UNIVERSITY OF GOTHENBURG
SCHOOL OF BUSINESS, ECONOMICS AND LAW

WORKING PAPERS IN ECONOMICS

No 529

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Jorge Bonilla, Jessica Coria and Thomas Sterner

April 2012

ISSN 1403-2473 (print)
ISSN 1403-2465 (online)

Department of Economics
School of Business, Economics and Law at University of Gothenburg
Vasagatan 1, PO Box 640, SE 405 30 Göteborg, Sweden
+46 31 786 0000, +46 31 786 1326 (fax)
www.handels.gu.se info@handels.gu.se

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

Jorge Bonilla[(1,2)], Jessica Coria[(1)], and Thomas Sterner[(1)]

University of Gothenburg

Abstract

In this paper, we explore the synergies and tradeoffs between abatement of global and local pollution. We built a unique dataset of Swedish heat and power plants with detailed boiler-level data 2001-2009 on not only production and inputs but also emissions of CO2 and NOx. Both pollutants are subject to strict policies in Sweden. CO2 is subject to multiple levels of governance using environmental instruments such as the EU ETS and Swedish carbon taxes; NOx – 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. The fact that generating units face a trade-off between the pollutants indicates a need for policy coordination.

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

Key Words: Interaction of environmental policies, shadow pricing, directional distance function, climate change, local pollution.

Acknowledgments: Jorge Bonilla gratefully acknowledges financial support from the Swedish International Development Cooperation Agency (SIDA) to the capacity building program at the Environmental Economics Unit, University of Gothenburg, and from the Universidad de los Andes, Colombia. Jessica Coria and Thomas Sterner gratefully acknowledge financial support from Göteborg Energi and the Swedish Energy Agency. Valuable comments from Rolf Färe are gratefully acknowledged. This paper is dedicated to the memory of our friend and colleague Lennart Hjalmarsson. We are grateful for his generous advice on an early version but he unfortunately passed away during the course of this project.

[(1)] Department of Economics, University of Gothenburg, Sweden. Email addresses: Jorge.Bonilla@economics.gu.se; Jessica.Coria@economics.gu.se; and Thomas.Sterner@economics.gu.se

[(2)] Universidad de los Andes, Bogotá, Colombia.

I. 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, subnational, national, and transnational levels. Usually, these decision makers are not fully coordinated with respect to goals and methods, and the existence of several layers of governance may encourage strategic behavior from powerful local actors trying to enhance their own positions (Nijkamp and Rietveld, 1981, Caillaud et al., 1996, Eichner and Pethig, 2009). 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 CO2 emissions might therefore create spillovers, i.e., 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 CO2 emissions is switching the fuel mix from fuel oil towards bio-fuels. However, while net CO2 emissions do fall dramatically with biofuels, they often imply an increase in nitrogen oxides (NOx), particulate matter (PM), carbon monoxide (CO), and volatile organic compound (VOC) emissions (Brännlund and Kriström, 2001, Burtraw et al., 2003).

The aim of the present paper is to study the effects of the interaction between the European Union Emissions Trading System (EU ETS), the Swedish CO2 tax and a refundable charge on NOx on the relative performance of Swedish plants with respect to CO2 and NOx emissions. For this purpose, we built a unique dataset of Swedish heat and power plants for the period 2001-2009 that contains detailed boiler-level data not only on production and inputs but also on the two pollutants: CO2 and NOx. We use a quadratic directional output distance function to study and compare patterns of technical progress, substitution between CO2 and NOx, 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 long been at the forefront of GHG emissions reduction, and the country is one of few that present emissions below the level recorded in 1990, which is mainly explained by a number of initiatives taken in order to reduce CO2 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. Thus, the tax is relatively high on coal, lower on oil, and lower still on natural gas, whereas biofuels such as wood and ethanol are exempt. Since 2005, most installations within the heat and power sector are part of the EU ETS – the cornerstone of the European Union's policy to combat climate change.

The main function of the carbon tax is to reduce carbon dioxide emissions by reducing the demand for the taxed fuels. The EU ETS has the same purpose, but unlike the carbon tax it does not cover all sectors of the economy. Hence, insofar as the carbon tax and the emissions trading scheme do not overlap, they complement one another. Nevertheless, from 2005 to 2008, heat and power installations were not exempt from the carbon tax; they continued to pay the tax (although at a reduced rate) and also took part in emissions trading. To address the potentially negative effect of policy overlapping on the competitiveness of the Swedish industry, the government took the first step towards abolishing the CO2 tax within the EU ETS on July 2008, a still ongoing process.

The most obvious effect of the carbon tax and the EU ETS has been the expansion of biomass use. Indeed, fuel substitution seems to have played the most important role for reduction of carbon emissions in Sweden (Löfgren and Muller, 2010), with the potential negative side effect of increased NOx emissions. Since Sweden has ecosystems that are naturally very sensitive to acidification, the country has a very aggressive policy on the precursors, notably NOx. Indeed, to reduce the emissions of this pollutant, a heavy refundable charge has been in place since 1992. A significant reduction in NOx emission was noted after the implementation of the NOx charge. In fact, NOx emissions decreased much faster than expected; the initial target of achieving a 35% reduction from 1990 levels by 1995 was achieved already in 1993. However, the path of reduction did not show significant progress during the last decade, and hence, the charge level was raised in 2008 to encourage further NOx 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, e.g., 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 SO2 and the output substitution between electricity and SO2 before and after the implementation of Phase I of the Acid Rain Program in United States. Unlike Färe et al. (2005), we focus on policy-induced substitutability *across* pollutants and the changes in the relative prices of emissions introduced by environmental multi-level governance.

To what extent could an increased relative CO2/NOx price have contributed to increased/decreased emissions of NOx? If the joint abatement creates some synergies, pollutants can be considered as complements in the abatement process. Instead, if CO2 emissions are reduced at the cost of increasing NOx emissions, they can be considered as substitutes. Theory shows that an increased relative CO2/NOx price might lead NOx emissions either to increase or decrease depending on whether NOx is a substitute or a complement to CO2 (see Ambec and Coria, 2011). Technological progress does, however, also play an important role as it might lead to reduced emissions of one or 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., CO2 and/or NOx saving) and on whether the technological progress outweighs the substitution effect.

This paper is organized as follows. Section II briefly describes the climate and NOx policies in Sweden and the changes in the relative price of CO2 to NOx over the period 2001-2009. Section III presents the theoretical and empirical framework of the joint production of heat and power, CO2 and NOx emissions. Section IV discusses the data and the empirical results. Finally, Section V concludes the paper.

II. Climate and NOx Policy in Sweden: Carbon Tax, EU ETS, and the Refundable NOx Charge

Since the climate challenge discussion in the 1980s had largely focused 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. For example, there is no carbon tax on electricity production, but non-industrial consumers have to pay a tax on electricity consumption. Initially, the general tax was equivalent to 25 €/ ton of CO2. However, it has increased steadily over 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 into context, the carbon dioxide permits on US markets such as RGGI and Chicago are trading at around 4 €/ ton; the EU ETS has varied around a mean of 15-20 €/ ton; and France tried to introduce a carbon tax of 17 €/ ton, but failed due to fears that such a level would be detrimental to the economy.

Since the tax is very high and Sweden is a small open economy, there has been quite some concern about the competitiveness of some energy-intensive industries. Thus, a number of deductions and exemptions have been created in sectors that are open to competition, and a series of reduced rates have been introduced. In the case of the heat and power sector, the carbon tax varies according to the type of generation. As mentioned in the introduction, until July 2008 the carbon tax and the EU ETS overlapped. At that time, 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%, while district heating plants (hereinafter DH) were granted a reduction of only 6%.[1] Since the level of the permit price is much lower than the Swedish tax level, this harmonization with the EU actually implies a sizeable fall in the price of carbon emissions for some Swedish plants.

[1] A CHP is a generating unit that produces heat and electricity simultaneously. A DH generating unit only produces heat (for district heating – and is thus almost completely sheltered from competition). Cogeneration has been promoted within the European Union as an effective means to increase the overall energy efficiency (EU Directive 2004/8/EC). 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 to electricity; the rest becomes heat (Svensk Fjärrvärme, 2011).

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 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 the very low allowance 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. After remaining at levels of 22-23 €/ton in early 2008, the CO_2 price dropped during the market crash. Although the fundamentals of the carbon market in the road ahead to Phase III remain 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 nuclear power. It then decreased after June of the same year when the European Commission proposed a mandatory target for energy efficiency improvements, and shot up again with increasing gas and power prices in August.

