollutants are close substitutes.

With regard to elasticities of pollution-output, since $\vec{D}_o(x, y, b; g)$ satisfies monotonicity, concavity and translation properties then $M_{b_i y} \leq 0$.[6] Like in the case of the indirect Morishima elasticities of substitution between pollutants, lower pollution-output elasticities in absolute value indicate larger substitution possibilities. Finally, regarding shadow prices, it can be shown that for a given vector $g = (g_y, g_b)$ and a differentiable directional output distance function, the relative shadow price between undesirable outputs is[7]:

$$\frac{q_1}{q_2} = \frac{D_{b_1}}{D_{b_2}} \geq 0 \qquad (4)$$

The ratio in (4) represents the trade-off between these two pollutants, i.e. the shadow marginal rate of transformation. Furthermore, the analog relative shadow price for pollution-good output is defined as $-q_i/p = D_{b_i}/D_y$. The shadow prices are calculated on the technically efficient frontier of P(x).[8]

3.2 Functional Form

We specify our directional output distance function with a quadratic form. This specification is twice differentiable, has a flexible structure that ensures the translation property, and has the advantage that it tends to have a better adjustment than trans-log specifications (Färe et al., 2010).

[6] This is because the elasticities are obtained on the boundary along the positive sloped portion of the output possibilities set, i.e. when emissions fulfill weak disposability. See Färe et al. 2005.

[7] For a detailed explanation of the shadow prices approach in the context of the output distance function see Färe et al. (1993). For an application in the case of directional output distance functions see Färe et al. (2005).

[8] It should be noted that due to the translation property of the directional output distance function, the first order derivatives and the relative shadow prices calculated at the observed output vectors and their corresponding projected output values are the same.

As in Färe et al. (2006) our choice of the directional vector is $g = (1,1)$, i.e. the component of the good output and the components of the two pollutants are equal to one, making the model parsimonious. For a generating unit k operating at period t, the directional output distance function has the following expression:

$$\vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1)$$

$$= \alpha + \sum_{n=1}^{3} \alpha_n x_{nk}^t + \beta_1 y_k^t + \sum_{i=1}^{2} \theta_i b_{ik}^t + \tfrac{1}{2}\sum_{n=1}^{3}\sum_{n'=1}^{3} \alpha_{nn'} x_{nk}^t x_{n'k}^t + \tfrac{1}{2}\beta_2 y_k^{2t}$$

$$+ \tfrac{1}{2}\sum_{i=1}^{2}\sum_{i'=1}^{2} \theta_{ii'} b_{ik}^t\, b_{i'k}^t + \sum_{n=1}^{3}\sum_{i=1}^{2} \eta_{ni} x_{nk}^t b_{ik}^t + \sum_{i=1}^{2} \mu_i y_k^t b_{ik}^t + \sum_{n=1}^{3} \delta_n x_{nk}^t y_k^t \quad (5)$$

with $k = 1,2,\dots,K$ and $t = 1,2,\dots,T$. The following parameter restrictions are imposed to satisfy the translation property:

$$\beta_1 - \sum_{i=1}^{2} \theta_i = -1\ , \beta_2 - \sum_{i=1}^{2} \mu_i = 0\,, \mu_i - \sum_{i'=1}^{2} \theta_{ii'} = 0\,, i$$

$$= 1,2,\ \ \delta_n - \sum_{i=1}^{2} \eta_{ni} = 0\,,\ \ n = 1,2,3 \quad (6)$$

Furthermore, symmetry conditions for cross-output and cross-input terms are also assumed:

$$\alpha_{nn'} = \alpha_{n'n} \text{ with } n, n' = 1,2,3, \text{ and } \theta_{ii'} = \theta_{i'i} \text{ for } i, i' = 1,2 \quad (7)$$

We estimate the directional output distance function using a deterministic method, i.e., Parametric Linear Programming (PLP) that allows us to impose parametric restrictions that are a result of the underlying technology such as monotonicity in good or bad outputs[9]. We follow

[9] The directional output distance function may also be estimated using stochastic approaches. Although stochastic frontier method has the advantage that statistical hypotheses can directly be tested within the model and deal with issues such as measurement errors, it requires the inclusion of distributional assumptions for the inefficiencies scores and residuals that are arbitrary and may be violated. Moreover, through the

Aigner and Chu (1968)'s procedure of minimizing the sum of the distance between the frontier technology and the actual observations of the generating units in each period. Hence, it chooses the parameters that make the generating units as efficient as possible subject to a set of restrictions associated to the technology properties already described (see Färe et al. 2001, 2005, 2006).[10] We derive estimates of the coefficients for the period of pre-introduction of the EU ETS (2001-2004) and the period post implementation (2006-2009). In order to control for possible yearly changes; e.g. variations in weather (during certain years some winters may be colder than others) or also general shocks of the economy that may affect heat and power production, we add a group of yearly dummy variables to equation (5). We wrote the code to solve the optimization problem in Matlab. All the variables (outputs and inputs) are expressed in normalized values with respect to their corresponding means in order to avoid convergence problems in the algorithm.

Using the estimated coefficients of the directional distance output function we compute the technical and environmental efficiencies and the Morishima elasticities of substitution between CO_2 and NO_x and pollutants-output according to equations (2) and (3), respectively. The relative shadow CO_2/NO_x prices are obtained applying equation (4). To identify changes in efficiencies, elasticities and relative shadow prices before and after the implementation of the EU-ETS we compare the density functions of these measures between periods. For that purpose we employ kernel-based methods to statistically test the difference between distributions. Our tests are conducted by computing the T_n-statistic of Li et al. (2009)[11], which is a nonparametric consistent test that assesses the equality between two density functions, $f(x)$ and $g(x)$ of a random variable x (see Li et al. 2009 for details). Following Hayfield and Racine (2008, 2011), we implement this procedure in the software R with 500 bootstrap repetitions and estimate the T_n-statistic using a

stochastic approach is not always possible to satisfy the properties and assumptions of the technology described above, e.g. monotonicity in good or bad outputs, because those properties cannot be simply introduced in the model as linear restrictions of the parameters.

[10] We solve the optimization problem subject also to the constraints of monotonicity of inputs at the mean level of input usage and a set of constraints that ensure the null-jointness and concavity properties and estimates in the positive slope of the output set

[11] The T_n-statistic of Li et al. (2009) can be used in a broader perspective to test equality of distributions with mixed and continuous data. The test of equality of two density functions is just a particular case of it. Unlike the T-statistic of Li (1996, 1999), the T-statistic of Li et al. (2009) is not sensitive to the ordering of the data.

standard normal kernel. The empirical p values of the consistent density equality test are computed after bootstrapping.

3.3 Data

Our analysis models production and emissions at the boiler level using data of the Swedish CHP plants during 2001-2009. We focus on CHP plants since approximately 75% of the plants on the heat and power sector belongs to this group. Moreover, CHP plants have been promoted within the European Union as an effective means of increasing the overall energy efficiency (EU Directive 2004/8/EC).[12]

Our measure of good output is the amount of useful energy (MWh) commercially sold. The two undesirable outputs, CO_2 and NO_x emissions, are expressed in tons, and as stated above, installed capacity (MW), fuel consumption (MWh) and labor (number of employees) are the inputs. NO_x emissions and useful energy are taken from the Swedish Environmental Protection Agency's (SEPA) NO_x charge database. These two variables are measured and reported to the SEPA directly by the generating units including as well information about energy fuel shares, installed capacity, and the available NO_x Post-combustion (PCT) and Combustion Technologies (CT) [13], among other variables; which makes this dataset unique in the sense that it is the most detailed longitudinal database at the boiler level of the Swedish heat and power sector. Installed capacity is employed as a proxy of capital in physical units. With regard to labor, data at company level were gathered from

[12] A highly-efficient CHP can use 10% less fuel than would be used by separate production of the same quantities of heat and electricity (Swedish Energy Agency, 2009). In Sweden approximately 30-50% of the total input energy of a CHP is converted into electricity and the rest in heat (Svensk Fjärrvärme, 2011).

[13] PCT consists of flue gas treatment that aims at cleaning up NO_x once it has been formed, usually through conversion to benign chemical species. It includes: 1) SCR Selective Catalytic Reduction, whose installation is rather large and costly but achieves highly efficient reduction. And 2) SNCR Selective non-catalytic reduction of chemicals (ex: ammonia, urea, sodium bicarbonate). SNCR is less costly than SCR in both capital and operating costs, but it achieves lower performance. CT involves combustion measures that seek to inhibit the formation of thermal and prompt NO_x. These strategies typically involve the optimal control of combustion parameters (temperature, pressure, stoichiometry, flame stability and homogeneity, and flue gas residence time) for minimal NO_x formation. See Sterner and Turnheim (2009) for further details.

Retriever Bolagsinfo. For multi-unit plants, we allocated labor to generating units according to their useful energy ratio.

Although we have CO_2 values accessed from the SEPA's EU ETS database, their aggregation at the installation level prevents us to recover the emissions for each boiler. Instead, CO_2 emissions are estimated based on boilers' available data on energy fuel shares and emission factors per fuel type. Hence, we can recover the total input energy that corresponds to the amount of fuel consumed per boiler. In addition, our dataset comprises a rich variation in fuel types: gas, oil, coal, peat, biofuel, and waste using emission factors for each fuel classification.[14] However, this method only considers emissions derived from fuel use for combustion and excludes emissions coming from raw materials, which – unlike other industries – are not significant in the case of the heat and power generation. [15]

We focus on boilers that operate every year. This group of generating units represents the operation of the sector under normal or standard conditions, i.e. we exclude boilers that may only work in certain circumstances (e.g. backup during episodes of very cold winters). In order to allow for adjustments at the early stage of the EU ETS, information for the year 2005 is excluded from the sample. Two boilers with lack of information on fuel shares are dropped. One boiler using a combination of mixed refinery gas, and gas converted during the process is also excluded due to the complexity of the fuel and its extremely high emissions. Finally, our sample consists of a panel of 82 boilers (656 observations).

Descriptive statistics of the variables are presented in Table 1. As we can see, there is a large variation in emission rates among boilers. CO_2 emission rates vary from zero to 428 tons/GWh,

[14] Each type of fuel has some sub-classifications. For instance gas types may include natural gas, LPG, biogas; oil types like fuel oil 1, 5, bio-oil; and in biofuel group we may include several kinds of residues from the forest and other types of biomass. To develop the estimations specific emission factors for every sub-classification have been considered (see emission factors in SEPA 2009).

[15] For comparison purposes we also estimated the CO_2 emissions using the total output per boiler, adjusting it by boiler efficiency to obtain the input energy, and distributing it across fuels by means of the energy fuel shares. Another check involved the comparison between the sum of the estimated emissions per installation and the corresponding aggregated emissions in the SEPA's EU ETS database for some installations where was possible to make such aggregation. In both cases, our estimations were in a similar order of magnitude.

while NO_x emission rates vary almost twenty fold from 0.03 to 0.6 tons/GWh. This reflects differences within the sector in fuel mix, fuel usage, size of the boiler, and CO_2 and NO_x abatement strategies.

4. Results

In this section we describe and analyze the results of our estimates of technical efficiencies, elasticities of substitutes and shadow prices.

4.1 Technical Efficiency and Technical Progress

The estimated coefficients of the quadratic directional output distance function are shown in Table 2.

Stringent environmental regulations not only have a positive effect on environmental quality, but may also have a positive effect on firms' efficiency if they induce a more efficient use of resources, as well as the development of new technologies. Compliance costs due to stricter environmental regulation make environmentally friendlier technological development relatively less costly. This should be represented by an outward shift of the production possibility curve or an inward shift in the input coefficient space, which means that with a given set of resources it is now possible to produce more goods and services without worsening the environmental quality or vice versa. (Xepapadeas and de Zeeuw 1999).

Our results from the Swedish combined heat and power plants are a case in point and indicate the existence of significant technical progress. We compute the frontier for a size range including the hypothetically efficient boiler for the years 2001 and 2009 (initial and final year in our sample). As can be seen in Figure 3, technological progress drives a significant movement of the frontier towards reduced emissions of both pollutants, though as expected, the overall reduction is biased towards CO_2 emissions reduction.[16] Figure 3 also shows the actual relative prices of CO_2 and NO_x emissions for each respective year[17] and we see that the optimal mix CO_2-NO_x emissions (in tons) for the period 2001-2004 and 2006-2009 is actually found as a corner-solution. This means that it is optimal (in both years) for plants to minimize CO_2 emissions even at the expense of increasing NO_x emissions (for example through higher furnace temperatures and through the use of biofuels). The movement of the frontier itself is also consistent with the fact that the amount of heat and power generation per unit of emissions increases for both pollutants between periods, but the efficiency increase is much larger in the case of CO_2. We could thus say that the technical change is "carbon-saving" in much the same way as traditionally technical changes often have been characterized as "labor saving". On average, CO_2 emissions per GWh by CHP plants decreased 16% between periods (from 81.4 in 2001-2004 to 68.6 ton/GWh in 2006-2009), and approximately 5% for NO_x (from 0.245 to 0.232ton/GWh).

Thus far we have analyzed technical progress at the frontier. We are also interested in the performance of all firms which is conveniently measured by studying technical efficiency relative to each respective frontier: We find that out of 82 boilers operating during the period 2001-2009, only 10 and 13 boilers were found to operate on the frontier during 2001-2004 and 2006-2009, respectively. The highest technical and environmental efficiencies are reached by boilers using biofuel while the highest inefficiencies are found in boilers burning fossil fuels[18]. The estimations of technical and environmental efficiency yield mean inefficiency values of 18.9% and 21.6% for pre and post-introduction periods of the EU ETS, respectively. This indicates that during 2001-2004 boilers, on average, could have expanded heat and power generation by 45.23 GWh (i.e. 239.3*0.189) or contracted emissions of CO_2 by 4744 tons (25098*0.189) and NO_x by 8.8 tons

[16] We also estimated the quadratic directional output distance function for the whole sample (pre and post EU ETS together) substituting the dummy variables for a time trend (in linear and quadratic form and its corresponding interactions with inputs and outputs). We found technological progress is supported by the negative sign of the time trend coefficients.

[17] The price of CO_2 emissions is given by the sum of the carbon permit price and tax - given the fuel consumption and fuel mix of a representative firm at the frontier.

[18] Mean inefficiencies for boilers using biofuel ranges between 16-21%. On the other hand, average inefficiencies in boilers employing fossil fuels vary between 23-35%.

(46.7*0.189) if they would have adopted the best practice of frontier generating units. Similarly, for 2006-2009 boilers could have increased their production by 52.41 GWh or decreased CO_2 and NO_x by 4397 and 9.1 tons, respectively if all the generating units were to operate on the frontier. Given these results, the amount of possible reduction in emissions and increase in heat and power generation that could have been achieved between periods following the practice of the most efficient generating units is of considerable magnitude. Moreover, the fact that relative inefficiency appears to increase would mean that technical progress was fastest at the frontier but slower for average plants. In the long run, this would suggest that considerable further progress is to be expected as average plants and laggards catch up with the frontier. However, point estimates are not enough if we want to be sure there is a difference in relative efficiency between the two periods. We find that the bootstrapped Li et al. (2009) Tn-statiststic that allows us to compare the inefficiency distributions is equal to -2.95 with an empirical p-value of 0.128. That is, the firms are statistically equally efficient in both periods at 10% significance level.

4.2 Output-CO_2 and Output-NO_x Elasticities

With regard to the elasticities pollution-output, Figures 4a and 4b illustrate the kernel distributions of pollution-output between the two periods. When it comes to CO_2, the mean CO_2-output elasticity changed from -0.623 to -0.374 between periods. Using the Li et al. test we are able to reject the null hypothesis of equality between the density functions of the elasticities pollution-output pre and post-introduction of the EU ETS for CO_2. i.e., the responsiveness of the ratio emissions/output to emission price has increased. This result indicates that it has been less costly to reduce CO_2 emissions during the period 2006-2009 than 2001-2004. An explanation to this is technological development. Indeed, the results in the previous section indicate that several technical measures have been implemented in order to reduce emissions. The trend towards phasing out of fossil fuels in Sweden has been quite stable over the sampled period and most firms in the sector have already switched to “carbon-free” fuels. For instance, in 2009 the fraction of boilers with biofuel shares greater than or equal to 80% was approximately 72%, while the fraction of boilers using mainly fossil fuels was around 11% (see Table 3).

The mean NO_x-output elasticity also decreased in absolute value between the pre and post-EU ETS periods (from -1.145 to -0.835). However, although the fraction of generating units without any NO_x abatement measure has declined from 21% to 15% and the simultaneous adoption of more than one NO_x reduction technology has raised from 18% to 32% (see Table 3), the achieved NO_x emission reductions have not been quite enough to support statistically significant difference between the density functions of the elasticities before and after the EU ETS (p-value of Tn-statiststic is 0.194). This is to say, the easiest emission reductions have already been undertaken, and further reductions of NO_x per unit of output will be very difficult to achieve and will only be supported by much higher charges on NO_x emissions. This may also be related with the nominal value of the NO_x charge, which kept constant for a long period since its implementation and then only slightly increased in 2008. The smooth pace of NO_x emission reductions relative to CO_2 during the last decade also reflects an inclination to rather favor CO_2 than NO_x emission reductions.

4.3 CO_2-NO_x substitution

One of the main purposes of this study is to assess the existence of substitutability between CO_2 and NO_x and the changes induced by regulatory changes. Our results indicate that CO_2 and NO_x are substitutes, and that - despite the reductions on the relative price CO_2-NO_x - substitution has increased during the period 2006-2009 in response to technological progress. For instance, the mean estimates of the Morishima elasticity CO_2-NO_x (M_{b_1,b_2}) fell from 0.906 in 2001-2004 to 0.411 in 2006-2009. The mean estimates of the corresponding elasticity NO_x-CO_2 (M_{b_2,b_1}) fell from 0.461 to 0.008. The asymmetry of the elasticities indicates that it is easier to substitute CO_2 by NO_x than NO_x by CO_2. In other words, firms are much sensitive to variations in CO_2 prices and willing to decrease CO_2 emissions at expenses of increasing NO_x emissions than the other way around.

Clearly, this is linked to the fact that CO_2 regulations have a much higher opportunity cost than the fee on NO_x. As mentioned before, a policy implication that could be derived from here is that the NO_x fee would have to be increased to a much larger extent than its current value in order to achieve large NO_x emission reductions. The increased substitution is also observed when classifying by fuel groups (that is, boilers using mainly biofuel, fossil fuels, or other fuels) and types of NO_x abatement technologies.

Kernel distributions of the elasticities between pollutants are depicted in Figures 5a and 5b. Using the bootstrapped Li et al. T_n-statiststics we tested the null hypothesis of equal density functions between the pre and post-implementation period of the EU ETS. We reject this null hypothesis; the elasticity of substitution CO_2-NO_x does tend to shift towards the left-hand side of the distribution. In the case of the elasticity NO_x-CO_2, we observe also that for many generating units the estimates are concentrated around zero. Further CO_2-NO_x substitution has increased because most firms implemented technical measures to allow for this substitution to take place (notably undertaken almost complete switching to biofuels). Given that after the implementation of the EU ETS some boilers are fully operating with biofuels (18% of the sample), we analyze the sensitivity of the estimates excluding those observations from the whole sample. As total fuel conversion to biofuels is a possible option for the remaining group, we should expect to observe a lower responsiveness of the relative intensity of emissions CO_2/NO_x to changes in relative prices, i.e., a higher value of the indirect Morishima elasticity of substitution. Indeed, our estimates indicate that elasticities of substitution between pollutants during 2006-2009 for this group are in general higher than those estimated previously for the same period. The values of M_{b_1,b_2} and M_{b_2,b_1} were on average 0.50 and 0.49, respectively.

4.4 Shadow Prices

Finally, we test for changes in the relative shadow price CO_2/NO_x. We find a reduced shadow price CO_2/NO_x for the period post introduction of the EU ETS. The relative shadow price CO_2/NO_x decreases from a mean of 0.000139 in 2001-2004 to a mean of 0.000097 in 2006-2009[19].

The Li et al. (2009) test leads us to reject the null hypothesis of equality in the relative shadow prices distributions between pre and post-implementation of the EU ETS. Figure 6 illustrates the kernel relative shadow prices distributions for CHP plants. The direction of the changes here in relative shadow prices is compatible with the constructed relative prices of the regulation for an average CHP calculated in section 2, which confirms that for CHP plants the CO_2 policy has become less stringent after the introduction of the EU ETS.

4.5 Synergies and Trade-offs between CO_2 and NO_x Emissions Reductions

Has the overlapping of climate policies led to increased NO_x emissions? Since pollutants are substitutes, we do find a tendency in this direction but NO_x emissions have not actually increased in practice for two reasons. First, there has been a significant technological development leading to reduced emissions for both pollutants. Second, contrary to what one could have expected- relative prices CO_2/NO_x prices have decreased for most firms in the combined heat and power sector due to the gradual reduction of the Swedish carbon tax for generating units within the EU-ETS and the simultaneous increase in the local NO_x charge in 2008. This is to say that in relative terms CO_2 policies in Sweden have become less stringent after the introduction of the EU ETS due to the variations in the levels of the local policies, and this has reduced the cost of compliance with both

[19] In our analysis, we have omitted year 2005 on the estimates. However, we also estimated the directional output distance function for a time window of four years during the period 2005-2008 to analyze the potential effects of earlier adjustments to the policies. For this period, the mean inefficiency was 0.189 and the CO_2-output and NO_x-output elasticities were -0.493 and -0.900, respectively. The mean CO_2-NO_x and NO_x-CO_2 elasticities were 0.444 and 0.046, respectively. Finally, the mean relative shadow price was equivalent to 0.000112. Note that though the magnitude of these elasticities is relatively similar to those found for 2006-2009, the value of the pollutant-output elasticities during the period 2005-2008 are slightly lower than those for the period 2006-2009, which confirm that substitution is relatively more limited in the shorter-run.

CO_2 and NO_x environmental regulations, as reflected by the reduced shadow prices and pollutant-output elasticities. A natural question that arises is what would have happened if the carbon tax was not phased out. In such a case, the relative price (CO_2/NO_x) during the period 2006-2009 would have been 60% larger than during the period 2001-2004 (i.e., 0.029 instead of 0.019). Based on our back of the envelope estimates for the CO_2-NO_x elasticity, we calculate that (NO_x/CO_2) emissions intensity would have increased from 0.186% to 0.341%, leading to 83% increased in NO_x emissions[20]. In reality, the emissions' intensity ratio (NO_x/CO_2) increased to a much lower extent than predicted (i.e., 0.207%) due to the combined effect of technological progress and the fact that the actual (CO_2/NO_x) relative price during the period 2006-2009 decreased sharply.

