Fiscal Federalism, Interjurisdictional Externalities and Overlapping Policies

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Abstract

In this paper, we analyze the effects of the interaction between national and local policies designed to reduce an environmental externality that causes environmental damages both nationally and locally. We formulate a theoretical model to develop hypotheses regarding the combined effects of such policies on the stringency of the local policies and on firms' emissions reductions. To test our hypotheses, we use actual data for Sweden, where emissions of nitrogen oxides from combustion plants are subject to a heavy national tax and to individual emissions standards set by county authorities. Our analytical findings suggest that it is unlikely that local regulators will impose emissions standards stringent enough to achieve further reductions than those induced by the national tax. This is confirmed in our data, where most emissions reductions can be attributed to the national tax and the effects of the emissions standards are not significant.

Key Words: environmental regulation, multi-governance, federalism, emission taxes, command-and-control, air pollution, NO_X, Sweden.

JEL classification: H77, Q58, H23, D62

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1 Introduction

The economic literature on regulatory federalism indicates that regulatory measures should, in principle, be the responsibility of the lowest level of government whose jurisdiction encompasses the relevant benefits and costs associated with the regulated activity. In some cases, like air quality, there are not only local effects of pollution control, but also spillover effects into other jurisdictions. The implementation of national policies may partially internalize spillovers, but impose uniform incentives ignoring local heterogeneity. When there are interjurisdictional spillovers, the literature typically finds that decentralized policymaking produces socially inefficient outcomes as interjurisdictional spillovers are simply more important than local heterogeneity (see e.g., Banzhaf and Chupp 2012 who find that decentralized policymaking of US air pollution loses 31.5% of the potential first-best benefits, whereas a uniform national policy loses only 0.2%).

Cooperation across jurisdictions is an alternative and appealing approach to addressing environmental spillovers. But such cooperation is not always easy to come by if "a race to the bottom" in environmental quality occurs as states compete for new investments or if states free ride (see e.g., Oates and Schwab 1998, and Markusen et al. 1995).[1] Thus, spillover effects across jurisdictions present us with a fairly complex set of policy alternatives in practice. The first-best measures of economic theory may simply not be feasible. The available alternatives then include policy mixes where not only central regulations but also local regulations that in some cases might override federal environmental regulations with tighter regulations of their own are implemented. This is the case in, e.g., the United States and Europe, where environmental regulations are enacted and managed at all levels of government: federal (i.e., national), state, and local. The regulations pertaining to many major environmental problems—for example, clean air, clean water, and hazardous waste management—are typically passed at the national level. Local regulators then pass regulations that may restrict pollution beyond national requirements to address environmental problems for which the local benefits are high (see, e.g., Chupp 2011, who studies the ability of US states to respond to local conditions through implementation of local policies).

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[1] See also List and Gerking (2000) and Sigman (2005). List and Gerking (2000) test whether environmental quality declined when President Reagan's policy of new federalism returned the responsibility for many environmental regulations to the states. They find no evidence of a race to the bottom. In contrast, Sigman (2005) estimates the costs of free riding among U.S. states under the Clean Water Act. Her results indicate that free riding gives rise to a 4% degradation of the water quality downstream of authorized states, costing about $17 million annually.

The interaction of policies raises important questions regarding the effects of national actions on the implementation and stringency of local regulations and viceversa. In particular, if the interaction between national and local policies affects a country's ability to achieve emissions reductions and whether the policy overlapping leads to potentially inefficient regulatory competition. The answers to such questions certainly depend on the nature of the overlap between the two regulations, their relative stringency as well as the types of instruments utilized. In the case of climate change, the literature has shown, for instance, that the interaction between national and local regulations becomes problematic if the national policy involves restrictions on aggregate emissions quantities. That is, since the emissions reductions accomplished by a subset of local governments reduce pressures on the constraints posed by the national policy, facilities are encouraged to increase emissions in other states leading to "emissions leakage" and a loss of cost-effectiveness at the national level. In contrast, when the national policy involves fixed prices, more aggressive local policies in a subset of states would generally lead to differing marginal abatement costs across local jurisdictions and reduced cost-effectiveness (see e.g., Goulder and Stavins 2011).

In this paper, we analyze the effects of the interaction between national and local policies designed to reduce local air pollution in Sweden, where emissions of nitrogen oxides (NO_x) from combustion plants are subject to a heavy national tax and most (but not all) are also subject to individual emissions standards set by county authorities. These policies were implemented to control the serious soil acidification and water eutrophication problems that Sweden faced in the 1980s, which were caused partly by NO_x emissions from combustion processes in transport, industry, and power generation. In 1992, Sweden introduced a high tax on NO_x emissions from large combustion sources (e.g., power plants, industrial plants, and waste incinerators). The tax was initially set at a rate of 40 SEK per kilogram of NO_x emitted from any stationary combustion plant producing at least 50 megawatt hours (MWh) of useful energy per year. This corresponds to 6,000 USD/metric ton, which is much higher than the hundreds of dollars commonly seen in the US programmes for NO_x permits (Sterner and Höglund-Isaksson, 2006). Individual emissions standards for NO_x (so-called emission limit values, ELVs) had also been introduced in the late 1980s, and thus for some combustion plants ELVs were already in place when the charge was introduced (Sterner and Höglund-Isaksson 2006). This is to say, the regulations were additive and the plants that were subject to both types of regulations had to comply with both. We set up a theoretical model to develop a series of hypotheses regarding the combined effects of such

policies on the stringency of the emissions standards and on firms' emissions reductions. We use actual data to test our hypotheses. Since there is no centralized comprehensive information on the emissions standards and how they have developed over time, we collected such information from each of the county authorities and merged it with the Swedish Environmental Protection Agency (SEPA) database, which covers the combustion plants monitored under the Swedish NO_x tax. Thus, we built a unique database that allows us to investigate the effects of the implementation of the national tax on the stringency of the ELVs and the combined effects of the national and local policies on abatement efforts.

To the best of our knowledge, the closest paper to ours is the one by Williams III (2012) who investigates the growing state-federal conflicts in environmental policy. He argues that the observed patterns of regulation in the United States where federal legislation serves as a minimum floor beyond which states can move ahead and set stricter regulations can be explained by a shift in the type of regulation used at the federal state, from command-and-control toward incentive-based regulation. This shift implies that a state imposing a tighter regulation will bear only part of the additional cost - and thus has a stronger incentive to tighten regulation than it does under federal command-and control. Thus, Williams III (2012) concludes that for a pollutant that causes any local damage, the average state will want to impose its own tax.

In contrast to Williams III (2012), who focuses on the case where both national and local governments regulate emissions via the same instrument, we study the case of a mixed policy instrument with a tax at the national level and command-and-control at the local level. In such a setting, we find that even though the stringency of the standards increases when standards are combined with a national tax, it is unlikely that local regulators will impose emissions standards stringent enough to achieve further reductions than those induced by the national uniform tax. Such a result is also observed in our data, where most emissions reductions can be attributed to the national tax and the effects of the ELVs are not significant. Hence, our paper provides evidence of the effects of interactions between national market-based policies and local regulations, showing that not all mixes of policies create positive interactions that lead for further levels of environmental protection, and thus the choice of policy mix has important implications in terms of environmental outcomes.

The paper is organized as follows. The next section sets out the theoretical model conceptualizing how the implementation of the national tax might affect the stringency of the local regulations.

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The empirical strategy is described in the subsequent section, followed by a description of the data used to set up the study case. Next, we describe the main results and conduct a number of robustness tests. The final section presents the main conclusions and discusses the policy implications of our results.

2   The Model

Assume one country made up of n counties. In each county there is a polluting firm that emits a pollutant that causes environmental damages both nationally and locally. Pollution damage in county i can be represented as $D_i(e_i, E)$, where E is the nationwide level of emissions given by $E = \sum_{i=1}^{n} e_i$. Thus, environmental regulation will have local and national benefits.[2] The function $D_i(e_i, E)$ is twice differentiable and convex in both arguments. The cost of emissions reductions in county i is given by the function $C_i(e_i)$, which is twice differentiable, decreasing and strictly convex in e_i.[3]

In this setting let us start by analyzing (i) first-best regulation, followed by the case where (ii) only national or local governments regulate emissions, and then conclude with the case of (iii) policy overlapping where both national and local regulators implement policies to reduce emissions. Like Williams III (2012), we assume that governments incur some cost of imposing a policy, yet this cost is arbitrarily small. To mirror the Swedish context, we focus on the case where the national regulator sets a uniform emissions tax and the local regulators set command-and-control regulations in the form of emissions limits. Moreover, the national regulator commits to maintain the tax rate once it has been set. Thus, local regulators cannot affect the level of the tax via their choices of

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[2] As described by Williams III (2012), making damages a function of both local and national emissions allows this model to represent cases where a pollutant has only local effects (in which case, $\partial D_i/\partial e_i > 0$ and $\partial D_i/\partial E = 0$), only national effects (in which case, $\partial D_i/\partial e_i = 0$ and $\partial D_i/\partial E > 0$), and both local and national effects (in which case, both partial derivatives are positive). This last case could be a single pollutant with both local and national effects, or two pollutants that are highly correlated and identically affected by efforts to reduce emissions. For instance, the environmental impact of chemically reactive air pollutants on both local air quality (smog) and the regional ecosystem (acid rain) is well established, i.e., NOx concentrations, in both polluted cities and the remote troposphere, play the dominant role in tropospheric ozone production. Moreover, ozone can be transported by wind currents and cause health impacts far from the sources.