(INSERT FIGURE 1 HERE)

The 1991 tax reform introduced not only carbon taxes but also other taxes, including 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.

The total collected NO_x 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. The fee was originally set at 4.3 €/kg – which again is an extremely high level compared to other countries. The figure corresponds to 4,300 €/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 the US of approximately 600 €/ton.[3] The tax 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, e.g., Höglund, 2005; Sterner and Höglund, 2006; and Sterner and Turnheim, 2009), and has then been shown to reduce emissions very effectively. The empirical findings suggest that extensive emission reductions have been achieved as a

[2] With the exception of the 0.7 percent that is kept for administrative costs.
[3] Price in 2008. NO_x prices have dropped dramatically since then to approx. 12 €/ton in 2011.

result of learning and technological development in abatement. NOx 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 NOx reduction through various technical measures (including changing the shape, temperature, or oxygen and moisture content of the combustion chamber), fuel switching, and other abatement strategies including the addition of ammonia or passing exhaust through catalytic converters (Millock and Sterner, 2004). Adoption of NOx 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, 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 CO2 and NOx emissions? To assess the overall picture is not easy. Despite the fact that 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 CO2 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 CO2 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 CO2 and NOx policies, we compute the relative opportunity cost of CO2 and NOx emissions per unit of output. As shown in Figure 2, it seems clear that Swedish policy signals have encouraged power companies to avoid fossil fuels. The opportunity cost of CO2 emissions is much larger than the cost of emitting NOx. For example, in 2003, an average CHP plant emitted 0.082 tons of CO2 and 0.248 kilograms of NOx to produce 1 Mwh of useful energy. Given the magnitude of the carbon tax and NOx 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 CO2 emissions per unit of output was 6.54 times the cost of NOx emissions. The figure is similar for DH plants, as the opportunity cost of CO2 emissions per unit of output in 2003 was 4.58 times the cost of NOx emissions.

Moreover, the variation observed in Figure 2 suggests that CO2 policy, on average, did become more stringent (relative to NOx) immediately after the introduction of the EU ETS, yet since 2008 it has become less stringent due to the carbon tax phase-out. Indeed, the relative CO2/NOx opportunity cost has decreased for both an average DH and an average CHP after 2008 as a result of the reduction in the share of the carbon tax paid by the heat and power sector, though this effect is rather modest for DH plants as they continued to pay 94% of the carbon tax. In the case of the CHP plants, the reduction started earlier as they were granted a significant carbon tax reduction already in 2004. Moreover, Sweden increased the NOx charge for all regulated boilers in 2008, adding to the effect of the reduced carbon tax on the relative CO2/NOx opportunity cost.

It is clear that most abatement efforts should focus on reducing CO2 emissions as the economic effect of CO2 regulations on firms' profitability is much larger than that of NOx regulation. However, the variations in the opportunity costs of CO2 and NOx emissions should induce some variations in the relative CO2-NOx 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 CO2-NOx emissions would depend, however, on a series of factors, including technological development and whether CO2 and NOx are substitutes or complements in abatement. For instance, in the absence of technological development, one would expect NOx emissions from CHPs to decrease to a relatively larger extent during the period 2006-2009 than during the period 2001-2004 if pollutants are substitutes, while the reverse should hold if pollutants are complements. For DH plants, the trend should be the opposite: the mean CO2/NOx opportunity cost was overall larger during the period 2006-2009, which should have triggered more efforts to reduce CO2 emissions compared to NOx if these pollutants are substitutes.

On the other hand, the high relative opportunity cost of CO2 emissions should also trigger technological fixes and fuel switching aiming to reduce them. For instance, CO2 emissions can be reduced by raising combustion temperature – but this increases NOx emissions. Alternatively, NOx emissions can be reduced through post combustion measures, yet such measures usually require energy and thus increase carbon emissions (or reduce output). Hence, given the relatively stringency of CO2-NOx regulations, we should expect technological development to be overall biased toward CO2 emissions reductions. In the following sections, we will 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 first we will describe the estimation strategy.

(INSERT FIGURE 2 HERE)

III. Estimation Strategy and Data

In our approach, we employ the directional output distance function to derive estimates of technical efficiencies, pollutant-output elasticities of substitution, elasticities of substitution between pollutants, and relative CO2/NOx shadow prices. This function 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 to reduce the bad outputs (emissions) while increasing, or holding 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 (see Färe et al., 2005 for details).

Let the set of output possibilities $P(x) = \{(y, b): x \text{ can produce } (y, b)\}$ represent all feasible input-output possibilities of boilers that jointly generate heat and/or power (*y*) and CO2 and NOx emissions (denoted $b = (b_1, b_2)$, 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 P(x) is compact and 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.[4] Moreover, all outputs are assumed to be jointly weakly disposable. That is, for a given input vector, reduction in emissions is feasible if good output is proportionally reduced.[5] In other words, emissions can be reduced at the expense of reduced output. Finally, we consider that the good output is strongly disposable[6], 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, the function finds the simultaneous maximum expansion in good output and contraction in undesirable outputs. In this sense, 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 $(b - \beta^* g_b, y + \beta^* g_y)$.

[4] Formally null-jointness implies that if $(y, b) \in P(x)$ and $b = 0$, then $y = 0$.
[5] Under weak disposability, if $(y, b) \in P(x)$ and $0 \leq \theta \leq 1$, then $(\theta y, \theta b) \in P(x)$.
[6] Free disposability of output can be expressed as: if $(y, b) \in P(x)$ and $(y', b) \leq (y, b)$, then $(y', b) \in P(x)$. In a similar manner, strong disposability of inputs can be written as: if $x' \geq x$, then $P(x') \supseteq P(x)$.

The directional output distance function has several properties: (i) It is non-negative for feasible output vectors.[7] 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, i.e., (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 by definition satisfies the translation property, expressed as $\vec{D}_o\left(x, y + \alpha g_y, b - \alpha g_b; g\right) = \vec{D}_o(x, y, b; g) - \alpha$, $\alpha \in \Re$.[8]

The directional output distance function approach allows us not only to account for technical efficiency but also to calculate the elasticity of substitution between CO2 and NOx emissions and the relative shadow prices of these pollutants. Regarding the former, the Morishima elasticity of substitution provides us with information about 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 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 it 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 CO2 and NOx 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 $M_{b_1 b_2} > 0$, then b_1 and b_2 are Morishima substitutes. That is, the pollutants are substitutes if the

[7] This feature of the directional output distance function is related with the property that P(x) or $\vec{D}_o(x, y, b; g)$ can equivalently describe the technology, which is valid if g-disposability is satisfied; i.e., if $(y, b) \in P(x)$, then $\left(y - g_y, b + g_b\right) \in P(x)$.

[8] The translation property is equivalent to homogeneity in outputs of the Shepard's (1970) output distance function (see Färe et al., 2006). Under this property, the inefficiency can decrease by the amount of α if bad outputs are contracted by αg_b and the good output is expanded by αg_y. For a description of the directional output distance function's properties, see Färe et al. (2005).

emission intensity of b_2 relative to 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 ($M_{b_1 b_2} \neq M_{b_2 b_1}$), and can be represented as the difference between two price elasticities, i.e., $M_{b_1 b_2} = \varepsilon_{b_2 b_1} - \varepsilon_{b_1 b_1}$, where $\varepsilon_{b_i b_j} = \partial ln b_i / \partial ln q_j$ (Stiroh, 1999, page 6). Indeed, consider the effect of a change in the price ratio q_1/q_2 on the emissions intensity b_1/b_2. A change in q_1 – holding q_2 constant – has two effects on the quantity ratio: the first term $\varepsilon_{b_2 b_1}$ shows the effect on q_2, and the second term $\varepsilon_{b_1 b_1}$, shows the effect on q_1. In contrast, a change in q_2 – holding q_1 constant – has two different effects on the input quantity ratio, which are given as $M_{b_2 b_1} = \varepsilon_{b_1 b_2} - \varepsilon_{b_2 b_2}$. There is no a priori reason that these effects should be the same. 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).

With regard to pollutant-output elasticities, since $\vec{D}_o(x, y, b; g)$ satisfies monotonicity, concavity, and translation properties, then $M_{b_i y} \leq 0$[9]. Higher pollutant-output elasticities in absolute values indicate that it is more difficult (or more costly) for the generating units to reduce emissions.

Regarding shadow prices, it can be shown that for a given vector $g = \left(g_y, g_b\right)$ and a differentiable directional output distance function, the relative shadow price between undesirable outputs is[10]:

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

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 $-q_{b_i}/p = D_{b_i}/D_y$. The shadow prices are calculated on the technically efficient frontier of P(x).[11]

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

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

[10] For a more 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).