5. Conclusions

The implementation of environmental policies to reduce greenhouse gasses emissions does not only have a global impact, but can also bring local co-benefits (or costs) by reducing (increasing) other air pollutants due to substitution. These interactions have clear implications for policy design as many European countries are committed to reach the Kyoto obligations (though real action has been slow up to now) and there are currently multiple policies in place aiming to reduce CO_2 emissions. The question is what ancillary benefits (costs) we can expect from pursuing GHG reduction policies and local air pollution policies simultaneously. We explore this question formally by analyzing the patterns of substitution between CO_2 and NO_x in Sweden induced by the interaction of national and international environmental policies.

We modeled the pollution technology of generating units in the Swedish heat and power sector as a non-separable production process in which CO_2, NO_x and production are treated as joint outputs. We use a directional output distance function that counts for the simultaneous expansion of good outputs and contraction of bad outputs, which is a fair representation of the problem that

[20] We use the 2001-2004 indirect Morishima elasticity of substitution and emissions intensity ratio (NO_x/CO_2) as the baseline.

many regulated firms deal with. We chose a quadratic representation of the technology and subsequently derived the estimates of elasticities between CO_2 and NO_x and heat and power generation. Through this method, we computed the impact of the introduction of the EU ETS on the relative shadow price and substitutability between CO_2 and NO_x, as well as the substitutability between emissions and power generation for generating units included in the EU ETS. To evaluate the change in elasticities or relative shadow prices, we compared the probability distributions between periods by means of a kernel consistent density test.

Our results indicate that there are important interactions between the abatement efforts of CO_2 and NO_x emissions. Indeed, we find that in the combined heat and power generation sector CO_2 and NO_x are Morishima substitutes. Overall, the degree of substitution for CHP plants between these two pollutant increases after the introduction of the EU ETS as a response to technological development and regulatory changes that led to a reduced CO_2/NO_x relative price.

Our results also indicate that CO_2 is more sensitive to prices that NO_x. The value of the asymmetric Morishima elasticity NO_x-CO_2 (M_{b_2,b_1}) is lower than the value of the elasticity CO_2-NO_x, which means that CO_2 emissions are more likely to be reduced at the expense of increased emissions of NO_x than vice versa. Moreover, if the regulator is to encourage a large reduction in NO_x emissions, the charge must be increased to a much larger level than its current value. Regarding elasticities pollution-output, our results tell us that firms were able to decrease CO_2 and NO_x emissions per unit of output between the periods 2001-2004 and 2006-2009. The fact that technological development has been biased towards CO_2 emissions reductions implies that it has become easier to reduce CO_2 emissions than NOx emissions per unit of output.

Finally, the fact that generating units respond to variation in the relative prices of emissions by changing the intensity of their abatement efforts suggests that there is a need for policy coordination to avoid unintended effects of one policy instrument on the emissions of other pollutants. This is an area where much research is needed since we are experiencing a polycentric approach to climate change with mitigation and adaptation activities undertaken by multiple policy actors at a range of different scales.

References

Aigner, D.J., Chu, S.J., 1968. On estimating the industry production function. American Economic Review 58: 826–839.

Ambec, S. Coria, J., 2013. Prices vs Quantities with Multiple Pollutants, forthcoming in Journal of Environmental Economics and Management.

Blackorby, C., Russell, R.R., 1981. The Morishima elasticity of substitution; symmetry, constancy, separability, and its relationship to the Hicks and Allen elasticities. Review of Economic Studies 48: 147-158.

Blackorby, C., Russell, R.R., 1989. Will the real elasticity of substitution please standup? (A comparison of the Allen/Uzawa and Morishima elasticities). American Economic Review 79: 882–888.

Brännlund, R., Kriström, B., 2001. Too hot to handle? Benefits and costs of stimulating the use of biofuels in the Swedish heating sector. Resource and Energy Economics 23: 343-358.

Burtraw, D., Krupnick, A., Palmer, K., Paul, A., Toman, M., Bloyd, C., 2003. Ancillary benefits of reduced air pollution in the U.S. from moderate greenhouse gas mitigation policies in the electricity sector. Journal of Environmental Economics and Management 45: 650-673.

Caillaud, B., Jullien, B., and Picard, P., 1996. Hierarchical organization and incentives. European Economic Review 40: 687-695.

Färe, R., Grosskopf, S., Lovell, C.A.K., Yaisawarng, S., 1993. Derivation of shadow prices for undesirable outputs: a distance function-approach. Review of Economics and Statistics 75: 374-380.

Färe, R., Grosskopf, S., Noh, D.W., Weber, W., 2001. Shadow prices of Missouri public conservation land. Public Finance Review 29: 444-460.

Färe, R., Grosskopf, S., Noh, D.W., Weber, W., 2005. Characteristics of a polluting technology: theory and practice. Journal of Econometrics 126: 469-492.

Färe, R., Grosskopf, S., Weber, W., 2006. Shadow prices and pollution costs in U.S. agriculture. Ecological Economics 56: 89– 103.

Färe, R., Martins-Filho, C., Vardanyan, M., 2010. On functional form representation of multi-output production technologies. Journal of Productivity Analysis 33: 81-96.

Färe, R., Grosskopf, S., Pasurka, C. A., Weber, W., 2012. Substitutability among undesirable outputs. Applied Economics 44: 39-47.

Hayfield, T., Racine, J.S, 2008. Nonparametric Econometrics: The np Package. Journal of Statistical Software 27: 1-32. Available at www.jstatsoft.org/v27/i05/ (2011-11-27).

Hayfield, T, Racine, J.S., 2011. Nonparametric kernel smoothing methods for mixed data types: Package ‘np’. Available at http://cran.r-project.org/web/packages/np/ (2011-11-27).

Höglund, L., 2005. Abatement costs in response to the Swedish charge on nitrogen oxide emissions. Journal of Environmental Economics and Management 50: 102-120.

Kumar, S., Managi, S., 2011. Non-separability and substitutability among water pollutants: evidence from India. Environment and Development Economics. In press.

Li, Q., 1996. Nonparametric testing of closeness between two unknown distribution functions. Econometric Reviews 15: 261–274.

Li, Q., 1999. Nonparametric testing the similarity of two unknown density functions: local power and bootstrap analysis. Nonparametric Statistics 11: 189–213.

Li, Q., Maasoumi, E., Racine, J.S., 2009. A nonparametric test for equality of distributions with mixed categorical and continuous data. Journal of Econometrics 148: 186-200.

Löfgren, Å., Muller, A.., 2010. Swedish CO_2 emissions 1993-2006: An application of decomposition analysis and some methodological insights. Environmental and Resource Economics 47: 221-239.

Millock, K., Sterner, T., 2004. NO_x Emissions in France and Sweden. In Choosing Environmental Policy, Comparing Instruments and Outcomes in the United States and Europe. W. Harrington, R.D. Morgenstern and T. Sterner (editors), RFF Press, Washington, DC, United States.

Murty, M.N., S. Kumar, Dhavala, K.K., 2007. Measuring environmental efficiency of industry: a case study of thermal power generation in India. Environmental and Resource Economics 38: 31-50.

SEPA, 2003. Kväveoxidavgiften ett effektivt styrmedel Utvärdering av NO_x-avgiften. Rapport 5335. http://www.naturvardsverket.se/Documents/publikationer/620-5335-3.pdf

SEPA, 2007. Economic Instruments in Environmental Policy. Report 5678. http://www.energimyndigheten.se/Global/Engelska/News/620-5678-6_webb.pdf

SEPA, 2009. Naturvårdsverkets föreskrifter och allmänna råd om utsläppsrätter för koldioxid. NFS 2007:5 Konsoliderad med NFS 2009:6. Stockholm.

Sterner, T., Höglund,L., 2006. Refunded emission payments theory, distribution of costs, and Swedish experience of NO_x abatement. Ecological Economics 57: 93-106.

Sterner, T., Turnheim, B., 2009. Innovation and diffusion of environmental technology: Industrial NO_x abatement in Sweden under refunded emission payments. Ecological Economics 68: 2996-3006.

Stiroh, K.J., 1999. Measuring Input Substitution in Thrifts: Morishima, Allen-Uzawa, and Cross-Price Elasticities. Journal of Economics and Business 51:145–157

Svensk Fjärrvärme, 2011. Combined heat and power. Available at www.svenskfjarrvarme.se (2011-10-05).

Swedish Energy Agency, 2009. Energy in Sweden 2009. Stockholm.

Xepapadeas, A. and de Zeeuw, A., 1999. Environmental Policy and Competitiveness: The Porter Hypothesis and the Composition of Capital. Journal of Environmental Economics and Management, 37: 165-182.

Figure 1. The CO_2 Tax in Sweden during 2001-2009 for Combined Heat and Power plants (CHP)

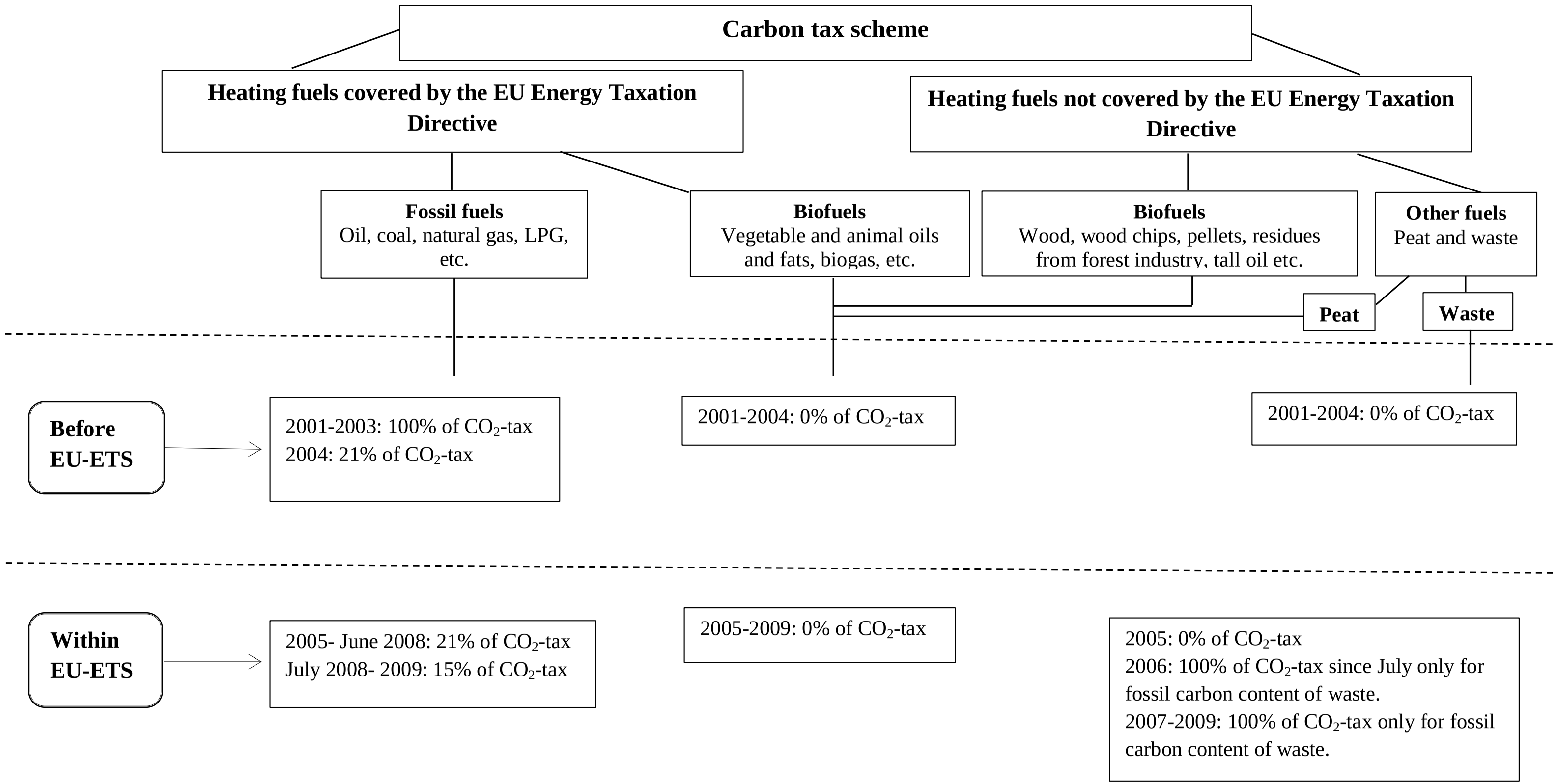
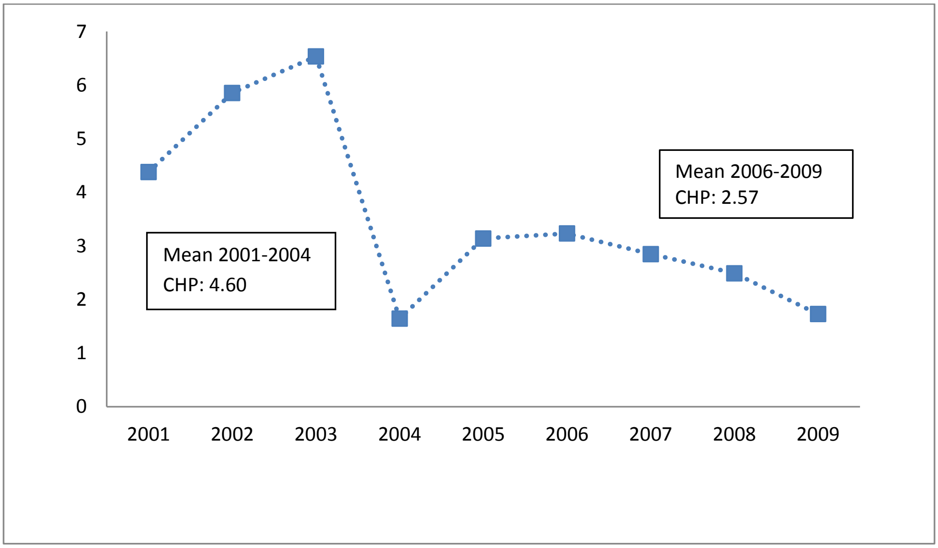


Figure 2. Average relative opportunity costs CO_2/NO_x per MWh for CHP plants[1,2].



(1) The opportunity cost of CO_2 emissions is calculated as the sum of the carbon tax plus the CO_2 EU ETS price (mean of forward contracts 2007-2013).
(2) We compute the relative opportunity cost CO_2/NO_x per MWh per boiler and average the values across boilers.

Figure 3: Frontiers for a hypothetical technically efficient boiler in 2001 and 2009

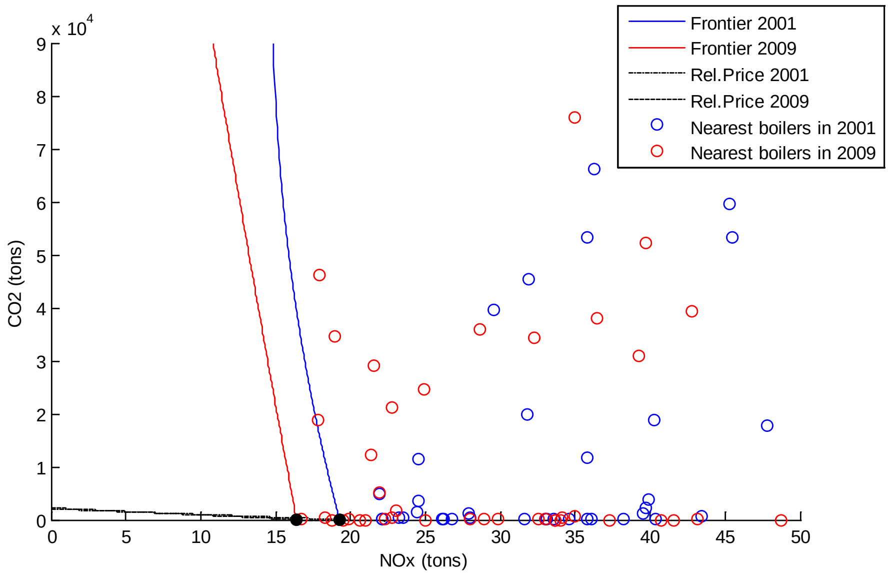


Figure 4a: Kernel distribution of elasticities CO_2-output[21]

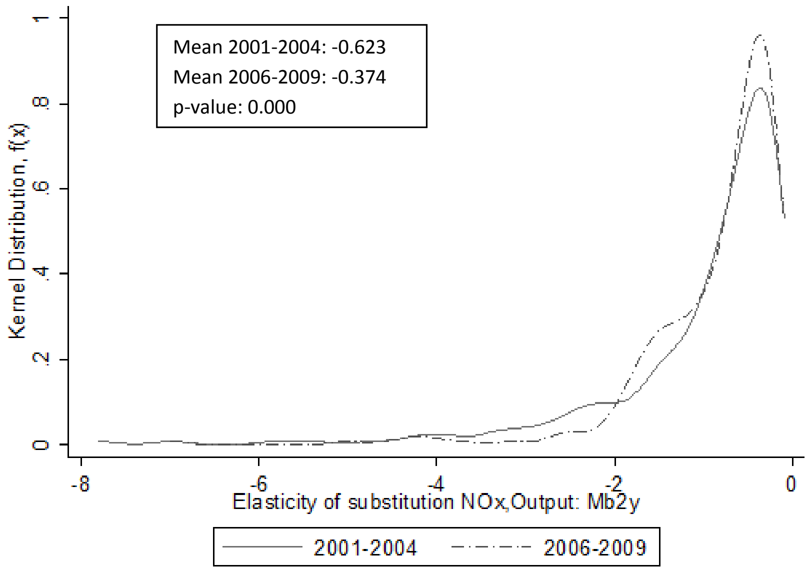
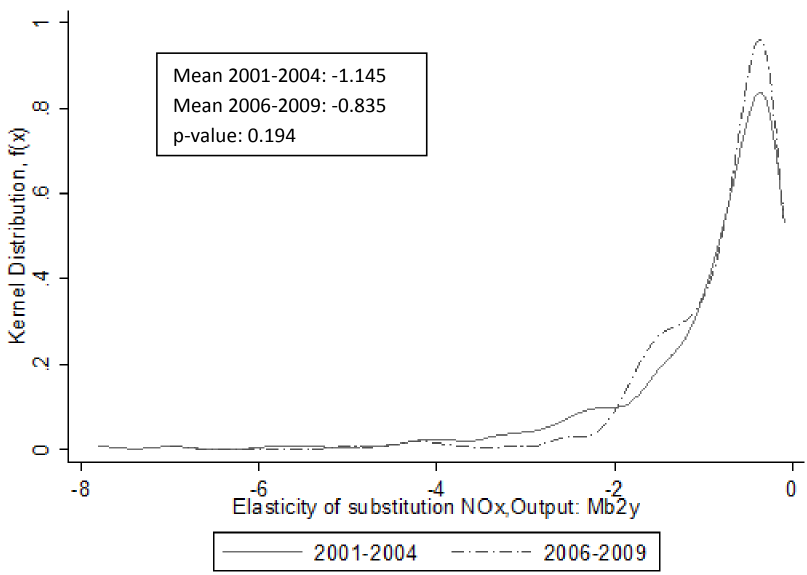


Figure 4b: Kernel distribution of elasticities NO_x-output



[21] For convenience in presentation some few elasticity estimates of CO_2-output lower than -5and NO_x-output less than -8 are not shown in the graphs.

Figure 5a: Kernel distribution of elasticities CO_2-NO_x[22]

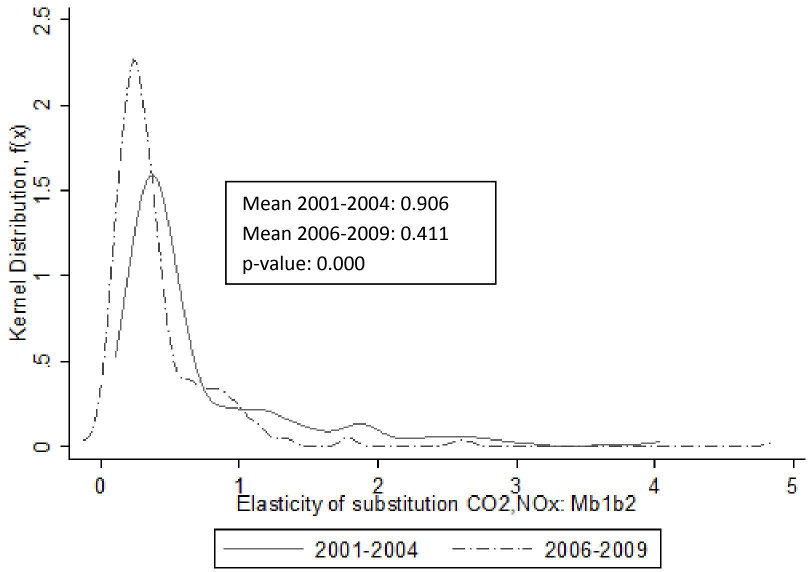
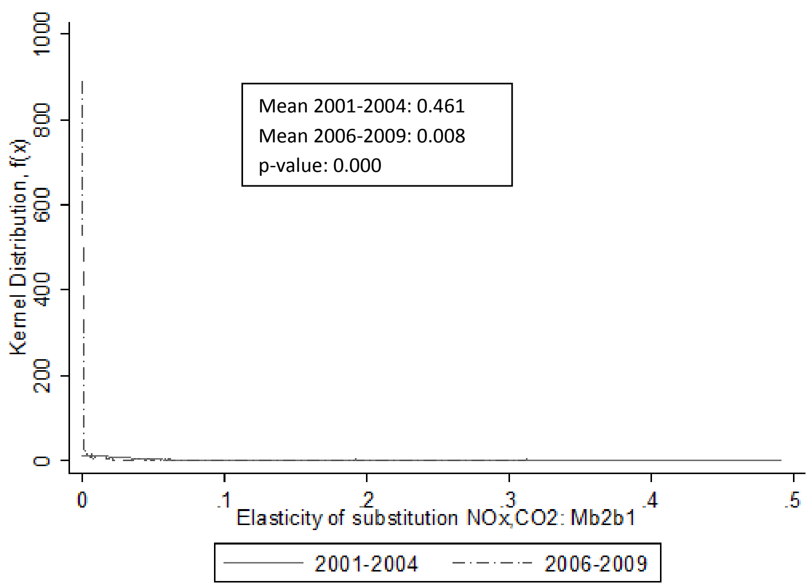


Figure 5b: Kernel distribution of elasticities NO_x-CO_2



[22] For convenience in presentation some few elasticity estimates of CO_2-NO_x greater than 5 and NO_x-CO_2 higher than 0.5 are not shown in the graphs.