[3] Note that the cost of emissions reductions in one state does not depend on emissions reductions in any other state, which rules out any cost-side spillovers.

stringency of local emissions standards (only the level of abatement by the local firm).

2.1 First-Best vs. Decentralized Policies

Let us assume that the national regulator's objective is to minimize the sum of all pollution damage and the costs of reducing emissions in all counties:

$$\min_{e_i} \sum_{i=1}^{n} \left[C_i(e_i) + D_i(e_i, E) \right],$$

which leads to the following FOC for e_i:

$$-\frac{\partial C_i(e_i)}{\partial e_i} = \frac{\partial D_i(e_i, E)}{\partial e_i} + \sum_{j=1}^{n} \frac{D_j(e_j, E)}{\partial E}, \tag{1}$$

for $j = 1, ..., n$. That is, optimal emissions in each county i are such that the marginal cost of emissions reductions in that county equals the marginal damage from pollution emissions in county i plus the national-level effects of emissions of county i on all counties. It is well known in the literature that the first-best regulation can be implemented through differentiated emissions standards per county - set at the level defined by equation (1) - or through differentiated tax rates τ_i equal to:

$$\tau_i = \frac{\partial D_i(e_i, E)}{\partial e_i} + \sum_{j=1}^{n} \frac{\partial D_j(e_j, E)}{\partial E}. \tag{2}$$

Note that under differentiated taxation, the stringency of the local regulations will be larger in counties where the local damages of pollution are greater and counties that cause larger inter-state spillover effects on other counties. However, as discussed before, differentiated first-best policies might not be politically or legally feasible. In practice, the majority of existing and planned market-based emissions regulations are implemented as spatially uniform undifferentiated policies where all regulated emissions are penalized at the same tax rate. Hence, let us assume that the national regulator decides on a uniform tax rate T and that polluting firms in each county want to minimize the sum of abatement costs $C_i(e_i)$ and tax payments Te_i. The national regulator solves:

$$\min_{e_i} \sum_{i=1}^{n} \left[C_i(e_i(T)) + D_i(e_i(T), E(T)) \right],$$

subject to the firms' implicit reaction function:

$$-\frac{\partial C_i(e_i)}{\partial e_i} = T \ \forall \ i. \tag{3}$$

It is easy to show that the uniform tax rate T corresponds to the average marginal damage across counties. Indeed, the FOC determining the optimal level of emissions in each county i is given by equation (1). Summing up these FOCs for all counties, we obtain that

$$\sum_{i=1}^{n} -\frac{\partial C_i(e_i)}{\partial e_i} = \sum_{i=1}^{n} \frac{\partial D_i(e_i, E)}{\partial e_i} + \sum_{i=1}^{n} \sum_{j=1}^{n} \frac{\partial D_j(e_j, E)}{\partial E}.$$

Furthermore, given the firms' implict reaction function in equation (3) we know that

$$\sum_{i=1}^{n} -\frac{\partial C_i(e_i)}{\partial e_i} = nT,$$

and thus:

$$T = \frac{1}{n} \sum_{i=1}^{n} \frac{\partial D_i(e_i, E)}{\partial e_i} + \sum_{j=1}^{n} \frac{\partial D_j(e_j, E)}{\partial E}. \tag{4}$$

Comparing equations (2) and (4) it is straighforward that the uniform tax rate is only optimal when the marginal damage from pollution is the same for all states. Otherwise, it will not yield the first best: the uniform tax would be too low for those counties where local damages are high and too stringent for those counties where local damages are low.

Let us now consider the case of decentralized policy implemented in the form of emissions standards $\widehat{e}_i$. County i's objective is to minimize the local cost of reducing emissions plus pollution damages to the county,

$$\min_{e_i} C_i(e_i) + D_i(e_i, E), \tag{5}$$

which yields the following FOC:

$$-\frac{\partial C_i(e_i)}{\partial e_i} = \frac{\partial D_i(e_i, E)}{\partial e_i} + \frac{\partial D_i(e_i, E)}{\partial E}. \tag{6}$$

From equation (6), we know that counties set emissions standards $\widehat{e}_i$ such that marginal cost equals marginal damage (including only damages within the county and not damage to other counties). Moreover, comparing the marginal costs in equations (4) and (6) of emissions reductions under a uniform tax and decentralized emissions standards, it is straightforward that the emissions reductions achieved in those cases where local governments implement decentralized emissions standards will generally be smaller than the emissions reductions achieved when only the national government regulates emissions by means of a uniform tax. Indeed, the only exception to this is the case where the local environmental damages of pollution are very high and the inter-jurisdictional

spillovers are small (e.g., the second term on the right hand side of equation (7) is negligible), such that the following condition holds:

$$\frac{\partial D_i(e_i, E)}{\partial e_i} > \frac{1}{n-1} \sum_{j \neq i}^{n} \frac{\partial D_j(e_j, E)}{\partial e_j} + \left[\frac{n}{n-1} \right] \sum_{j \neq i}^{n} \frac{\partial D_j(e_j, E)}{\partial E} \qquad (7)$$

That is, the local damages to jurisdiction i must be larger than the average local damages in all other jurisdictions plus the sum of damages of the interjurisdictional spillovers caused by jurisdiction i on all the other jurisdictions. Consistent with the empirical literature that finds that decentralized policymaking produces larger welfares losses than a uniform national policy since interjurisdictional spillovers are more important than local heterogeneity of damages, we argue that cases where condition (7) holds are rare and characterized by large heterogeneity in damages across counties (e.g., pollution hot spots) and small spillovers. In contrast, if we would assume that all counties are symmetric in terms of the damages of air pollution and costs of emissions reductions, it would be easy to show that the emissions reductions would be larger under the uniform national tax (see Appendix A).

2.2 Policy Overlapping

Let us now analyze the case where the national regulator decides on a uniform tax rate T and the local regulator sets (or revises) emissions standards $\widehat{e}_i$ after observing the uniform tax rate set by the national regulator. Thus, the timing of the regulatory game is as follows. First, the national regulator sets the uniform tax rate T to minimize nationwide pollution damages and costs of pollution control. Second, local regulators set emissions standards $\widehat{e}_i$. Finally, firms decide on pollution abatement. By backwards induction, we know that if both levels of government implement regulations, firms' emissions will be determined by the tighter regulation, and thus $e_i = \min [e_i(T), \widehat{e}_i]$, where $e_i(T)$ corresponds to the solution to the firm's implicit reaction function in equation (3) and $\widehat{e}_i$ corresponds to the local emissions standards.[4]

Given the tax T, the local regulators set emissions standards $\widehat{e}_i$ to minimize the cost of reducing emissions and pollution damages to the county, plus the national emissions tax paid by the local firm to the national government, net of any revenue returned to it as a lump sum. Let r_i represent the share of national emissions tax revenues that are returned to county i (which is assumed to be exogenous). County $i's$ objective function can now be represented as:

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[4]Firms now minimize the sum of abatement costs $C_i(e_i)$ and tax payments Te_i subject to the constraint $e_i \leq \widehat{e}_i$.

$$\min_{e_i} C_i(e_i) + D_i(e_i, E) + Te_i - r_i \sum_{j=1}^{n} Te_j, \tag{8}$$

which yields the following FOC:

$$-\frac{\partial C_i(e_i)}{\partial e_i} = \frac{\partial D_i(e_i, E)}{\partial e_i} + \frac{\partial D_i(e_i, E)}{\partial E} + T[1 - r_i]. \tag{9}$$

From equation (9), we know that counties set emissions standards $\widehat{e}_i$ such that the marginal cost equals the marginal damage (including only damages within the county and not damages to other counties) plus the national emissions tax rate (net of return transfers to the county). Thus, compared with the FOCs in the case of decentralized policy given by equation (6), we should expect emissions standards to be more stringent in the case of overlapping policies since a more stringent standard incentivizes increased emissions reductions, allowing counties to reduce the tax payment to the national government. The increased incentive due to tax payments is the largest when there is no tax redistribution (and thus, $r_i = 0$, implying that the county pays the full tax) and disappears if $r_i = 1$, since then the county gets the tax fully returned.