[11] 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.

advantage that it tends to have a better adjustment than trans-log specifications (Färe et al., 2010). 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:

$$\begin{aligned}\vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1) \\ &= \alpha + \sum_{n=1}^{3} \alpha_n x_{nk}^t + \beta_1 y_k^t + \sum_{i=1}^{2} \theta_i b_{ik}^t + \tfrac{1}{2} \sum_{n=1}^{3} \sum_{n'=1}^{3} \alpha_{nn'} x_{nk}^t x_{n'k}^t + \tfrac{1}{2} \beta_2 y_k^{2\,t} \\ &+ \tfrac{1}{2} \sum_{i=1}^{2} \sum_{i'=1}^{2} \theta_{ii'} b_{ik}^t b_{i'k}^t + \sum_{n=1}^{3} \sum_{i=1}^{2} \eta_{ni} x_{nk}^t b_{ik}^t + \sum_{i=1}^{2} \mu_i y_k^t b_{ik}^t + \sum_{n=1}^{3} \delta_n x_{nk}^t y_k^t \qquad (5)\end{aligned}$$

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:

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

We estimate the directional output distance function using a deterministic method, i.e., Parametric Linear Programming (PLP), which allows us to impose parametric restrictions that are a result of the underlying technology, e.g., monotonicity in good or bad outputs[12]. We follow Aigner and Chu's (1968) 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 with the technology properties already described (see Färe et al., 2001, 2005, 2006). We derive estimates of the coefficients for the period of pre-introduction of the EU ETS (2001-2004) and for the post implementation period (2006-2009). In order to control for possible yearly changes, e.g., variations in weather (some winters may be colder than others) and general shocks of the economy that may affect

[12] The directional output distance function may also be estimated using stochastic approach. Although the stochastic frontier method has the advantage of dealing with issues such as measurement errors and also allows testing statistical hypotheses within the model, it requires the inclusion of distributional assumptions for the inefficiencies scores and residuals that are arbitrary and may be violated. Moreover, with the implementation of the 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; properties that cannot be simply introduced in the model as linear restrictions of the parameters.

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 (see Appendix A for details of the minimization problem).

Using the estimated coefficients of the directional distance output function, we compute the technical and environmental efficiencies and the Morishima elasticities of substitution between CO2 and NOx and between pollutants and output according to equations (2) and (3), respectively. The relative CO2/NOx shadow prices are obtained by applying equation (4). To identify changes in efficiencies, elasticities, and relative shadow prices from before to after the EU ETS implementation, 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)[13], 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 Appendix B for details on the T_n-statistic of Li et al., 2009). 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 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 heat and power sector in 2001-2009. Our measure of good output is the amount of useful energy (MWh) commercially sold. The two undesirable outputs, CO2 and NOx emissions, are expressed in tons, and as stated above, the inputs are installed capacity (MW), fuel consumption (MWh), and labor (number of employees). NOx emissions and useful energy are taken from the Swedish Environmental Protection Agency's (SEPA) NOx charge database. These two variables are measured and reported to the SEPA directly by the generating units along with information about energy fuel shares, installed capacity, and the availability of NOx post-combustion (PCT)[14] and combustion technologies (CT)[15],

[13] 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.

[14] PCT are flue gas treatments that aim to clean up NOx once it has been formed, usually through conversion to benign chemical substances. It includes: 1) Selective Catalytic Reduction (SCR), the installation of which is rather large and costly but yields highly efficient reduction, and 2) Selective Non-Catalytic Reduction (SNCR) of chemicals (ex: ammonia, urea, sodium bicarbonate), which implies lower costs than SCR, since there is no need to cool the gases or use a catalyst. Although SNCR is less costly than SCR in both capital and operating costs, it is less effective (Sterner and Turnheim, 2009).

[15] CT involves combustion measures that seek to inhibit the formation of thermal and prompt NOx. These strategies typically involve optimal control of combustion parameters (temperature, pressure, stoichiometry, flame stability and homogeneity, and flue gas residence time) for minimal NOx formation. Fine tuning of operational conditions has turned out to be a significant means of abatement (Sterner and Turnheim, 2009). This

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 used as a proxy for capital in physical units. With regard to labor, company-level data were gathered from Retriever Bolagsinfo. For multi-unit plants, we allocated labor to generating units according to their useful energy ratios.

Although we have CO2 values from the SEPA's EU ETS database, its aggregation at the installation level prevents us from recovering the emissions for each boiler. Instead, CO2 emissions are estimated based on available data of boilers about energy fuel shares and emission factors per fuel type. Hence, we can recover the total energy input 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[16]. However, this method only considers emissions from fuel used for combustion[17] and excludes emissions coming from raw materials, which are not significant in the case of heat and power generation – unlike other industries such as metal, refineries, and chemistry. [18]

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, the information for the year 2005 is excluded from the sample. Two boilers for which there is a lack of labor data and two boilers for which there is a lack of information on fuel shares are dropped. Finally, 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 111 boilers (888 observations).

Table 1 presents descriptive statistics of the variables. As we can see, there is large variation in emissions among boilers. The yearly CO2 and NOx emissions per generating unit vary from zero to

technology may include: trimming the burner and air supply, LNB (Low NOx burners), OFA (Over Fire Air), ROFA (Rotating Over Fire Air), and RGA (flue gas recirculation).

[16] Each fuel consists of some sub-classifications. For instance, gas may include natural gas, LPG and biogas. Oil types are fuel oil 1, fuel oil 5 and bio-oil. In the biofuels 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). In the case of waste, besides its emission factor, it is also assumed that the fossil fraction of the fuel corresponds to 12.6% of the total weight.

[17] The term combustion means combustion of fuel in the presence of oxygen, in contrast to processes defined as reactions between substances, reduction processes, thermic decomposition of substances, and formation of substances used as products or process material (see also law NFS 2007:5 3 § point 11 and 29 in SEPA, 2009).

[18] For comparison purposes, we also estimated the CO2 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. We conducted a statistical mean comparison test of the two procedures and did not find evidence to reject the null hypothesis that the means are equal at the 1% significance level. Although the two estimation approaches are equivalent, we preferred the first procedure. 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 it was possible to make such aggregation. Our estimations were in a similar order of magnitude.

383,000 tons and from 3.9 to 358.5 tons, respectively. This reflects differences within the sector in fuel mix, fuel usage, boiler size, and NOx abatement strategies. The amount of useful energy produced by an average boiler is around 213 GWh per year in both periods.

(INSERT TABLE 1 HERE)

IV Results

In this section we describe and analyze the results of our estimates of technical efficiencies, substitution elasticities, and shadow prices. We present the results for the pooled sample including combining CHP and district heating plants. However, we report results by subgroup when there are statistically significant differences in the direction and magnitude of the estimates for CHP and DH plants.

4.1 Technical Efficiency and Technical Progress

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

(INSERT TABLE 2 HERE)

Stringent environmental regulations have a positive effect not only 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 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 shown 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 toward CO2 emissions reduction.[19] Figure 3 also shows the actual relative prices of CO2 and NOx emissions for each year[20] and we see that the optimal mix of CO2-NOx 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 CO2 emissions even at the expense of increasing NOx 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, yet the efficiency increase is much larger in the case of CO2. We could thus say that the technical change is "carbon saving" in much the same way as technical changes have traditionally often been characterized as "labor saving." On average, CO2 emissions per MWh in the heat and power sector decreased by 20% between the periods (from 9.4 MWh/ton in 2001-2004 to 11.88 MWh/ton in 2006-2009); and the corresponding reduction for NOx was approximately 9% (from 4,822 to 5,337 MWh/ton).

(INSERT FIGURE 3 HERE)

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 111 boilers operating during the period 2001-2009, only 12 and 11 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.[21] The estimations of technical and environmental efficiency yield mean inefficiency values of 23.7% and 33.2% for the pre- and post-introduction periods of the EU ETS, respectively. This indicates that during 2001-2004, boilers on average could have expanded their heat and power generation by 50.71 GWh (i.e., 213.625∗0.237) and contracted CO2 emissions by 5,392 tons (22,716.4∗0.237) and NOx emissions by 11 tons (44.3∗0.237) had they adopted the best practice of frontier generating units. Similarly, for 2006-2009, boilers could have increased their production by 71.07 GWh and decreased their CO2 and NOx emissions by 5,984 and 13 tons, respectively, had the generating units worked at the frontier. Given these results, the amount of possible reduction in emissions and increase in heat and power generation that could have

[19] 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, and found technical progress supported by a negative coefficient of the time trend.

