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Essays on Environmental Taxation and Climate Policy

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Kristina Mohlin
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Popular summary

A tax on pollution is one of the environmental economist's standard policy recommendations for correcting polluting activities which cause unintended harm to people and the environment. In theory, at least, the economist would like to set the tax at the optimal level at which the value of the damages from one more unit of pollution is equal to the cost of reducing pollution by one unit. This is the so-called Pigouvian tax which would maximize social well-being or welfare. The basic idea is that by putting the right price on pollution, individuals and firms will have an incentive to make choices which are in line with what is best for society as whole. Like most theories, the Pigouvian tax can work perfectly only if some very specific assumptions hold true.

In practice, of course, the costs and especially the benefits of reducing pollution are never fully known. Determining the benefits of reducing pollution requires first an understanding of the complex biogeochemical processes of pollution, including where, how long, and in which form the polluting compounds remain in the environment. Secondly, even when we have a good idea of what happens with the substance after its release, there is still the question of how to value the damages in terms of adverse effects on people's health and livelihoods, and damages to plant and animal life. Although economists have developed methods for doing this sort of valuation, it remains a very challenging task. On the cost side, policy makers usually cannot know very well what the polluters' actual costs of reducing pollution are. In addition, there is the issue of which sources of pollution can actually be monitored and effectively regulated. When there are sources of pollution which cannot be regulated, there is always the risk of so called "emission leakage" - in other words, that introducing a pollution policy will simply push some of the polluting activities to relocate.

In the end, environmental policies are not the outcome of the maximization of any abstract social welfare function. Instead, they are the result of a political process likely to be governed by, on the one hand, how much pollution is acceptable considering impacts on public health and ecosystems and, on the other, which polluters can be effectively regulated and how much they can spend on reducing pollution without reducing employment or economic growth. It is important to understand also what sort of incentives these policies provide to the businesses and consumers who make decisions about polluting activities.

This thesis consists of four self-contained papers on environmental taxes designed to

deal with the monitoring and emission leakage problems that are often relevant to the actual implementation of environmental policies. It covers two main themes. Papers I and II focus on greenhouse gas emissions from agriculture. Papers III and IV focus on diffusion of environmental technology and analyze when businesses decide to adopt emission-reducing technologies.

Starting with the first theme, an environmental tax in the form of a price on emissions of carbon dioxide (CO_2) is a solution to climate change. However, Sweden is one of few countries with a high CO_2 tax (approximately €110 or 1000 SEK per tonne of CO_2). Due to international competition and the risks of emission leakage, a large number of exceptions and deductions from the Swedish CO_2 tax have been granted to industry. Primarily, it is Swedish households and the service sectors which are facing the full Swedish CO_2 tax on fuels and heating.

The different tax rates across industry sectors is a cause for concern and is related to the main problem that most other countries are not pursuing a similar climate policy. Globally, most CO_2 emission sources still go unregulated. Under the Pigouvian theory, for the world to reach an emission target at the lowest cost, the price of emitting CO_2 would need to be the same irrespective of where the emissions come from and which activity produces them, so that reductions are made where they are the least costly. This ideal arrangement is not in place even in one country and is definitely not in place on a global level. Furthermore, to achieve the least costly solution to climate change, other greenhouse gases (GHGs), such as nitrous oxide (N_2O) and methane (CH_4), would need to be taxed at an equivalent rate.

Papers I and II are concerned with the fact that current climate policies primarily cover CO_2 emissions from the energy and transportation sectors, even though the agricultural sector is responsible for 25-30% of global GHG emissions. In the EU, the great majority of agricultural GHG emissions are in the form of nitrous oxide from fertilized agricultural soils and methane from manure and digestion by ruminants (cows, sheep, goats). However, monitoring emissions of nitrous oxide and methane from agriculture is much more difficult than monitoring CO_2 emissions from fossil fuel use in the energy and transportation sectors. To accurately monitor methane from digestion, emissions from a significant sample of animals at each farm would have to be measured regularly. An accurate monitoring of nitrous oxide emissions would similarly require virtually continuous measurement for a great fraction of the fields. Obviously, in these circumstances, an actual emission tax on methane and nitrous oxide is not feasible since the emission sources cannot be monitored at reasonable cost.

One alternative to an emission tax, which avoids the monitoring problem and reduces the risk of emission leakage, is to tax the consumption of the most emission-intensive agricultural products. These are generally food products from livestock production and, in particular, ruminant meat, such as beef. ***Paper I*** analyzes a greenhouse gas tax on the consumption of animal food products. It aims to answer the question of how much emissions could be reduced if there were a GHG tax in the EU on the consumption of meat and dairy products on the same order of magnitude as the Swedish CO_2 tax. By lowering demand for agricultural land used for meat production, this tax could also contribute to further emission reductions by expanding the opportunities for biofuel production. The paper, therefore, also explores different scenarios for how much emissions could be reduced by promoting substitutes for fossil fuels through an expansion in bioenergy production. The results suggest that the tax could lower emissions from EU agriculture by 7% - primarily by reducing the demand for ruminant meat and encouraging substitution to poultry and pig meat, which give rise to much less GHG emissions in production. Emission reductions could be six times higher if the agricultural land no longer used for animal food production were used instead to grow bioenergy crops that substitute for coal in power generation.

An environmental tax on consumption can have the additional advantage of clearly signaling to consumers which products are more environmentally friendly and thereby possibly influencing what they choose to buy, not merely by increasing the price of the polluting goods, but also by informing them. A disadvantage is that a consumption tax does not provide any incentives to producers to change their production practices in ways which reduce the environmental impact.

Paper II considers another alternative to an emission tax which does encourage producers to change their practices: taxes on polluting inputs to production. Fertilizer is an example of a polluting input to agricultural production. However, taxing fertilizer generally does not target agricultural pollution very well. Take the example of a tax on nitrogen in fertilizers. The problem is that one unit of reactive nitrogen contributes differently to nitrous oxide emissions (as well as water pollution) depending on when, how and where it is used. Still, with the intention of reducing water pollution, Sweden introduced a uniform tax on nitrogen in commercial fertilizer in 1989. This tax was removed in 2010, possibly to compensate farmers for planned future increases in the CO_2 tax for the agricultural sector; this is another illustration of political concerns over polluters costs.

The research question this paper aims to answer is to what extent the Swedish nitrogen tax helped to reduce nitrous oxide emissions from Swedish agriculture. The results suggest a relatively modest impact of a 2% reduction of N_2O emissions from Swedish agricultural soils. Nevertheless, it appears that increases in N_2O emissions resulting from the political decision to remove the nitrogen tax can possibly fully offset the decreases in CO_2 emissions that can be expected from the future increase in the CO_2 tax for the agricultural sector. This paper also illustrates some of the challenges in constructing a model that gives a good representation of emissions resulting from complex processes and at the same time can be linked to a simple model of the economic driving forces.

In the second part of the thesis, the focus is shifted to pollution resulting from energy production. The long-run impact of environmental policies is determined mainly by the incentives they provide for innovation and diffusion of environmentally friendly technologies. To manage climate change over the long run, innovative technologies that can drastically reduce CO_2 emissions will be required. However, the technologies that already exist can break the path of increasing CO_2 emissions and, in the next decades, reduce them to levels which would make it possible to avert dangerous changes in the climate system. What is missing are the policies which make broad-scale investments in these technologies profitable and thus encourage their diffusion.