So far we have shown that emissions standards become more stringent when they are combined with a uniform tax rate, but if they are not stringent enough to be binding, they will have no effect on the level of emissions reductions by the firm. This is to say, $\min[e_i(T), \widehat{e}_i] = e_i(T)$. In such a case, the emissions reductions will be determined by equation (3), and the unique Nash-equilibrium tax rate T is then defined by equation (4).

Let us now analyze the case where the emissions standards $\widehat{e}_i$ are binding (at least for a subset of counties). Then, the marginal cost of abatement to the firm in county i is equal to $-\frac{\partial C_i(e_i)}{\partial e_i}\lfloor_{\widehat{e}_i} > T$. Making use of such an inequality allows us to represent equation (9) as:

$$\frac{\partial D_i(e_i, E)}{\partial e_i} + \frac{\partial D_i(e_i, E)}{\partial E} > Tr_i. \tag{10}$$

By substituting equation (4) into equation (10) we find a condition determining when emissions standards are binding under overlapping policies compared with the case where only standards are in place:

$$\frac{\partial D_i(e_i, E)}{\partial e_i} > \left[\frac{r_i}{n - r_i}\right] \sum_{j \neq i}^{n} \frac{\partial D_j(e_j, E)}{\partial e_j} + \left[\frac{nr_i}{n - r_i}\right] \sum_{j \neq i}^{n} \frac{\partial D_j(e_j, E)}{\partial E} - \left[\frac{n[1 - r_i]}{n - r_i}\right] \frac{\partial D_i(e_i, E)}{\partial E}. \tag{11}$$

Note that if $r_i = 1$, condition (11) simplifies to that in equation (7). Hence, policy overlap would have no effect on the stringency of the emissions standards since there are no additional benefits of increased stringency in the form of reduced tax payments because the tax is fully refunded. If we instead assume that $r_i = 0$, the emissions standards become binding as increased stringency reduces pollution damages and tax payments. Finally, let us assume that counties are symmetric in terms of the damages of air pollution and costs of emissions reductions. As shown in Appendix B, under such assumptions there is a critical refund r^* that determines whether the emissions standards become binding. If $r_i > r^*$, the emissions standards will not be binding, while the opposite holds for $r_i < r^*$.

All in all, our analytical results suggest that emissions standards are more likely to be binding when combined with a uniform tax that is not fully refunded to the county. As explained before, under such policy overlapping, local regulators have an additional incentive to increase the stringency of standards to incentivize more abatement and a reduced tax payment to the national government. Thus, in line with Williams III (2012), our results suggest that the use of a market-based national regulation leads local regulators to impose tigher regulations. However, the mechanism is slightly different in his paper since when local regulators implement taxes, local regulators imposing a tighter regulation will only bear part of the additional cost of the tax. In contrast, in our case, local regulators implement tighter emissions standards (than they would have in the absence of the national regulation) to reduce the tax payment to the national government.

Hence, we can conclude that when the national and local regulators implement regulations to restrict emissions, the emissions standards set by local regulators might become more stringent than in the case where they are the only policy in place. It is unclear, however, whether they will be stringent enough to achieve further reductions beyond those induced by the national uniform tax. This is likely to happen in cases where local environmental damages are large and pollution spillovers are small.

3   NO$_x$ emissions regulations in Sweden

For geological reasons, Sweden is particularly vulnerable to acidification, causing negative impacts on lake and forest ecosystems. Consequently, NO$_x$ emissions have been an important environmental policy target in Sweden. For instance, in 1992, a tax on NO$_x$ emissions from large combustion plants

was introduced. The tax was accompanied by a refund according to the amount of useful energy generated to ensure that facilities with low NO_x emission intensities would be net beneficiaries of the scheme (see Bonilla et al. 2018). At the time when the tax was implemented, close to 25% of Swedish NO_x emissions came from stationary combustion sources and the tax was seen as a cost-efficient way to reduce NO_x emissions. Regulated entities belong to the heat and power sector (from 1992 to 2012 on average of 50.5% of all boilers), the pulp and paper industry (14.1%), the waste incineration sector (11.8%), and the chemical (6.8%), wood (9.8%), food (5.5%) and metal industries (1.5%). Initially, the tax requirement only covered boilers and gas turbines with a yearly production of useful energy of at least 50 GWh. However, because of its effectiveness in reducing emissions and simultaneously falling monitoring costs, in 1996 the tax system was extended to include all boilers producing at least 40 GWh of useful energy per year, and in 1997 the limit was again lowered to 25 GWh (Sterner and Höglund-Isaksson 2006). From 1992 to 2007, the tax was 40 SEK/kg NO_x, but in 2008 the tax was raised to 50 SEK/kg NO_x following a series of reports from the Swedish EPA indicating that the impact of the tax system had diminished over the years (SEPA 2012). In real terms, the tax had decreased over time and the increase to 50 SEK in 2008 was in practice a mere restoration of the tax to the real level in 1992.

SEPA manages the tax on NO_x emissions at a small administrative cost amounting to 0.2–0.3% of the total revenues. The entire remaining tax revenue is refunded to the same collective of polluters in proportion to their output of useful energy. Useful energy produced has been accepted as a relevant and neutral yardstick for measuring output from this heterogeneous group of industries since the main goal is to affect combustion technologies. For power plants and district heating plants, it is equal to the energy sold. For other industries, the energy is defined as steam, hot water, or electricity produced in the boiler and used in production processes or heating of factory buildings (see Sterner and Höglund-Isaksson 2006).

Individual emissions standards for NO_x emissions from stationary sources (hereinafter, ELVs) were introduced in the late 1980s and thus, for many boilers (but not all), they were already in place when the tax was introduced. They are determined case-by-case either by either one of the 21 regional County Administrative Boards (CABs) or one of the five Environmental Courts.[5] There

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[5]We restricted the period under study to 1980–2012 because after June 1, 2012, coordination across counties was enhanced since only 12 CABs were responsible for issuing the operating licenses (instead of 21: see, e.g., SFS 2011:1237).

is no legal limit for how long an ELV specified in an operating license is valid, though the common practice seemed to have been that operating licenses and ELVs are revised about every ten years. Firms must, however, apply for a new operating license if they make large changes to the operations (e.g., if they install a new boiler or retrofit a boiler to use a different type of fuel). In addition, there can be appeals that change the original permissions, or postpone the implementation of requirements specified in the operating license. Regarding enforcement, if a firm violates the ELV specified in the operating license, it risks criminal charges and could face fines determined in court.

Unfortunately, there is no comprehensive data on the ELVs and how they have developed over time. Therefore, for the purpose of the present research we collected information about boiler-specific ELVs (specified in the operating licenses of the plants) for the firms regulated under the NO_x tax. In particular, the gathering of ELVs data proceeded as follows. We gathered information for all firms included in the NO_x database over the period 1992–2012. These corresponded to 392 firms whose operations were distributed across 434 plants located all over Sweden. We contacted all of the CABs hosting the 434 plants from October 2013 to June 2014. We provided each CAB with a list of the plants of interest and asked them to provide all the permissions connected to all boilers operating at that plant over the period 1980–2012.[6]

Figure 1 shows the average highest wintertime annual ambient concentration of NO_x per county 1986–2012. As can been seen, the ambient concentration of NO_x varies across counties, with the highest concentrations found in the most populated counties (e.g., Stockholm, Västra Götaland and Skåne) and the lowest concentrations found in Jämtland and the southeastern part of the country (e.g., Blekinge, Kalmar, Kronober, Södermanland, and Gotland). According to the Swedish Environmental Quality Standards, the binding upper limit for ambient NO_2 concentration is 40 $\mu g/m^3$ as a yearly average (SEPA, 2014). From 1986 to 2012, this limit was exceeded in five Swedish counties, namely, Stockholm, Västra Götaland, Skåne, Västerbotten och Västernorrland.