[20] The price of CO2 emissions is given by the sum of the carbon permit price and relevant tax – given the fuel consumption and fuel mix of a representative firm at the frontier.

[21] Mean inefficiencies for boilers using biofuel range from 18 to 29%. On the other hand, average inefficiencies in boilers employing fossil fuels vary from 49 to 53%.

been achieved between the periods following the practice of the most efficient generating units is of considerable magnitude. Moreover, the fact that relative inefficiency appears to increase implies that technical progress was fastest at the frontier and slower for average plants. In the long run, this suggests that considerable further progress can 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-statistic, which allows us to compare the inefficiency distributions, is equal to 19.84 with an empirical p-value of 0.924. Hence, by this test, we cannot reject the null hypothesis of equality between the two density functions of inefficiencies.

4.2 CO2-Output and NOx-Output Elasticities

With regard to the pollution-output elasticities, Figures 4a and 4b illustrate the kernel distributions of pollution-output between the two periods. When it comes to CO2, the mean output elasticity changed from -0.666 to -0.299 between the periods. The mean NOx-output elasticity changed from -1.335 to -0.456. Using the Li et al. test, we are able to reject the null hypothesis of equality between the density functions of the pollution-output elasticities pre- and post-introduction of the EU ETS for both pollutants, i.e., the responsiveness of the ratio of emissions/output to emission price decreased. These results indicate that the easiest emission reductions have already been undertaken, and that further reductions of CO2 and NOx emissions per unit of output will be very difficult to achieve and will only be supported by much higher charges on CO2 and NOx emissions.

(INSERT FIGURE 4a HERE)

(INSERT FIGURE 4b HERE)

The results in the previous section indicate that several technical measures have been implemented in order to reduce emissions. Regarding CO2 emissions, the trend toward 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 74%. The fraction of boilers using mainly fossil fuels was very low (11%). Regarding NOx emissions, the fraction of generating units without any abatement measure has declined from 24% to 16% during 2001-2009. Instead, there is now a clear trend toward simultaneous adoption of more than one NOx reduction technology (i.e., the fraction of units using more than one technology has increased from 16% to 32% in 2001-2009; see Table 3).

(INSERT TABLE 3 HERE)

4.3 CO2-NOx substitution

One of the main purposes of this study is to assess the existence of substitutability between CO2 and NOx and the changes induced by regulatory changes. Our results indicate that CO2 and NOx are substitutes, though substitution decreased during the period 2006-2009 in response to variations in relative prices. For instance, the mean estimates of the Morishima CO2-NOx elasticity (M_{b_1,b_2}) fell from 0.680 in 2001-2004 to 0.169 in 2006-2009. The mean estimates of the corresponding elasticity NOx-CO2 (M_{b_2,b_1}) fell from 0.319 to 0.114. The asymmetry of the elasticities indicates that the substitutability of NOx for CO2 is lower than the substitutability of CO2 for NOx. In other words, CO2 emissions are more sensitive than NOx emissions to price changes. Clearly, this is linked to the fact that CO2 regulations have a much higher opportunity cost than the NOx fee. A policy implication that could be derived from this is that the NOx fee would have to be increased to a much higher level than today in order to achieve a substantial reduction in NOx emissions.

The decreased substitution elasticity is also observed when classifying by fuel groups (biofuel>80%, biofuel>rest, and others) and types of NOx abatement technologies. When it comes to type of generation (i.e., DH vs. CHP), we observe that for DH plants the mean CO2-NOx elasticity (M_{b_1,b_2}) actually increased slightly between the periods, from 0.626 in 2001-2004 to 0.728 in 2006-2009, which seems logical given that this group faced a higher opportunity cost of CO2 emissions after the implementation of the EU ETS (since these plants still have to pay 94% of the Swedish carbon tax). Like for the whole sample, the NOx-CO2 elasticity (M_{b_2,b_1}) for DH plants is lower in the second period – down from 0.383 in 2001-2004 to 0.237 in 2006-2009.

Kernel distributions of the elasticities between pollutants for the entire sample are depicted in Figures 5a and 5b. Using the bootstrapped Li et al. T_n-statistics, 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 CO2-NOx does tend to shift towards the left-hand side of the distribution. In the case of the NOx-CO2 elasticity, we also observe that for many generating units the estimates are concentrated around zero. Further CO2-NOx substitution has become less likely since most firms have already implemented the technical measures that are economically feasible (notably, they have almost completely switched to biomass). This trend should be clearer when we exclude from the sample all the boilers that are fully operating with bio-fuels (approx. 40% of the sample). As total fuel conversion to biofuels is a possible option for this group we should expect to observe a higher responsiveness of the relative intensity of emissions CO2/NOx to changes in relative prices, i.e., a higher elasticity of substitution. Indeed, our estimates indicate that the elasticities of substitution between pollutants for this group are generally higher than those estimated for the whole sample, though they still statistically decreased during 2006-2009. For instance, M_{b_1,b_2} and M_{b_2,b_1} fell on average from 1.06 to 0.45 and from 0.82 to 0.04, respectively.

(INSERT FIGURE 5a HERE)

(INSERT FIGURE 5b HERE)

4.4 Shadow Prices

Finally, we test for changes in the relative CO2/NOx shadow price. Firstly, we estimate the shadow prices for the whole sample. Secondly, we estimate the shadow prices by subgroups (CHP and DH). For the whole sample, we find a reduced CO2/NOx shadow price for the post-EU ETS introduction period. The relative CO2/NOx shadow price per ton decreases from a mean of 0.000085 in 2001-2004 to a mean of 0.000047 in 2006-2009.

When we estimate the shadow prices for subgroups, we find that – in line with Figure 2 – for CHP plants the relative CO2/NOx shadow price per ton decreased after the introduction of the EU ETS, from a mean of 0.000420 in 2001-2004 to 0.000061 in 2006-2009. Yet, not surprisingly, the reduction in relative prices does not hold for DH plants. For these generating units, the CO2 shadow price became relatively higher than the NOx shadow price (changing from a mean of 1.19e-13/ton in 2001-2004 to 3.15e-13/ton in 2006-2009), which is in line with the fact that for this group the phasing out of the carbon tax has been negligible. Overall, the group of CHP plants dominates on the magnitude and direction of the change in the relative prices. A considerable fraction of the plants belong to this group (around 74% of the whole sample).

The Li et al. (2009) test leads us to reject the null hypothesis of equality in the relative shadow price distributions between pre- and post-implementation of the EU ETS. Figures 6a and 6b illustrate the kernel relative shadow price distributions for CHP and DH plants, respectively. The direction of the changes here in relative shadow prices is compatible with the constructed relative prices of the regulation for an average DH and an average CHP calculated in Section 2, which confirms that for CHP plants the CO2 policy became less stringent since the introduction of the EU ETS, while the reverse holds for DH plants.

(INSERT FIGURE 6a HERE)

(INSERT FIGURE 6b HERE)

V. Conclusions

The implementation of environmental policies to reduce greenhouse gas emissions certainly has a global impact, but it can also bring local co-benefits (or costs) by reducing (or increasing) other air pollutants due to complementarity/substitutability between pollutants. These interactions have clear implications for policy design as many European countries are committed to reaching the Kyoto obligations (though real action has been slow up to now), and there are currently multiple policies in place aiming to reduce CO2 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 CO2 and NOx 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 CO2, NOx, 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 many regulated firms deal with. We chose a quadratic representation of the technology and subsequently derived the estimates of elasticities between CO2 and NOx 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 CO2 and NOx, 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 density functions between periods by means of a kernel-consistent density test.

Our results indicate that there are important interactions between the abatement efforts of CO2 and NOx emissions. Indeed, we find that in the heat and power generation sector, CO2 and NOx are Morishima substitutes. Overall, the degree of substitution between these two pollutants decreased after the introduction of the EU ETS as a response to the regulatory changes that led to a reduced relative CO2/NOx price. Nevertheless, for the subgroup of DH plants, for which the carbon tax reduction has been so far very small, CO2-NOx substitution increased due to the high relative carbon price they pay. The mean estimates of the CO2-NOx elasticities (M_{b_1,b_2}) for the entire sample ranged from 0.680 in 2001-2004 to 0.169 in 2006-2009, while for DH they changed from 0.626 to 0.728.

Our results also indicate that CO2 is more sensitive than NOx to prices. The asymmetric Morishima NOx-CO2 elasticity (M_{b_2,b_1}) is lower than the CO2-NOx elasticity, which means that if the regulator is to encourage a large reduction in NOx emissions, the charge must be increased to a much higher level than today. As in the previous case, the overall mean NOx-CO2 elasticity decreased after the introduction of the EU ETS, from 0.319 to 0.114.