Paper III and IV analyze diffusion of emission-reducing technologies under one type of emission tax which Swedish policy makers successfully introduced in the 1990s to overcome the problems of political resistance from polluters and the risks of emission leakage. What was introduced in 1992 was a refunded tax (or charge) on NO_x* emissions from large combustion plants. By refunding the tax revenues back to the regulated firms in proportion to how much useful energy they produce, the producers as a group pay a zero net tax. The plants that are dirtier than average pay a net tax on their energy production while the plants that are cleaner than average receive a net subsidy. This refunding scheme made a high tax more acceptable to the regulated firms and also avoids emission leakage to plants whose emissions are too small and costly to monitor.

Paper III uses a theoretical model to compare a refunded emission tax to a non-refunded

*NO_x is a generic term for the nitrogen oxides NO and NO_2 which contribute to acid rain, among other things, and should not be confused with nitrous oxide (N_2O), which contributes to global warming.

tax in terms of how they stimulate investments in emission-reducing technologies. It aims to answer the question of whether the refunding of taxes would speed up diffusion vis-a vis non-refunded taxes. The results suggest that refunding can speed up the diffusion of emission-reducing technologies, but this depends on how competitive the market for the firms' output is. There is no general result on which type of emission tax will stimulate faster technology diffusion.

Paper IV is an empirical study of which factors determine when the firms covered by the Swedish NO_x charge invest in environmental technologies. It aims to answer the question of what drives diffusion of different emission-reducing technologies. The paper analyzes investments in three types of technologies - first, technologies which reduce the formation of NO_x at combustion; second, end-of-pipe technologies which are add-on measures that curb emissions after their formation; and lastly, technologies which improve energy efficiency. The results indicate that a firm which pays a higher NO_x charge, net of the refund, is more likely to invest, but this is only true for end-of-pipe technologies. The combustion plants also belong to different industrial sectors, and the results show that firms in some sectors are more likely to invest than firms in other sectors. End-of-pipe NO_x technologies and technologies that improve energy efficiency are more likely to be adopted in the heat and power and waste incineration sectors, which is possibly linked to less competition and more public ownership in these sectors.

In sum, this thesis analyzes three environmental taxes designed to deal with the monitoring and emission leakage problems often relevant to the implementation of environmental policies. It assesses impacts of these taxes relevant to their evaluation and contributes to the discussion about appropriate policies for reducing pollution from agriculture and energy production.

Abstracts

Paper I: *Greenhouse gas taxes on animal food products - rationale, tax scheme and climate mitigation effects*

Agriculture is responsible for 25- 30% of global anthropogenic greenhouse gas (GHG) emissions but has thus far been largely exempted from climate policies. Because of high monitoring costs and comparatively low technical potential for emission reductions in the agricultural sector, output taxes on emission-intensive agricultural goods may be an efficient policy instrument to deal with agricultural GHG emissions. In this study we assess the emission mitigation potential of GHG weighted consumption taxes on animal food products in the EU. We also estimate the decrease in agricultural land area through the related changes in food production and the additional mitigation potential in devoting this land to bioenergy production. Results indicate that agricultural emissions in the EU27 can be reduced by approximately 32 million tons of CO_2-eq with a GHG weighted tax on animal food products corresponding to €60 per ton CO_2-eq. The effect of the tax is estimated to be six times higher if lignocellulosic crops are grown on the land made available and used to substitute for coal in power generation. Most of the effect of a GHG weighted tax on animal food can be captured by taxing the consumption of ruminant meat alone.

Paper II: *The Swedish nitrogen tax and greenhouse gas emissions from agriculture*

The Swedish tax on nitrogen in synthetic fertilizers was abolished in 2010, possibly to compensate farmers for planned future increases in the CO_2 tax for the agricultural sector. This study estimates the effect of the nitrogen tax on agricultural emissions of nitrous oxide (N_2O), another greenhouse gas (GHG) that is more potent than CO_2. Price elasticities of nitrogen fertilizer use are estimated from county-level panel data and combined with the standard GHG accounting approach for international reporting of N_2O emissions, as well as an alternative emission function suggested in the literature, to estimate the impact of the tax on emissions. The results suggest that annual direct N_2O emissions from agricultural soils in Sweden would have been on average 160 tons higher without the tax. Results also indicate that higher N_2O emissions from the removal of the N tax has the potential to fully offset the decreases in GHG emissions that can be expected from the future tax increase on CO_2 from agricultural diesel use.

Paper III: ***On refunding of emission taxes and technology diffusion***

We analyze diffusion of an abatement technology under a standard emission tax compared to an emission tax which is refunded in proportion to output market share. The results indicate that refunding can speed up diffusion if firms do not strategically influence the size of the refund. If they do, it is ambiguous whether diffusion is slower or faster than under a non-refunded emission tax. Moreover, it is ambiguous whether refunding continues over time to provide larger incentives for technological upgrading than a non-refunded emission tax, since the effects of refunding dissipate as the overall industry becomes cleaner.

Paper IV: ***Diffusion of NO_x abatement technologies in Sweden***

This paper studies how different NO_x abatement technologies have diffused under the Swedish system of refunded emissions charges and analyzes the determinants of the time to adoption. The policy, under which the charge revenues are refunded back to the regulated firms in proportion to energy output, was explicitly designed to affect investment in NO_x-reducing technologies. The results indicate that paying a higher net NO_x charge increases the likelihood of adoption, but only for end-of-pipe post-combustion technologies. We also find some indication that market power considerations in the heat and power industry reduce the incentives to abate emissions through investment in post-combustion technologies. Adoption of post-combustion technologies and the efficiency improving technology of flue gas condensation is also more likely in the heat and power and waste incineration sectors, which is possibly explained by a large degree of public ownership in these sectors.

Paper I

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Greenhouse gas taxes on animal food products: rationale, tax scheme and climate mitigation effects

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Abstract Agriculture is responsible for 25–30% of global anthropogenic greenhouse gas (GHG) emissions but has thus far been largely exempted from climate policies. Because of high monitoring costs and comparatively low technical potential for emission reductions in the agricultural sector, output taxes on emission-intensive agricultural goods may be an efficient policy instrument to deal with agricultural GHG emissions. In this study we assess the emission mitigation potential of GHG weighted consumption taxes on animal food products in the EU. We also estimate the decrease in agricultural land area through the related changes in food production and the additional mitigation potential in devoting this land to bioenergy production. Estimates are based on a model of food consumption and the related land use and GHG emissions in the EU. Results indicate that agricultural emissions in the EU27 can be reduced by approximately 32 million tons of CO_2-eq with a GHG weighted tax on animal food products corresponding to €60 per ton CO_2-eq. The effect of the tax is estimated to be six times higher if lignocellulosic crops are grown on the land made available and used to substitute for coal in power generation. Most of the effect of a GHG weighted tax on animal food can be captured by taxing the consumption of ruminant meat alone.