Table 1 summarizes the number of boilers subject to ELVs and the NO_x tax. In total, there are 935 boilers included in the dataset. Out of these, 240 have only been subject to ELVs, 116

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[6]Moreover, in Sweden, all environmentally hazardous activities that require environmental permission for operation must on a yearly basis provide the monitoring agency with an annual environmental report in which they describe their environmental impact and in what ways they comply with the regulations and restrictions that apply to them. For the purpose of tracking permissions over the years, these reports were used to double check that no permissions had been overlooked by the CAB or other authorities.
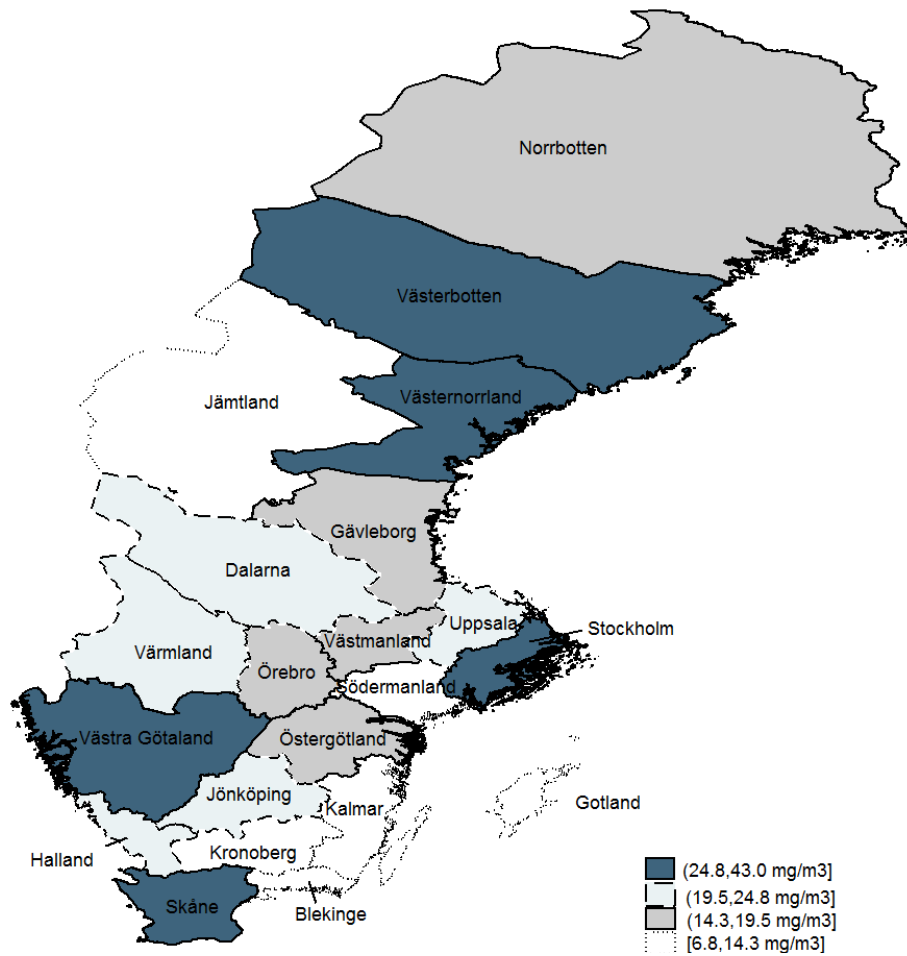
Figure 1: Average Highest Wintertime Ambient NO$_x$ Concentration per County 1986-2012

have only been subject to the tax, while 579 have been subject to both ELVs and the NO_x tax at least one year since 1992. ELVs are, however, expressed in different units and in order to perform our empirical analysis we focus mainly on emissions standards expressed in mg NO_x per MJ added energy to make it comparable to the unit used for the NO_x tax. Thus, our empirical analysis will be based on 867 boilers, out of which 741 have been subject to ELVs and 516 have been subject to both regulations.

As shown in Table 1, the stringency of ELVs has increased significantly over time (by about 44%, decreasing from an average of 187.05 mg/MJ before the implementation of the tax in 1992 to 104.86 mg/MJ after that). Moreover, the increased stringency is more pronounced for the group of boilers subject to both regulations than for those only subject to ELVs , e.g., a 48% vs a 31% reduction, respectively (see also Figure 4 in the Appendix for a plot of the trends over time).

ELV Stringency (mg/MJ)				
	N	$\mathbf{N}_{mg/MJ}$	**Before 1992**	**After 1992**
ELV-Tax	579	516	193.23	101.05
Only ELV	240	225	165.17	113.90
Total	819	741	187.05	104.86
Actual Emissions Concentration (mg/MJ)				
	$\mathbf{N}_{mg/MJ}$		**1992-2012**	
ELV-Tax	516		65.82	
Only Tax	116		80.88	
Total	632		68.2	

Table 1: Number of boilers subject to ELVs/Tax

Regarding actual emissions concentrations, unfortunately we only observe this information from the year 1992 onward for the boilers subject to the NO_x tax (as such information is only available from the SEPA NO_x tax database). We can see, however, that the actual emissions concentrations for the boilers subject to both regulations is about 19% lower than that for the boilers subject only to the tax.

Figure 2 illustrates the distribution of the values of ELVs and actual emissions concentration
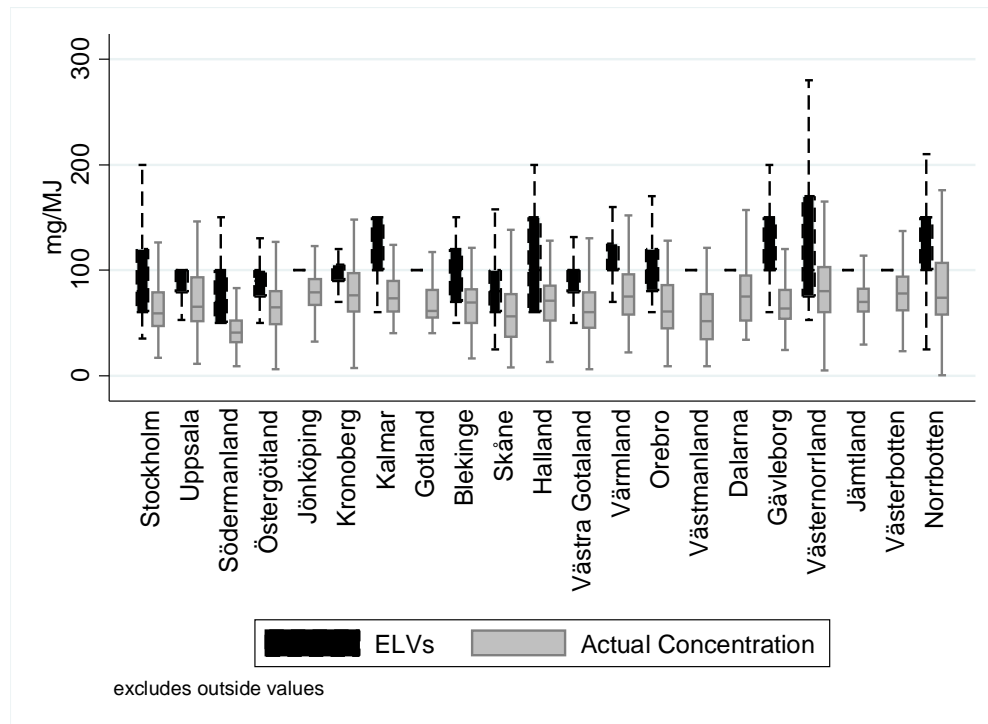
Figure 2: ELVs vs. Actual Concentration by County 1992-2012.

(both expressed as mg/MJ) per county for all observations 1992–2012. In the sample of 857 boilers, the counties with the largest numbers of boilers are Västra Götaland (136 boilers), Skåne (117) and Stockholm (99). Gotland, Jämtland and Uppsala are the counties with the fewest boilers (4, 12 and 18, respectively).

Figure 2 is consistent with a fair deal of variation in the stringency of ELVs across counties. The average ELV stringency for all boiler observations within counties varies between approximately 95 and 213 mg/MJ, with a mean of about 104 mg/MJ. Boilers in Jämtland seem to be, on average, the most leniently regulated, whereas boilers in Södermanland and Östergötland seem to be slightly more stringently regulated. We also observe that there is a lot of within-county variation in the stringency as well. In Jämtland, for example, the stringency ranges from 60 to 300 mg/MJ. The smallest variation in stringency is observed for Gotland, yet this is also the county with the fewest observations.