Has the overlapping of climate policies led to increased NOx emissions? We do find a tendency in this direction, but NOx emissions did not actually increase for two reasons. First, there was significant technological development leading to reduced emissions of both pollutants. Second, contrary to what one may expect, the relative CO2/NOx prices decreased for most firms in the 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 NOx charge in 2008. This is to say that in relative terms, CO2 policies in Sweden became less stringent since 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 CO2 and NOx regulations, as reflected by the reduced shadow prices and pollutant-output elasticities. Nevertheless, our results tell us that although firms were able to decrease CO2 and NOx emissions per unit of output between the periods 2001-2004 and 2006-2009, further decreases in the CO2/output and NOx/output ratios will be more difficult to achieve as most firms have already implemented all the technical measures available, and will have to be supported by much higher charges on CO2 and NOx emissions.

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 as stated by Elinor Ostrom, we are likely to experience a polycentric approach to climate change with mitigation and adaptation activities undertaken by multiple policy actors at a range of different scales.

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Appendix A. The optimization problem

In order to estimate the directional output distance function in (5), we solve the following optimization problem:

$\min \sum_{t=1}^{T} \sum_{k=1}^{K} \left[\vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1) - 0\right]$

subject to

(a) $\vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1) \geq 0, k = 1, \dots, K,\ t = 1, \dots, T,$

(b) $\partial\, \vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1)/\partial b_i \geq 0, i = 1,2,\ k = 1, \dots, K,\ t = 1, \dots, T,$

$\partial\, \vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1)/\partial y \leq 0, k = 1, \dots, K, t = 1, \dots, T,$

(c) $\partial \vec{D}_o^t\left(\overline{x}, \overline{y}, \overline{b}, 1,1\right)/\partial x_n \geq 0, n = 1,2,3$

(d) $\partial^2\, \vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1)/\partial y^2 \leq 0, k = 1, \dots, K, t = 1, \dots, T,$

$\partial^2\, \vec{D}_o^t(x_k^t, y_k^t, b_k^t; 1,1)/\partial b_i \partial y \leq 0, k = 1, \dots, K, t = 1, \dots, T,$

(e) $\vec{D}_o^t\,(x_k^t, y_k^t, 0; 1,1)/\partial x_n < 0, k = 1, \dots, K, t = 1, \dots, T,$

(f) $\beta_1 - \theta_1 - \theta_2 = -1$, $\beta_2 - \mu_1 - \mu_2 = 0$, $\mu_1 - \theta_{11} - \theta_{12} = 0,$ $\mu_2 - \theta_{21} - \theta_{22} = 0,$

$\delta_n - \eta_{n1} - \eta_{n2} = 0,\ n = 1,2,3$

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

Constraint (a) is the feasibility condition for each generating unit's observation and corresponds to property (i). Restrictions in (b) are imposed to satisfy the monotonicity conditions of emissions and good output described in (ii) and (iii), respectively. Monotonicity of inputs at the mean level of input usage is included in constraint (c), which satisfies property (iv). In order to ensure estimates in the positive slope of the output set, we incorporate restrictions in (d); note that the first of these constraints is related as well with the concavity property in (v) since the first leading principal minor of the hessian matrix is non-positive. The null-jointness property in (vii) is imposed by constraint (e). This restriction implies in other words that if $\vec{D}_o^t(x_k^t, y_k^t, 0; 1,1) < 0$ then the output vector $(y, 0)$ is not feasible. Condition (f) ensures the translation property in (viii), which is parametrically specified by equation (6) (see Färe et al., 2006). Finally, the symmetry conditions defined in (7) are imposed by constraint (g).

Appendix B. The T_n-statistic of Li et al (2009)

The T_n-statistic is a nonparametric consistent test that assesses the equality between two density functions, $f(x)$ and $g(x)$ of a random variable x. Let $\{X_i\}_{i=1}^{K_1}$ and $\{Y_i\}_{i=1}^{K_2}$ be i.i.d. observations from populations with density functions $f(\cdot)$ and $g(\cdot)$, and $w\left(\frac{X_i - x}{h}\right)$ be a univariate kernel function associated with the continuous variable x, where h is the corresponding smoothing parameter[22]. The product kernel is represented by $Q_{x_i,x} = h^{-1}w\left(\frac{X_i - x}{h}\right)$. Thus, the density functions $f(x)$ and $g(x)$ are estimated as follows:

$$\hat{f}(x) = \frac{1}{K_1}\sum_{i=1}^{K_1} Q_{x_i,x} \quad \text{and} \quad \hat{g}(x) = \frac{1}{K_2}\sum_{i=1}^{K_2} Q_{y_i,x} \quad .$$

Li et al. (2009) constructed the T_n-statistic based on the integrated squared difference, I_n , that serves as a measure of the closeness between $f(x)$ and $g(x)$:

$$I_n = \frac{1}{K_1(K_1-1)}\sum_{i=1}^{K_1}\sum_{j=1,j\neq i}^{K_1} Q_{x_i,x_j} + \frac{1}{K_2(K_2-1)}\sum_{i=1}^{K_2}\sum_{j=1,j\neq i}^{K_2} Q_{y_i,y_j} - \frac{1}{K_1K_2}\left[\sum_{i=1}^{K_1}\sum_{j=1}^{K_2} Q_{x_i,y_j} + \sum_{i=1}^{K_2}\sum_{j=1}^{K_1} Q_{x_j,y_i}\right]$$

The T_n-statistic is calculated by $\hat{T}_n = (K_1K_2h)^{1/2} I_n/\hat{\sigma}_n \xrightarrow{d} N(0,1)$, where the variance is obtained by:

$$\hat{\sigma}_n^2 = 2(K_1K_2h)\left[\frac{1}{K_1^2(K_1-1)^2}\sum_{i=1}^{K_1}\sum_{j=1,j\neq i}^{K_1}\left(Q_{x_i,x_j}\right)^2 + \frac{1}{K_2^2(K_2-1)^2}\sum_{i=1}^{K_2}\sum_{j=1,j\neq i}^{K_2}\left(Q_{y_i,y_j}\right)^2 \right.$$
$$\left. + \frac{1}{K_1^2K_2^2}\sum_{i=1}^{K_2}\sum_{j=1}^{K_1}\left(Q_{x_i,y_j}\right)^2 + \frac{1}{K_1^2K_2^2}\sum_{i=1}^{K_1}\sum_{j=1}^{K_2}\left(Q_{x_j,y_i}\right)^2\right] \quad .$$

[22] Given that the selection of the smoothing parameter is very important in nonparametric estimation, it is suggested to estimate h by cross validation using the pooled sample (see Li et al., 2009).