Figure 6: Kernel distribution of relative shadow prices CO_2/NO_x - CHP plants

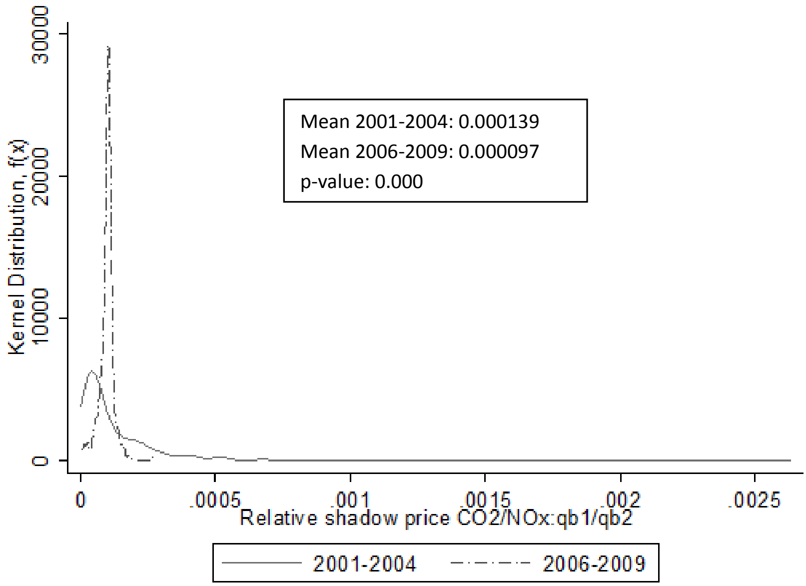


Table 1. Descriptive statistics

Variable	Description	Mean	Standard Deviation	Min	Max
		2001-2004			
Y	Useful energy (MWh)	239286	227590	26119	1292804
b_1	CO_2 (tons)	25098	53406	6.5	350320
b_2	NO_x (tons)	46.7	33.9	4.4	185.8
b_1/Y	CO_2(tons)/GWh	81.4	120.7	0.2	428.5
b_2/Y	NO_x(tons)/GWh	0.245	0.093	0.032	0.561
x_1	Installed capacity (MW)	72.3	80.2	6	600
x_2	Fuel consumption (MWh)	253096	237637	23642	1247052
x_3	Labor (nr. of employees)	37.7	46.2	1	220
		2006-2009			
Y	Useful energy (MWh)	242639	258072	25091	1232591
b_1	CO_2 (tons)	20355	57016	0	382960
b_2	NO_x (tons)	42.1	31.9	5.0	167.7
b_1/Y	CO_2(tons)/GWh	68.6	118.5	0.0	427.3
b_2/Y	NO_x(tons)/GWh	0.232	0.099	0.027	0.683
x_1	Installed capacity (MW)	70.5	76.9	6	600
x_2	Fuel consumption (MWh)	255411	272454	23726	1334877
x_3	Labor (nr. of employees)	44.6	50.8	2	266

Table 2. Parameter estimates of the quadratic directional output distance function

Coefficients	Variable	Before EU ETS 2001-2004	After EU ETS 2006-2009
α	Constant	-0.005	0.004
α_1	x_1	-0.002	-0.002
α_2	x_2	0.592	0.616
α_3	x_3	0.033	0.021
β_1	y	-0.740	-0.773
θ_1	b_1	-0.001	0.010
θ_2	b_2	0.261	0.217
α_{11}	x_1^2	0.004	0.014
α_{12}	x_1x_2	-0.228	-0.339
α_{13}	x_1x_3	0.018	0.133
α_{22}	x_2^2	0.041	-0.284
α_{23}	x_2x_3	-0.042	0.067
α_{33}	x_3^2	-0.004	-0.085
β_2	y^2	-0.149	-0.152
θ_{11}	b_1^2	-0.004	-0.0002
θ_{22}	b_2^2	-0.146	-0.149
η_{11}	x_1b_1	0.000	0.007
η_{21}	x_2b_1	0.015	-0.005
η_{31}	x_3b_1	0.005	0.001
η_{12}	x_1b_2	0.104	0.090
η_{22}	x_2b_2	0.138	0.295
η_{32}	x_3b_2	0.022	-0.017
μ_1	yb_1	-0.003	-0.001
μ_2	yb_2	-0.145	-0.151
θ_{12}	b_1b_2	0.001	-0.001
δ_1	x_1y	0.104	0.097
δ_2	x_2y	0.152	0.291
δ_3	x_3y	0.027	-0.016
ϕ_1	Year2	-0.004	-0.010
ϕ_2	Year3	-0.004	-0.005
ϕ_3	Year4	-0.005	-0.002

Note: x_1 is installed capacity, x_2 is fuel consumption, x_3 is labor, y is useful energy, b_1 is CO_2 and b_2 is NO_x. Year2, Year3, and Year4 are yearly dummies.

Table 3. CO_2 and NO_x abatement technology adoption

Year	% of boilers according to fuel type					% boilers with NOx technology			
	Biofuel[a] (>80%)	Mostly Biofuel[b]	Fossil fuel[a] (>80%)	Mostly Fossil Fuel[b]	Mostly Others[b]	None	Only PCT	Only CT	PCT and CT
2001	69.5	76.8	12.2	12.6	15.9	20.7	19.5	41.5	18.3
2002	61.0	72.0	12.2	12.6	14.6	20.7	19.5	41.5	18.3
2003	64.6	75.6	14.6	13.5	15.9	20.7	18.3	41.5	19.5
2004	65.9	76.8	13.4	12.6	15.9	20.7	18.3	40.2	20.7
2006	74.4	78.0	12.2	12.6	14.6	17.1	18.3	41.5	23.2
2007	69.5	76.8	12.2	10.8	13.4	17.1	15.9	39.0	28.0
2008	72.0	78.0	11.0	10.8	13.4	15.9	14.6	39.0	30.5
2009	72.0	78.0	11.0	10.8	13.4	14.6	14.6	39.0	31.7

[a] Biofuel and fossil fuel (>80%) includes all the boilers with fuel share greater than or equal to 80%.
[b] Mostly biofuel contains all the boilers with fuel share greater than fossil fuel share and the rest. Mostly fossil fuel includes boilers with fuel share higher than biofuel share and the rest. Mostly others is defined in a similar manner for other fuels (peat and waste).

Chapter IV

Diffusion of NO_x abatement technologies in Sweden*

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Abstract

Though economists argue for the use of single instruments, we often observe the use of multiple instruments in actual regulations. These may include permit schemes, taxes, fees, subsidies and emission standards. In order to evaluate these combinations and to better understand their effects, we need more empirical analysis of how they interact. They might, for example, be either complements or substitutes; this might even vary between different types of instrument. As a case study we look at detailed data of NO_x emissions from large combustion plants in Sweden. These are regulated both by a refunded charge and at the same time plant level emission standards. We study the adoption of abatement technologies under the combined effect of these charges and standards. The results indicate that the net charge has an effect and one that is complementary to the standards, but only for end-of-pipe post-combustion technologies.

Keywords: technology diffusion, NO_x abatement technologies, environmental regulations, refunded emission charge

JEL Classification: H23, O33, O38, Q52

1 Introduction

The long-run impact of emission regulations is mainly determined by the incentives they provide for innovation and diffusion of more environmentally benign technologies. In Sweden, a charge on NO_x emissions from large

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combustion plants was introduced in 1992, as a complement to the existing system of individual emission standards (SEPA, 2003). The regulation, under which the charge revenues are refunded back to the regulated firms in proportion to energy output, was explicitly designed to affect technology investment (Sterner & Turnheim, 2009). Judging from the significant reductions in emission intensities achieved since the introduction of the policy, this objective would appear to have been reached.

However, changes in emission intensities is the combined result of upfront investments in abatement technology, fuel switching, and improved knowledge of how to optimize the combustion process (Höglund-Isaksson & Sterner, 2009). Sterner & Turnheim (2009) sought to disentangle the separate contributions of technology diffusion and innovation to reductions in emission intensities among the regulated plants and found both factors very important. In this paper, we focus on one of these factors: the diffusion process of NO_x-reducing technology. The purpose is to analyze the factors that affect the timing and decision to invest in NO_x abatement technologies.

Technology diffusion generally follows an S-shaped pattern over time, in which the number of adopters initially increases slowly until a point in time at which adoption starts to increase rapidly, followed by a period of leveling off when most potential adopters have already invested. Early literature such as Griliches (1957) tried to explain this pattern with epidemic models capturing the spread of knowledge and information about the new technology (Popp, 2010).

More recent literature has attempted to find mechanisms which explain differences in preferred dates of adoption among potential adopters. Karshenas & Stoneman (1993) described three such mechanisms: rank, stock and order effects. The rank effects results from the different inherent characteristics of the firms, such as size and industrial sector, which affects the returns from adoption of the new technology and in turn the preferred adoption dates. A stock effect is present if the benefit of the marginal adopter decreases as the number of previous adopters increases, e.g., because previous adopters increase their output and thereby depress industry prices and the return for the next adopter. An order effect is present if early adopters can achieve a higher return than the late adopters, e.g., because the first adopters can preempt the pool of skilled labor. Karshenas & Stoneman (1993) also developed a decision-theoretical model which they linked to a proportional hazard model to empirically assess the influence of rank, stock, order and epidemic effects on the pattern of diffusion.

Popp (2010) used the framework in Karshenas & Stoneman (1993) to analyze adoption of NO_x control technologies at coal-fired power plants in the US. He found that expectations of future technological advances slow down the diffusion of existing combustion modification technologies. Due to the differences in regulation across states, he could also identify environmental regulations as the dominating determinants behind adoption. Other studies have also found environmental regulations to have a major influence on technology adoption. Gray & Shadbegian (1998) found that clean technology investment among paper mills was significantly more likely in states with more stringent water and air pollution regulation while Kerr & Newell (2003) concluded that increasing regulatory stringency for leaded gasoline significantly increased the propensity for lead-reducing technology adoption. Snyder *et al.* (2003) on the other hand found that environmental regulation did not have a significant impact on the adoption of cleaner technology in chlorine manufacturing but did induce technological change by reducing the demand for chlorine and thereby accelerating closures of dirtier facilities.

The majority of these studies analyze the impact on environmental technology adoption of different regulatory regimes or differences in quantitative regulations. In this paper we analyze instead the effect of a market-based instrument in combination with quantitative regulations on the propensity to adopt environmental technology. Moreover, unlike most previous studies analyzing diffusion of NO_x abatement technologies, which focus on diffusion within one industry, we compare environmental technology adoption across different industry sectors.

In the next section, we describe the Swedish NO_x policies in more detail and the incentives they provide for the regulated plants. Section 3 gives a brief overview of different NO_x-reducing technologies. Section 4 introduces the theoretical framework and our empirical model. Section 5 describes the data and section 6 the results. Section 7 concludes.

2 NO_x policies

This section describes the Swedish NO_x policies for large combustion plants in the form of the refunded NO_x charge and individual emission standards. It also includes a section describing the incentives for emission reductions provided under this combination of policies.

2.1 The Swedish NO_x charge

The Swedish charge on NO_x emissions from large combustion plants was introduced in 1992. At the time, close to 25% of Swedish NO_x emissions came from stationary combustion sources and the charge was seen as a faster and more cost-efficient way of reducing NO_x emissions than the existing system of individual emission standards (SEPA, 2003).

Because NO_x emissions vary significantly with temperature and other combustion parameters (Sterner & Höglund-Isaksson, 2006), continuous measurement of the flue gas was required to implement the charge. The installation of the measuring equipment was judged too costly for smaller plants and the charge therefore only imposed on larger boilers. In order not to distort competition between larger plants and smaller units not subjected to the charge, a scheme was designed to refund the charges back to the regulated plants in proportion to energy output. Energy output within the NO_x charge system is measured in terms of so-called useful energy, which can be either in the form of electricity, steam or hot water depending on end-use[1] Regulated entities belong to the heat and power sector (between 1992 and 2009 on average 52% of total useful energy production in the system), the pulp and paper industry (on average 23% of useful energy production), the waste incineration sector (15%) and the chemical (5.5%), wood (3.1%), food (1.7%) and metal (1.0%) industries. Initially the charge only covered boilers and gas turbines with a yearly production of useful energy of at least 50 GWh, but in 1996 the threshold was lowered to 40 GWh and in 1997 further lowered to 25 GWh per year (Höglund-Isaksson & Sterner, 2009).

From 1992 to 2007, the charge was 40 SEK/kg NO_x[2]. In 2008, the charge was raised to 50 SEK/kg NO_x following a series of reports from the Swedish Environmental Protection Agency which indicated that the impact of the charge system had diminished over the years (SEPA, 2012). In real terms, the charge had decreased over time and the increase to 50 SEK in 2008 was in practice a restoration of the charge to the real level in 1992.

2.2 Individual emission standards

Individual emission standards for NO_x emissions from stationary sources were introduced in 1988 and thus were already in place when the charges were introduced (Höglund-Isaksson & Sterner, 2009). Any quantitative emission limits are determined by county authorities[3] and may vary with industry sector. Emission limits commonly cover nitrogen oxides, carbon dioxide, carbon monoxide, sulfur, ammonia and particulate matter (SEPA, 2012).

In 2003, the Swedish Environmental Protection Agency (SEPA) conducted an evaluation of the effect of the emission standards compared to the NO_x charge during the period 1997-2001[4], finding that emission intensities for boilers not subject to emission restrictions were higher than for

[1]In the heat and power industry, useful energy is most often the amount of energy sold. For other industries, it is defined as the steam, hot water or electricity produced in a boiler and used in the production process or for heating of plant facilities (SEPA, 2003).

[2]Approximately 4€/ kg $NO_x\dot{N}O_x$ is measured in kilograms of NO_2. In air, NO is naturally converted to NO_2 and vice versa and the equilibrium ratio of NO to NO_2 is determined by atmospheric conditions. A kilogram of NO is converted into units of kilograms of NO_2 by multiplying by the factor 46/30 (the molecular mass ratio).

[3]They evaluate the plants with respect to the Environmental Code and issue permits which may entail quantitative restrictions on emissions of polluting substances.

[4]SEPA (2003) analyzed 228 boilers (of a total of 448 at the time) that were subject to charges during the period 1997-2001. Among the 228 boilers, 140 were subject to restrictions on NO_x emissions. The restrictions were most often in terms of yearly averages and in units of mg NO_x per megajoule (MJ) of fuel but sometimes in other units, e.g., mg NO_x per m^3 of fluegas.

boilers with restrictions. Emission intensities also remained unchanged for boilers without restrictions during those years. In contrast, emission intensities were 11% lower in 2001 compared to 1997 for boilers with restrictions. Relevant to note is that boilers without emission standards often belonged to smaller plants and that fewer boilers in the wood and pulp and paper industry were subject to restrictions, while restrictions were more common for boilers in the waste incineration and heat and power sector. Because emissions were generally much below the quantitative restrictions, the conclusion from SEPA (2003) is that, for boilers in the heat and power and waste incineration sectors, the NO_x charge was more effective than the restrictions in reducing NO_x emissions. Figure 1 illustrates that the emission standards do not appear to have been the binding factor limiting emissions in 2001 for any of the boilers which were part of the NO_x charge system in both 1997 and 2001 and which were subject to an emission standard in terms of mg NO_x per MJ of fuel at the time when SEPA did its audits.

Since the SEPA (2003) evaluation, there has been no comprehensive survey of the emission standards and how they have developed over time. SEPA kindly supplied us with data on emission standards in place in 2012 for 42 out of 50 firms in the NO_x system, randomly selected for an interview survey for the SEPA (2012) report. The majority of the quantitative restrictions were in terms of mg NO_2 per MJ of fuel[5]. Figure 2 illustrates the emission standards for firms in this subsample with an emission standard in equivalent units of mg NO_x per MJ of fuel. In most cases, we do not know in which year the standard came into effect and for the comparison with actual emissions we therefore illustrate emissions as an average over the period 1992-2009. Nevertheless, from Figure 2 it appears as if on average the standard has not been binding over the period. However, we cannot rule out the possibility that the standard was binding at some point in time.

In the interviews at the surveyed firms, some respondents viewed the standards as a more important factor than the NO_x charge. Respondents also said that the standards made it more difficult to trade off different emission-reducing measures. The often strong negative correlation between CO and NO_x emissions makes quantitative restrictions on carbon monoxide especially relevant. It appears that authorities generally have increased the stringency of restrictions on CO since the 1990s, making it more difficult in later years to trade off NO_x emissions for emissions of CO. Some of the interview respondents also claimed that authorities in some counties issue more stringent emission standards compared to other counties (SEPA, 2012) - an observation which we attempt to control for in our estimations.

[5]Out of 81 different forms of quantitative restrictions for the boilers at these firms, 52 were in terms of mg NO_2 per MJ of fuel, 25 in terms of mg NO_2 per m^3 of flue gas or ton of NO_2 per year and 4 in terms of mg NO_2 per ton of pulp or paper. One heat and power plant and a production line at one waste incineration plant instead had technology standards mandating SNCR (or equivalent) and SCR, respectively.

2.3 Incentives provided by NO_x charge and standards

To describe the incentives provided by the NO_x charge and the most common form of emission standard, we consider a firm (with only one boiler for expositional clarity) which faces a refunded NO_x charge at the level of σ per unit of emissions. The cost of generating q_i units of useful energy with emission intensity ε_i is $C_i(q_i, \varepsilon_i)$ for firm i. Firm-level emissions is given by $e_i = \varepsilon_i q_i$ and total emissions from all firms covered by the NO_x charge by $E = \sum_i e_i$. With total production of useful energy $Q = \sum_i q_i$ over all firms and boilers, we define the average emission intensity $\overline{\varepsilon} = \frac{E}{Q}$. The firm chooses the level of useful energy production and emission intensity which minimize the cost of the NO_x regulation and satisfy a minimum level of useful energy, $\overline{q}_i$, and an emission standard, $\overline{\xi}_i$ (equal to infinity in case of the absence of a standard). Since the emission standards are often expressed in terms of units of emissions per unit of input energy, we write input energy as $\frac{q_i}{\varphi_i}$ where φ_i is the energy efficiency of the boiler, and define the standard as a constraint on $\frac{e_i \varphi_i}{q_i}$.

This minimization problem can be written

$$\min_{q_i, \varepsilon_i} C_i(q_i, \varepsilon_i) + \sigma \left[\varepsilon_i - \overline{\varepsilon}\right] q_i \tag{1}$$

subject to

$$q_i \geq \overline{q}_i \tag{2}$$

$$\varepsilon_i \varphi_i \leq \overline{\xi}_i \tag{3}$$

The average net NO_x charge per unit of useful energy, $\sigma \left[\varepsilon_i - \overline{\varepsilon}\right]$ is positive for a firm with an emission intensity higher than the average emission intensity $\overline{\varepsilon}$, i.e., $\varepsilon_i^* > \overline{\varepsilon}$, and negative for a firm with an emission intensity which is lower than average, i.e., $\varepsilon_i^* < \overline{\varepsilon}$ The two first-order conditions for the cost-minimizing energy production, q_i^*, and emission intensity, ε_i^*, are

$$\frac{\partial C_i(q_i^*, \varepsilon_i^*)}{\partial q_i} = \lambda_{\overline{q}_i} - \sigma \left[\varepsilon_i^* - \overline{\varepsilon}\right] \left[1 - \frac{q_i^*}{Q}\right] \tag{4}$$

$$-\frac{\partial C_i(q_i^*, \varepsilon_i^*)}{\partial \varepsilon_i} \frac{1}{q_i} = \lambda_{\overline{\xi}_i} \frac{\varphi_i}{q_i} + \sigma \left[1 - \frac{q_i^*}{Q}\right] \tag{5}$$

with the complementary slackness conditions

$$\lambda_{\overline{q}_i} \geq 0,\ \lambda_{\overline{q}_i}[q_i^* - \overline{q}_i] = 0$$

$$\lambda_{\overline{\xi}_i} \geq 0,\ \lambda_{\overline{\xi}_i}[\varepsilon_i^* \varphi_i - \overline{\xi}_i] = 0$$

The firm should choose the useful energy production and emission intensity that makes marginal cost equal to the shadow price of useful energy, net of the net marginal payment for the NO_x charge (condition (4)), while also fulfilling condition (5), which sets the marginal abatement cost equal to the sum of the shadow price on emissions related to constraint (3) and the marginal NO_x charge[6]. It is quite natural to assume that constraint (2) is binding with a shadow price of useful energy which is larger than zero. If a standard is so lax that (3) is not binding or no standard exists then (5) reduces to

$$-\frac{\partial C_i(q_i^*, \varepsilon_i^*)}{\partial \varepsilon_i}\frac{1}{q_i} = \sigma\left[1 - \frac{q_i^*}{Q}\right] \tag{6}$$

Comparing (5) and (6), we see that if the marginal cost is non-decreasing in the emission intensity, a boiler with a binding individual emission standard (i.e., $\lambda_{\bar{\xi}_i} > 0$) should choose a lower emission intensity than a comparable boiler without a binding emission standard (operating at the same level of efficiency and producing the same level of output). This is to say, a binding standard induces the boiler to operate at a marginal cost of abatement which is higher than the NO_x charge.

3 NO_x abatement technologies

As shown by Sterner & Höglund-Isaksson (2006) and demonstrated in the previous section, the system of refunded emission charges taxes firms which have higher than average emission intensities and therefore pay more in charges than they receive in refunds and it rewards firms which have lower than average emission intensities and receive a net refund. Therefore, the refunded NO_x charge encourages competition among the regulated plants for the lowest emissions per unit of useful energy. The policy should therefore spur adoption of technologies which decrease emission intensities. Such technologies include both purely emission reducing technologies and technologies which improve energy efficiency.

NO_x-emission reducing technologies can be divided into combustion and post-combustion technologies. Combustion technologies are designed to inhibit the formation of NO_x in the combustion stage, e.g., by lowering temperature, controlling air supply or enhancing the mixing of flue gases. Examples of such technologies installed at the Swedish plants are flue gas recirculation, ECOTUBE technology, injection technology, low-NO_x burner, reburner, over-fire-air, rotating over-fire-air and ROTAMIX technology (Höglund-Isaksson & Sterner, 2009).