Furthermore, from Figure 2, it appears as if ELVs have not been binding over the period, on average (e.g., the average actual emissions concentration across counties varies from 45 to 87
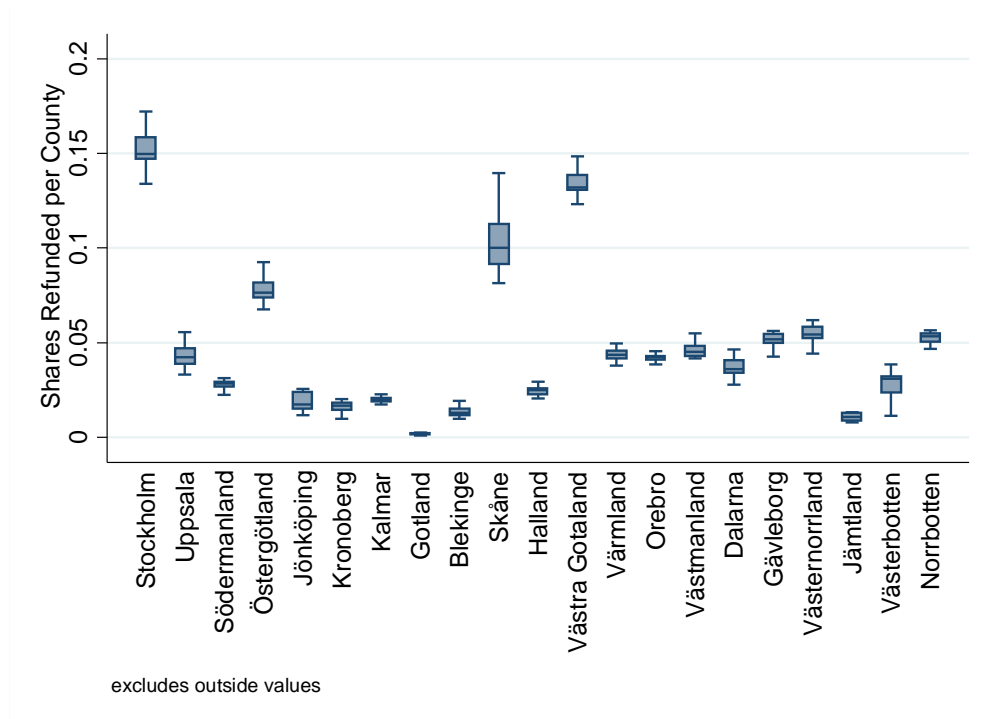
Figure 3: Total NO_x Tax Revenues Refunded to Counties 1992-2012.

mg/MJ, with a mean of about 68.2 mg/MJ). However, we cannot rule out the possibility that the ELVs were binding for some boilers at some point in time.[7]

Finally, Figure 3 illustrates the distribution of shares of the tax revenues refunded to the counties. Note that if tax revenues were to be distributed evenly across counties, each should receive around 5% of the tax revenues. From the figure, it is clear that the distribution is uneven, with the counties with the largest number of boilers (Västra Gotaland, Skåne and Stockholm) receiving much larger shares of the refunds. The figure also shows that the distribution of tax revenues refunded to each county varies across the years.

What hypotheses can be derived from our analytical framework? One is that ELVs are more

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[7]Indeed, out of 4,616 observations for which an ELV (mg/MJ) and the NO_x tax apply simultaneously, the actual NOx concentration exceeds the maximum limit specified by the ELV in a given year only for about 485 observations (10.5%). This observation is consistent with evidence from the Swedish Protection Agency (SEPA), which in 2003 evaluated of the effect of the ELVs for the period 1997–2001 and concluded that, because emissions were generally much below the ELVs, the NO_x tax was more effective than the ELVs in reducing NO_x emissions.

stringent for boilers subject to both regulations than for boilers only subject to ELVs. A second hypothesis is that since ELVs are not expected to be binding, most emissions reductions have been achieved thanks to the implementation of the national tax.

In what follows, we verify the empirical validity of these hypotheses.

4 Econometric Model

We wish to evaluate i) the impact of the introduction of the NO_x tax on the stringency of the ELVs. Moreover, ii) we wish to analyze whether the actual emissions concentrations are driven mainly by the NO_x tax or the ELVs. Below, we explain our methodological approach in each case.

4.1 Evaluating the effects of the NO_x tax on the stringency of the ELVs

Since different boilers were affected by the NO_x tax at different points in time (as the size threshold changed in 1996 and 1997), we divide our boilers into three groups. Group 1 (G_1) includes all boilers with a yearly production of useful energy of at least 50 GWh. G_1 began being regulated by the tax already in 1992. Group 2 (G_2) includes all boilers with a yearly production of useful energy in the $]50, 40]$ GWh range. G_2 became regulated by the tax in 1996. Finally, Group 3 (G_3) includes all boilers with a yearly production of useful energy in the $]40, 25]$ GWh range, which became subject to the tax in 1997. Table 2 presents statistics of the mean ELV stringency before and after the treatment for each group. In all cases, the stringency of ELVs after the treatment is statistically higher than the stringency of the ELVs after the treatment. Nevertheless, as shown in the table, the increased stringency is larger for G_1 than for boilers in G_3 and G_2.

ELV Stringency (mg/MJ)				
Group	N	Before Treatment	After Treatment	Diff.
G_1	380	199.65	100.03	99.62***
G_2	37	123.54	106.64	16.90***
G_3	99	123.25	101.21	24.03***

Table 2: Number of Boilers per Treatment

For the sample of all boilers in operation by 1992 (hereinafter, OldBoilers), we estimate a

difference-in-difference model to analyze the effect of the implementation of the tax on G_1 by using the information available from 1980 to 1995. For this group, we observe the ELVs in two time periods, $D\_92 = 0, 1$, where 0 indicates years before G_1 received treatment (i.e., before 1992) and 1 indicates a time period after G_1 began receiving treatment (i.e., 1992–1995). The control group includes all boilers in operation by 1992 that were never subject to the tax plus the boilers in G_2 and G_3 (before they became subject to the tax).

The outcome variable Y_{ijt} corresponds to the ELV that applies to boiler i located in county j at time t (expressed in mg NO_x per MJ added energy). It is modeled as:

$$Y_{ijt} = \alpha + \beta D\_92_i + \gamma G_1 + \delta D\_92_i * G_1 + \sum_{l=1}^{L} \kappa_l Z_{il} + \sum_{m=1}^{M} \varkappa_m X_{im} + \sum_{p=1}^{P} \varphi_p W_{jp} + \zeta_j + \varepsilon_{ijt}, \ (12)$$

where β is the time trend common to control and treatment groups, γ is the treatment group specific effect (which accounts for average permanent differences between treatment and control), and δ is the true effect of treatment defined as the difference in average outcome in the treatment group before and after treatment minus the difference in average outcome in the control group before and after treatment. Moreover, Z is a vector of L boiler characteristics (e.g., size and type of fuel), X is a vector of M firm characteristics (e.g., industrial sector, number of boilers and plants owned by the firms, and ownership type), and W is a vector of P county characteristics (e.g, severity of NO_x pollution and importance of regulated firms for economic activity).[8] Moreover ζ_j are county-fixed effects that account for non-observable characteristics of the county that may affect the stringency of the ELVs, and ε_{ijt} is the error term.

For the sample of all boilers that were in operation by 1997, we estimate a difference-in-difference model with multiple time periods, which allows us to verify that the effect of the treatment is not driven by a particular set of treated firms and that it is similar across time periods. The outcome variable Y_{ikjt} (i.e., boiler i in group $k = 1, 2, 3$, located in county j at time t) is now modeled as:

$$Y_{ikjt} = \alpha + \gamma G_k + \delta D_{ikt} + \sum_{l=1}^{L} \kappa_l Z_{il} + \sum_{m=1}^{M} \varkappa_m X_{im} + \sum_{p=1}^{P} \varphi_p W_{jp} + \zeta_j + \varepsilon_{ijt}, \qquad (13)$$

where G_k are group fixed effects and D_{ikt} is an indicator on whether boiler i in group k was subject to the tax by time t (i.e., interaction between treatment and treatment having occurred by

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[8]There are some indications from SEPA (2003) that the individual emissions standards are set differently for different industry sectors, with the heat and power sector and waste incineration possibly being subject to more demanding emissions standards.

period t).

Regarding boiler characteristics, from interviews with decision makers at the CABs, it was suggested that larger boilers (in terms of installed capacity) generally provide better combustion conditions and can have more advanced abatement technology installed enabling larger emissions reductions. We therefore hypothesize that larger boilers obtain more stringent emissions standards than smaller boilers. We control for this effect by means of the variable *Size*, which measures the installed capacity in MW. We also control for number of boilers located in the plant and for the number of plants owned by the firm. Our hypothesis is that ELVs for the boilers located in plants with several boilers (or owned by firms that have several plants) generally are more stringently regulated.