Figure 1. The CO2 Tax in Sweden during 2001-2009 for the heat and power sector

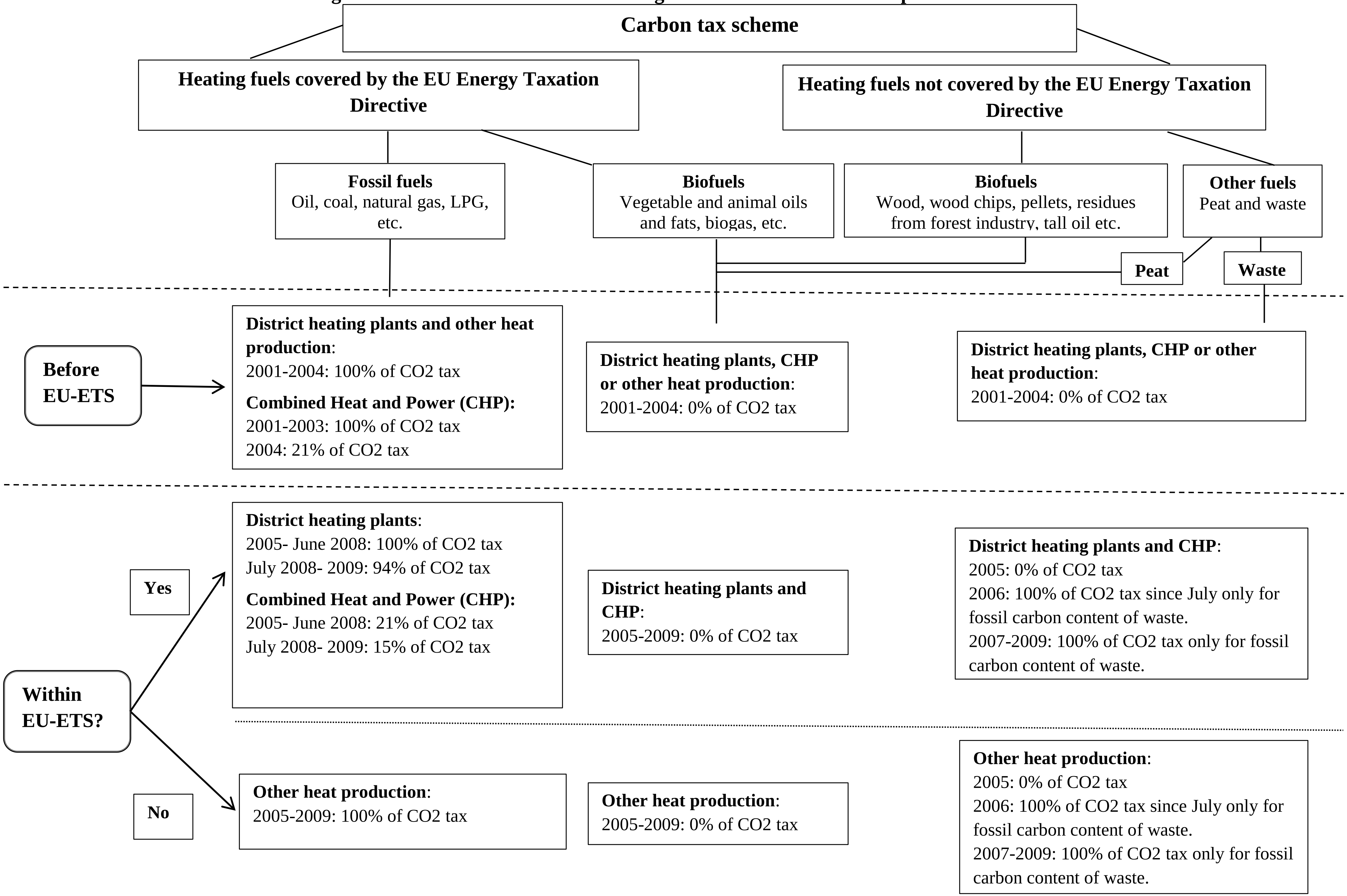
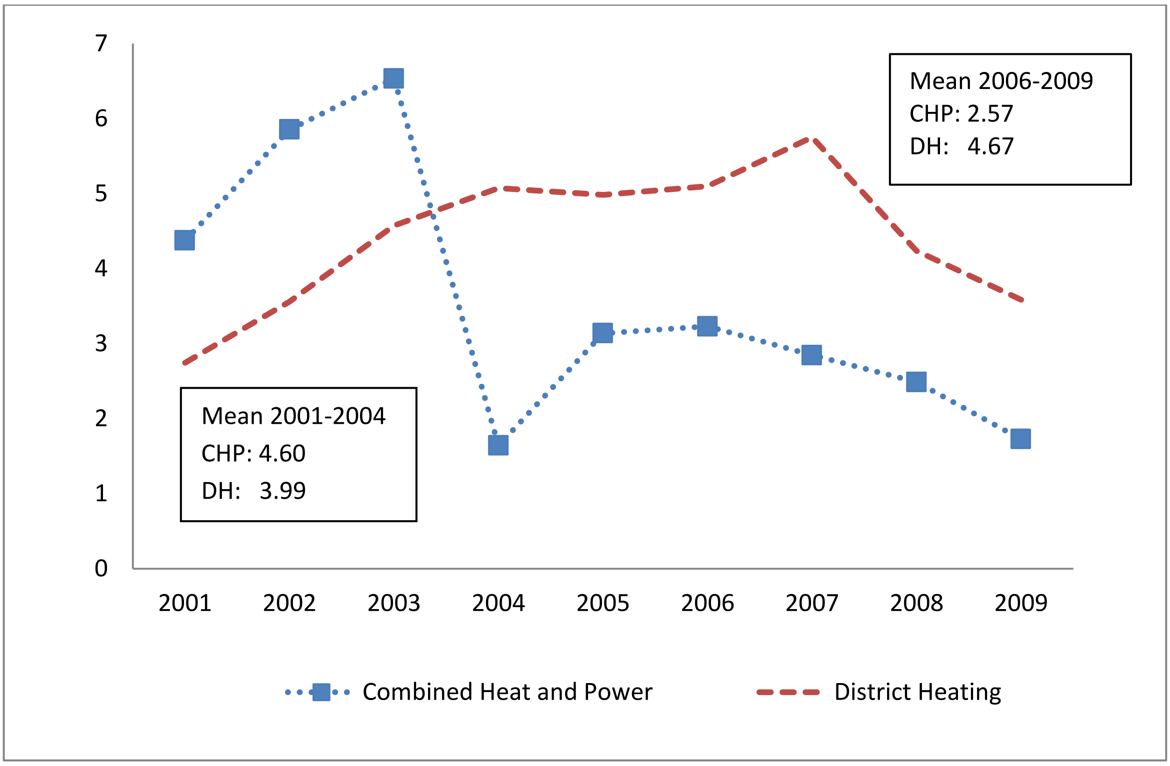


Figure 2. Average relative opportunity costs CO2/NOx per MWh for CHP and DH plants[1,2].



[1] The opportunity cost of CO2 emissions is calculated as the sum of the carbon tax plus the CO2 EU ETS price (mean of forward contracts 2007-2013).
[2] We compute the relative CO2/NOx opportunity cost per MWh per boiler and average the values across boilers.

Table 1: Descriptive statistics

Variable	Description	Mean	Standard Deviation	Min	Max
		2001-2004			
Y	Useful energy (GWh)	213.6	226.4	26.1	1292.8
b_1	CO2 (tons)	22716	51772	0.0	350320
b_2	NOx (tons)	44.3	42.7	3.9	358.5
b_1/Y	CO2(tons)/GWh	75.6	127.1	0	695.2
b_2/Y	NOx(tons)/GWh	0.25	0.09	0.03	0.57
x_1	Installed capacity (MW)	68.3	93.0	6	600
x_2	Fuel consumption (GWh)	227.7	240.1	23.6	1247.1
x_3	Labor (no. of employees)	33.7	41.6	1.0	220.4
		2006-2009			
Y	Useful energy (GWh)	213.9	246.1	25.1	1232.6
b_1	CO2 (tons)	18012	51993	0.0	382960
b_2	NOx (tons)	40.1	37.8	5.0	258.6
b_1/Y	CO2(tons)/GWh	63.1	124.7	0	722.7
b_2/Y	NOx(tons)/GWh	0.24	0.09	0.03	0.68
x_1	Installed capacity (MW)	66.8	90.9	5	600
x_2	Fuel consumption (GWh)	226.1	259.8	23.7	1334.8
x_3	Labor (no. of employees)	39.6	46.4	2.2	265.9

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	0.023	-0.045
α_1	x_1	-0.002	0.051
α_2	x_2	0.323	0.449
α_3	x_3	-0.026	0.052
β_1	Y	-0.567	-0.609
θ_1	b_1	-0.001	0.001
θ_2	b_2	0.434	0.390
α_{11}	x_1^2	-0.012	-0.017
α_{12}	x_1x_2	-0.153	-0.099
α_{13}	x_1x_3	0.021	-0.065
α_{22}	x_2^2	0.040	-0.053
α_{23}	x_2x_3	-0.016	0.085
α_{33}	x_3^2	0.020	-0.052
β_2	y^2	-0.180	-0.063
θ_{11}	b_1^2	-0.002	-0.001
θ_{22}	b_2^2	-0.182	-0.064
η_{11}	x_1b_1	0.007	0.018
η_{21}	x_2b_1	-0.003	-0.001
η_{31}	x_3b_1	0.012	0.003
η_{12}	x_1b_2	0.066	0.047
η_{22}	x_2b_2	0.273	0.142
η_{32}	x_3b_2	0.001	0.016
μ_1	yb_1	0.000	0.000
μ_2	yb_2	-0.180	-0.063
θ_{12}	b_1b_2	0.002	0.001
δ_1	x_1y	0.073	0.065
δ_2	x_2y	0.270	0.133
δ_3	x_3y	0.014	0.019

Year dummies are not shown in the table.