[6] The adjustment in (5) reflects the fact that a firm with a larger share in total useful energy $\frac{q_i^*}{Q}$ pays a lower effective charge on emissions than a firm with a smaller share. In (4), it also implies that an above average emitter pays a lower net NO_x charge and a below average emitter gets a lower marginal net subsidy with a larger market share (Fischer, 2011). See Fischer (2011) for more details on the incentives provided by the refunded charge. In practice, the average boiler share in total useful energy is 0.3% with a maximum of 4.1%. At firm level, the share is on average 2.1%, with a maximum of 11.7%, suggesting that the market share distortion is perhaps relevant only for the largest heat and power producers.

Post-combustion technologies instead reduce NO_x in the flue gases after the combustion stage, either through catalytic or non-catalytic reduction of NO_x compounds. Selective catalytic reduction (SCR) uses ammonia or urea to reduce NO_x into water and molecular nitrogen (N_2) on catalytic beds at lower temperatures. SCR is highly efficient in reducing NO_x emissions but is a large and costly installation. Selective non-catalytic reduction (SNCR) on the other hand does not require catalysts and cooling of the flue gases and is therefore less costly but also less efficient (Höglund-Isaksson & Sterner, 2009).

Flue gas condensation is a technology which improves energy efficiency and has been adopted by many of the regulated Swedish plants. It recovers heat from the flue gases and improves energy efficiency without increasing NO_x emissions (Höglund-Isaksson & Sterner, 2009). This installation would therefore help to reduce a boiler's emission intensity and thereby increase the firm's net refund.

One important determinant of adoption is naturally investment cost. The cost of installing combustion technologies are highly variable across different boilers. Costs depend on size, purification requirements, system of injection, type of chemicals used and the complexity of the control system. Investment costs for the post-combustion technologies SCR and SNCR are also boiler and plant specific and vary with boiler capacity, among other things (SEPA, 2012). Moreover, some technologies are not commercially available below certain size thresholds (Sterner & Turnheim, 2009). Technology adoption also depends on access to information and degree of involvement in R&D and innovation activities (Sterner & Turnheim, 2009), which would seem to support the existence of learning effects.

This brief overview illustrates that there is a wide variety of technologies for plant managers to choose from when responding to the NO_x regulation. Moreover, because post-combustion allows firms to choose emissions independently from output to a much larger extent than the combustion technologies, the adoption of these two types of technologies might differ in responsiveness to the charge[7]. In our empirical analysis, we follow Popp (2010) and group the NO_x abatement technologies into two main categories to separately analyze the determinants of adoption for combustion technologies versus post-combustion technologies. Additionally, we also analyze investment in flue gas condensation because the NO_x charge system's focus on emission intensities may have increased the attractiveness of not only emissions-reducing but also energy efficiency improving technologies.

[7] Sterner & Turnheim (2009) found that, as expected from their characteristics, SCR followed by SNCR provided the most significant and sizable reductions in emission intensities.

4 Model of the investment decision

We use the framework in Karshenas & Stoneman (1993) and consider a situation in which a firm has the choice to install a new technology in a boiler i which is included in the refunded NO_x charge system. The cost of doing the installation at time t is $I(Z_i(t), L_i(t), S_i(t), t)$ where $Z_i(t)$ is a vector of boiler-specific characteristics which may affect investment costs. $L_i(t)$ is a vector of the number boilers at the plant and firm that unit i belongs to and that may give rise to internal learning effects which decrease investment costs. $S_i(t)$ is the stock of boilers already installed with the new technology in the industry of unit i which could affect investment costs if there are external learning effects.

By switching to the new technology, the gross profit gain of the boiler in period t increases by $g_i(t) = g(R_i(t), Z_i(t), L_i(t), S_i(t), t)$, where $R_i(t)$ is the level of regulatory stringency for boiler i in period t before adoption. The net present value of making the investment at time t is

$$V_i(t) = \int_t^{\infty} g(R_i(\tau), Z_i(\tau), L_i(\tau), S_i(\tau), \tau)e^{-r(\tau-t)}d\tau - I(Z_i(t), L_i(t), S_i(t), t)$$

Following Karshenas & Stoneman (1993), we specify the conditions which determine the investment decision: the profitability condition and the arbitrage condition. Clearly, for adoption to be considered at all, it is necessary that the investment yields positive profits, i.e.,

$$V_i(t) > 0 \tag{7}$$

Furthermore, for it not to be profitable at time t to wait longer to adopt, it is necessary that

$$y_i(t) \equiv \frac{d(V_i(t)e^{-rt})}{dt} \leq 0$$

Differentiating with respect to t we get

$$\begin{aligned} y_i(t) = -g(R_i(t), Z_i(t), L_i(t), S_i(t), t) + rI(Z_i(t), L_i(t), S_i(t), t) \\ -\frac{dI(Z_i(t), L_i(t), S_i(t), t)}{dt} \leq 0 \end{aligned} \tag{8}$$

There are various factors that we cannot observe which also affect the timing and decision to adopt. We therefore introduce the stochastic term ε which represents these unobserved factors. If we assume that ε is identically distributed across the firms and over time with the distribution function $F_\varepsilon(\epsilon)$, the condition that it must not be profitable to postpone adoption to a later date becomes

$$y_i(t) + \varepsilon \leq 0$$

If we also consider the optimal time of adoption for firm i, t_i^*, a random variable with distribution function $F_i(t)$, we can write

$$F_i(t) = \Pr\{t_i^* \leq t\} = \Pr\{\varepsilon \leq -y_i(t)\} = F_\varepsilon(-y_i(t)) \qquad \forall\, i, t$$

To estimate $F_i(t)$, we start from the hazard rate $h_i(t) = \frac{f_i(t)}{1-F_i(t)}$, where $f_i(t)$ is the probability distribution of t_i^*. The hazard rate is defined as the conditional probability of adoption at time t, given that the firm has not adopted before t[8].

As is common in the adoption literature (e.g., Karshenas & Stoneman (1993), Popp (2010) and Kerr & Newell (2003)), we estimate a proportional hazard model of the form

$$h_i(t) = \lambda_0(t) \exp(X_{it}'\beta)$$

where $\lambda_0(t)$ is the so called baseline hazard and X_{it} is composed of the vectors $R_i(t)$, $Z_i(t)$, $L_i(t)$ and $S_i(t)$ which are likely to affect whether the arbitrage condition in (8) is fulfilled. A variable which negatively affects $y_i(t)$ should increase the hazard rate and vice versa.

The baseline hazard $\lambda_0(t)$ is common to all units. We estimate semi-parametric Cox proportional hazard models because the Cox model has the advantage that it does not require any assumptions about the shape of the baseline hazard. The Cox model is estimated using the method of partial likelihood. A fully parametric proportional hazard model can be more efficiently estimated by maximum likelihood but is less robust because it entails the risk of misspecifying the baseline hazard (Cleves *et al.*, 2004).

For simplicity, we have so far only discussed the adoption of a single technology. The situation we are considering is however one where the plant managers can choose between three different types of technologies. Similar to Stoneman & Toivanen (1997) who consider diffusion of multiple technologies, we define $g_{i,k}(t)$ to be the gross profit gain at t from adoption of technology k relative to the no adoption scenario and $v_{i,k}(t, \tau_k)$ to be an additive synergistic gross profit gain, which is the increase in gross profit at time t from adoption of technology k at τ_k ($\tau_k \leq t$) relative to the prior technological state[9]. The total gross profit gain in time t from adoption of technology k at τ_k is then given by $g_{i,k}(t) + v_{i,k}(t, \tau_k)$.

We now specify that $g_{i,k}(t)$ is a function of the previously discussed explanatory variables with $g_{i,k}(t) = g_k(R_i(t), Z_i(t), L_{i,k}(t), S_{i,k}(t), t)$ with $L_{i,k}(t)$ the number of boilers at the plant and firm with technology k installed

[8]The hazard rate is an event rate per unit of time. In the case of technology adoption, a hazard rate might be intuitively thought of as the number of adopters divided by the number of units that have not still adopted, i.e., the survivors, at time t.

[9]This additive profit gain could e.g. be the additional net decrease in production and regulatory costs from installing a post-combustion technology when the boiler is already equipped with a combustion technology relative to when it is not.

and $S_{i,k}(t)$ the number of boilers in the industry of unit i installed with technology k. Further, we specify $v_{i,k}(t, \tau_k)$ as a function of the prior technological state $D_i(\tau_k)$ at the time of adoption of technology k as well as rank, stock and learning effects, so that we can write $v_{i,k}(t, \tau_k) = v(D_i(\tau_k), Z_i(t), L_{i,k}(t), S_{i,k}(t))$. We can now specify our technology-specific hazard function as

$$h_{i,k}(t) = h_k(R_i(t), D_i(t), Z_i(t), L_{i,k}(t), S_{i,k}(t), t) \tag{9}$$

We separately estimate a proportional hazard model for combustion, post-combustion and flue gas condensation technology, respectively. We expect the sign on the dummy variables indicating prior technological state to be positive if the technologies are complements, and negative if they are substitutes. We expect the sign of $R_i(t)$ to be positive so that a boiler under more stringent regulation would be prone to adopt earlier. We would expect the signs on $L_{i,k}(t)$ and $S_{i,k}(t)$ to be positive if there are internal and external learning (as long as their effect on the rate of decrease in investment costs is non-positive or not too large) and the sign on $S_{i,k}(t)$ to be negative if there is an industry stock effect. Expectations on the sign of the coefficients on the boiler-specific characteristics in $Z_i(t)$ are discussed in more detail in the next section.

5 Data and explanatory variables

The data covers the boilers monitored under the Swedish NO_x charge system and is a panel collected over the period 1992-2009 by SEPA. It contains the information on NO_x emissions and production of useful energy necessary to establish the charge liabilities and refunds. It also includes survey information that covers which technologies are installed at each boiler as well as information on boiler capacity and the share of different types of fuels in the fuel mix. There is unfortunately no information on investment costs. Differences in investment costs are therefore proxied by boiler and firm characteristics.

Our sample consists of 556 boilers for which the information required to estimate at least one of the three econometric models is available. Descriptive statistics are presented in Tables 1 [10]. As pointed out in section 2, the

[10] The number of boilers paying the NO_x charge has varied over the years because of the change of the production threshold in 1996 and 1997 but also because of entrances of new boilers in other years and the option to produce below the threshold even though emissions are monitored. 669 boilers paid the charge in at least one year between 1992 and 2009 with 182 boilers paying the charge in 1992 and 427 boilers in 2009. The boilers not included in the sample but paying the charge in at least one year have on average significantly lower production of useful energy and a significantly lower share of boilers with any of the technologies installed in any year between 1992 and 2009. These are not surprising results seeing that a boiler for which information is missing for the estimations is likely to have produced below the threshold and not been part of the charge and refunding

number of regulated boilers under the NO_x scheme increased in 1996 and 1997. Moreover, new boilers have become part of the scheme over the period under analysis. Because the boilers which entered later into the system may be different from the early entrants in their propensity to adopt, we estimate the empirical model for the full sample and for the subsample of boilers that have been part of the NO_x scheme (and our panel) since 1992 (See Table 6 in the appendix) and compare the results.

5.1 Dependent variables - technology adoption

The dependent variable is an indicator variable equal to one if the boiler has the particular type of technology installed (combustion, post-combustion or flue gas condensation, respectively) and zero otherwise[11].

Figure 3 illustrates the diffusion pattern of the three technologies in our sample. There is a sharp increase in the adoption of both combustion and post-combustion technologies from 1992 to 1993. The decline in the rate of adoption in the years 1995-1997 is due to the entrance of many smaller units without the technologies installed. The number of boilers changes each year depending on how many of the boilers paid the NO_x charge in that particular year. For example, in our sample there were 174 boilers paying the charge in 1992 and 400 boilers in 2009. Starting in 1998 there is a steady increase in the share of boilers with one of the three technologies installed and in 2009 close to 80% of the boilers in our sample had at least one of the technologies.

5.2 Explanatory variables

Regulatory stringency

To capture the effect of the NO_x charge in our econometric model, we include the variable *Net NOx charge level*$_{t\text{-}1}$, which is the average net NO_x charge paid at the individual boiler denoted $\sigma\left[\varepsilon_i - \overline{\varepsilon}\right]$ in the cost function in (1). We expect the benefits of adoption $g_i(t)$ to be larger for a boiler subject to more stringent regulation and hypothesize that a higher net NO_x charge would expedite adoption since the gains from making such a boiler cleaner should be larger.

As discussed previously, a confounding factor is the emission standards issued for some of the boilers by county authorities. Unfortunately, we only have information about the emission standards for a small randomly selected subsample of 42 firms. Anecdotal evidence tells us that some county

scheme for most of the period. The profitability of installing the technologies at such boilers which can strategically produce just below the threshold should reasonably be low. Due to these observable differences, our results are likely not representative for boilers producing around the production threshold.

[11] Because we are estimating hazard models, a boiler is only included in the estimation sample as long as it is at risk of adopting or actually adopts. After the technology is installed, the boiler is dropped from the sample.

authorities apply more stringent standards than others. To try to capture some of the difference in regulatory stringency across regions, we use the county location of the boilers for which we know the emission standard to construct county average standards. We use these averages to divide the counties into two halves: one with relatively more stringent standards, and one with relatively lax standards. We estimate our model separately for the two groups.

Technology substitutes or complements

To analyze how the prior technological state affects the investment decision, we include dummy variables indicating whether the boiler was already equipped with the other two types of technologies in the previous year, *Combustion tech.*$_{t\text{-}1}$, *Post-comb. tech.*$_{t\text{-}1}$ and *Flue gas cond. tech.*$_{t\text{-}1}$, respectively. If the technologies are substitutes, we would expect a negative sign, while we would expect a positive sign if they are complements. We lag these variables to avoid the issue of simultaneity in the investment decision across the three technologies.

Biofuel use

On the one hand, biofuel use releases high levels of NO_x emissions relative to coal and gas, which might lead to earlier adoption of NO_x abatement technologies. On the other hand, biofuel use might entail a lower cost as it is exempted from the Swedish CO_2 tax and other regulations (Brännlund & Kriström, 2001). To capture these dimensions, we include a measure of relative expected cost of burning biofuels compared to other fuels, *Bio/fossil fuel cost*$_{t+1}$. In calculating this ratio, the numerator is computed as the product of the biofuel share in the total fuel mix in time t and the price of biofuel in $t+1$. The denominator is the product of the fuel shares for other fuels (oil, gas, coal, peat and waste) in t and their respective pre-carbon tax price in $t+1$ plus the total CO_2 cost of these fuels. The forwarded prices are used as proxies of the expected prices. We employed the price of forest fuels (skogsflis) as the price of biofuels, the price of EO1 oil as the oil price, and the gate fees for burning waste as the price for waste[12]. Regarding the CO_2 fuel costs, climate policy in Sweden includes a carbon tax and the price of carbon allowances in the European Union Emission Trading Scheme (EU ETS). We calculate CO_2 fuel costs at the boiler level considering the CO_2 emissions of the fuel mix and the differences in the carbon tax that apply to different sectors[13].

[12] These are yearly market and forecasted fees obtained from interviews conducted by Projektinriktad forskning och utveckling to representatives of several waste incineration plants (see Profu (2011)). A heating value of 2.8 re/kWh (approximately 0.3 €-cents/kWh) is used to express the fees in the same units of the other fuel prices.

[13] For CO_2 emission factors see SEPA (2009). The carbon tax is based on the carbon content of the fuel; a number of deductions and exemptions from the carbon tax have been

Entry effects
We include the indicator variables *Entrant 1996-1997* and *Entrant 1998-2009*, which are equal to one if the boiler entered the NO_x charge system in those years, and zero otherwise. Because *Capacity* is already controlling for differences in size between earlier and later entrants, *Entrant 1996-1997* and *Entrant 1998-2009* should be capturing any other potential effect of late entry into the system on the profitability of adopting cleaner technologies, such as lower investment cost or the redistribution of charge revenues that might have occurred when smaller and dirtier than average units entered and increased charge revenues in 1996 and 1997.

Rank effects
With respect to rank effects, i.e., inherent characteristics of the firm and boiler which may affect adoption, we consider industry sector and boiler capacity. Firms in different industry sectors face different economic conditions and levels of competition which may affect the propensity and ability to adopt new technologies. The NO_x charge may also affect the heat and power industry differently than the other sectors since useful energy as it is defined is the end product of the heat and power sector but mainly an intermediate input for the other industries. There are also some indications from SEPA (2003) that the stringency of quantitative restrictions may vary between industry sectors, with the heat and power sector and waste incineration possibly being subject to more stringent regulation. We include the dummy variables *Pulp-paper sector* and *Waste incineration* and, due to the relatively small sample of boilers in the remaining industries, a common dummy, *Other sectors*, for the food, chemical, wood and metal industries. We use heat and power as our reference sector, mainly because this is the dominant sector in the NO_x system.

Boiler capacity, *Capacity*, is expected to increase the benefits of adoption and possibly also lower the cost of adoption through economies of scale, at least for the post-combustion technologies.

Stock effects or external learning
To test for a potential stock or external learning effect, we further include the total number of boilers in the industry sector that had the technology installed in the previous year, *Sector comb.*$_{t\text{-}1}$, *Sector post-comb.*$_{t\text{-}1}$ and *Sector flue gas.*$_{t\text{-}1}$, respectively. If there are learning effects, we would expect benefits to increase and/or the cost of adoption to decrease with a larger

introduced in different sectors, and this also varies according to the type of generation in the case of the heat and power sector. Additionally, not all the plants are part of the EU ETS system and the overlapping process with the carbon tax has added other tax exemptions for the plants within the EU ETS. The EU ETS CO_2 price employed to compute the CO_2 fuel costs is the yearly average spot price.

stock of boilers installed with the technology. In contrast, if there is a stock effect, the benefit of adoption would decrease with the stock of boilers in the industry already equipped with the technology.

Internal learning

We also include a measure of plant and firm experience with the relevant technology, indicated by the number of boilers at the plant and firm that had the technology installed in the previous year, *Plant comb.*$_{t-1}$, *Plant post-comb.*$_{t-1}$ and *Plant flue gas.*$_{t-1}$ and *Firm comb.*$_{t-1}$, *Firm post-comb.*$_{t-1}$ and *Firm flue gas.*$_{t-1}$, respectively. We would also expect more boilers at the plant and firm equipped with the new technology to possibly decrease the cost of adoption. We lack information on the financial situation of the firms which the boilers belong to, but previous investments at the plant or firm may also proxy for greater financial strength. We lag these variables as well to avoid the issue of simultaneity in the investment decision at plant and firm level.

6 Econometric results

In this section we present the results of the Cox proportional hazard model for the adoption of combustion, post-combustion and flue gas condensation technologies (see Tables 2, 3 and 4, respectively). We conducted the regressions for different subsamples to analyze how the effect of the NO_x charge differs across sectors and the effect of joint regulations on technology adoption. Thus, in Tables 2, 3, and 4, column (1) shows the estimates for the pooled sample, column (2) for the heat and power sector, column (3) for boilers in non-heat and power sectors, and columns (4) and (5) for counties with indications of stringent and lax emission intensity standards, respectively.

As robustness checks, we also estimated the regressions using the Weibull parametric proportional hazard model. Although the Weibull model is more restrictive than the Cox model because it assumes a monotonic function of time of the baseline hazard, it can tell us about the sensitivity of our estimates to changes in the specification. The results for the Weibull regression are presented in the Appendix in Table 5.

All the tables present the estimated coefficients and therefore their sign is indicative of whether an explanatory variable speeds up or retard the adoption decision. Given the nonlinearity of the hazard function and to ease interpretation of the magnitude of these coefficients throughout the text, the effect of a covariate on the conditional probability of adopting is calculated as $exp(\beta)$, i.e., as the hazard ratio. In that manner, exponentiated coefficients larger than one imply that the covariate increases the hazard of adoption, whereas values lower than one mean that it decreases the hazard.

For example, a hazard ratio of 1.02 indicates that a one unit increase in the explanatory variable increases the hazard of adopting the technology by 2%. This effect is interpreted as a proportional shift of the hazard rate relative to the baseline, all other things equal. Likewise, if a hazard ratio is equal to 3, this would imply that boilers in the analysis group (e.g., belonging to a particular sector) are three times more likely to adopt compared to the reference group. In this case, it is usual to say that the likelihood of adoption increases by a factor of 3.

6.1 Adoption of combustion technologies

The estimated proportional hazard models explaining the diffusion pattern of combustion technologies in Table 2 indicate that the *Net NOx charge level*$_{t-1}$ does not influence the hazard of adoption. Although we found a positive coefficient of the net charge for the pooled sample, the heat and power sector, and the counties with stringent emission intensity standards, the effect of the net NO_x charge is not statistically significantly different from zero at conventional levels across all subsamples. There are some possible explanations for this result. On the one hand, as mentioned previously, the emission reductions with these technologies occur during the combustion process and might not be easy for the firm to fully control, which make firms less responsive to increases in the charge. On the other hand, the implementation of combustion measures might be governed by other factors such as compliance with emission standards for other pollutants than NO_x[14].

The results in general also show a low explanatory power of the covariates across subsamples. Few of the variables are statistically significant. In the pooled regression, the only variable that plays a role in the adoption decision is *Entrant 1998-2009*. The hazard of adopting the technology for those boilers that entered the program in that period tends to be 66% higher than the hazard of the boilers that entered in 1992-1995. This variable is also statistically significant and positive for the heat and power sector, but statistically insignificant for the rest of the subsamples. Although the entry effects are not consistently observed across specifications, for the heat and power sector this may have been important. As previously discussed, NO_x-reducing measures may have greater priority in the heat and power sector. For these potentially relatively new boilers, the cost of installation may be lower, leading them to adopt soon after entry into the system.

As expected, internal learning increases adoption; however these effects are only statistically significant for *Plant comb.*$_{t-1}$ in the non-heat and power sectors regression and in counties with lax emission intensity standards. For

[14] Höglund-Isaksson (2005) found that out of 162 NO_x-reducing measures undertaken by the plants during the initial phase of the charge in 19921995 (post-combustion, combustion and trimming measures), 31% were stated to be primarily adopted because of other reasons.

instance, in the counties with lax standards, having one additional boiler within the same plant already equipped with a combustion technology increases the likelihood of adoption of another boiler in that plant by 50%. One reason why internal learning might not have influenced the likelihood of adopting a combustion technology in the counties with stringent standards is because combustion measures are less effective in reducing emissions. Although there seems not to be any statistical difference in the sector dummies for the pooled model and the counties with stringent standards, boilers belonging to the pulp and paper sector within the counties with lax standards were less likely to adopt than boilers in the heat and power sector. This result indicates that boilers in the pulp and paper industry are relatively less prone to adoption unless induced to invest in NO_x-reducing measures by stringent emission standards.