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Paper II

The Swedish nitrogen tax and greenhouse gas emissions from agriculture*

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Abstract

The Swedish tax on nitrogen in synthetic fertilizers was abolished in 2010, possibly to compensate farmers for planned future increases in the CO_2 tax for the agricultural sector. This study estimates the effect of the nitrogen tax on agricultural emissions of nitrous oxide (N_2O), another greenhouse gas (GHG) that is more potent than CO_2. Price elasticities of nitrogen fertilizer use are estimated from county-level panel data and combined with the standard GHG accounting approach for international reporting of N_2O emissions, as well as an alternative emission function suggested in the literature, to estimate the impact of the tax on emissions. The results suggest that annual direct N_2O emissions from agricultural soils in Sweden would have been on average 160 tons higher without the tax. Results also indicate that higher N_2O emissions from the removal of the N tax has the potential to fully offset the decreases in GHG emissions that can be expected from the future tax increase on CO_2 from agricultural diesel use.

Keywords: nitrogen tax, agriculture, greenhouse gas emissions

JEL Classification: H23, Q11, Q18, Q54

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1 Introduction

Agriculture is responsible for 25-30% of total global greenhouse gas (GHG) emissions and is the largest contributor to emissions of nitrous oxide (N_2O) and methane (Smith *et al.*, 2007; Houghton, 1999; Steinfield *et al.*, 2006). In Europe, most countries do not have agricultural policies designed to reduce GHG emissions. Emission reductions in the agricultural sector have instead mostly occurred through non-climate policies (Smith *et al.*, 2007), such as agri-environmental schemes which have helped to increase soil carbon stocks (Freibauer *et al.*, 2004). Another non-climate policy is the tax on nitrogen (N) in synthetic fertilizers which was introduced in Sweden in 1984 with the purpose of reducing water pollution. Because the use of nitrogenous fertilizers is also a driver of emissions of N_2O, a very potent greenhouse gas, the N tax may also have had an impact on GHG emissions. Nevertheless, in 2010, the Swedish tax of 1.80 SEK kg^{-1} N was abolished, possibly to compensate farmers for planned future increases in the CO_2 tax for the agricultural sector[1].

The purpose of this paper is to estimate the effect of the Swedish nitrogen tax on direct N_2O emissions from agricultural soils. Few previous studies have analyzed the link between taxes on nitrogenous fertilizers and N_2O emissions[2]. Many analyses which discuss fertilizer taxes, such as Shortle *et al.* (1998), Shortle & Horan (2001) and Claassen & Horan (2001), focus on impacts on water pollution. Fertilizer demand in general, on the other hand, has been studied quite extensively. Previous studies of fertilizer demand in Sweden include Brännlund & Gren (1999a), in which demand elasticities for six Swedish drainage basins were found to be between -0.3 and -1.23. Brännlund & Gren (1999b) used the seemingly unrelated regression (SUR) estimator on a similar dataset and found that own-price elasticities of nitrogen demand for seven different regions in Sweden were between -0.1 and -0.5 and Ingelsson & Drake (1998) estimated a national own-price elasticity of -0.3.

Studying N_2O emissions in particular is also relevant in view of the large uncertainties

[1]The tax rate is approximately 0.24 USD kg^{-1} using an exchange rate of 7 SEK USD^{-1}. The tax was initially a charge and from 1994 onwards 1.80 SEK kg^{-1} N (on average 20% of the nitrogen price) and later designated a tax (SOU, 2003:9). The government's stated intention with the proposal to abolish the N tax in 2010 was to improve the competitiveness of Swedish farmers (Proposition 2009/10:41, p. 136). From reading the bill, a natural interpretation is also that abolishing the N tax was intended to compensate farmers for, in the same bill, reducing the previously generous deductions on the CO_2 tax on diesel use in agriculture.

[2]Cara *et al.* (2005) assesses the potential GHG abatement at a range of CO_2-equivalent prices using a linear programming model for EU agriculture and could be said to implicitly analyze N taxes since they, like this study, use linear relationships between N_2O emissions and N use based on the IPCC methodology. Results are, however, difficult to compare due to the differences in the level of aggregation and modelling approach.

surrounding their relationship to nitrogen use. N_2O emissions display large spatial and temporal variability, and much uncertainty remains around the impact of nitrogen application to soils (Grant *et al.*, 2006). The IPCC (Intergovernmental Panel on Climate Change) method for international reporting of N_2O emissions under the United Nations Framework Convention on Climate Change (UNFCCC) relies on linear relationships between emissions and total nitrogen use, while studies such as Bouwman *et al.* (2002) and Van Groenigen *et al.* (2010) suggest that N_2O emissions may increase progressively with the rate of nitrogen application. That the intensity of N use is a determinant of agricultural pollution is a natural and almost implicit view in the literature on nitrogen pollution (see e.g., Galloway *et al.* (2008) in *Science*). This study therefore also compares results based on the IPCC method for international reporting of GHG emissions to results using an alternative emission function suggested in the literature that takes the intensity of N use into account.

To analyze the impact of the N tax on N_2O emissions, we first analyze farmers' response to changes in the price of nitrogen. We estimate price elasticities of total N demand as well as price elasticities of N use disaggregated into changes on the intensive margin (N application rates) and extensive margin (cropland allocation) from county-level panel data. We combine the price elasticity of total N demand with the IPCC accounting method used in the Swedish GHG inventory reports to find our first estimate of the impact of the N tax on direct N_2O emissions from agricultural soils. Second, we use an emission function for which emissions is a function of the rate of N application. We combine this emission function with the estimates of price elasticities of N application rates and cropland allocation to find a second estimate of the impact of the N tax. To take account of the uncertainty surrounding the value of the emission function parameters, we perform a Monte Carlo simulation of emissions with and without the N tax. The mean estimate for the results based on the emission function is comparable in magnitude to the estimate based on the IPCC accounting method. The average estimated impact is a reduction in direct N_2O emissions of 160 tons or 2% of direct emissions from agricultural soils in Sweden.

The paper is organized as follows. Section 2 briefly describes the relationship between agricultural N use and N_2O emissions. Section 3 presents a simple framework for assessing how direct N_2O emissions from soils change in response to a nitrogen tax. Section 4 describes the data and econometric results on the price elasticities of nitrogen and land use. Section 5 presents the simulation results on the impact of the N tax on N_2O emissions. Section 6 discusses the results and concludes.

2 Nitrogenous fertilizers and N_2O emissions

Nitrous oxide (N_2O) is a product of soil microbial processes and a natural component of the Earth's atmosphere. It is a potent greenhouse gas with a capacity to absorb infrared radiation that is close to 300 times greater than that of carbon dioxide[3]. In the stratosphere, N_2O also contributes to the loss of ozone, with consequences for human health (Smith, 2010b). Since 1850, the atmospheric concentration of N_2O has increased by 20%, indicating changes to the sources and sinks of N_2O in the global nitrogen cycle (Smith *et al.*, 2010). The likely cause is human activity and the major driver the application of nitrogenous fertilizers and animal manure to agricultural land (Stehfest & Bouwman, 2006).

N_2O from agricultural soils can be decomposed into background emissions as well as direct and indirect emissions. Background emissions arise from unfertilized agricultural fields. Direct emissions are the emissions from fertilized fields which are additional to the background emissions. Lastly, indirect emissions occur when N_2O eventually forms from nitrogen lost from the farm system due to nitrate leaching or ammonia volatilization (Mosier *et al.*, 1998).