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

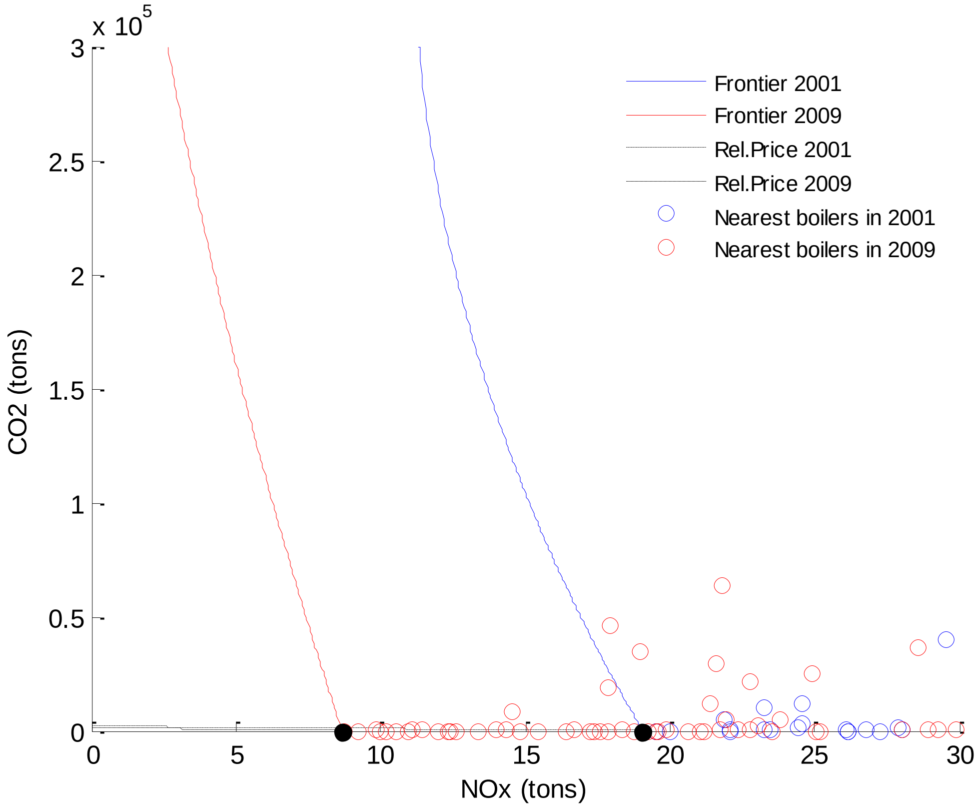


Figure 4a: Kernel distribution of CO2-output elasticities[23]

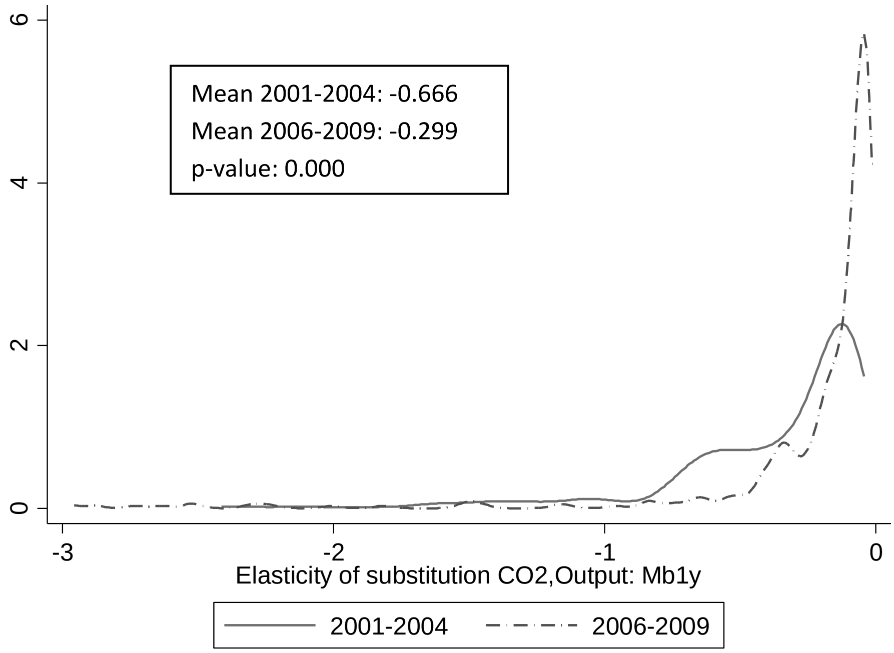
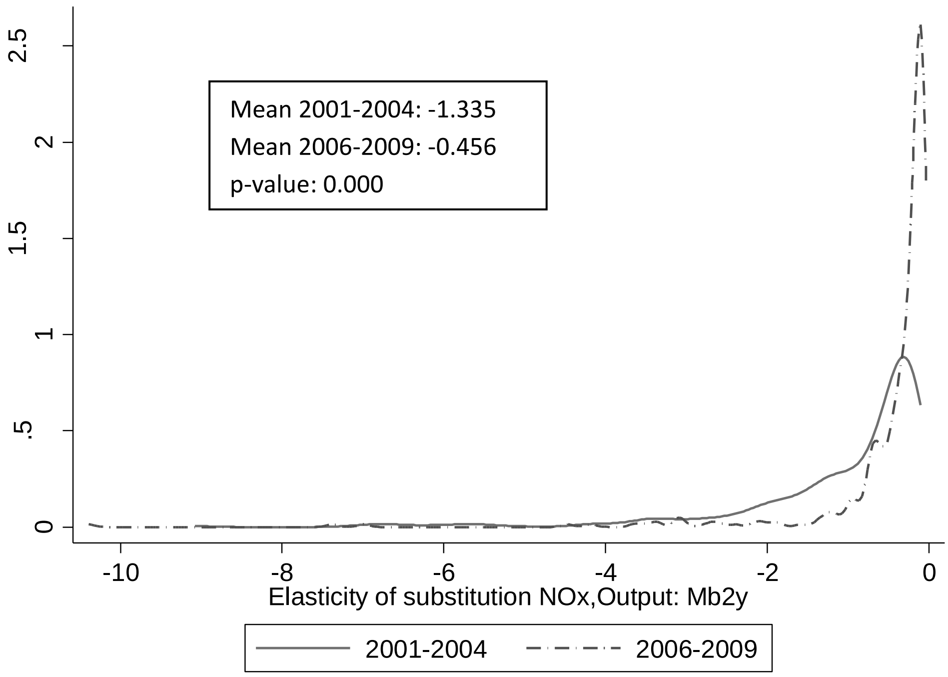


Figure 4b: Kernel distribution of NOx-output elasticities



[23] For convenience in presentation, some elasticity estimates of CO2-output lower than -3 and of NOx-output less than -10 are not shown in the graphs.

Table 3: CO2 and NOx 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	71.2	80.2	9.9	12.6	7.2	24.3	16.2	43.2	16.2
2002	64.0	75.7	9.9	12.6	10.8	22.5	16.2	45.0	16.2
2003	67.6	78.4	11.7	13.5	7.2	21.6	15.3	45.9	17.1
2004	68.5	79.3	10.8	12.6	7.2	21.6	15.3	45.0	18.0
2006	74.8	81.1	9.9	12.6	6.3	18.0	15.3	45.0	21.6
2007	71.2	79.3	9.9	10.8	9.0	18.0	12.6	41.4	27.9
2008	73.9	82.0	9.0	10.8	7.2	17.1	11.7	40.5	30.6
2009	73.9	82.9	9.0	10.8	6.3	16.2	11.7	40.5	31.5

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

Figure 5a: Kernel distribution of CO2-NOx elasticities[24]