The conditional probability of adopting combustion technologies seems to be independent of installing the most expensive technology in the previous year, *Post-comb.*$_{t\text{-}1}$. Having another of the technologies such as flue gas condensation installed does not increase the likelihood of adoption, except in the case of the subsample for the heat and power sector. A boiler in this sector that already has flue gas condensation technology installed increases the probability that the boiler will implement combustion technology in the current year by 60%. This result is mainly present when the combustion technology is flue gas recirculation, indicating complementarity of these two technologies. That relationship is in line with the European commission directives that encourage the implementation of energy efficient technologies in the energy sector, which also may be associated with other combustion measures implemented by the plant to improve cost effectiveness.

Bio/fossil fuel cost$_{t+1}$, *Capacity*, and *Sector comb.*$_{t\text{-}1}$ do not appear to play a role in inducing adoption of combustion technology. We might have expected a positive and significant coefficient of the relative fuel cost if investment in NO_x-reducing technologies should be more profitable for a boiler which uses more biofuel. However, our price variable does not capture individual agreements between firms and fuel suppliers regarding fuel prices. For instance, the presence of middle-term contracts could counteract this effect, making firms less responsive to changes in the prices in the next year. Regarding *Capacity*, because combustion technologies are installed at a relative low cost, they are also a viable alternative at smaller boilers, and this seems to be confirmed in our data.

Neither stock effects nor external learning were observed in our estimation of combustion technologies. Our results for the general model are qualitatively similar when we estimate the regressions using a Weibull parametric model. An important difference with the estimates in the Cox model is that *Sector flue gas.*$_{t\text{-}1}$ is statistically significant at 1% level, suggesting the presence of stock effects. Moreover, there seem to be differences across sectors in the hazard of adopting combustion technologies. Statistical testing indicates that we cannot reject the hypothesis that the Weibull shape parameter is equal to one at the 1%, 5%, and 10% significance level, implying that the baseline hazard is constant over time. This result is also consistent with Popp (2010)'s argument that unmeasured learning effects are small because the technologies have been well known for a long time. Our results also remain largely unchanged when we study the subsample of boilers who have been part of the NO_x scheme since 1992 (see Appendix Table 6). Therefore, it appears that the net NO_x charge did not affect the adoption of combustion technologies.

6.2 Adoption of post-combustion technologies

The estimated proportional hazard models explaining the diffusion pattern of post-combustion technologies are presented in Table 3. Compared to combustion technologies, the covariates for post-combustion technologies have a better explanatory power. The results show that the *Net NOx charge level*$_{t-1}$ is one of the major factors encouraging the adoption of post-combustion technologies. The effect of the charge is statistically significantly different from zero either at the 5% or 10% level across specifications, except for the subsample of boilers within counties with lax emission intensity standards. The effect is relatively higher for boilers in the heat and power sector and for boilers in the counties with stringent standards. For instance, an increase of 10 SEK/MWh of the net NO_x charge raises the hazard of adoption by 44%-98%, other things equal.

Post-combustion technologies can be characterized as end-of-pipe abatement technologies. To a much larger extent than combustion technologies, post-combustion technologies allow firms to choose emissions independently of output. Hence, it seems reasonable to find that the decision to adopt a post-combustion technology is more responsive to the net NO_x charge.

It is interesting to note the significant effect of the net NO_x charge on the likelihood of adoption for boilers in counties with stringent standards compared to the non-significant effect for boilers in counties with lax standards. It indicates that the individual standards and NO_x scheme are complementary. Possibly a stringent binding standard brought the boiler over the profitability threshold for investment and paying a higher net charge helped to expedite the investment. Nevertheless, since we cannot fully control for the individual stringency of the standard nor the year of implementation, we cannot rule out the possibility of temporal correlation across the instruments. For example, it could be the case that more stringent standards are the result of larger past reductions due to the charge.

Whether the boiler has combustion or flue gas condensation technology already installed has no statistically significant effect in the pooled and sector subsamples. However, there are some differences in the effect of these technologies when classifying counties by stringency of standards. The effect of a combustion technology installed in the previous year, *Comb.*$_{t\text{-}1}$, is positive and statistically significant at the 5% level in counties with lax standards. In those counties, a combustion technology already installed increases the hazard of adopting by a factor of 2.1. Although combustion technologies seem to be independent of the adoption decision of post-combustion technologies in counties with stringent standards, in counties with lax standards the low cost investments of NO_x reducing technologies appear to have been first exhausted before plants moved to more expensive abatement investments. In the case of flue gas condensation technology, *Flue gas cond.*$_{t\text{-}1}$ is statistically significant in counties with stringent standards. Installing this technology in the previous year raises the hazard of adopting post-combustion technology by a factor of 3.6. Therefore, complementarities between technologies seem to be associated more with the stringency of the emission standards: post-combustion and combustion technology in counties with lax standards, and post-combustion and flue gas condensation technology in counties with stringent standards. Unfortunately, we do not have the information on individual emission standards at the boiler level to properly control for this.

As expected, boiler capacity, *Capacity* has a significant and positive effect on the conditional probability of adoption in most of the subsamples. A 10 MW increase in the boiler capacity increases the hazard by 2.2%-4.6%, which is consistent with the economies of scale related to SCR and SNCR technologies. A boiler belonging to the *Waste incineration* sector is more likely to be equipped with a post-combustion technology than a boiler in the heat and power industry (see columns 1 and 5), while boilers in *Other sectors* such as the wood, metal food, and chemical sectors are significantly less likely to be equipped with a post-combustion technology (see columns 1 and 4). The fact that in general waste incineration boilers are more likely to be equipped with post-combustion technologies could possibly be explained by the ownership structure of this sector. Waste incineration boilers often belong to public utilities which may have motives other than pure profitability for investing in emission reducing technologies.

Just as in the combustion technology regressions, the expected relative cost of burning biofuels *Bio/fossil fuel cost*$_{t+1}$ is also not statistically significant in the post-combustion technology estimations. There are no robust indications of a general stock or external learning effect across the subsamples. Nevertheless, in counties with stringent emission intensity standards there is some indication of a stock effect.

Consistent with expectations, boilers in the heat and power sector and in counties with stringent standards that entered in the period 1996-1997 were less likely to adopt than boilers entering in 1992-1995. Internal learning seems to have been a relevant factor explaining the conditional likelihood of adoption. The variable *Plant post-comb.*$_{t\text{-}1}$ was positive and statistically significant in most of the subsamples, while *Firm post-comb.*$_{t\text{-}1}$ was negative and statistically significant in only two of them. The highest internal learning effect of *Plant post-comb.*$_{t\text{-}1}$ is present in the heat and power sector. The decreased hazard induced by *Firm post-comb.*$_{t\text{-}1}$ may indicate that internal learning is counteracted by financial constraints, if firms could not afford to buy the most expensive technologies for a great number of boilers in the short run.

Our results appear not to be very sensitive when we estimate the Weibull parametric model and when the sample is restricted to the boilers that have been in the the NO_x charge system since 1992 (see Appendix Table 6). The net NO_x charge is consistently positive and statistically significant at the 5% significance level. The magnitude of the effect of an increase in the NO_x charge is roughly similar to that found in the pooled sample for the Cox model. Therefore our results are robust to changes in the specification of the baseline hazard. We cannot reject the hypothesis at the 1% significance level that the baseline hazard is constant over time. As mentioned earlier, this also supports the idea that the unmeasured learning effects are not significant.

For the entrant boilers in 1992, the effect of the charge is much larger than in the general Cox model. An increase of 10 SEK/MWh in the net NO_x charge increases the hazard rate by 78%. Interestingly, after controlling for boiler capacity, these boilers tend to be more responsive to the charge. The fact that they have faced the regulation for a longer time than other boilers might be an explanation of this result.

6.3 Adoption of flue gas condensation technology

The models explaining diffusion of flue gas condensation technologies are found in Table 4. The *Net NOx charge level*$_{t\text{-}1}$ has no statistically significant effect in either specification. This indicates that the cost related to the NOx charge is not a major driver for investments in flue gas condensation technology.

Having a post-combustion technology already installed has a positive but non-significant effect for the pooled sample. However, the sign of *Post-comb. tech.*$_{t\text{-}1}$ is positive and significant for the heat and power sector and for the non-heat and power sectors when estimated separately in specification (2) and (3), indicating that the two technologies are somehow complementary. This may also be an indicator for boilers belonging to less capital constrained firms. *Post-comb. tech.*$_{t\text{-}1}$ also has a large positive and significant effect for boilers in counties assumed to have more stringent emission standards but not for boilers in the counties with assumed more lax standards. One possible explanation to this result could be that already being equipped with a post-combustion technology in the stringent counties is an indicator of being subject to a relatively more stringent individual standard, further raising the incentives to become more energy efficient.

Having a combustion technology installed has a positive but non-significant effect for both the pooled sample and the subsamples, but it is only significant for the sample of non-heat and power sectors. Among the non-heat and power sectors, it appears that having either of the two other technologies already installed significantly increases the likelihood of also investing in flue gas condensation.

Bio/fossil fuel cost$_{t+1}$ is significant but only weakly so at the 10% level in the pooled sample. Generally, since flue gas condensation can greatly improve heat output, a positive effect could have been expected, but it is not consistently supported across subsamples. For *Capacity*, it seems that, at least in the non-heat and power sectors, smaller boilers are more likely to adopt flue gas condensation. This is also true for boilers in the counties with indications of more stringent emission standards.

Boilers in the *Waste incineration* sector do not significantly differ from boilers in the heat and power sector in their likelihood of being equipped with flue gas condensation technology controlling for the other explanatory variables. However, boilers in the *Pulp-paper sector* and other non-heat and powers sectors are significantly less likely to be installed with flue gas condensation technology than a boiler in the heat and power sector. Belonging to the *Pulp-paper sector* in the pooled sample on average reduces the hazard of adoption by 79% while belonging to the *Other sectors* reduces the hazard by 88%.

Entrant 1996-1997 has a consistently negative effect which is significant both in the pooled sample and the non-heat and power sector. It is also significant for the boilers in the counties with stringent standards. This is an indication of an entry effect in the sense that the profitability of investing in flue gas condensation appears to be larger for boilers entering the system before 1996. However, the negative coefficient on *Entrant 1998-2009* is not significant in either specification, not supporting a general negative effect of late entry.

The negative sign on *Sector flue gas.*$_{t-1}$ is consistent with a stock effect but it is very small even in the pooled sample where it is significant. One more boiler in the same sector with flue gas condensation installed would reduce the hazard of adoption by 1.3%. When it comes to the internal learning variables, *Plant flue gas.*$_{t-1}$ is only positive for the non-heat and power sample in (3) and *Firm flue gas.*$_{t-1}$ only positive and significant in (5), not really supporting the existence of any general internal learning effects.

Comparing the results for the pooled sample in (1) with results for the parametric estimation in the Appendix Table 5, there are just slight differences in the magnitude of the coefficients but signs remain the same. This seems to indicate that a Weibull distributed baseline hazard is not an unreasonable assumption. The Weibull shape parameter is significantly larger than 1, indicating a baseline hazard which increases over time. Generally, the level of significance of the coefficients increases with the Weibull specification which reflects the fact that a fully parametric estimation tend to be more efficient.

Looking at the separate estimation in the Appendix Table 6 for the boilers that entered the NO_x charge system in 1992, we note that results are different for specification (1) in Table 4. For this sample of boilers, the *Net NOx charge level*$_{t-1}$ has a weakly significant negative effect while the effect of having a post-combustion technology installed in the previous year is significant and increases the hazard of adoption. Further exploring this result, it appears that the significantly negative effect of the *Net NOx charge level*$_{t-1}$ only exists for the subsample of boilers that had a post-combustion technology installed in any previous year. Given that a boiler has a post-combustion technology, it appears that it is actually more likely to be equipped with flue gas condensation technology the cleaner it already is (as indicated by the negative sign). Possibly non-observed individual emission standards could be an explanation if, among the boilers with a post-combustion technology that entered the system in 1992, the ones that produced with a lower emission intensity also were subject to a more stringent individual standard that raised the incentives to adopt flue gas condensation technology.

7 Conclusions

The refunded emission payment scheme has been in place in Sweden since 1992 to reduce NO_x emissions from large combustion plants. Previous studies have shown that the charge induced a sizable reduction in emission intensities in the earlier years of implementation. In this paper, we investigate the factors affecting the decision to invest in NO_x-reducing and energy efficiency improving technologies.

The results indicate that the drivers of adoption vary across technologies. The net NO_x charge does not seem to encourage adoption of combustion or flue gas condensation technologies. The net NO_x charge only plays a role in stimulating adoption of the most expensive technologies: post-combustion installations. These types of technologies can be characterized as end-of-pipe solutions which allow firms to choose emissions independently from output to a much larger extent than the other technologies, possibly explaining why firms are more responsive to the charge.

Interestingly, we find that the emission standards and the charge tend to be complementary: a higher net charge promotes adoption among boilers in counties with more stringent emission standards. This complementary effect is clear for the expensive end-of-pipe post-combustion technologies undertaken only to reduce NO_x emissions. We have not been able to identify the same effect for process investments such as combustion and flue gas condensation technologies.

The regulations are complex, as are the causalities and timing involved, and this topic would perhaps benefit from a more detailed future study of the dynamics of plant regulations. Adoption might also have been determined by other factors such as improved cost-effectiveness or compliance with emission standards for pollutants other than NO_x.

References

Brännlund, Runar, & Kriström, Bengt. 2001. Too hot to handle?: Benefits and costs of stimulating the use of biofuels in the Swedish heating sector. *Resource and Energy Economics*, **23**(4), 343 – 358.

Cleves, Mario A., Gould, William W., & Gutierrez, Roberto G. 2004. *An introduction to survival analysis using STATA Revised edition.* College station, Texas, US: STATA Press.

Fischer, Carolyn. 2011. Market power and output-based refunding of environmental policy revenues. *Resource and Energy Economics*, **33**(1), 212 – 230.

Gray, Wayne B., & Shadbegian, Ronald J. 1998. Environmental Regulation, Investment Timing, and Technology Choice. *The Journal of Industrial Economics*, **46**(2), pp. 235–256.

Griliches, Zvi. 1957. Hybrid Corn: An Exploration in the Economics of Technological Change. *Econometrica*, **25**(4), pp. 501–522.

Höglund-Isaksson, Lena. 2005. Abatement costs in response to the Swedish charge on nitrogen oxide emissions. *Journal of Environmental Economics and Management*, **50**(1), 102 – 120.

Höglund-Isaksson, Lena, & Sterner, Thomas. 2009. Innovation effects of the Swedish NOx charge. *In: OECD Joint Meeting of Environment and Tax Experts, (COM/ENV/EPOC/CTPA/CFA(2009)8/FINAL).*

Karshenas, Massoud, & Stoneman, Paul L. 1993. Rank, Stock, Order, and Epidemic Effects in the Diffusion of New Process Technologies: An Empirical Model. *The RAND Journal of Economics*, **24**(4), pp. 503–528.

Kerr, Suzi, & Newell, Richard G. 2003. Policy-Induced Technology Adoption: Evidence from the U.S. Lead Phasedown. *The Journal of Industrial Economics*, **51**(3), 317–343.

Popp, David. 2010. Exploring Links Between Innovation and Diffusion: Adoption of NOx Control Technologies at US Coal-fired Power Plants. *Environmental and Resource Economics*, **45**, 319–352. 10.1007/s10640-009-9317-1.

Profu. 2011. *Mottagningsavgifter för Avfallsförbränning 2011 (Gate fees for waste incineration 2011).* Report Projektinriktad forskning och utveckling.

SEPA. 2003 (November). *Kväveoxidavgiften - ett effektivt styrmedel (Reducing NOx Emissions - An Evaluation of the Nitrogen Oxide Charge).* Naturvårdsverket: Rapport 5335.

SEPA. 2009. *Naturvårdsverkets föreskrifter och allmänna råd om utsläppsrätter för koldioxid (The Swedish Environmental Protection Agency's provisions and general advice on emission permits for carbon dioxide).* NFS 2007:5 konsoliderad med NFS 2009:6.

SEPA. 2012 (November). *Utvärdering av 2008 års höjning av kväveoxidavgiften (An Evaluation of the 2008 increase in the NOx Charge).* Naturvårdsverket: Rapport 6528.

Snyder, Lori D., Miller, Nolan H., & Stavins, Robert N. 2003. The Effects of Environmental Regulation on Technology Diffusion: The Case of Chlorine Manufacturing. *The American Economic Review*, **93**(2), pp. 431–435.

Sterner, Thomas, & Höglund-Isaksson, Lena. 2006. Refunded emission payments theory, distribution of costs, and Swedish experience of NOx abatement. *Ecological Economics*, **57**(1), 93 – 106.

Sterner, Thomas, & Turnheim, Bruno. 2009. Innovation and diffusion of environmental technology: Industrial NOx abatement in Sweden under refunded emission payments. *Ecological Economics*, **68**(12), 2996 – 3006.

Stoneman, Paul, & Toivanen, Otto. 1997. The Diffusion Of Multiple Technologies: An Empirical Study. *Economics of Innovation and New Technology*, **5**(1), 1–17.

Figures and Tables

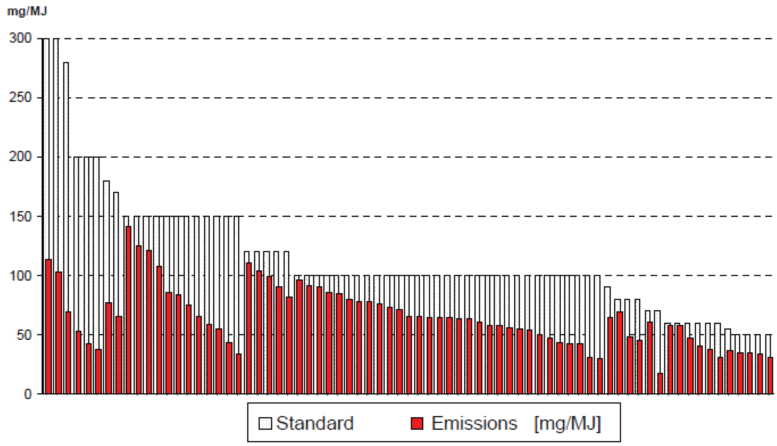

Figure 1: Emission standards and actual emissions (in terms of mg NO_x per MJ of fuel) in 2001 for boilers which were in the NO_x charge system in both 1997 and 2001 (SEPA, 2003).

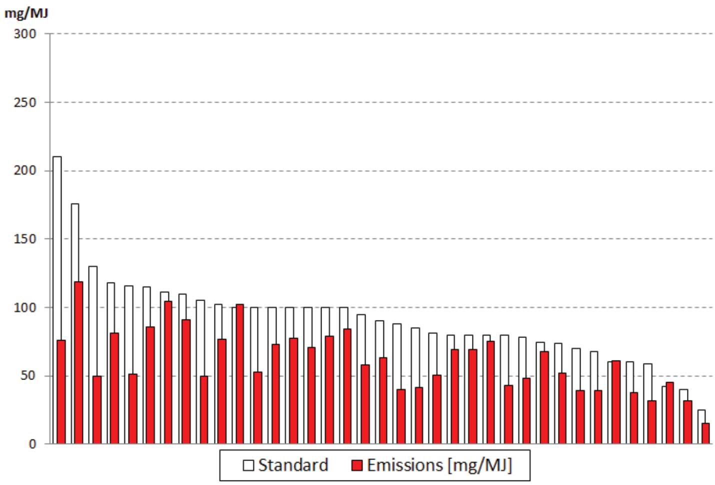

Figure 2: Emission standards and actual emissions (average over 1992-2009) for boilers which were randomly sampled for the SEPA(2012) report and also subject to emission standards in terms of mg NO_x per MJ of fuel. Data supplied by SEPA. Averages at plant level.

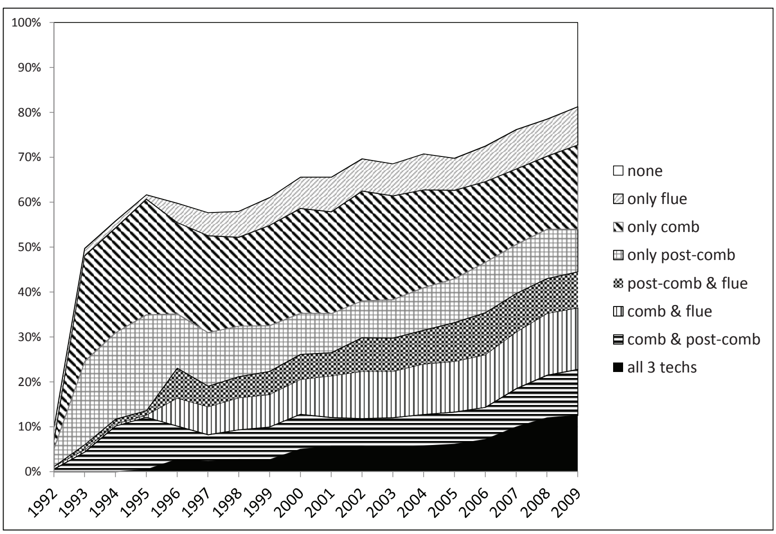

Figure 3: Diffusion of post-combustion, combustion and flue gas condensation technology among the boilers in the joint sample for all three proportional hazard models.

Table 1: Descriptive statistics.