Direct emissions account for the largest share of N_2O from agricultural soils in Sweden. According to Sweden's national GHG inventory report for 2009, submitted under the UNFCCC (SEPA, 2011), direct N_2O emissions from agricultural soils were $7.8 \cdot 10^3$ tonnes in 2009, equivalent to 2.4 million tonnes of CO_2-equivalents or 31% of the official Swedish figure on GHG emissions from agriculture. In comparison, indirect emissions were $3.4 \cdot 10^3$ tonnes of N_2O and background emissions from the cultivation of mineral soils were $2.4 \cdot 10^3$ tonnes. In this study, we focus on the direct N_2O emissions from agricultural soils.

The above figures for direct N_2O emissions from agricultural soils are based on the IPCC (2006) methodology recommended for countries' reporting under the UNFCCC. They are, in principle, assessed by multiplying a constant emission factor by total national additions of nitrogen to soils, with different emission factors used for different sources such as synthetic

[3]The global warming potential (GWP) is a measure of the radiative forcing capacity of another greenhouse gas relative to CO_2 over a given time horizon. A recent estimate of the global warming potential of N_2O for a 100 year horizon is 298 (Forster *et al.*, 2007). In GHG national inventory reports submitted under the UN Framework Convention on Climate Change (UNFCCC), an older estimate of 310 is used for N_2O, which means that 1 kg of N_2O is equivalent to 310 kg of CO_2 in the GHG accounts (SEPA, 2011). For comparability with official GHG accounts, we use a global warming potential of 310 for N_2O in this study.

and organic fertilizer nitrogen. The Swedish Environmental Protection Agency (SEPA) uses a factor of 0.8% for synthetic fertilizer nitrogen, implying that 1 tonne of nitrogen added to soil is assumed to result in 80 kg of N_2O-N[4] released from the fields to the atmosphere. Direct emissions, E, of N_2O-N related to synthetic fertilizer nitrogen is thus approximately[5] simply given by

$$E = c_{N_2O} N \tag{1}$$

where the emission factor $c_{N_2O} = 0.8\%$ and N is the total amount of nitrogen sold nationally.

We use the relationship in (1) to find our first estimate of the impact of the N tax, and refer to this as the emission factor approach. However, N_2O emissions display large spatial and temporal variability, and much uncertainty remains around the impact of nitrogen application to soils (Grant *et al.*, 2006). Indeed, the Swedish GHG inventory report for 2009 lists direct soil emissions as the largest source of uncertainty (SEPA, 2011).

Some studies (e.g., Bouwman et al., 2002, Grant et al., 2006, and Van Groenigen et al., 2010) suggest that N_2O emissions may increase progressively with the rate of nitrogen application. A potential explanation is that emissions of N_2O increase as more nitrogen is applied than is taken up by the crop, because nitrogen not taken up by the crop (residual nitrogen) may be lost to the environment. Due to decreasing yield returns, residual nitrogen should be a convex function of the rate of nitrogen application. A convex relationship between nitrogen runoff and nitrogen application is also a common assumption in the literature on non-point source pollution (Claassen & Horan, 2001). If the relationship between emissions and N application is non-linear, the linear emission factor approach may give a poor approximation of the impacts on emissions from changes in N application.

In a meta-analysis of studies of N_2O emissions from agricultural soils in primarily temperate climates, Van Groenigen *et al.* (2010) estimated yearly direct emissions per hectare as a convex function of residual nitrogen. The proposed relationship was:

$$e(z) = \kappa + \psi \exp(\rho z), \tag{2}$$

where $e(z)$ is direct N_2O emissions [kg N_2O-N ha^{-1} year^{-1}], z is residual nitrogen in kg N per hectare and the estimated parameters $\kappa > 0, \psi > 0$ and $\rho > 0$, are shown in Table 1.

[4]Since the relative atomic mass of nitrogen is 14 and the relative molecular mass of N_2O is 44, 1 kg of N_2O-N is equivalent to $44/28 \approx 1.57$ kg of N_2O.

[5]In the official national inventory figures on direct N_2O emissions from agricultural soils, some additional smaller adjustments are made related to, inter alia, nitrogen lost as ammonia (SEPA, 2011).

$\kappa + \psi$ represent average background emissions of 1.5 kg N_2O-N ha^{-1} year^{-1} and $\rho > 0$ is a scale parameter which determines the convexity of the function, i.e., at what rate emissions increase with a marginal unit of residual nitrogen.

An association between N_2O emissions and residual nitrogen may also explain observed differences in emissions for different crop types, given comparable levels of nitrogen application and other conditions. Bouwman *et al.* (2002) and Bouwman (1996) found that emissions from grasslands appear to be lower than those from croplands. Grasses take up nitrogen quickly and have a longer growing season than crops, which could lead to a higher uptake of nitrogen and less denitrification in grasslands than for annual crops (Bouwman, 1996).

We take account of differences in crop nitrogen uptake by disaggregating cultivation into three broad crop types: cereals, ley (here referring to fields sown with grass not used for grazing) and other crops. Furthermore, we approximate crop nitrogen uptake by multiplying estimated crop yield with a constant share of nitrogen in crop yield, since there is a strong linear relationship between total crop nitrogen and grain yield (Cassman *et al.* (2003) in Bouwman *et al.* (2002)). The residual nitrogen can then be seen as a function of the N application rate and the type of crop according to

$$z(r_j) = r_j - k_j f_j(r_j), \tag{3}$$

where r_j is the N application rate for crop type j, k_j is the share of nitrogen in crop yield and $f_j(r_j)$ is the yield function for crop type j .

Combining (2) and (3), we get emissions per hectare, e, with crop type j as a function of the N application rate for crop type j, r_j:

$$e(r_j) = \kappa + \psi \exp(\rho(r_j - k_j f_j(r_j))). \tag{4}$$

Figure 1 illustrates general relationships between the N application rate and crop N uptake, residual nitrogen and the N_2O emission rate for a concave yield function. Due to decreasing yield returns, marginal residual nitrogen is increasing in the rate of nitrogen application. But whether an increase in residual nitrogen results in increased emissions of N_2O depends on the level of residual nitrogen. Up to levels of 50 kg N per hectare, there is practically no change in the level of N_2O emissions according to the estimated relationship in (2), which is also reflected in the example in Figure 1, since emissions only start to increase for

N application rates of more than 100 kg ha^{-1}.

Using the emission function in (4), total emission can be approximated[6] by

$$E = \sum_j e(\bar{r}_j) A_j, \tag{5}$$

where A_j is the number of hectares cultivated with crop type j and $\bar{r}_j$ is the average application rate for crop type j. We use the relationship in (5) to find our second estimate of the impact of the N tax on N_2O emissions, and refer to this as the emission function approach.