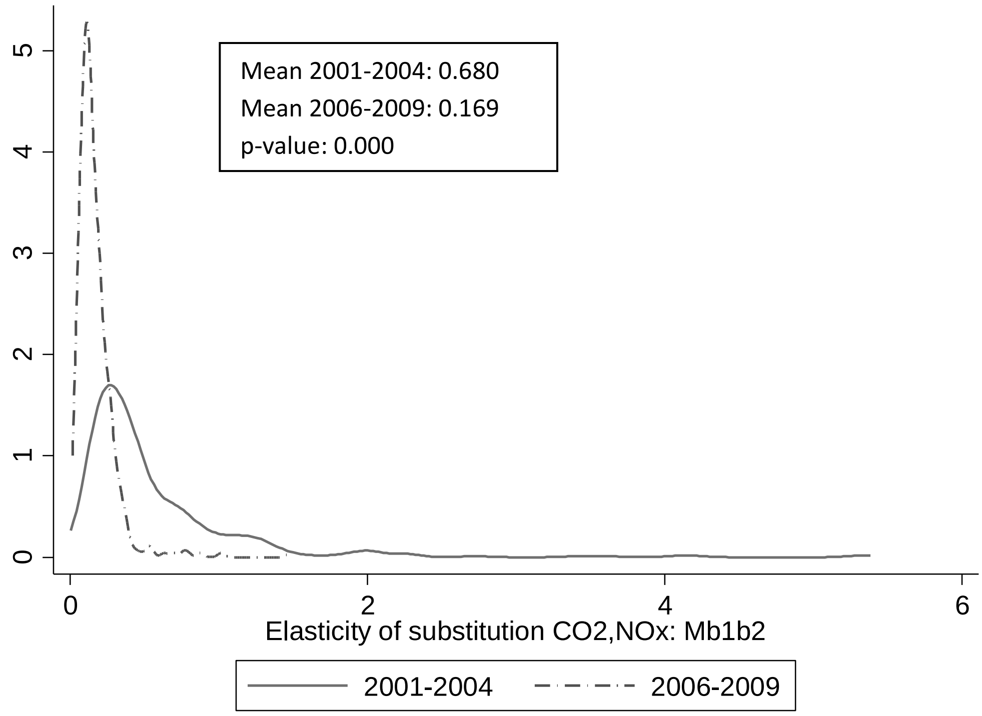
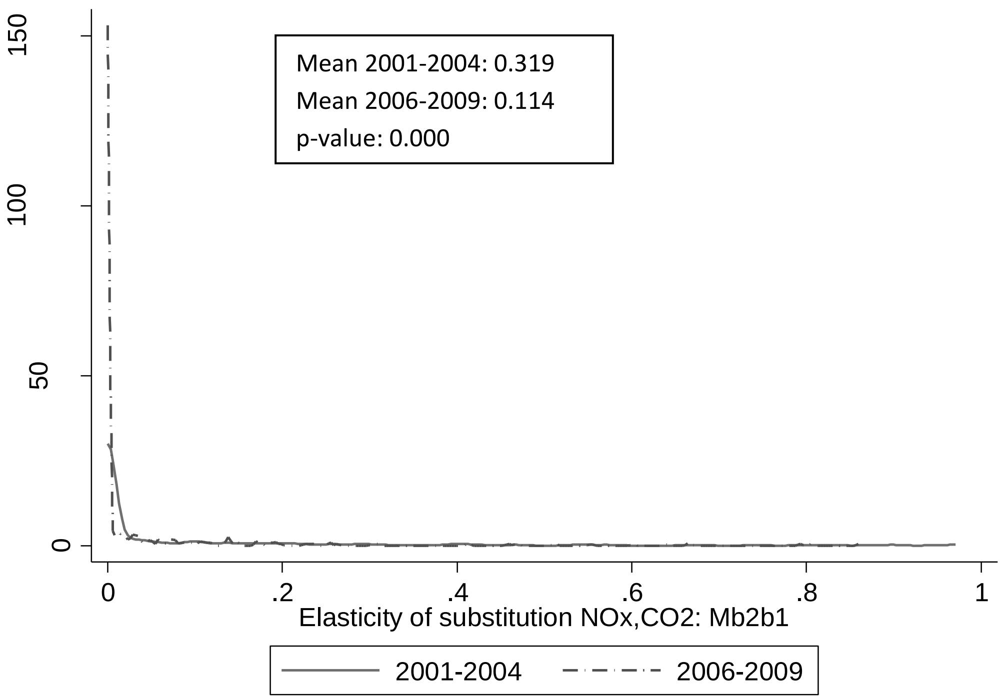


Figure 5b: Kernel distribution of NOx-CO2 elasticities



[24] For convenience in presentation, some elasticity estimates of CO2-NOx higher than 5 and of NOx-CO2 higher than 1 are not shown in the graphs.

Figure 6a: Kernel distribution of relative CO2/NOx shadow prices – CHP plants[25]

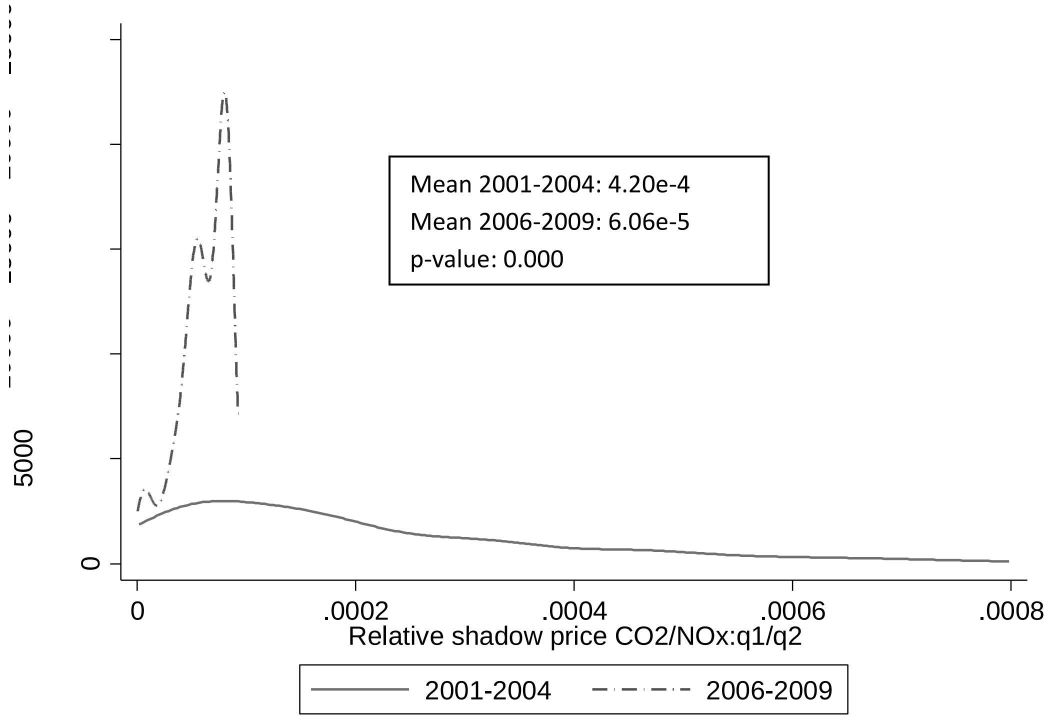
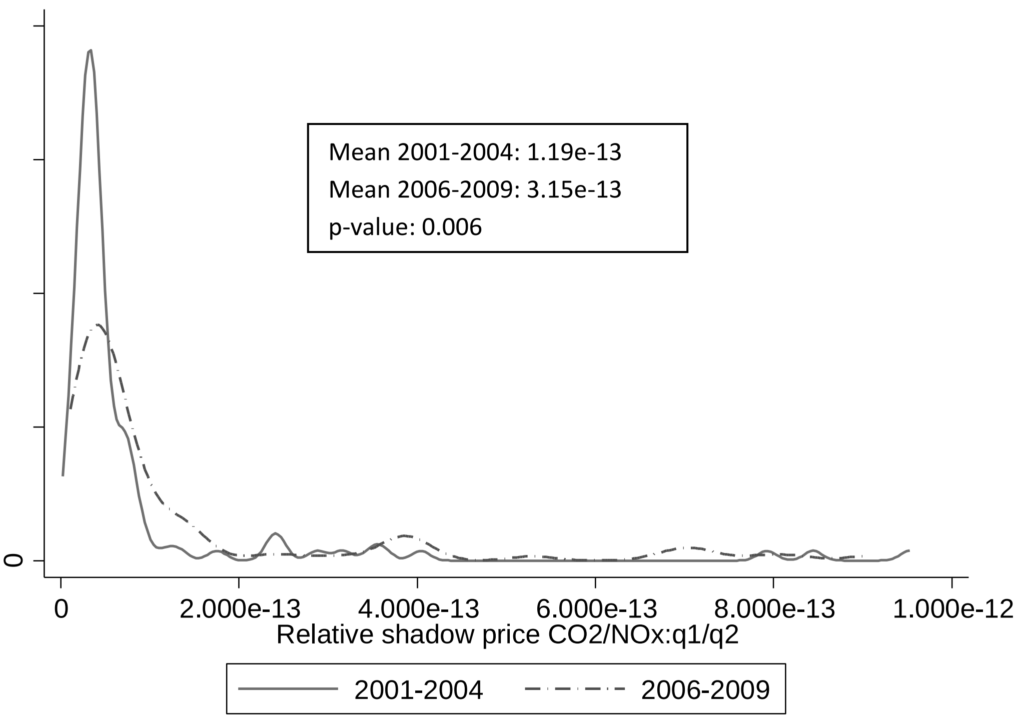


Figure 6b: Kernel distribution of relative CO2/NOx shadow prices – DH plants



[25] For convenience in presentation some few relative shadow price estimates greater than 0.008 for CHP and higher than 1e-12 for DH are not shown in the graphs.