The type of fuel and the sector in which the boiler is operated might also affect the stringency of the ELV. In particular, the EU directives for boilers used in waste incineration and at large combustion plants specify that gas fuels (which are associated with relatively lower levels of NO_x emissions) are more stringently regulated than solid and liquid fuels (which are associated with relatively higher levels of NO_x emissions). The reason for this is that a larger share of emission reduction percentage-wise can be achieved in boilers that use less NO_x intensive fuels. We therefore hypothesize that boilers that use relatively less NO_x intensive fuels are more stringently regulated.

Firm ownership might also affect environmental performance. The theoretical literature has identified three major determinants of environmental performance associated with ownership: economic efficiency, willingness to internalize environmental externality, and bargaining power with governments (see e.g., Wang and Wheeler 2005). Compared with public companies, private firms may exhibit greater economic efficiency, less bargaining power to elicit the stringency of environmental regulations, and fewer incentives to internalize environmental externalities. To capture differences in the regulatory process between private and public firms we make use of the variable *Public,* which takes a value of one for boilers owned by municipal/national firms, and zero otherwise.

Regarding variation across counties, we hypothesize that decision makers impose more stringent emissions standards in counties with more severe NO_x associated environmental damages. The decision makers may also impose less stringent standards in counties where there is greater concern about economic activity and employment. We proxy the first effect by means of the highest wintertime yearly average urban NO_x concentration in the air measured in $\mu g/m^3$. We argue

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that it is a good proxy variable since it captures variations in the exposure of people and the environment to NO_x emissions. As a variable intended to capture the potential bargaining power of a firm, we use the share of employees per sector out of all employed people in the county. The assumption is that in counties where the sector in which a firm's boiler belongs is of relatively large importance employment-wise, the firm has a higher bargaining power vis-à-vis the decision maker. Although the importance of a single firm (measured by the number of employees) would have been a more direct measure, it was not possible to use due to lack of data.

Since a great deal of technological development in the areas of NO_x reducing technologies has taken place over time (see e.g., Bonilla et al. 2015 and 2018), we should expect the stringency of ELVs to have increased across revisions. To account for this we include fixed effects per order of the ELV (i.e., first ELV, second ELV, etc.) and also run separate regressions for the subsample of first ELVs (as the factors affecting regulatory stringency might be more prominent the first time a boiler is regulated). Finally, we estimate equations (12) and (13) with robust standard errors clustered at county level to account for the potential correlation of the ELV decisions made by a given county.

4.2 Evaluating the effects of ELVs on actual emissions concentrations

Regarding the methodological approach to address our second research question, as previously described, we only observe information regarding emissions concentration from the year 1992 onward as such information is *only* available for the panel of firms subject to the NO_x tax. Thus, we cannot control for the value of the outcome variable before the treatment and only have information for treated boilers. Hence, to test the hypothesis that emissions concentrations are mainly driven by the NO_x tax, we modify our methodological approach to a simple regression analysis. Our dependent variable, which now corresponds to the actual concentration of NO_x (expressed in mg NO_x per MJ added energy) is modeled as:

$$C_{ijt} = \alpha + \phi D\_Y_{ijt-1} + \sum_{l=1}^{L} \kappa_l Z_{il} + \sum_{m=1}^{M} \varkappa_m X_{im} + \sum_{p=1}^{P} \varphi_p W_{jp} + \zeta_j + \varepsilon_{ijt}, \qquad (14)$$

where C_{ijt} corresponds to the actual emissions' concentration for boiler i located in county j at time t (expressed in mg NO_x per MJ added energy). Our variable of interest is D_Y_{ijt-1}, which corresponds to the lagged value of a dummy variable that takes a value of one if the boiler was

subject to an ELV in the previous year and zero otherwise. If the ELVs have an effect incentiviz-ing further reductions on actual emissions concentration that those induced by the NO_x tax, the coefficient ϕ should be negative and statistically significative.

As before, we control for boiler, firm, and county characteristics as well as county fixed effects. Moreover, we estimate equation (14) with robust standard errors clustered at boiler level, and run separate regressions for the sample of boilers already in operation by 1992 and the sample of all boilers in operation by 1997.

Regarding boiler characteristics, the Swedish NO_x database contains information about the availability of technologies to reduce NO_x emissions. Therefore, among the covariates we also include a dummy variable accounting for the availability of such technologies.

4.3 Data

The data consists of an unbalanced pooled cross-section over time panel of boilers, where boilers are observed every year from the year when they obtain the first ELV (or from the first year in the NO_x dataset in case of no ELV). In our sample, each boiler has received (on average) 1.80 ELVs, and 294 boilers have obtained only one ELV during the whole sample period. The boilers that have received more than one ELV have received (on average) 2.6 ELVs, and the average number of years between revisions is 6.59.

Regarding the data sources, information about ELVs over the period 1980–2012 specified in the operating licenses of combustion plants was obtained from county authorities. The information on NO_x emissions and production of useful energy necessary to establish the NO_x tax liabilities and refunds over the period 1992–2012 comes from the Swedish NO_x database, which is a panel covering all boilers monitored under the tax system. The NO_x database also includes information on whether NO_x abatement technologies are installed at each boiler, boiler capacity, and shares of different fuels in the fuel mix. Information about firm ownership was obtained from the Orbis database.

County-level variables were collected from several different sources. A variable for the ambient NO_x concentration in a county was obtained from the Swedish Research Institute IVL Air Quality Database (IVL, 2015). It includes data for most counties and years from 1986 and 2012. Data concerning the share of employees per sector per county was also obtained from Statistics Sweden.

See Table 2 for a description of the variables.

Variable		N	Mean	Std.Dev.	Min	Max
ELV	mg/NO$_x$	11477	110.77	50.22	21.90	300
NO$_x$ Concentration	mg/NO$_x$	7276	70.70	35.58	0.25	277
Treated	1 if subject to NO$_x$ tax; 0 otherwise	18386	0.66	0,47	0	1
OldBoiler	1 if existed in 1992; 0 otherwise	18386	0.57	0.49	0	1
Entrants92-96	1 if entered 1992-1996; 0 otherwise	18386	0.16	0.36	0	1
Entrants96-97	1 if entered 1996-1997; 0 otherwise	18386	0.06	0.24	0	1
# ELV	# of ELV (1: first ELV)	11477	1.67	0.91	1	7
Boiler Characteristics						
Size	Installed boiler effect in MegaWatts	15503	51.42	89.92	1.3	825
Technology	1 if technology installed; 0 otherwise	11479	0.55	0.50	0	1
Gas Fuel	1 if boiler uses mainly gas fuel; 0 otherwise	12275	0.18	0.38	0	1
Liquid Fuel	1 if boiler uses mainly liquid fuel; 0 otherwise	12275	0.19	0.39	0	1
Solid Fuel	1 if boiler uses mainly solid fuel; 0 otherwise	12275	0.16	0.36	0	1
Biofuel	1 if boiler uses mainly biofuel; 0 otherwise	12275	0.47	0.50	0	1
Firm Characteristics						
Waste	1 if waste sector; 0 otherwise	18386	0.10	0.30	0	1
Food	1 if food sector; 0 otherwise	18386	0.06	0.23	0	1
Heat and Power	1 if heat and power sector; 0 otherwise	18386	0.55	0.50	0	1
Pulp and Paper	1 if pulp and paper sector; 0 otherwise	18386	0.11	0.32	0	1
Metal	1 if metal sector; 0 otherwise	18386	0.03	0.16	0	1
Chemicals	1 if chemicals sector; 0 otherwise	18386	0.06	0.24	0	1
Wood	1 if wood sector; 0 otherwise	18386	0.09	0.28	0	1
# Boilers	No. of boilers per plant	18386	4.07	3.64	1	21
#Plants	No. of plants per firm	18386	3.64	5.61	1	29
Public	1 if publicly owned; 0 otherwise	18386	0.46	0.50	0	1
County Characteristics						
NO$_x$ County	Highest wintertime concentration air, μg/m3	17053	27.38	14.68	0.6	63.4
EmployeesSector	% employees in sector in which boiler is used	18386	6.66	2.11	0.82	12.8

Table 3: Summary Statistics

Table 3 shows that 70% of our observations have been subject to the NO_x tax at some point, and that 79% were in operation by 1997. Moreover, there is large variation in the stringency of ELVs and NO_x emissions concentrations. Such variation reflects differences in, e.g., boiler size, fuel mix, sector, and availability of technology. Most boilers have NO_x-reducing technologies installed (55%), the most common fuel used by boilers is biofuel (47%), and most of the boilers in the dataset belong to the heat and power sector. Furthermore, 46% of the boilers in the sample are owned by public companies (i.e., by municipalities or the national government). Regarding county characteristics that might affect the stringency of the ELV setting process, the highest wintertime ambient NO_x concentration ranges from 0.6 to 63.4 $\mu g/m^3$ across counties, with a mean of about 27 $\mu g/m^3$. As described before, values over 40 $\mu g/m^3$ are consistent with exceedances of the Swedish Environmental Quality Standards. Finally, on average, 6.66% of the people employed in each county works in the sectors in which the boilers in the sample are used.