3 Estimating the impact of the N tax on N_2O emissions

To estimate a counterfactual level of emissions using the emission factor, we need to know the price responsiveness of aggregate nitrogen demand. Aggregate nitrogen demand N can be seen as a function of the price of fertilizer nitrogen, η, availability of organic nitrogen (from animal manure and urea), M, as well as other input prices, $\boldsymbol{w}$, crop prices, $\boldsymbol{p}$ and total area of land cultivated, A, i.e., we can write $N(\eta, M, \boldsymbol{w}, \boldsymbol{p}, A)$. The price of fertilizer nitrogen is in turn a function of the nitrogen tax, τ, i.e., $\eta(\tau)$. Using the linear function in (1), a change in total emissions, dE, from a marginal change in the nitrogen tax, $d\tau$, can then be approximated by

$$dE = c_{N_2O} \frac{\partial N}{\partial \eta} \frac{\partial \eta}{\partial \tau} d\tau. \tag{6}$$

Here, we employ a short run demand function and assume that the nitrogen tax only impacts nitrogen demand directly through the price of nitrogen, given a fixed level of organic nitrogen as well as total area of arable land. In the simulations, we will also assume that a change in the nitrogen tax is fully transmitted to the buyers of fertilizer nitrogen, i.e., $\frac{\partial \eta}{\partial \tau} = 1$, which is an assumption largely supported by a report from the Swedish Agency for Public

[6] On the basis of (2), total emissions would be more precisely estimated by $E = A \int e(z) f(z) dz \approx \sum_j A_j \int e(r_j) g_j(r_j) dr_j$, with $f(z)$ being the density function representing the distribution of N surplus rates across all cropland A and $g_j(r_j)$ the density function representing the distribution of N application rates for crop type j across A_j. Owing to the convexity of the emissions function, the approximation made here of using the emission rate for the average application rate will be an underestimate of the average emissions rate. Furthermore, as pointed out by Van Groenigen *et al.* (2010), N_2O emission variability remains large even after accounting for the N application rate because of the wide variety of agroecosystems represented in their meta-analysis. Although the majority of the studies included were conducted in temperate climates, additional factors such as weather, crop residue quality, soil type and fertilizer type will also affect N_2O emissions. Due to limitations in the data, we cannot address this variation and the underlying assumption is that (2) approximates the average relationship across the different agri-ecological conditions for crop production in Sweden.

Management, see Statskontoret (2011:31).

In contrast, for finding counterfactual emissions for the non-linear relationship in (5), we need to consider that nitrogen use may change in response to a changed nitrogen price both on the intensive margin (changes in application rates) and on the extensive margin (changed allocation of land across crop types). In particular, we think of the (average) nitrogen application rate for crop type j, $\bar{r}_j$, as a function of the nitrogen price, availability of organic nitrogen per unit area of agricultural land, m, other input prices and the price of crop type j, p_j, i.e., $\bar{r}_j(\eta, m, \boldsymbol{w}, p_j)$. Similarly, the area of land allocated to crop type j, A_j, can be seen as a function of the nitrogen price, availability of organic nitrogen, M, other input prices, crop prices, and the total area of agricultural land, A, i.e., $A_j(\eta, M, \boldsymbol{w}, \boldsymbol{p}, A)$.

To illustrate the intensive versus extensive margin effects, we can use the fact that the change in total emissions, dE, as expressed in (5) is, to a first order approximation, given by

$$dE = \sum_j \Big[\frac{\partial e(\bar{r}_j)}{\partial \bar{r}_j}\frac{\partial \bar{r}_j}{\partial \eta}\frac{\partial \eta}{\partial \tau}A_j + e(\bar{r}_j)\frac{\partial A_j}{\partial \eta}\frac{\partial \eta}{\partial \tau}\Big]d\tau \qquad (7)$$

where A_j is the number of hectares cultivated with crop type j and $\bar{r}_j$ is the average application rate for crop type j.

Expression (7) illustrates the two price effects: the first on the intensive margin and the second on the extensive margin. The intensive margin effect comes from the effect of the tax-induced change in the price of N fertilizers on nitrogen application rates. The effect on the extensive margin comes from the potential effect of the N tax on the relative profitability of different crop types and the related changes in the allocation of land between crops. The separation requires an assumption of constant returns to scale with respect to land, which is in line with previous literature on non-point source pollution (see Claasen and Horan, 2001)[7]. We will test this assumption in the following section.

[7]Additionally, this separation into an extensive and an intensive margin effect is based on an assumption of input non-jointness, also used by Moore *et al.* (1994) in modeling water use in irrigated agriculture. Input non-jointness would here imply that nitrogen application rates are determined separately for cereals, ley and other crops. More details on deriving nitrogen use and land allocation functions based on these assumptions are provided in Appendix A.

4 Econometric analysis

4.1 Data

We use panel data on nitrogen sales and nitrogen use for the 21 counties of Sweden over the period 1989-2009. The data record annual sales of fertilizer nitrogen in each county. The average nitrogen application rate for the three crop categories (cereals, ley and other crops) is also available for every second year of cultivation. The application rates are the total of the nitrogen in synthetic fertilizer plus estimated mineralized nitrogen from applied manure and urea. The nitrogen data are aggregated over the crop seasons from June in one year to May the following year. For each year in the same period, we also have data on the acreage cultivated with cereals, ley and other crops.

For the main independent variable of interest, we use the real price of ammonium nitrate per kilo of nitrogen as a proxy for the nitrogen price. As proxies for variation in the prices of cereals, ley and other crops, we use real price indices for cereals, cattle and crops, respectively. Prices refer to calendar years, meaning that prices lag four months behind the crop season data on nitrogen use. Other input prices in the form of labor costs are approximated by the real wage for unskilled labor in agriculture. The additional control of organic nitrogen availability in each county was estimated from data on animal numbers. Descriptive statistics for all variables are presented in Table 2.

4.2 Econometric models and results

To find the price responsiveness of aggregate nitrogen demand to the nitrogen price, we estimated the following econometric model for the time period 1989-2009:

$$ln\, N_{it} = \gamma_0 + \gamma_\eta ln\, \eta_t + \gamma_M ln\, M_{it} + \gamma_w ln\, w_t + \gamma_p' \, ln\, p_t + \gamma_A ln\, A_{it} + \gamma_{time} time + \nu_i + \varepsilon_{it},$$

where N_{it} is sales of synthetic fertilizer nitrogen in county i in year t, η_t is the nitrogen price inclusive of the N tax in year t, M_{it} is the amount of organic nitrogen and A_{it} is the area of arable land in county i in year t, w_t is the farm labour wage and p_t is a vector of crop price indices in year t, *time* is a linear time trend and ν_i is a county fixed effect. Because all variables are log-transformed, the coefficients are directly interpretable as elasticities. Models were estimated with fixed effects[8] to capture the effects of time-invariant agricultural conditions

[8]Because the regular Hausman test for comparing fixed and random effects is invalid in the presence of cross-sectional dependence (Hoechle, 2007), we chose to use the fixed effects estimator, which is less restrictive and

that differ between counties and may affect nitrogen use.

Results are in Table 3. The estimate of the own-price elasticity of aggregate N sales is -0.27 and statistically significant at the 10%-level. The standard errors are adjusted to control for heteroscedasticity and cross-sectional and temporal dependence. This estimate is in line with elasticities from the previous study on aggregate Swedish nitrogen demand in Ingelsson & Drake (1998).

We estimated the following econometric model explaining the nitrogen application rate for each of the crop types[9] for every second year between 1989 and 2009:

$$ln\, \bar{r}_{j,it} = \beta_0 + \beta_\eta ln\, \eta_t + \beta_m ln\, m_{it} + \beta_w ln\, w_t + \beta_{p_j} ln\, p_{j,t} + \beta_{A_j} ln\, A_{j,it} + \beta_{time} time + v_{j,i} + \epsilon_{j,it},$$

where $\bar{r}_{j,it}$ is the average application rate of mineralized N and m_{it} is the amount of organic nitrogen per hectare in county i in year t, $p_{j,t}$ is the output price related to each crop type j in year t, $A_{j,it}$ is the acreage cultivated with crop type j in county i in year t and $v_{j,i}$ is the county fixed effect for crop type j. The logarithmic functional form[10] here is consistent in principle with a Cobb-Douglas production function. However, because we are using county-level data, we do not make any structural interpretation of our parameter estimates other than as point elasticities for the particular price ranges in the data.