Variable	Mean	Std. Dev.	Min.	Max.	N
Combustion tech.	0.45	0.5	0	1	5343
Post-comb. tech.	0.32	0.47	0	1	5343
Flue gas cond. tech.	0.29	0.45	0	1	5343
Plant comb.$_{t-1}$	0.9	1.09	0	4	5343
Firm comb.$_{t-1}$	3.26	4.81	0	21	5343
Sector comb.$_{t-1}$	50.2	39.09	0	137	5343
Plant post-comb.$_{t-1}$	0.68	1.08	0	5	5343
Firm post-comb.$_{t-1}$	2.26	3.51	0	14	5343
Sector post-comb.$_{t-1}$	33.12	24.72	0	87	5343
Plant flue gas.$_{t-1}$	0.53	0.9	0	5	5343
Firm flue gas.$_{t-1}$	1.91	3.41	0	16	5343
Sector flue gas.$_{t-1}$	37.42	39.5	0	114	5343
Net NOx charge level$_{t-1}$	0.16	0.55	-1.27	4.51	5343
Bio/fossil fuel cost$_{t+1}$	1.21	1.55	0	7.82	5291
Capacity	49.251	81.799	4	825	556
Heat-power sector	0.5	0.5	0	1	556
Pulp-paper sector	0.153	0.36	0	1	556
Waste incineration	0.115	0.319	0	1	556
Wood industry	0.106	0.308	0	1	556
Chemical industry	0.072	0.259	0	1	556
Food industry	0.041	0.199	0	1	556
Metal industry	0.013	0.112	0	1	556
Entrant 1996-1997	0.243	0.429	0	1	556
Entrant 1998-2009	0.372	0.484	0	1	556

Notes: *Net NOx charge level* is in units of 10 SEK per MWh of useful energy (real values with reference year 1992). *Capacity* in units of 10 MW. *Bio/fossil fuel cost* is a relative cost in units of 10. *Plant post-comb.$_{t-1}$*, *Plant comb.$_{t-1}$*, *Plant flue gas.$_{t-1}$*, *Firm post-comb.$_{t-1}$*, *Firm comb.$_{t-1}$*, *Firm flue gas.$_{t-1}$*, *Sector post-comb.$_{t-1}$*, *Sector comb.$_{t-1}$* and *Sector flue gas.$_{t-1}$* are in units of number of boilers. All other variables are dummy variables.

Table 2: Adoption of combustion technologies.

Variable	(1) Pooled	(2) Heat & Power	(3) Non-Heat & Power	(4) Stringent counties	(5) Lax counties
Net NO_x charge level$_{t-1}$	0.042	0.284	-0.089	0.128	-0.124
	(0.135)	(0.214)	(0.153)	(0.178)	(0.160)
Post-comb. tech.$_{t-1}$	-0.013	0.060	0.100	-0.429	0.289
	(0.229)	(0.316)	(0.281)	(0.291)	(0.323)
Flue gas cond. tech.$_{t-1}$	0.316	0.482**	0.089	0.315	0.280
	(0.195)	(0.226)	(0.392)	(0.240)	(0.341)
Capacity	-0.007	0.002	-0.049	0.010	-0.016
	(0.012)	(0.010)	(0.031)	(0.015)	(0.018)
Entrant 1996-1997	0.142	0.228	0.031	-0.134	0.498
	(0.229)	(0.369)	(0.303)	(0.331)	(0.342)
Entrant 1998-2009	0.505*	0.714*	0.228	0.558	0.363
	(0.294)	(0.423)	(0.386)	(0.401)	(0.395)
Bio/fossil fuel cost$_{t+1}$	-0.007	0.029	-0.021	-0.002	-0.008
	(0.050)	(0.077)	(0.066)	(0.079)	(0.075)
Plant comb.$_{t-1}$	0.181	0.089	0.301*	0.010	0.440***
	(0.143)	(0.210)	(0.170)	(0.167)	(0.158)
Firm comb.$_{t-1}$	0.022	0.029	0.019	0.024	0.027
	(0.015)	(0.024)	(0.031)	(0.019)	(0.022)
Sector comb.$_{t-1}$	-0.007	-	-	-0.003	-0.011
	(0.005)			(0.006)	(0.008)
Pulp-paper sector	-0.427	-	-	0.112	-0.895**
	(0.283)			(0.396)	(0.435)
Waste incineration	-0.367	-	-	-0.041	-0.652
	(0.309)			(0.438)	(0.437)
Other sectors	-0.544	-	-	-0.107	-0.986
	(0.384)			(0.455)	(0.724)
Observations	3016	1324	1692	1702	1314
No. of subjects	464	221	243	269	195
No. of failures	194	88	106	108	86
Log likelihood	-987.64	-370.61	-481.45	-489.06	-357.25
Chi-squared	21.08	16.36	7.89	14.43	40.45
P-value	0.07	0.06	0.55	0.34	0.00

Notes: This table shows the coefficients of the Cox proportional hazard model from five sub-samples for the period 1992-2009. The dependent variable is an indicator variable equal to one if the boiler has a combustion technology installed; zero otherwise. Continuous variables are in units of 10. (1) General model for the pooled sample with sector dummies. The variable "Other sectors" is a dummy variable for boilers in the wood, metal, food, and chemical industries, while the reference group is heat & power sector. (2) Estimates for heat and power sector, (3) estimates for other sectors, (4) estimates for counties with stringent emission intensity standards, and (5) estimates for counties with lax emission intensity standards. Standard errors, in parentheses, are robust to heteroskedasticity and arbitrary correlation within firm-level clusters. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 3: Adoption of postcombustion technologies.

Variable	(1) Pooled	(2) Heat & Power	(3) Non-Heat & Power	(4) Stringent counties	(5) Lax counties
Net NO_x charge level$_{t-1}$	0.367***	0.685**	0.491***	0.631***	-0.013
	(0.126)	(0.290)	(0.108)	(0.171)	(0.162)
Combustion tech.$_{t-1}$	0.249	0.253	0.388	-0.336	0.729**
	(0.272)	(0.408)	(0.331)	(0.519)	(0.300)
Flue gas cond. tech.$_{t-1}$	0.313	0.450	0.502	1.278***	-0.680
	(0.360)	(0.403)	(0.603)	(0.479)	(0.517)
Capacity	0.023***	0.022**	0.045**	0.007	0.026***
	(0.007)	(0.010)	(0.020)	(0.018)	(0.007)
Entrant 1996-1997	-0.298	-1.410**	-0.182	-2.542**	0.223
	(0.427)	(0.665)	(0.471)	(1.171)	(0.514)
Entrant 1998-2009	0.301	-0.670	0.773	-0.339	0.617
	(0.386)	(0.513)	(0.514)	(0.557)	(0.603)
Bio/fossil fuel cost$_{t+1}$	-0.017	0.046	-0.115	0.107	-0.173
	(0.085)	(0.114)	(0.114)	(0.099)	(0.137)
Plant post-comb.$_{t-1}$	0.355**	0.938***	0.247	0.440*	0.731***
	(0.172)	(0.210)	(0.159)	(0.242)	(0.223)
Firm post-comb.$_{t-1}$	-0.060	-0.195***	0.072	-0.120**	0.032
	(0.037)	(0.047)	(0.048)	(0.061)	(0.041)
Sector post-comb.$_{t-1}$	-0.015	-	-	-0.032*	0.007
	(0.014)			(0.019)	(0.020)
Pulp-paper sector	-0.085	-	-	-0.188	-0.043
	(0.360)			(0.505)	(0.500)
Waste incineration	1.001***	-	-	0.777	1.205***
	(0.330)			(0.555)	(0.344)
Other sectors	-1.719***	-	-	-1.885***	-1.364
	(0.629)			(0.688)	(1.141)
Observations	3680	1744	1936	2113	1567
No. of subjects	484	237	247	275	209
No. of failures	110	44	66	47	63
Log likelihood	-533.86	-179.12	-292.51	-192.38	-248.41
Chi-squared	139.05	38.68	63.83	109.59	96.81
P-value	0.00	0.00	0.00	0.00	0.00

Notes: This table shows the coefficients of the Cox proportional hazard model from five sub-samples for the period 1992-2009. The dependent variable is an indicator variable equal to one if the boiler has a post-combustion technology installed; zero otherwise. Continuous variables are in units of 10. (1) General model for the pooled sample with sector dummies. The variable "Other sectors" is a dummy variable for boilers in the wood, metal, food, and chemical industries, while the reference group is heat & power sector. (2) Estimates for heat and power sector, (3) estimates for other sectors, (4) estimates for counties with stringent emission intensity standards, and (5) estimates for counties with lax emission intensity standards. Standard errors, in parentheses, are robust to heteroskedasticity and arbitrary correlation within firm-level clusters. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Table 4: Adoption of flue gas condensation technology.

Variable	(1) Pooled	(2) Heat & Power	(3) Non-Heat & Power	(4) Stringent counties	(5) Lax counties
Net NO_x charge level$_{t-1}$	-0.304	0.011	-0.380	-0.613	-0.170
	(0.304)	(0.441)	(0.434)	(0.459)	(0.456)
Post-comb. tech.$_{t-1}$	0.394	0.559*	1.302***	1.117***	-0.836
	(0.263)	(0.335)	(0.335)	(0.276)	(0.667)
Combustion tech.$_{t-1}$	0.297	0.083	0.726**	0.081	0.293
	(0.219)	(0.297)	(0.303)	(0.380)	(0.412)
Capacity	-0.043	-0.055	-0.090*	-0.083***	-0.003
	(0.029)	(0.042)	(0.050)	(0.025)	(0.027)
Entrant 1996-1997	-0.786**	-0.810	-1.046**	-2.162**	-0.672
	(0.348)	(0.537)	(0.502)	(0.905)	(0.461)
Entrant 1998-2009	-0.308	-0.121	-0.759	-0.552	-0.463
	(0.400)	(0.547)	(0.584)	(0.534)	(0.836)
Bio/fossil fuel cost$_{t+1}$	0.128*	0.012	0.166	-0.046	0.206
	(0.065)	(0.080)	(0.103)	(0.079)	(0.135)
Plant flue gas.$_{t-1}$	-0.157	-0.526	0.471**	-0.017	-0.317
	(0.250)	(0.733)	(0.238)	(0.230)	(0.494)
Firm flue gas.$_{t-1}$	0.038	0.045	0.025	0.027	0.081*
	(0.026)	(0.039)	(0.029)	(0.037)	(0.042)
Sector flue gas.$_{t-1}$	-0.013*	-	-	-0.011	-0.013
	(0.007)			(0.008)	(0.013)
Pulp-paper sector	-1.576***	-	-	-1.280*	-1.992***
	(0.524)			(0.697)	(0.764)
Waste incineration	0.353	-	-	-0.008	0.791
	(0.400)			(0.383)	(0.822)
Other sectors	-2.139***	-	-	-2.805***	-2.139**
	(0.621)			(0.993)	(0.901)
Observations	3797	1461	2336	2124	1673
No. of subjects	448	191	257	264	184
No. of failures	89	44	45	50	39
Log likelihood	-445.08	-190.42	-203.21	-209.29	-161.32
Chi-squared	67.59	7.30	30.29	106.50	68.33
P-value	0.00	0.61	0.00	0.00	0.00

Notes: This table shows the coefficients of the Cox proportional hazard model from five sub-samples for the period 1992-2009. The dependent variable is an indicator variable equal to one if the boiler has flue gas condensation technology installed; zero otherwise. Continuous variables are in units of 10. (1) General model for the pooled sample with sector dummies. The variable "Other sectors" is a dummy variable for boilers in the wood, metal, food, and chemical industries, while the reference group is heat & power sector. (2) Estimates for heat and power sector, (3) estimates for other sectors, (4) estimates for counties with stringent emission intensity standards, and (5) estimates for counties with lax emission intensity standards. Standard errors, in parentheses, are robust to heteroskedasticity and arbitrary correlation within firm-level clusters. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Appendix

Table 5: Adoption of technologies. Parametric estimations using Weibull distribution.

Variable	Post-combustion	Combustion	Flue gas condensation
Net NO_x charge level$_{t-1}$	0.321**	-0.005	-0.375
	(0.139)	(0.152)	(0.331)
Post-comb. tech.$_{t-1}$	-	-0.212	0.469
		(0.252)	(0.288)
Combustion tech.$_{t-1}$	0.005	-	0.235
	(0.273)		(0.246)
Flue gas cond. tech.$_{t-1}$	0.363	0.348*	-
	(0.360)	(0.204)	
Capacity	0.025***	-0.005	-0.047
	(0.008)	(0.013)	(0.031)
Entrant 1996-1997	-0.664*	-0.169	-1.092***
	(0.390)	(0.210)	(0.365)
Entrant 1998-2009	0.293	0.436*	-0.523
	(0.342)	(0.258)	(0.404)
Bio/fossil fuel cost$_{t+1}$	-0.009	-0.012	0.163**
	(0.087)	(0.050)	(0.072)
Plant post-comb.$_{t-1}$	0.294	-	-
	(0.237)		
Firm post-comb.$_{t-1}$	-0.057	-	-
	(0.044)		
Sector post-comb.$_{t-1}$	-0.058*	-	-
	(0.030)		
Plant comb.$_{t-1}$	-	0.114	-
		(0.152)	
Firm comb.$_{t-1}$	-	0.022	-
		(0.016)	
Sector comb.$_{t-1}$	-	-0.021***	-
		(0.008)	
Plant flue gas.$_{t-1}$	-	-	-0.328
			(0.247)
Firm flue gas.$_{t-1}$	-	-	0.044*
			(0.025)
Sector flue gas.$_{t-1}$	-	-	-0.047***
			(0.010)
Observations	3680	3016	3797
No. of subjects	484	464	448
No. of failures	110	194	89
Log likelihood	-230.98	-324.72	-170.07
Chi-squared	125.34	28.79	66.85
P-value	0.00	0.01	0.00
Weibull shape parameter	1.34	0.96	3.09
P_value	0.45	0.87	0.00

Notes: This table shows the coefficients of the parametric proportional hazard model with Weibull distribution for (1) post-combustion technologies, (2) combustion technologies, and (3) flue gas condensation technology for the pooled sample 1992-2009. The dependent variable is an indicator variable equal to one if the boiler has one of the technologies installed described above and zero otherwise. Continuous variables are in units of 10. Sector dummies are not

Table 6: Adoption of technologies for boilers that entered the NO_x charge system since 1992.

Variable	Post-combustion	Combustion	Flue gas condensation
Net NO_x charge level$_{t-1}$	0.576***	-0.064	-0.768*
	(0.128)	(0.225)	(0.419)
Post-comb. tech.$_{t-1}$	-	0.223	0.579**
		(0.356)	(0.263)
Combustion tech.$_{t-1}$	0.047	-	-0.104
	(0.321)		(0.287)
Flue gas cond. tech.$_{t-1}$	-0.107	0.365	-
	(0.469)	(0.401)	
Capacity	0.017***	-0.001	-0.065
	(0.006)	(0.013)	(0.051)
Bio/fossil fuel cost$_{t+1}$	0.012	-0.129	0.130
	(0.088)	(0.097)	(0.087)
Plant post-comb.$_{t-1}$	0.816***	-	-
	(0.227)		
Firm post-comb.$_{t-1}$	-0.033	-	-
	(0.051)		
Sector post-comb.$_{t-1}$	0.022	-	-
	(0.014)		
Plant comb.$_{t-1}$	-	0.292	-
		(0.232)	
Firm comb.$_{t-1}$	-	0.085*	-
		(0.051)	
Sector comb.$_{t-1}$	-	-0.027***	-
		(0.010)	
Plant flue gas.$_{t-1}$	-	-	-0.271
			(0.703)
Firm flue gas.$_{t-1}$	-	-	0.065
			(0.050)
Sector flue gas.$_{t-1}$	-	-	0.005
			(0.007)
Observations	1066	1027	1437
No. of subjects	112	115	114
No. of failures	63	73	51
Log likelihood	-267.18	-313.95	-222.12
Chi-squared	54.18	13.39	12.92
P-value	0.00	0.10	0.11

Notes: This table shows the coefficients of the Cox proportional hazard model for (1) post-combustion technologies, (2) combustion technologies, and (3) flue gas condensation technology for the sample of boilers that entered the NO_x charge system in 1992 and continued operating until 2009. The dependent variable is an indicator variable equal to one if the boiler has one of the technologies installed described above and zero otherwise. Continuous variables are in units of 10. Sector dummies are excluded due to few observations in some sectors. Standard errors, in parentheses, are robust to heteroskedasticity and arbitrary correlation within firm-level clusters. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Chapter V

Air Quality Combination Forecasting with an Application to Bogota*

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Abstract

The bulk of existing work on the statistical forecasting of air quality is based on either neural networks or linear regressions. The present paper shows how forecast combination can be used to produce more accurate results. This is accomplished using both Monte Carlo simulation and an extensive application to air quality in Bogota, one of the largest and most polluted cities in Latin America.

JEL Classification: C45, C53, Q53

Keywords: Air quality forecasting, pollution, Bogota, forecast combination, neural networks.

1 Introduction

Air quality modeling and forecasting have become a rapid growth areas in recent years. The main reason for this is the increased awareness of the adverse effects of a wide range of pollutants such as carbon monoxide (CO), particulate matter (PM_{10}), ground level ozone (O_3),

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nitrogen dioxide (NO_2), and sulfur dioxide (SO_2). Epidemiological studies, for instance, suggest that health impacts may vary from subtle cardiovascular and respiratory diseases at low concentrations to an increased risk of mortality under severe exposure to high concentrations (see Lippmann, 2003; Brunekreef and Holgate, 2002; Kassomenos et al., 2008). Since air quality is a public good its socially optimal level of provision cannot be ensured through markets. Therefore, the responsibility of this provision is generally given to the environmental authorities, which have to set policies and regulations to mitigate pollution externalities. Air quality forecasting is one of the tools available to these institutions to manage health effects and air pollution events.

In their recent overview of the literature, Zhang et al. (2012) divide the main air quality forecasting approaches in three; (i) physically-based deterministic approaches, (ii) empirical approaches, and (iii) statistical approaches. Deterministic "3-D" air quality forecasting (also referred to as chemical transport forecasting) combines models of emissions with those of meteorological and chemical atmospheric processes, and has been shown to lead to accurate forecasts. Unfortunately, this approach involves scarce emission inventories and meteorological data, it is computationally very costly, difficult to operate, and requires a high level of expertise. Empirical approaches such as climatology and persistence forecasting are, by contrast, reasonably simple, and inexpensive to operate. Persistence forecasting simply says that tomorrow's pollution levels will be the same ones that applied today, and is at the same time both the quickest and most unreliable approach. Since pollution concentrations are highly weather dependent, climatology can also be used. However, as with persistency forecasting, climatology forecasts are very rough. Statistical forecasting requires relatively less detailed data, it is inexpensive and easy to operate, and, in contrast to empirical approaches, can actually be used to produce accurate forecasts even under changing weather conditions. In fact, statistical approaches has been shown to outperform even advanced deterministic approaches (see, for example, Diaz-Robles et al., 2008).

But while attractive in many regards, statistical forecasting also has a drawback in that it is not equipped to handle model uncertainty. Forecasters often encounter uncertainty about what variables to include in their models. As a result, they frequently end up estimating a variety of models before deciding on the one to use. This practice leads to a number of pathologies. First, it understates the uncertainty about the forecast. Basing the forecast on a single model implicitly assumes that the probability that the reported model generated the data is one, an assumption that is surely mistaken. Second, some forecasters search the model space until they find a specification with good forecasting performance, a practice that has led to indications of publication bias. As a result, reported results are often fragile to slight variations in model specification.

One way to circumvent these difficulties altogether is to use forecast combination (FC). The idea here is that, with multiple competing models at hand, each having its own strengths and weaknesses, rather than insisting on finding the one best forecast, it should be possible to combine the individual forecasts into a single forecast that is at least as good as any of the individual forecasts. This approach has been shown to work well in econometrics and it will be used also in the present paper. The purpose is to propose combination forecasts as an alternative to more common statistical air quality forecasting approaches such as linear regressions (LR) and neural networks (NN), and in so doing we will consider a unique data set for Bogota.

There are many reasons for looking particularly at Bogota. First, as the capital city of Colombia, Bogota is the fifth most populated city in Latin America with around 7.4 million inhabitants.[1] Second, although industrial emissions have long been considered to be one of the most severe pollution problems[2], emissions from traffic are nowadays an increasing concern. During the last decade Bogota experienced an enormous increase in vehicles. In fact, between 1999 and 2009 the vehicle fleet increased by no less than 100%, reaching about 1.5 million. Emissions of nitrogen oxides (NO_x), for instance, are much higher in traffic than in industry. Moreover, although diesel vehicles are relatively few they contribute significantly to the emissions of fine particulate matter.[3] Third, air pollutant concentrations have at times been well above the national air quality standard (AQS), specially for PM_{10} and O_3.[4] Needless to say, such high levels of pollution are very costly for society. In fact, the price of PM_{10} alone is estimated at a staggering USD 46 million per year, a cost that could be avoided if PM_{10} emissions had been cut by half (Lozano, 2004). Fourth, at present the environmental

[1]Population estimated for 2010 using the 2005 Census (DANE, 2006).

[2]Annual industry emissions amounts 1,400 tons of PM_{10}, 2,600 tons of NO_x and 2,200 tons of sulfur oxides (SOx) (see Uniandes, Secretaria de Ambiente, 2009).

[3]The vehicle fleet contributes 1,100 tons of fine particulate matter, 30,000 tons of nitrogen oxides, 450,000 tons of CO and 60,000 tons of hydrocarbons (see Uniandes, Secretaria de Ambiente, 2009).

[4]For example, in 2009 on no less than 190 occasions did PM_{10} surpass the daily norm of 150 $\mu g/m^3$ and the hourly O_3 standard of 61 ppb was exceeded even more often, 210 times (see Secretaria de Ambiente, 2010, for a recent account of air quality). Resolution 601 of 2006 issued by the Ministry of Environment provides a complete description of the AQS for several pollutants.

and health agencies of Bogota lack an air pollution forecasting system.

We begin by providing some Monte Carlo evidence of the relative forecasting ability of the statistical approaches considered. The main conclusion here is that, unless one is fortunate enough to pick the correct model, FC generally outperforms both LR and NN. These findings are confirmed by our empirical results. The monitoring data we use are daily and cover the period July 10, 2005, to April 30, 2010. The data set includes observations on six pollutants, PM_{10}, CO, NO_2, NO_x, SO_2, and O_3, and five meteorological variables. Our main findings can be summarized as follows. First, FC is always better than using NN, the benchmark statistical approach. Second, the best performing individual forecast is generally dominated by the best performing FC. Third, the combinations that performed relatively well in the Monte Carlo study also did a good job in forecasting pollution. Fourth, while PM_{10} and NO_x are relatively easy to forecast, forecasting SO_2 is more difficult. Fifth, while forecasting accuracy is generally increased by the inclusion of the information contained in other pollutants, the value added by the meteorological variables is less clear. The implications of these findings for policy and future forecasting practices are discussed to some length.

The rest of the paper is organized as follows. Section 2 introduces the concept of FC and reviews some of the most popular FC approaches. Sections 3 and 4 present the results from the Monte Carlo and empirical studies, respectively. Section 5 provides some concluding implications for policy and practice.