5   Results

In what follows, we discuss the results of the econometric models.

5.1   ELV$_S$ are more stringent for boilers subject to the national tax

We first compare the stringency of ELVs in G_1 to the control group to verify the hypothesis that ELVs are more stringent for boilers subject to both the standards and the tax. In cols (1) and (2) of Table 4, we present the results of the difference-in-difference model specified in equation (12), focusing on the boilers already in operation by 1992 (using only the information available over the period 1980–1995). In col (1) we present the results for the subsample of first ELVs regulating the boilers in operation by 1992, while col (2) shows the results for the whole sample of ELVs for the boilers in operation by 1992 (controlling for the order of the ELVs). Moreover, cols (3)–(5) present the results of the difference-in-difference model specified in equation (13), focusing on the boilers already in operation by 1997 (and using the information available for the period 1980–2012). Col (3) refers again to the sample of first ELVs while cols (4)–(5) refer to the whole sample of ELVs for the boilers in operation by 1997. Col (4) refers to boilers in all counties, while

col (5) includes only boilers located in the five counties where the Swedish Environmental Quality Standards have been exceeded (e.g., it includes only the ELVs for the counties where ambient NO_x emissions concentration are higher than the legal limit to investigate whether the results are more salient in counties with higher levels of pollution and industrial activity).

A negative sign of the coefficient indicates that the determinant makes the ELV more stringent. We observe that the difference-in-difference estimator Did_G_i is negative and statistically significant in almost all specifications in cols (1)–(5). This is to say, the stringency of the ELVs for those boilers in G_1 subject to the NO_x tax increased after they became subject to the tax, and the same holds for the stringency of the ELVs for boilers in G_2 in specifications (4)–(5) and the stringency of the ELVs for boilers in G_3 in specifications (3)–(4).

Let us focus on the analysis of the results in cols (1)–(2). As shown in the table, after the treatment, the stringency of the ELVs for the boilers subject to the tax increased by 41.34 mg in col (1) and by 42.53 mg in col (2). Moreover, the simple pre versus post estimator $\beta + \delta$ in cols (1) and (2) is also negative and statistically significant. This finding is in line with our expectations as the stringency of the environmental legislation is likely to have increased over time and the average ELV stringency should follow this trend. Note that γ is positive and statistically different from zero, indicating permanent average differences in ELV stringency between the treatment groups (in particular, in the absence of treatment, boilers exceeding the threshold of production of useful energy to become subject to the NOx tax were allowed less stringent ELVs than boilers that produce lower amounts of useful energy).[9] However, such a difference disappeared after the treatment since $\gamma + \delta$ (the simple treatment versus control estimator that ignores pre-treatment outcomes) is not statistically different from zero.

Regarding cols (3)–(5), in col (3) the difference-in-difference estimator Did_G_i is negative not only for boilers in G_1 but also for boilers in G_2 and G_3. However, the increased ELV stringency for boilers regulated under the tax is only statistically significant for boilers in G_1 and G_3. In relative terms, the size of G_2 is small (6.6% of our observations, vs. 64% and 14% for G_1 and G_3), and in col (3) we only analyze the subsample of first ELVs. In cols (4) and (5), the sample includes all ELVs for the boilers in operation by 1997, and the difference-in-difference estimator becomes

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[9]Note that in randomized experiments, where subjects are randomly selected into treatment and control groups, γ should be zero as both groups should be nearly identical. However, in most program evaluation problems seen in economics there is selection into the treatment.

statistically significant also for boilers in G_2. Moreover, in line with the summary statistics in Table 2, the results for the difference-in-difference estimators in col (4) indicate that the increased stringency of ELVs after the treatment is larger for G_1 than for G_3, and larger for G_3 than for G_2. Thus, even if the effect of the treatment is observed for all our three groups of boilers, the effect is the largest for the first group of boilers being treated, followed much behind in magnitude by G_3.

Regarding covariates, in cols (3)–(5) we observe that the lagged value of the county-level concentration of NO_x is a statistically significant determinant of ELV stringency in period t, but this effect is not observed in cols (1)–(2). Interestingly, the effect of the ambient concentration of NO_x on ELV stringency is larger in the counties where the Swedish Environmental Quality Standards have been exceeded; see col (5).

The share of employees per sector working in the regulated firms seem to only have mattered when setting the first ELV; see col (3). In such case, a larger share of employees working in regulated firms translated into less stringent ELVs. In contrast, boiler size is an important determinant of the ELVs in all specifications. The results indicate that larger boilers obtain more stringent ELVs, probably because they have better capacity to reduce emissions.

Regarding fixed effects for fuel, using the least NO_x-intensive fuels (gas fuel) as the reference dummy variable, we observe that, on average, the more NO_x-intensive a fuel is, the less strictly regulated it tends to be. Finally, in specifications (2), (4), and (5) we also control for the order of the ELVs. These fixed effects are all negative and statistically significant, indicating that the stringency of the ELVs has increased across revisions. As discussed before, this is to be expected due to learning, technological progress, and adoption of NO_x dedicated abatement technologies.

ELVs	(1)	(2)	(3)	(4)	(5)
D_92	-3.59	4.58			
G_1	34.70**	43.12***	86.47***	87.65***	78.36***
Did_G_1	-41.34***	-42.53***	-67.51***	-78.48***	-71.93**
G_2			34.14***	28.99**	18.58
Did_G_2			-17.31	-11.90*	-11.45**
G_3			42.70**	22.95*	4.64
Did_G_3			-24.94**	-21.12**	-12.81
Size	-0.06**	-0.07**	-0.07**	-0.07***	-0.11***
Boilers per Plant	1.67	-2.29	0.43	-0.25	1.16
Plants per Firm	-0.21	-2.73	0.02	-0.83**	-0.42
Public Firm	-10.38	-8.74	-10.54	-8.18	-1.84
NO_x County_1	-0.02	0.91	-0.55***	-0.36**	-0.67***
EmployeesSector_1	-2.56	-0.04	12.45**	-1.38	-8.31
FE Fuel	YES	YES	YES	YES	YES
FE Sector	YES	YES	YES	YES	YES
FE County	YES	YES	YES	YES	YES
FE # ELV	—	YES	—	YES	YES
#Obs	590	821	2137	5020	2837
#Boilers	135	159	220	327	175
R^2	0.68	0.56	0.56	0.46	0.48

Table 4: Difference-in-Difference Model

Thus, we can conclude that the results provide empirical support to our hypothesis that the ELVs are more stringent for boilers that are subject to the national tax. Moreover, the magnitude

of the effect of the tax on the ELV stringency seems to be larger for the group of boilers that first became subject to the tax.

5.2 Emissions Reductions are not driven by ELVs

Table 5 presents the results of the regression model specified in equation (14). In cols (1)–(4), we present the results for the sample of boilers subject to the NO_x tax that were in operation by 1992, while cols (5)–(6) present the results for the sample of boilers subject to the NO_x tax in operation by 1997.

Regarding cols (1)–(4), emissions concentrations are expected to decrease either through the adoption of abatement technologies or if revisions of ELVs would impose further requirements on boilers. In col (2), we control for the effect of the availability of technology at time $t-1$ on emissions concentrations at time t. In col (3), we also control for the effects of ELV revisions through fixed effects accounting for the order of the ELVs. Finally, since our theoretical model suggests that ELVs are more likely to be binding when the share of tax revenues refunded to a county is small, in col (4) we run the specification in col (3) only for the subsample of counties with lower shares of the refunds (i.e., we exclude boilers located in Stockholm, Västra Götaland and Skåne since as shown in Figure 3, they receive much larger shares of the refunds than other counties). Finally, in col (5), we run the same specification as in col (3) for all boilers in operation by 1997, while col (6) only includes the sample of boilers in operation by 1997 located in counties with low shares of the NO_x refunds.

The results in cols (1)–(4) consistently indicate that when analyzing the sample of boilers in operation by 1992, the only two variables affecting firms' emissions reductions are size and the existence of installed abatement technology. Indeed, in all specifications, we observe no statistically significant emissions-reducing effect of the ELVs. These results also hold when analyzing the full sample of boilers in operation by 1997 (see col 5), but the effect of the ELV on emissions concentrations turns out to be slightly statistically significant in col (6), which corresponds to the sample of boilers in operation by 1997 in counties that receive lower shares of the NO_x tax revenues. Thus, the results seem to confirm our theoretical prediction that it is unlikely that ELVs induce further emissions reductions than those induced by the NO_x tax, and if so, this might happen in counties that receive a low share of the tax refunds.