Results for the N application rate for cereals, ley and other crops are found in column (1), (2) and (3), respectively, in Table 4. The elasticity with respect to the nitrogen price for cereals is -0.17. The coefficient is not statistically significantly different from zero with standard errors robust to autocorrelation and heteroscedasticity[11]. For ley, the estimate for the N price elasticity is close to zero (-0.021) and also not statistically significant[12]. The estimate for the N price elasticity for other crops is, in contrast, -0.45 and significant at the 10% level [13].

allows for correlation between the county effect and the regressors.

[9]For readability, we suppress in the following the subscript j for crop type on the parameters.

[10]The application rates, which are averages such that equal marginal effects in levels across counties of different size could still be reasonable, were also tested and gave comparable results.

[11]We reject both the absence of autocorrelation and heteroscedasticity as well as cross-county correlation. To correct for cross-county correlation, we also estimated the model with Driscoll & Kraay (1998) standard errors. The Driscoll-Kraay standard errors are, however, consistently smaller (potentially indicating unexpected negative cross-county correlation in the residuals), and we therefore present the more conservative robust standard errors.

[12]We cannot reject the null hypotheses of no autocorrelation and no cross-county correlation but we can reject the null hypotheses of no heteroscedasticity. We therefore present the results with standard errors robust to heteroscedasticity and autocorrelation.

[13]Again, we cannot reject the null hypotheses of no autocorrelation and no cross-county correlation but we can reject the null hypotheses of no heteroscedasticity and therefore present the results with standard errors robust to heteroscedasticity and autocorrelation.

To be consistent with our assumption of constant returns to scale with respect to land, the coefficient on the area of land allocated to crop type j, β_A, should not be significantly different from zero. For all three crop categories, the coefficient of crop acreage is small and not statistically significant, which indicates that we cannot reject the null hypothesis of constant returns to scale.

Similarly, to find the price responsiveness of the land allocation, we estimated the following econometric models for each of the crop types, cereals, ley and other crops for the years 1989-2009:

$$ln\,A_{j,it} = \alpha_0 + \alpha_\eta ln\,\eta_{t-1} + \alpha_M ln\,M_{it} + \alpha_w ln\,w_{t-1} + \alpha'_p\,ln\,p_{t-1} + \alpha_A ln\,A_{it} + \alpha_{time} time + \varsigma_{j,i} + \mu_{j,it}$$

where $\varsigma_{j,i}$ is the county fixed effect for crop type j. Prices are lagged one period because we assume that the land allocation decisions are taken early in the season with expectations about prices based on price levels in the previous year.

The results on land allocation are presented in Table 5. A priori, we would not expect the price of synthetic nitrogen to have a pronounced effect on the allocation of land to different crops since synthetic fertilizer nitrogen is a relatively small share of variable production costs for Swedish farmers. From column (1) in Table 5, we also see that the elasticity of cereals acreage with respect to the lagged N price is practically zero (-0.046) and not statistically significant[14], in line with this hypothesis. Similarly, in column (2) the estimate for the elasticity of ley acreage with respect to the lagged N price is 0.054 and also not statistically significant. The estimate in column (3) for the elasticity of the acreage planted with other crops with respect to the lagged N price is, on the other hand, larger (0.97) and statistically significant.

The land allocation models generally have coefficients of the expected signs for the respective crop prices and an acceptable level of explanatory power. In contrast, the models for the nitrogen application rates generally have few variables that are statistically significant. The coefficients on the nitrogen price for the application rates are negative, as expected, but the lack of statistical significance is possibly due to the relatively small number of observa-

[14]For all three crop categories, we reject the absence of autocorrelation and heteroscedasticity as well as cross-county correlation and therefore present the Driscoll & Kraay (1998) standard errors that are heteroscedasticity consistent and robust to very general forms of cross-sectional and temporal dependence (Hoechle, 2007).

tions, since data on the dependent variable is only available for every second year.

5 Simulations on the impact of the N tax

In this section, we assess the impact of the nitrogen tax on N_2O emissions during the past decade using the parameter estimates presented in the previous sections. We present the estimated difference in N_2O emissions with and without the N tax for the years of cultivation 1998/99, 2000/01, 2002/03, 2004/05, 2006/07 and 2008/09. We present results based on the emission factor for N_2O used in the Swedish GHG inventory reports as well as the emission function suggested by Van Groenigen *et al.* (2010).

For the emission function, residual nitrogen was estimated by using yield functions for representative crops for cereals, ley and other crops, respectively, and a constant share of nitrogen in yield. Owing to the uncertainties surrounding the parameter values for the emission function, we present the results based on the emission function as a range from a Monte Carlo simulation. Distribution moments for the parameters in the emission function were chosen with the help of an expert in soil N_2O emissions. We also took account of the imprecision with which the price elasticities for the N application rates and the land allocation were estimated and used the point estimates and standard errors as measures of the mean and standard deviation, respectively, for what we assume are normally distributed parameters. Further details on the simulation model and the assumptions are described in Appendices B and C.

5.1 Simulation results

The results on the difference in N_2O emissions with and without the N tax are summarized in Figure 2. The black line shows the difference in N_2O emissions with and without the N tax, estimated with the emission factor approach and a point estimate for the price elasticity of N sales of -0.27. The average estimated difference over the six years is 160 tonnes of N_2O with lower estimates for more recent years, owing to the fact that the real value of the tax has decreased over time. 160 tonnes of N_2O is 2% of the mean estimate of annual total direct N_2O emissions from agricultural soils over the period and equivalent to $50 \cdot 10^3$ tonnes of CO_2-equivalents[15].

[15] CO_2-equivalents are calculated by multiplying by 310 - the GWP of N_2O.

The dotted lines show the results from the Monte Carlo simulation using the emissions function. Mean estimates from the Monte Carlo simulation with the emissions function and the estimates with the emission factor approach are of the same magnitude. Over the six years, the average estimated mean difference in emissions is 160 tonnes of N_2O.

However, we also see that the range of values for the estimated difference in emissions is wide, with the 5th percentile being lower than 10 tonnes for all years, while the 95th percentile is as high as 780 tonnes N_2O for 2000/01. The main driver of the wide range is the parameter ρ in the exponential in the emission function. This parameter is highly uncertain. In discussion with an expert in soil N_2O emissions, it was set to vary between 0 and 0.06 with a mode at the point estimate of 0.04. The range between the 5th and the 95th percentile also varies significantly between years. One reason is the low price of synthetic nitrogen in the early years, which results in a larger percentage change in the price without the tax. On average, this implies larger changes in the estimate of residual nitrogen and hence larger differences in emissions for earlier years.