2 Combining forecasts

As alluded in the introduction, users of modern forecasting techniques in environmental sciences are faced with an abundance of predictor variables and a plethora of methods for generating forecasts. An important issue is therefore whether to adopt a forecasting strategy that seeks out a single best forecasting method or, alternatively, attempt to combine forecasts generated by different models.

Most statistical work in the environmental literature are based on using single forecasting methods. In fact, as far as we know, this is the first study to consider FC. Among the many single forecasting methods available, NN have received most attention by far (see, for example, Perez and Reyes, 2006). The main reason for this is its ability to approximate virtually any nonlinear function and its generality when it comes to the allowable data generating processes (Slini et al., 2006). It can therefore be seen as the benchmark statistical approach in the air quality forecasting literature.

In order to describe NN more formally let us denote by y_t the pollutant to be forecasted using the observed sample $t = 1, ..., T$. The one-step ahead value of this pollutant is denoted by y_{T+1}. The goal is to construct a point forecast f_{T+1} of y_{T+1} given $\mathbf{x}_{T+1}$, a set of regressors believed to be able to predict y_{T+1}. In case of NN, this means setting

$$f_{T+1} = \mathbf{x}'_{T+1}\beta + \sum_{j=1}^{J} \alpha_j G(\mathbf{x}'_{T+1}\beta_j), \tag{1}$$

where G is the so-called "transfer" function, and β, α_j and β_j are coefficients of the model. Of course, since these quantities are not really known, f_{T+1} is unavailable and we will therefore consider replacing it by an estimate, $\hat{f}_{T+1}$ say. The idea is to, for a given choice of G, use the observed sample on $(y_t, \mathbf{x}_t)$ to obtain estimates $\hat{\beta}$, $\hat{\alpha}_j$ and $\hat{\beta}_j$ of β, α_j and β_j, respectively, which can be accomplished using nonlinear least squares (NLS), and then to make a one-step-ahead forecast from the resulting estimated model. However, for this to be possible there are a number of important choices that have to be made. The most obvious choice is that of G. While a wide range of transfer functions has been considered, the logistic function $G(x) = 1/(1 + e^{-x})$ is perhaps the most popular.[5] The idea is that by allowing the number of logistic components, here denoted J, to increase one can approximate any nonlinear relationship that might exist between y_t and $\mathbf{x}_t$. However, while the network can be made arbitrarily flexible by setting J large enough, this also increases the risk of in-sample overfitting. That is, as J increases so does the probability that some of the logistic components will capture noise or outliers that do not have any predictive power.

Another problem with NN is that even if J known, the NLS objective function is known to possess many local minima (see, for example, Nunnari et al., 2004; Franses and van Dijk, 2000, chapter 5). Thus, even if the estimation algorithm converges, there are no guarantees that it will be to the global minimum.

Because of these problems it is common to also consider simpler models. A very common choice is LR, which is (1) with $\alpha_1, ..., \alpha_J$ put to zero. Thus, in case of LR, one sets

$$f_{T+1} = \mathbf{x}'_{T+1}\beta, \tag{2}$$

which reduces significantly the complexity of the estimation problem. Indeed, since the

[5] Thus, with this choice $G(\mathbf{x}'_t\beta_j)$ is nothing but the familiar binary logit probability model.

model is now linear, the estimator $\hat{\beta}$ of β can be obtained from a simple ordinary least squares (OLS) regression of y_t onto $\mathbf{x}_t$. The resulting estimated forecast is given by $\hat{f}_{T+1} = \mathbf{x}'_{T+1}\hat{\beta}$.

Despite its simplicity LR has been found to perform quite well, even in comparison to more general models (see Perez and Reyes, 2006; Slini et al., 2006). One reason for this is that of parsimony. Indeed, it is a well-known fact that estimating additional parameters can raise the forecast error variance above what might be obtained when using relatively simpler models. This is obviously true when the additional parameters are all equal to zero. But the same can happen also when the additional parameters are nonzero, if the information content of the associated variables is just low enough. In such cases, the increased estimation uncertainty may cause the forecast error variance to raise more than the additional information lowers it. Thus, while excluding variables whose parameters are nonzero can adversely affect forecasting accuracy, adding them might lead to an increase in the forecast error variance.

In light of the above trade-off between forecast accuracy and error variance, it is clear that combining forecasts from unrestricted and restricted models could lead to higher forecast accuracy. For non-nested models, because of the larger information set on which they are built, FC should be advantageous even if the parameters are known.

In order to describe formally FC, assume that there are M competing forecasts available, denoted $f_{T+1}(1), ..., f_{T+1}(M)$. Let us further denote by $w(m)$ a weight assigned to the m-th forecast. The combination forecast of y_{T+1} is given by

$$f_{T+1} = \sum_{m=1}^{M} w(m) f_{T+1}(m). \tag{3}$$

The forecasts derive from different forecasting models and can be obtained in the same way as described above. For example, if model m linearly relates y_t to some model-specific regressor set $\mathbf{x}_t(m)$, then $f_{T+1}(m) = \mathbf{x}_{T+1}(m)'\beta$, where β can be estimated by applying OLS to a regression of y_t onto $\mathbf{x}_t(m)$, giving $\hat{f}_{T+1}(m) = \mathbf{x}_{T+1}(m)'\hat{\beta}$.

As for the estimation of $w(m)$ there are several possibilities, ranging from simple equal weights, which is equivalent to setting $\hat{w}(m) = 1/M$, to more sophisticated choices such as the one of Bates and Granger (1969, BG), in which the weights are set proportional to the inverse of the estimated forecast error variance. In order to describe this latter approach in more detail, it is convenient to divide the sample in two; a "training" sample containing the first $T - P$ observations, and a "calibration" sample containing the last P observations.

The idea is to use the training sample to estimate the model parameters, and then to make a series of P one-step ahead forecasts $\hat{f}_{T-P+1}(m), ...\hat{f}_T(m)$ (in the calibration sample), which can be used to obtain the estimated forecast error variance, $\hat{\sigma}^2(m)$ say. The BG estimator of $w(m)$ is now given by

$$\hat{w}(m) = \frac{1/\hat{\sigma}^2(m)}{\sum_{j=1}^{M} 1/\hat{\sigma}^2(j)},$$

which can be viewed as a smoothed version of the predictive least squares (PLS) weight that simply selects the forecast corresponding to the smallest forecast error variance.

The recent forecasting literature has devoted considerable attention to forecasts based on Bayesian model averaging. When the priors are diffuse, such that the researcher has no prior information regarding which model to use, the Bayesian estimator of $w(m)$ is approximately equal to the following "smoothed" Bayesian information criterion (SBIC) weight:

$$\hat{w}(m) = \frac{\exp(-\text{BIC}(m)/2)}{\sum_{j=1}^{M} \exp(-\text{BIC}(j)/2)},$$

where $\text{BIC}(m)$ is the value of the Bayesian information criterion (BIC) for model m based on the last P observations. A related proposal, called the "smoothed" Akaike information criterion (SAIC), is to use the same estimator but with $\text{BIC}(m)$ replaced by $\text{AIC}(m)$, the value of the Akaike information criterion (AIC).

Yet another possibility is to follow the Granger and Ramanathan (1984, GR) approach and to set $\hat{w}(m)$ equal to the OLS slope estimator in a regression of y_t onto $\hat{f}_t(1), ..., \hat{f}_t(M)$ using the last P observations. The Mallows model averaging approach of Hansen (2008) is very similar in spirit but here the sum of squared residuals is penalized and the weights are restricted to be non-negative and to sum up to one.

Among the FC approaches described above two are asymptotically root mean squared forecast error (MSFE) optimal in the sense that the expected out-of-sample MSFE converges to the lowest possible value. These are the GR and Mallows approaches. The GR estimator is based on OLS, which is consistent, thus making it asymptotically MSFE optimal. Because the objective function of the Mallows estimator is equal to that of the GR estimator less an asymptotically negligible penalty, Mallows is asymptotically equivalent to GR. The other approaches are MSFE suboptimal.

3 Monte Carlo investigation

This section reports the results from a small-scale simulation study to assess the relative performance of the three forecasting approaches considered; NN, LR and FC. Two (univariate) simulation designs are considered; (i) an autoregressive (AR) design, and (ii) a moving average (MA) design. In the AR design, y_t is generated as $y_t = \rho y_{t-1} + u_t$, whereas in the MA design, it is generated as $y_t = u_t + \theta u_{t-1}$. In both cases, $u_t \sim N(0,1)$ and $T = 500$. Thus, while in the first design the true model is an AR(1), in the second design it is an AR(∞).

The forecasts of y_{T+1} are based on AR models, which, except for the NN approach, are estimated by OLS.[6] In particular, in case of NN, we set $J = 1$ and $\mathbf{x}_t = (1, y_{t-1}, ..., y_{t-4})'$, whereas in case of LR, we set $\mathbf{x}_t = (1, y_{t-1})'$. In case of FC, we set $\mathbf{x}_t(m) = (1, y_{t-1}, ..., y_{t-m+1})'$ for $m = 2, ..., 5$ and $\mathbf{x}_t(1) = 1$. A wide set of weighting schemes were compared, including median selection, equal weights, AIC selection, SAIC, BIC selection, SBIC, PLS, BG, GR and Mallows. We compare the forecasting methods based on out-of-sample MSFE, which is averaged across 3,000 simulation draws.

Comparing the forecast methods across the parameter settings, we find that five of the combination methods, namely, median selection, equal weights, PLS, GR and Mallows, are strictly dominated by one of the others. To keep our graphs uncluttered, only the undominated combination methods are displayed.

Figures 1 and 2 display the results for the AR and MA designs, respectively. A close inspection reveals that there is no one method that uniformly dominates the others. In the AR design the AR(1) model forecast performs best, which is just as expected since the AR(1) is also the true model in this case. However, information criteria-based FC performs almost just as well. In fact, for $\rho \geq 0.3$ the MSFE of the BIC and SBIC are almost indistinguishable from that of the AR(1). The relative performance of the remaining two forecasting methods depends on the value of ρ; if $\rho < 0.5$, then BG tend to perform best, whereas if $\rho > 0.5$, then NN performs best.

As expected, the performance of the AR(1) model forecast is much poorer in the MA design when the true model is an AR(∞). In fact, unless $\rho < 0.3$, we see that it is strictly dominated by FC, and for $\rho > 0.4$, it is dominated also by NN. The overall best performance

[6] The NLS estimator is based on the steepest decent, or backpropagation, algorithm and the objective function is penalized to prevent unduly large parameter estimates (Franses and van Dijk, 2000). All variables are scaled such that they have zero mean and unit variance.

is obtained by using one of the two AIC methods, usually SAIC, which is not that surprising given the general tendency of the AIC to overparameterize. However, the SBIC is typically not far behind with roughly the same performance.

The above results confirm our expectations. In particular, unless one is fortunate enough to pick the correct model, FC generally leads to lower MSFE than both NN and LR. Two feasible and easy-to-apply combination methods with particularly low MSFE are SAIC and SBIC. In our simulations, they performed well in both simulations designs considered and across a range of parameter values, suggesting that it is hard to do better than these methods. They are therefore going to be used in our empirical application to pollution forecasting in Bogota.

4 Empirical results

4.1 Data

Air quality in the city of Bogota is measured and administrated by the Air Quality Monitoring Network of the Secretary of Environment of Bogota (RMCAB). The RMCAB is a system of 15 automated monitoring stations that measure pollutants concentrations and/or weather conditions. Our sample originates with one such monitoring station, St5, which is located in the central area of the city ($04^\circ\ 39'\ 30.5''$, $74^\circ\ 05'\ 2.3''$ on the ground level and approximately 400 meters from the major street), and as such it is influenced mainly by vehicular emissions. St5 records information on six pollutants, PM_{10}, CO, NO_2, NO_x, SO_2, and O_3, and five meteorological variables, wind speed (WS), wind direction (WD), rainfall accumulation (RAIN), temperature (TEMP) and solar radiation (SR). The sample is hourly and stretches the period from July 10, 2005, to April 30, 2010. Unfortunately, while the data coverage at St5 is relatively high, between 78% to 94%, it is not complete, which in turn requires imputation.

The imputation scheme considered here is implemented in two steps. The first step amounts to employing the site-dependent effect method (SDEM) of Plaia and Bondi (2006, 2010). This technique considers both the spatial and temporal correlation of the data, and has been shown to outperform not only single imputation methods like hourly mean, and last and next row-mean methods, but also multiple imputation techniques. In implementing the SDEM we employ data from additional monitoring stations. The stations to use for each variable were selected based on their coverage and correlation with St5. The characteristics of the eight stations considered in the SDEM are presented in Table 1.

Unfortunately, the data coverage of the other stations is relatively low, which is also why we in this study focus on St5. Hence, whenever data on the other stations are not available for the periods where St5 data are missing we need to consider a second imputation approach. This is the second step. We experimented with several methods but opted for the cyclostationary temporal covariablility approach (see Schneider, 2001; Anttila and Tuovinen, 2010). In contrast to the first, this second-step approach only considers the temporal variation of the variable under consideration. It replaces the missing values by the means at each hour of the day for the week-day, quarter and year in question. While still rather flexible, the second approach uses less information than the first. Fortunately, it is only very rarely that SDEM cannot be used. In fact, less than 2.1% of the observations were imputed by the second approach.

We examined whether there was any systematic changes and/or anomalies in the sample after imputation. However, the hourly means and standard deviations before and after imputation were almost identical, suggesting that the effect of the imputation is not very large. To formally test the equality of the pre- and post-imputation densities, we applied the T_n-test of Li et al. (2009).[7] According to the results (not reported) the equal density null could not be rejected at the 10% level, thus confirming the preliminary evidence that the imputation has not damaged the structure of the data.

The complete (imputed) hourly sample is transformed by taking daily means, which is the most common aggregation method by far (see, for example, Slini et al., 2006). Hence, the data to be used as input in the forecasting exercise are daily means covering the period July 10, 2005, to April 30, 2010, leading to a total of $T = 1,765$ observations.

4.2 Preliminary data analysis

Some descriptive statistics of the daily data are shown in Table 2. As expected, there is considerable variation across pollutants. For instance, while PM_{10} ranges from 6 to 125 μg/m^3, O_3 from 2 to 48 ppb.[8] We also see that the weather at St5 is characterized by relatively mild

[7] The T_n-test is based on 500 bootstrap repetitions and the standard normal kernel (see Hayfield and Racine, 2011).

[8] The city average during 2009 ranged between 24 and 101 μg/m^3 for PM_{10} and between 10 and 17 ppb for O_3.

winds (light air), mild ambient temperatures and low rainfall.[9]

Standard forecasting techniques assume that the variable to be forecasted is (trend) stationary. Any trends in the data must therefore be appropriately accounted for. There are two types of trends, deterministic and stochastic. The main difference, at least for our purposes, is that while deterministic trends can be forecasted, stochastic trends cannot, and must therefore be removed before the forecasting can begin. The most common way to distinguish between the two types of trends is to apply a unit root test that tests the null hypothesis of a stochastic trend against the alternative of (trend) stationarity. Such tests are very common in econometrics, but have also been used in other field such as climatic times series (see, for example, Kaufmann et al., 2010).

A large number of unit root tests exists. However, in the current paper we only consider the augmented Dickey–Fuller (ADF), Phillips–Perron (PP) and GLS–ADF tests, which are the workhorses of the industry (see Phillips and Xiao, 1998, for a review). All tests test the same hypotheses, but differ in the way the short-run dynamics and deterministic trend component are taken into account. While the ADF and GLS–ADF tests are parametric with respect to the short-run dynamics and require a choice of lag augmentation order, the PP test is non-parametric and require a choice of bandwidth. In the current paper, while the appropriate lag length is selected by the BIC, the bandwidth is set as an increasing function of T, which is the most common approach in the literature (see Phillips and Xiao, 1998). The maximum number of lags is set to eight.[10] As for the treatment of the deterministic trend components, while the ADF and PP tests are based on conventional OLS detrending, GLS–ADF is based on generalized least squares (GLS) detrending under the (locally) stationary alternative, which is expected to lead to higher power. As for the choice of the appropriate deterministic component, the approach used here involves pre-testing for the presence of a (linear) trend. If the trend is significant (at the 5% level), the data are detrended, whereas if the trend is insignificant, the data are demeaned. The results from all three tests are reported in Table 3. The first thing to note is that presence of a stochastic trend is unanimously rejected. The same is true when the test is applied to the first differences of the variables. We therefore conclude that the variables are (trend) stationary, as required.

[9] According to the Beaufort wind force scale, the wind speed at St5 is "light" (light air). RMCAB classifies rainfall levels between 0–5 mm as "low", while temperatures between 7 and 23 °C are considered as "mild".

[10] The appropriateness of the choice of the maximum number of lags was checked by applying the BIC to an even larger maximum.

4.3 Issues of implementation

For each pollutant, a large number of model specifications were considered. The models can be classified in three broad classes; (1) AR models, (2) distributed lag (DL) models, and (3) autoregressive distributed lag (ADL) models. Thus, while in the class of AR models, $\mathbf{x}_t$ is simply a vector of lags of the pollutant to be forecasted, in the class of DL models, $\mathbf{x}_t$ is not made up of lagged values of the dependent variable but rather by other predetermined regressors such as lags of the other pollutants and meteorological variables. The class of ADL models is the most general, and includes both sets of regressors.

AR models have been shown to be very useful in forecasting; however, for most air pollutants the true data generating process is probably much more complex (see, for example, Van der Wal and Janssenn, 2000; Sanchez et al., 1990). Many factors are known to influence pollution concentration levels. For instance, sunlight and temperature determine the chemical and photochemical reaction rate of pollutants (Pleijel et al., 2009). Similarly, rainfall may wash out atmospheric particles and gases (Huo et al., 2010), and stagnant air and slow winds, which usually predominate under high pressure systems, as well as temperature inversions may hinder dispersion and dilution of pollutants (see Aldrin and Hobæk, 2005). The weather conditions at St5 (see Table 2) are suggestive of low air mixing and dispersion, features that tend to be exacerbated during temperature inversion episodes, which then alter the pollutant residence time in the ambient air. Clearly, concerns like these motivate the use of DL and ADL models.

The total list of regressors considered includes the pollutants (PM_{10}, CO, NO_2, NO_x, SO_2 and O_3), the meteorological variables (WS, WD, RAIN, SR and TEMP or its first difference, denoted DTEMP) and their squares (to account for possible non-linear weather effects). The following four combinations of variables are considered (for the DL and ADL models): (1) level meteorological variables; (2) level and squared meteorological variables; (3) other pollutants (meaning all other pollutants than the one being forecasted) and level meteorological variables; (4) other pollutants, and level and squared meteorological variables. All four combinations include a full set of monthly and day-of-the-week dummy variables.

The obvious drawback of the models considered is that they can be very large, especially when wanting to entertain the possibility of an unrestricted lag structure. Three restricted lag structures are therefore considered; (1) a parsimonious structure with only one lag, (2) a relatively unrestricted structure with eight lags, and (3) a flexible structure in which the number of lags for each explanatory variable is determined in a data-driven fashion by using the software Autometrics (Doornik, 2009).

The idea behind Autometrics can be summarized as follows. The starting point is a general unrestricted model whose dimension is sufficiently large to capture all the features of the data, which in this case means a model with eight lags. Autometrics then uses a tree search to explore different reduction paths. It removes variables that are insignificant and each removal (that is later back-tested) constitutes a new branch of the tree. The reduction is considered final if no further variables can be removed. If several models are found (as possible with a tree search) these are then combined to form the final model. Autometrics is often used in econometrics but has recently proven to be useful also for modeling atmospheric CO_2 (Hendry and Pretis, 2011).[11]

No less than 51 specifications where estimated, and in most cases Autometrics lead to substantially simplified models. To take one example, while an DL(8) model including the lags of both other pollutants and meteorological variables had 99 parameters, the corresponding Autometrics reduced model only retained 11 variables in addition to the dummy variables.

4.4 Forecasting results

The complete sample comprises 1,765 observations. The models were estimated on the first sub-sample ending in April 2, 2009, and a sequence of one-step-ahead forecasts is then computed for the last 450 observations. The first 300 of these are used to compute the weights used in the forecast combinations and the remaining 150 observations are used for the forecasting comparison.

Parallelling the presentation of Section 3, we focus on MSFE, and as in that section the FC results are compared to the AR(1) and NN forecasts, where the latter is implemented using $J = 1$ logistic components and Autometrics to pick the dimension of the linear part of the model.[12] The results for the best and worst performing individual forecasts (used to compute the combinations) excluding the AR(1) forecast are also reported.

The results are presented in Tables 4 and 5. The information content in these tables can be summarized as follows:

[11]Slini et al. (2006) use a similar model reduction technique to forecast PM_{10} in Greece.

[12]Since $J > 1$ did not lead to any improvements in performance, we only report the results for $J = 1$.

- FC is always better than using NN (the benchmark statistical approach).
- Except when forecasting CO, the best performing individual forecast (except for the AR(1) forecast) is dominated by the best performing FC.
- Whenever FC is beaten by the competition, it is by the AR(1) forecast.
- Consistent with the Monte Carlo evidence, the SAIC-based FC performs relatively well.
- The overall best performance, also when compared to the other combinations, is obtained by using GR, which is in agreement with its theoretical optimality property (see Section 2). Interestingly, except for the case of SO_2, in which NN performs marginally better, one is always better off using GR than NN.
- SO_2 seem to be relatively difficult to predict, leading to high MSFE values across the board. This result may be due to the omission of some atmospheric reactions involving SO_2, such as photooxidation (see Özbay, 2012). Another explanation may be that in Bogota SO_2 is mainly emitted by industries, and industry chimneys are far from St5 and are located higher up. More accurate forecasts would therefore require the use of information of SO_2 concentrations and meteorological variables at heights and points closer to the source.
- The mean FC performs relatively well for PM_{10} and NO_x, which are also the pollutants that seem to be relatively easy to predict. However, for the other pollutants, the performance of the mean forecast is much poorer, especially for SO_2.
- Autometrics generally leads to relatively better forecasting performance when compared to the models using a fixed lag structure.
- The predictive power of the other pollutants varies depending on how the forecasts are combined. For example, while for the mean and median forecasts excluding the other pollutants is generally good for performance, for the other combinations this is typically not the case, that is, accuracy is increased by the inclusion of the information contained in the other pollutants.
- As with the other pollutants, the predictive power of the quadratic meteorological variables varies across the combinations. However, in this case the evidence is more clear

cut; the quadratic meteorological variables are not very useful for forecasting. Thus, assuming that the quadratic approximation is valid, this means that the effect of weather is mainly linear, a conclusion that is broadly consistent with the poor performance of NN. Similar results have been found by, for example, Özbay (2012).