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Regarding covariates, in all specifications we observe that boilers making using liquid and solid fuels generate higher emissions concentration than boilers using natural gas and biofuels and boilers in the pulp and paper sector generate higher emissions concentrations than boilers in other sectors.

The fact that emissions concentrations are mainly driven by boiler size, technology, and fuel type is consistent with the firms' implicit reaction function described in equation (3), which provides support to our hypothesis that the NO_x tax is the policy driving emissions reductions.

NO_x Concentration	(1)	(2)	(3)	(4)	(5)	(6)
Lag_D_ELVt	-6.79	-4.00	-6.89	-5.70	-6.79	-8.24*
Size	-0.06***	-0.05***	-0.05**	-0.04*	-0.06***	-0.07**
Boilers per Plant	-1.31***	-1.08***	-1.20***	-1.17	-0.73	-0.75
Plants per Firm	-0.28	-0.19	-0.11	-0.03	-0.43*	-0.08
Technology Installed_l		-7.76***	-8.14***	-11.71***	-7.69***	-9.13***
Public Firm	-2.19	0.96	1.18	-0.72	-3.31	-5.70
NO_x County_1	-0.07	-0.03	-0.05	0.09	-0.11	-0.03
EmployeesSector_1	0.56	-0.28	-0.35	-4.90	2.22	-1.12
FE Fuel	YES	YES	YES	YES	YES	YES
FE Sector	YES	YES	YES	YES	YES	YES
FE County	YES	YES	YES	YES	YES	YES
FE # ELV	NO	NO	YES	YES	YES	YES
#Obs	3297	3297	3156	1662	4388	2514
#Boilers	274	274	263	143	403	233
R^2	0.24	0.27	0.27	0.35	0.21	0.26

Table 5: Drivers of Emissions Reductions

6   Conclusions

Air quality control is often undertaken by means of policy mixes where not only national but also local regulations are implemented. In this paper, we study the effects of the interaction of national and local policies on NO_x emissions in Sweden, analyzing whether the coexistence of national and local policies has increased the country´s ability to achieve emissions reductions. We conclude that the implementation of the national tax has brought along stricter emissions standards over time. Counties set more stringent emissions standards in response to the implementation of a national uniform tax since stricter standards incentivize increased emissions reductions by local firms, allowing counties to reduce the tax payment to the national government. However, the increases in stringency of the local emissions standards have not been aggressive enough to induce any reductions in emissions beyond those induced by the national uniform tax.

Why is the tax more effective reducing emissions? Because a national uniform tax is set to a level that is high enough to internalize air pollution spillovers across counties. In contrast, when a county decides on the stringency of the local policies, it takes into account only local pollution damages and abatement costs. Thus, the tax is simply more stringent than the standards, except in the cases when the local damages of air pollution are quite high and the spillovers quite small. When both policies are combined, local regulators have an incentive to increase the stringency of the local standards as a means to reduce tax payments to the national government. Therefore, the national tax leads local governments to internalize part of the effects of their pollution in other counties. However, for this mechanism to work, the revenues from taxing emissions should not be fully refunded to the county. Otherwise, the incentive to increase the stringency of the local standards in order to reduce the payment of the national tax disappears from the local regulators' objective.

The model in this paper is simplified in a number of respects to keep the analysis tractable. For example, in reality, pollution spillovers are characterized by a much more complex pattern of interregional relations than assumed in this paper. Such relations can be represented through a pollution dispersion matrix that is a function of several climatic, geographic and technical factors, such as the direction and average speed of the wind, the size of the region, the height of the source of the emissions, and air turbulence. In this paper we disregard such complexities. Although this simplification might affect the conditions that define when the standards are binding (as pollution

from close neighbors would have a larger impact in a county than pollution from counties located farther away), the analysis provides a useful starting point for assessing the economic incentives provided by policy mixes. Finally, our paper provides a fiscal motivation to the response of the local regulators to a national tax on polluting emissions. However, when deciding on the level of stringency regulators face asymmetric information regarding the firms' abatement costs. Thus, a further area of research concerns the informational value of the policy overlap. In particular, how local regulators could make use of the information on the firms' costs provided by the implementation of the tax to better target the standard on the firms' true cost.

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Appendix A

To compare aggregate emissions reductions under a uniform national tax vs. decentralized emissions standards, we assume that counties are symmetric in terms of the damages of air pollution and costs of emissions reductions. Thus, we assume that $D_i(e_i, E) = D(e, E)$ and $C_i(e_i) = C(e)$ $\forall$ i, such that $e_i = e$ and $E = ne$. Under such assumptions, the uniform tax rate T in equation (4) can be represented as:

$$T = \left[1 + n^2\right] \frac{\partial D(e, E)}{\partial e}, \tag{15}$$

and county $i's$ emissions reductions under the uniform tax rate T are determined by the solution to:

$$-\frac{\partial C(e)}{\partial e} = \left[1 + n^2\right] \frac{\partial D(e, E)}{\partial e}. \tag{16}$$

In contrast, from equation (6) it holds that emissions reductions under decentralized emissions standards will be determined by the solution to:

$$-\frac{\partial C(e)}{\partial e} = \left[1 + n\right] \frac{\partial D(e, E)}{\partial e}. \tag{17}$$

Comparing equations (16) and (17), it is straightforward to say that emissions reductions under the uniform tax rate are larger than under decentralized emissions standards, as the stringency of the uniform tax takes into account the existence of interjurisdictional externalities.

Appendix B

To investigate whether ELVs become the most restrictive policy instrument when combined with a uniform national tax, we assume that counties are symmetric in terms of the damages of air pollution and costs of emissions reductions. Thus, we assume that $D_i(e_i, E) = D(e, E)$ and $C_i(e_i) = C(e) \; \forall \; i$, such that $e_i = e$ and $E = ne$. Under such assumptions, the uniform tax rate T can be represented as:

$$T = \left[1 + n^2\right] \frac{\partial D(e, E)}{\partial e}, \tag{18}$$

Moreover, county $i's$ emission reductions under policy overlapping in equation (9) are determined by the solution to:

$$-\frac{\partial C(e)}{\partial e} = \left[1 + n\right] \frac{\partial D(e, E)}{\partial e} + T\left[1 - r_i\right]. \tag{19}$$

Substituting equation (15) into equation (19), it holds that $-\frac{\partial C(e)}{\partial e} > T$ if:

$$\left[1 + n\right] > \left[1 + n^2\right] r_i,$$

or:

$$\frac{1 + n}{1 + n^2} > r_i.$$

Let $r^* = \frac{1+n}{1+n^2}$. ELVs will become binding when $r_i < r^*$, while the uniform tax would be the binding policy if $r_i > r^*$. Moreover, note that since $r^* > \frac{1}{n}$, refunding tax liabilities uniformly allocated across counties will lead to binding ELVs.
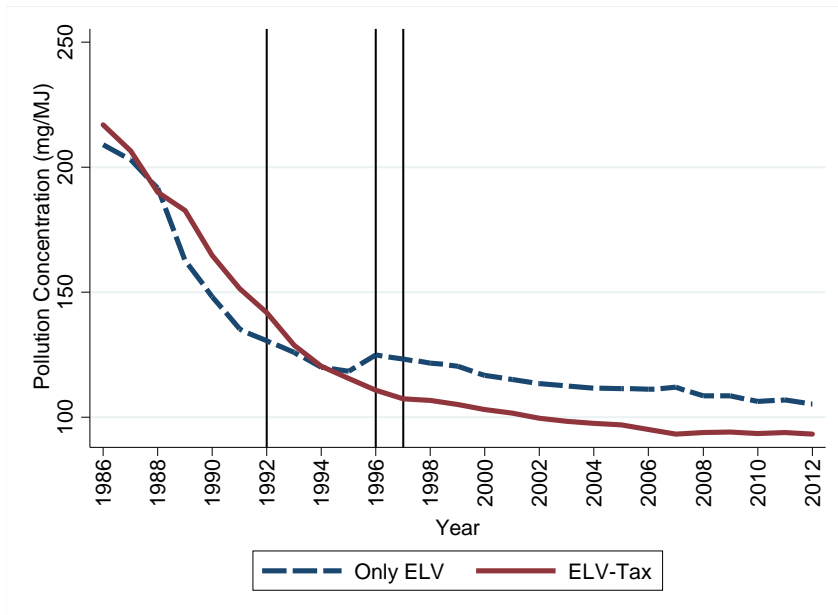
Figure 4: Stringency of ELVs Over Time

Appendix C