6 Discussion and conclusions

The objective of this study was to assess the effect of the Swedish nitrogen tax on direct N_2O emissions from agricultural soils. The results suggest that, on average over the last decade, annual direct N_2O emissions from agricultural soils in Sweden would have been 160 tonnes N_2O or $50 \cdot 10^3$ tonnes of CO_2-equivalents higher without the N tax. This is 2% of the mean estimate of annual direct N_2O emissions from agricultural soils over the analyzed period.

However, much uncertainty remains around the relationship between nitrogen fertilization and soil N_2O emissions. The Monte Carlo results indicate that the abolishment of the nitrogen tax could potentially result in significant increases in N_2O emissions if the relationship is a convex one. The reason is that marginal increases in residual nitrogen could lead to a more than proportional increase in N_2O emissions. However, we see from the wide range for the estimates of the policy impact that insignificant effects on N_2O emissions are also possible.

The small difference in the mean estimates of the tax impact using the emission factor compared to the emission function approach is due to the low estimates of residual nitrogen in Swedish agriculture. Swedish farmers are already using fertilizers quite efficiently and the nitrogen use efficiency[16] has increased from an average of 55% in 1995 to 70% in 2009 (SCB, 2011). This may be partly due to the longstanding nitrogen tax and the information and advisory services on efficient use of nitrogen and phosphorus, which the tax revenues have helped to finance.

As mentioned in the introduction, when the Swedish government removed the N tax, it also increased the CO_2 tax in the agricultural sector[17]. A more stringent climate policy for the agricultural sector may thereby have substituted for a non-climate policy that had the effect of reducing GHGs. This results in an ambiguous net effect on total GHG emissions from Swedish agriculture. It is therefore interesting to assess the net effect on GHG emissions of the Swedish policy shift from a nitrogen tax to an increased CO_2 tax on diesel in agriculture. According to our back-of-the-envelope calculations[18], a CO_2 tax increase in 2007 equivalent to the planned change in the CO_2 tax for the agricultural sector would have implied a reduction in CO_2 emissions on the order of $30 \cdot 10^3$ tonnes of CO_2. In comparison, the mean estimate of the increase in N_2O emission from removing the N tax in the cultivation year 2006/2007 is 170 tonnes of N_2O or $52 \cdot 10^3$ tonnes of CO_2-equivalents. It therefore appears that the removal of the N tax has the potential to fully offset the CO_2 emission reductions that can be achieved with the planned increase in the CO_2 tax for Swedish agriculture.

In practice, the removal of the N tax and the increase in the CO_2 tax have relatively small impacts on national GHG emissions[19]. However, both taxes are likely to provide other environmental benefits - the N tax in the form of reduced water pollution and the tax on diesel in the form of reduced emissions of particulate matter and NO_x gases, inter alia. It therefore seems unfortunate if, for political reasons, one of these environmental taxes was removed to compensate for increasing another.

Still, a uniform tax on nitrogen is far from being the theoretically optimal policy for dealing with agricultural pollution. This is because the contribution of a marginal unit of nitrogen to soil nitrogen surplus and subsequent pollution will vary by crop, but also by time

[16] Efficiency is measured as the amount of nitrogen in crop yield as a share of total nitrogen additions to soils.

[17] More specifically, the generous deductions on the CO_2 tax on diesel for farmers are being reduced gradually in three steps, to be completed by 2015 (Proposition, 2009/10:41).

[18] Following Hammar & Sjöström (2011), we use a long-run own price elasticity of -0.2 for diesel use in agricultural machinery. In 2007, diesel use in Swedish agriculture amounted to $3 \cdot 10^5$ m^3. We use the actual 2007 diesel use and price as baseline. For simplicity, we change the deduction on the CO_2 tax from the actual 77 % to 0.90 SEK per liter (the planned nominal deduction in 2015), which may lead to an underestimate of the impact.

[19] Total national GHG emissions in 2009 were 59.8 million tonnes of CO_2-equivalents, excluding land use change and forestry activities. A total of 8.2 million tonnes of CO_2-equivalents came from the agricultural sector in the form of nitrous oxide as well as methane (SEPA, 2011).

and location specific conditions that affect crop yield as well as nutrient runoff and denitrification. However, in the long run, a nitrogen tax should provide benefits in terms of reduced soil nitrogen surpluses and thereby both lower N_2O emissions and less nitrogen runoff to lakes and rivers, albeit at a cost of slightly reduced yields.

Globally, more than half of the fertilizer nitrogen is currently lost by denitrification, volatilization as ammonia and leaching of nitrate. All of these pathways lead to either direct or indirect emissions of N_2O (Smith, 2010a). To reduce the contribution from agriculture to climate change, eutrophication and other environmental problems related to excess nitrogen loads, policy makers need to find ways to increase the efficiency of fertilizer use. Although the direct impacts of the nitrogen tax are difficult to identify, Swedish farmers are using nitrogen more efficiently than before. The Swedish policy package, consisting of information on efficient use of fertilizers and a tax that may have helped to raise awareness, may therefore still be an alternative to consider in other countries which face problems of excessive fertilizer use and nutrient pollution.

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Figures and Tables

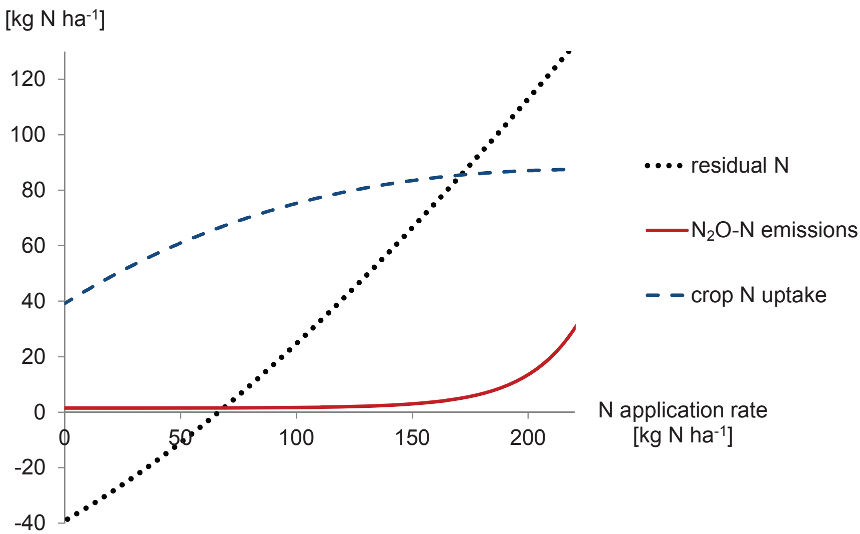

Figure 1: Crop N uptake, residual N and N_2O emissions as functions of the nitrogen application rate

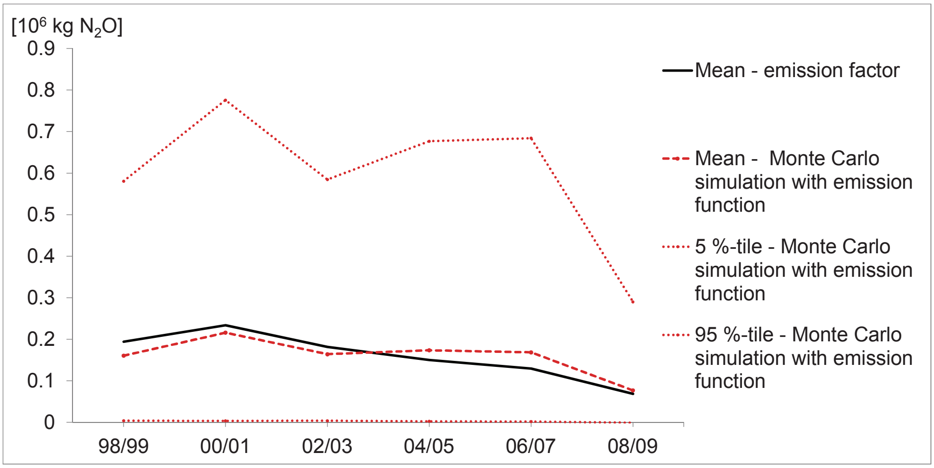

Figure 2: Simulation results on the difference in N_2O emissions with and without the N tax. Dotted lines show results from a Monte Carlo simulation with the emissions function. The black line shows results estimated with the emission factor approach.