5 Implications for policy and practice

The results reported herein have many policy implications.

- Our approach can be used to identify and foresee episodes of high pollution in Bogota. The more reliable the forecast, the more effective it is. This tool can be used to inform in advance contingency plans that reduce the adverse impacts of air pollution on population, especially on young children and elderly, which are more prone to asthma, respiratory and cardiac diseases – known effects of air pollution. For example, Arciniegas et al. (2006) document how a 10 $\mu g/m^3$ increase in PM_{10} emissions increased the number of hospital admissions for acute respiratory diseases in Bogota by 4%. High air pollution levels have even been detected inside classrooms in some public schools near busy roads (Franco et al., 2009). The fact that PM_{10} and O_3 frequently exceed the AQS and that wind direction helps spread pollution to vulnerable (low income) areas, such as the Kennedy neighborhood, makes the situation quite critical.

- Our results can feed the Air Quality Index of Bogota. Although this index uses information in real time, it only provides information of health alerts for past events. According to the Resolution 0601 of 2006 of the Ministry of Environment, modified by Resolution 0610 of 2010, the environmental authority must emit health alerts in three levels, namely, prevention, warning, and emergency. FC can be used to create health alerts that can be issued in advance. Those alerts could be informed through public media such as television, radio or internet.

- FC can supplement the existing information of air quality and emission control programs in Bogota. The availability of reliable air pollution forecasts would give RMCAB the option of identifying emission reductions on intermittent days. One application of particular interest would be the evaluation of the so-called "car free day" program, in which the use of some 1.2 million cars is restricted.[13] Another possibility involves

[13] The "car free day" program was implemented through Decree 1098 of 2000, and prohibited circulation of

using forecasts to complement inventory studies that are usually conducted every 5 or 10 years.

- Forecasts can be used as a guide to the general public as to which parts of the city that are relatively "safe". Even though our application focuses on a site highly influenced by traffic, forecast combination may be used to forecast pollution at other sites, including industry.

- Under the environmental law, citizens of Bogota can complain about high levels of pollution and declare their dissatisfaction to the authority. Reliable and up-to-date notification of pollution levels to the public would increase awareness and reduce complaints.

Our results are interesting not only for what they imply for policy but also for what they imply for future pollution forecasting practices in Bogota.

- Since the effect of weather on daily air pollution is mainly linear, it simplifies the forecast specifications because it reduces the need to model non-linearity.

- The relatively poor performance of NN, even in comparison to simple linear models, suggests that precision can be gained by the use of relatively simple models. This is an important finding, as NN represents the benchmark statistical approach. Many of the results reported in the literature can therefore probably be improved upon by using FC.

- Unless one has strong prior beliefs regarding the underlying data generating process, FC is expected to work well when compared to both NN and LR. In fact, FC is expected to outperform also the best performing individual forecast used in the combination. This is an advantage because it means that it is unlikely to obtain a better forecast than when using FC.

- While PM_{10} and NO_x are relatively easy to predict, SO_2 is more difficult, suggesting that alternative forecasting approaches might be needed, or that FC might be implemented for each local monitoring station taking into account the main pollutant source

private vehicles on the first Thursday of February of every year between 6:30 AM and 7:30 PM with the purpose of reducing vehicle emissions.

in the neighborhood. Forecasting PM_{10} is the priority since it is one of the most important air pollutants affecting human health in Bogota.

- The information contained in other pollutants is usually useful for forecasting. Several of these pollutants are strongly linked either through the same emission source or through the formation process by a chain of chemical reactions.
- Parsimony is important for accuracy. In particular, model selection prior to forecasting seems to be quite beneficial.

References

Aldrin, M., Hobæk, I. (2005). Generalised additive modelling of air pollution, traffic volume and meteorology. *Atmospheric Environment* **39**, 2145–2155.

Anttila, P., Tuovinen, J.-P. (2010). Trends of primary and secondary pollutant concentrations in Finland in 1994–2007. *Atmospheric Environment* **44**, 30–41

Arciniegas, A., Rodriguez, C., Pachon, J., Sarmiento, H., Hernandez, L. J. (2006). Relacion entre enfermedad respiratoria aguda en niños menores a cinco años y contaminacion atmosferica en Bogota. Secretaria Distrital de Salud. *Investigaciones en Seguridad Social y Salud* **8**, 121-135

Bates, J. M., Granger, C. M. W. (1969). The combination of forecasts. *Operations Research Quarterly* **20**, 451–468.

Brunekreef, B., Holgate, S. T. (2002). Air pollution and health. *Lancet* **360**, 1233–1242.

Diaz-Robles, L. A., Ortega, J. C., Fu, J. S., Reed, G. D., Chow, J. C., Watson, J. G., Moncada-Herrera, J. A. (2008). A hybrid ARIMA and artificial neural networks model to forecast particulate matter in urban areas: the case of Temuco, Chile. *Atmospheric Environment* **42**, 8331–8340.

DANE (2006). Series de Población 1985–2020. Available at www.dane.gov.co (April 25, 2012).

Doornik, J. A. (2009). Autometrics. In Castle, J. L. and Shephard, N. (Eds.), *The Methodology and Practice of Econometrics*, Oxford University Press.

Franco, J.F., Rojas, N., Sarmiento; O., Hernndez, L., Zapata, E., Maldonado, A., Matiz, L., Behrentz, E. (2009). Particulate matter concentrations at public schools located near major urban roads in Bogota, Colombia: a pilot study. *Revista Facultad de Ingeniera Universidad de Antioquia* **49**, 101–111.

Franses, P. H., van Dijk, D. (2000). *Non-linear time series models in empirical finance*. Cambridge University Press, UK.

Granger, C. W. J., Ramanathan, R. (1984). Improved methods of combining forecast accuracy. *Journal of Forecasting* **19**, 197–204.

Hansen, B. (2008). Least-squares forecast averaging. *Journal of Econometrics* **146**, 342–350.

Hayfield, T., Racine, J. S. (2011). Nonparametric kernel smoothing methods for mixed data types: Package 'np'. Available at http://cran.r-project.org/web/packages/np/ (November 27, 2011).

Hendry, D. F., Pretis, F. (2011). Anthropogenic influences on atmospheric CO2. Economics Series Working Papers 584, University of Oxford, Department of Economics.

Huo, M., Sun, Q., Bai, Y., Xie, P., Liu, Z., Li, J., Wang, X., Lu, S. (2010). Chemical character of precipitation and related particles and trace gases in the North and South of China. *Journal of Atmospheric Chemistry* **67**, 29–43.

Kassomenos, P., Papaloukas, C. Petrakis, M., Karakitsios, S. (2008). Assessment and prediction of short term hospital admissions: the case of Athens, Greece. *Atmospheric Environment* **42**, 7078–7086.

Kaufmann, R. K., Kauppi, H., Stock, J. (2010). Does temperature contain a stochastic trend? Evaluating Conflicting Statistical Results. *Climatic Change* **101**, 395–405.

Li, Q., Maasoumi, E., Racine, J. S. (2009). A nonparametric test for equality of distributions with mixed categorical and continuous data. *Journal of Econometrics* **148**, 186–200.

Lippmann, M. (2003). Air pollution and health – Studies in the Americas and Europe. In McGranahan, G., Murray, F. (Eds.), *Air pollution and health in rapidly developing countries*, Stockholm Environment Institute, 35–48.

Lozano, N. (2004). Air pollution in Bogota, Colombia: A concentration-response approach. *Desarrollo y Sociedad* **54**, 133–177.

Nunnari, G., Dorling, S., Schlink, U., Cawley, G., Foxall, R., Chatterton, T. (2004). Modelling SO_2 concentration at a point with statistical approaches. *Environmental Modelling & Software* **19**, 887–905.

Perez, P., Reyes, J. (2006). An integrated neural network model for PM10 forecasting. *Atmospheric Environment* **40**, 2845–2851.

Phillips, P. C. B., Xiao, Z. (1998). A primer on unit root testing. *Journal of Economic Surveys* **12**, 423–469.

Plaia, A., Bondi, A. L. (2006). Single imputation method of missing values in environmental pollution data sets. *Atmospheric Environment* **40**, 7316–7330.

Plaia, A., Bondi, A. L. (2010). Regression imputation for space–time datasets with missing values. In Palumbo, F., Natale Lauro, C., Greenacre, M. J. (Eds.), *Data Analysis and Classification: Studies in Classification, Data Analysis, and Knowledge Organization*, Springer–Verlag Berlin Heidelberg **11**, 465–472.

Pleijel, H., Klingberg, J., Bäck, E. (2009). Characteristics of NO2 Pollution in the City of Gothenburg, South-West Sweden-relation to NOx and O3 Levels, Photochemistry and Monitoring Location. *Water, Air, & Soil Pollution* **9**, 15–25.

Sanchez, M. L., Pascual, F., Ramos, C., Perez, I. (1990). Forecasting particulate pollutant concentrations in a city from meteorological variables and regional weather patterns. *Atmospheric Environment* **24A**, 1509–1519.

Schneider, T. (2001). Analysis of incomplete climate data: estimation of mean values and covariance matrices and imputation of missing values. *Journal of Climate* **14**, 853–871.

Secretaria de Ambiente (2010). Informe anual de calidad del aire de Bogota 2009. Red de Monitoreo de Calidad del Aire de Bogota, RMCAB, 106p.

Secretaria de Ambiente (2011). Manual de Procesos y procedimientos. Proceso: Evaluacion Control y Seguimiento. Procedimiento: Metodos de medicion, validacion y calculo de incertidumbre de medicion en la RMCAB, 20p.

Slini, T., Kaprar, A., Karatzas, K., Moussiopoulos, N. (2006). PM_{10} forecasting for Thessaloniki, Greece. *Environmental Modelling & Software* **21**, 559–565.

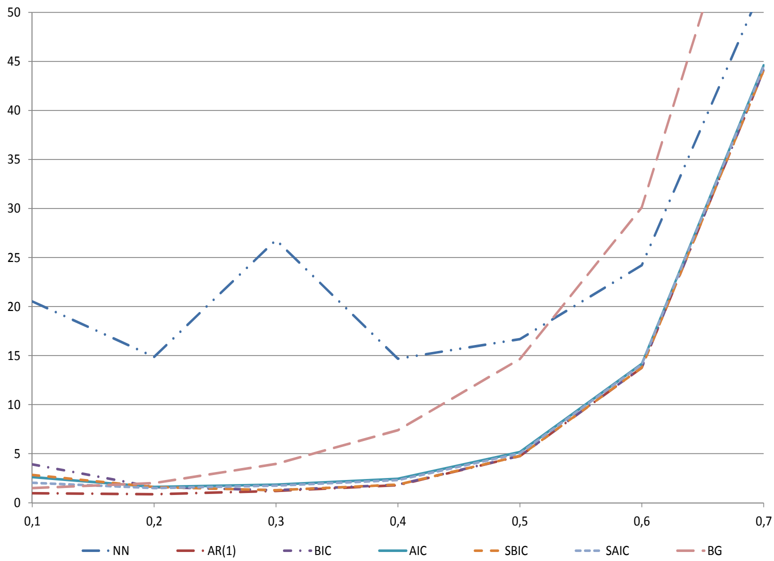
Uniandes, Secretaria de Ambiente (2009). Elementos Tecnicos del Plan Decenal de Descontaminacion de Bogota. Parte 2. Inventario de Emisiones Provenientes de Fuentes Fijas y Moviles. Bogota D.C. 98p.

Van der Wal, J. T., Janssen, L. H. J. M. (2000). Analysis of spatial and temporal variations of PM10 concentrations in the Netherlands using Kalman filtering. *Atmospheric Environment* **34**, 3675–3687.

Zhang, Y., Bocquet, M., Mallet, V., Seigneur, C., Baklanov, A. (2012). Real-time air quality forecasting, part I: History, techniques, and current status. *Atmospheric Environment* **60**, 632–655.

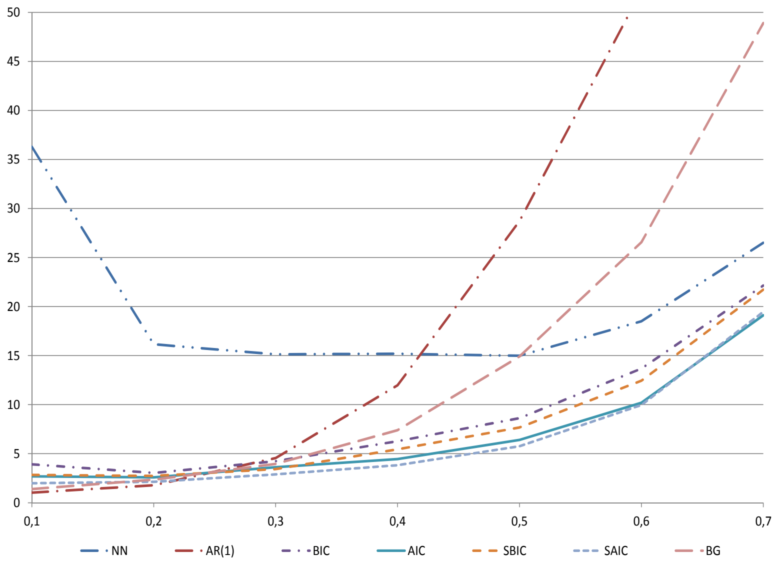
Özbay, B. (2012). Modeling the effects of meteorological factors on SO_2 and PM_{10} concentrations with statistical approaches. *Clean Soil, Air, Water* **40**, 571–577.

Figure 1: MSFE as a function of ρ when y_t follows an AR(1) process.



Note: The horizontal and vertical axes display the AR parameter ρ and MSFE, respectively.

Figure 2: MSFE as a function of θ when y_t follows an MA(1) process.



Note: The horizontal and vertical axes display the MA parameter θ and MSFE, respectively.

Table 1: Description of RMCAB monitoring stations.

								Zone classification				
Site	Name	Latitude	Longitude	Altitude	Location	Height	Area	Vehicles	Residential	Commerce	Institutions	Industry
St3	Carvajal	$4^\circ 35' 55.5''$	$74^\circ 09' 07.5''$	2,541 m	Rooftop	4 m	South	High	X	X		X
St5	IDRD	$4^\circ 39' 30.5''$	$74^\circ 05' 2.3''$	2,577 m	Ground	0 m	Center	High	X	X	X	
St6	Ferias	$4^\circ 41' 37.0''$	$74^\circ 05' 09.5''$	2,563 m	Ground	0 m	Northwest	High	X	X		
St7	Cazuca	$4^\circ 35' 57.1''$	$74^\circ 11' 17.4''$	2,546 m	Rooftop	8 m	Southwest	High	X			X
St8	Guaymaral	$4^\circ 47' 13.0''$	$74^\circ 02' 52.0''$	2,575 m	Ground	0 m	North	High	X			
St9	Kennedy	$4^\circ 37' 29.9''$	$74^\circ 09' 40.7''$	2,569 m	Ground	0 m	Southwest	High	X			X
St14	Fontibon	$4^\circ 40' 23.1''$	$74^\circ 08' 42.2''$	2,576 m	Rooftop	20 m	Southwest	High	X			X
St16	Vitelma	$4^\circ 34' 32.2''$	$74^\circ 04' 24.1''$	2,789 m	Ground	0 m	South	High	X			

Notes: Sites 3, 7, 9 and 14 are located in industrial areas with high vehicular traffic. Sites 6 and 8 are located in areas influenced mainly by vehicular traffic. Site 16 is a weather station while the other sites report information of both pollutants and weather. The stations used to apply SDEM for pollutants are: St7 and St8 for PM_{10}, St6 and St14 for CO, St6 and St14 for NO_2, St6 and St14 for NO_x, St9 and St14 for SO_2, and St3 and St14 for O_3. The stations used to apply SDEM for meteorological variables are: St8 and St14 for WS, St8 and St14 for WD, St8 and St14 for RAIN, St8 and St14 for TMP, and St8 and St16 for SR.

Table 2: Descriptive statistics of daily series for St5.

Variable	Units	Mean	STD	Min	Max
		Pollutants			
PM_{10}	$\mu g/m^3$	42.5	19.6	6.2	125.2
CO	ppm	0.8	0.4	0.1	4.6
NO_2	ppb	17.6	9.1	1.1	67.5
NO_x	ppb	41.4	22.0	3.0	167.1
SO_2	ppb	6.4	3.6	0.3	19.4
O_3	ppb	13.8	5.5	2.1	47.5
		Meteorological variables			
SR	W/m^2	167.4	52.3	43.6	347.9
WD	degrees	204.1	49.6	71.1	308.3
WS	m/s	1.3	0.7	0.2	9.2
TEMP	$^\circ C$	14.4	1.3	9.5	21.2
RAIN	mm	2.4	5.2	0	49.4

Notes: STD refer to the estimated standard deviation.

Table 3: Unit root tests.

Variable	ADF	PP	ADF–GLS
	Pollutants		
PM_{10}	−7.58	−21.79	−7.53
CO	−8.69	−21.72	−7.02
NO_2	−6.08	−13.62	−4.40
NO_x	−7.52	−21.16	−5.93
SO_2	−6.32	−15.28	−5.92
O_3	−5.98	−19.77	−5.85
	Meteorological variables		
SR	−16.44	−33.24	−7.59
WD	−22.42	−24.18	−22.06
WS	−9.35	−20.28	−9.29
TEMP	−5.47	−20.03	−4.38
RAIN	−13.29	−37.44	−12.25

Notes: All test values in the table are insignificant at the 1% level.

Table 4: MSFE and relative MSFE for forecast combinations based on all models.

	Forecast combinations										Individual forecasts			
Pollutant	Median	Mean	BIC	AIC	SBIC	SAIC	BG	PLS	GR	Mellows	Min	Max	NN	AR(1)
							MSFE							
PM_{10}	0.140	0.138	0.155	0.155	0.155	0.154	0.137	0.148	0.143	0.152	0.142	0.230	0.153	0.142
CO	0.067	0.073	0.068	0.068	0.068	0.067	0.068	0.063	0.060	0.064	0.059	0.166	0.069	0.058
NO_2	0.067	0.070	0.071	0.068	0.071	0.068	0.068	0.071	0.065	0.068	0.065	0.193	0.072	0.066
NO_x	0.155	0.152	0.164	0.163	0.164	0.162	0.152	0.162	0.153	0.155	0.155	0.188	0.164	0.159
SO_2	0.255	0.451	0.212	0.214	0.212	0.212	0.282	0.208	0.213	0.217	0.208	1.371	0.213	0.203
O_3	0.083	0.096	0.077	0.077	0.077	0.077	0.085	0.076	0.076	0.077	0.076	0.209	0.076	0.078
							Relative MSFE							
PM_{10}	1.017	1.003	1.126	1.126	1.126	1.122	1.000	1.078	1.041	1.105	1.031	1.674	1.112	1.038
CO	1.157	1.258	1.165	1.165	1.162	1.155	1.167	1.079	1.029	1.102	1.009	2.850	1.193	1.000
NO_2	1.025	1.080	1.083	1.041	1.083	1.040	1.037	1.095	1.000	1.038	1.002	2.960	1.106	1.018
NO_x	1.018	1.000	1.080	1.071	1.080	1.070	1.001	1.071	1.010	1.018	1.019	1.239	1.080	1.051
SO_2	1.252	2.221	1.044	1.051	1.044	1.041	1.385	1.022	1.050	1.066	1.022	6.748	1.048	1.000
O_3	1.105	1.266	1.019	1.019	1.020	1.020	1.126	1.011	1.000	1.020	1.007	2.763	1.011	1.026

Notes: Median, mean, BIC, AIC, SBIC, SAIC, BG, PLS, GR and Mellows refer to different forecast combination methods. Min and max refer to the minimum and maximum MSFE of the individual forecasts used for constructing the forecast combinations. The relative MSFE is the MSFE of each forecasting method divided by the MSFE of the best performing method for each pollutant.

Table 5: Relative MSFE for forecast combinations based on subsets of models.

Pollutant	Median	Mean	BIC	AIC	SBIC	SAIC	BG	PLS	GR	Mellows
Models based on Autometrics										
PM_{10}	1.032	1.017	1.000	1.000	1.000	1.000	1.022	1.000	1.011	1.000
CO	0.972	1.025	1.000	1.000	1.000	1.000	1.016	1.040	1.045	1.000
NO_2	0.999	0.984	1.000	1.000	1.000	1.000	0.994	0.934	1.009	1.000
NO_x	0.993	0.998	1.000	1.000	1.000	1.000	0.999	0.989	0.994	0.997
SO_2	0.857	0.934	1.000	1.000	1.000	1.000	0.914	1.000	1.000	1.000
O_3	0.940	0.951	1.000	1.000	1.000	1.000	0.956	1.000	1.013	1.001
Models without other pollutants										
PM_{10}	1.014	1.009	1.000	1.000	1.000	1.003	1.004	1.009	1.016	1.000
CO	0.981	1.015	1.000	1.000	1.003	1.009	1.027	1.000	1.057	1.047
NO_2	0.987	0.947	1.000	1.040	1.000	1.041	0.966	0.962	1.029	1.016
NO_x	1.001	1.001	1.002	1.010	1.004	1.013	1.001	0.994	1.010	1.014
SO_2	0.986	0.919	1.000	0.993	1.000	1.003	0.975	1.022	1.019	1.000
O_3	0.948	0.939	1.000	1.000	0.999	0.997	0.962	1.000	0.995	0.995
Models without quadratic meteorological variables										
PM_{10}	0.998	0.997	1.000	1.000	1.000	1.001	0.999	1.019	1.006	1.001
CO	0.979	1.026	1.000	1.000	1.000	1.000	1.018	0.994	1.002	0.998
NO_2	1.003	0.956	1.000	1.000	1.000	1.000	0.975	1.000	0.986	0.982
NO_x	1.008	1.001	1.000	1.000	1.000	1.000	1.002	1.000	0.989	0.995
SO_2	0.969	0.929	1.000	0.993	1.000	1.003	0.965	1.034	1.021	1.000
O_3	0.975	0.962	1.000	1.000	0.999	0.999	0.979	1.034	1.009	1.000

Notes: The numbers in the table are the MSFE of each forecast combination method for a particular subset of models divided by the MSFE of that method when using all models. See Table 4 for an explanation of the rest.

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