Table 1: Emission function parameters

$e(z) = \kappa + \psi \exp(\rho z)$	
κ [kg N_2O-N· ha^{-1} year^{-1}]	1.44
ψ [kg N_2O-N· ha^{-1} year^{-1}]	0.08
ρ [ha·year·(kg N)$^{-1}$]	0.04

Table 2: Descriptive statistics

Variable	Mean	Std. Dev.	Min.	Max.	Obs
sales of fertilizer nitrogen [10^6 kg]	8.93	11.74	0.4	56.3	439
average N app rate, cereals [kg ha^{-1}]	86.84	20.72	35	136	203
average N app rate, ley [kg ha^{-1}]	99.14	19.93	54	149	200
average N app rate, other crops [kg ha^{-1}]	81.66	23.07	40	137.71	178
cereals acreage [10^3 ha]	54.71	61.63	2.22	261.65	462
grass acreage [10^3 ha]	49.01	33.15	13.19	194.21	462
acreage with other crops [10^3 ha]	13.44	23.46	0.56	135.68	462
total arable land acreage [10^3 ha]	129.98	119.52	31.58	507.54	462
total organic N [10^6 kg]	8.91	8.74	1.97	39.58	441
organic N per hectare [kg ha^{-1}]	71.16	30.41	27.26	138.96	441
price of nitrogen [SEK (kg $N)^{-1}$, ref year 2010]	8.85	1.62	6.57	13.98	21
cereal real price index	156.14	41.77	108	266	21
cattle real price index	253.28	73.81	172.9	452.3	21
crops real price index	188.9	25.16	162	261	21
farm labour wage [SEK $hour^{-1}$, ref year 2010]	108.71	11.85	91	128	21

Table 3: Sales of nitrogen in synthetic fertilizers

	(1)
Price of nitrogen	-0.269**
	(0.101)
Organic N	-0.255
	(0.217)
Price of labour	0.694
	(0.616)
Cereal price index	0.417**
	(0.174)
Cattle price index	0.0188
	(0.134)
Crops price index	-0.584**
	(0.274)
Total arable land	1.488**
	(0.373)
Time trend	-0.0207*
	(0.0102)
Constant	-15.46**
	(5.135)
Observations	439
R^2	0.486

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$
All variables are log transformed.
Model estimated with county fixed effects and Driscoll-Kraay standard errors.

Table 4: N application rates

	(1) Cereals	(2) Ley	(3) Other crops
Price of nitrogen	-0.165	-0.0209	-0.446*
	(0.107)	(0.135)	(0.214)
Organic N per hectare	0.124	-0.0399	-0.00306
	(0.136)	(0.163)	(0.197)
Price of labour	2.163**	-3.615**	3.490*
	(0.677)	(1.057)	(1.771)
Cereal price index	0.121		
	(0.110)		
Cattle price index		0.0219	
		(0.129)	
Crops price index		0.149	1.266**
		(0.275)	(0.510)
Cereals acreage	-0.0410		
	(0.100)		
Ley acreage		-0.266	
		(0.171)	
Acreage-other crops			0.0155
			(0.0341)
Time trend	-0.0243**	0.0689**	-0.0478
	(0.00834)	(0.0183)	(0.0282)
Constant	-3.292	22.38**	-15.18*
	(2.594)	(4.984)	(7.741)
Observations	203	200	178
R^2	0.253	0.159	0.059

Standard errors in parentheses

* $p < 0.10$, ** $p < 0.05$

All variables are log transformed.

Models estimated with county fixed effects with robust standard errors.

Table 5: Land allocation

	(1) Cereals acreage	(2) Ley acreage	(3) Acreage -other crops
Price of nitrogen, t-1	-0.0457	0.0537	0.966**
	(0.146)	(0.0892)	(0.344)
Organic N	-0.204**	0.185**	0.321
	(0.0643)	(0.0620)	(0.257)
Price of labour, t-1	0.416	-0.698**	2.952**
	(0.443)	(0.288)	(1.069)
Cereals price index, t-1	0.237*	0.293**	-0.361
	(0.131)	(0.0898)	(0.348)
Cattle price index, t-1	-0.522**	0.372**	-0.184
	(0.0738)	(0.0496)	(0.309)
Crops price index, t-1	0.0709	-0.598**	0.117
	(0.235)	(0.103)	(0.853)
Total arable land	0.919**	1.163**	-0.228
	(0.255)	(0.190)	(0.902)
Time trend	-0.0347**	0.0376**	-0.0747**
	(0.00653)	(0.00482)	(0.0247)
Constant	1.447	-1.847	-6.045
	(4.510)	(3.030)	(11.58)
Observations	420	420	420
R^2	0.534	0.592	0.240

Standard errors in parentheses.

* $p < 0.10$, ** $p < 0.05$

All variables are log transformed.

Models estimated with county fixed effects and Driscoll-Kraay standard errors.

Appendices

A Deriving nitrogen demand and land allocation functions

To derive functions for nitrogen use and land allocation at the farm level, we follow Moore *et al.* (1994) in modeling the production decisions of a multi-output competitive farm. Moore *et al.* (1994)'s assumption of input non-jointness makes it possible to define separable crop-level profit functions.

Land and organic fertilizer nitrogen are assumed to be fixed inputs which are allocated across crop types to maximize total short-run profits over all crops. At the beginning of each crop season, farmers choose the levels of the variable inputs for each crop type to maximize crop-level profits based on crop and input prices in the previous year and a given allocation of land. Next, farmers allocate the land across the different crop types to maximize total short-run profits over all crops. Later in the season, when the land allocation is fixed across crops, the farmers can update their nitrogen input decision based on current prices.

With these assumptions, the crop-level profit functions can be written $\Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j,)$, where $\mathbf{w}$ is a vector of prices for inputs other than synthetic fertilizer nitrogen, p_j is the price of crop type j and M_j and A_j are the organic nitrogen and area of land allocated to crop j, respectively.

The decision on land and organic nitrogen allocation can then be written

$$\Pi(\eta, \mathbf{w}, \mathbf{p}, M, A) = \max\{\sum_j \Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j) : \sum_{j=1}^{J} A_j = A | \sum_{j=1}^{J} M_j = M\}$$

The $2J$ first-order conditions take the forms $-\frac{\partial \Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j)}{\partial A_j} = \lambda_A$ and $-\frac{\partial \Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j)}{\partial M_j} = \lambda_M$ where λ_A and λ_M are the shadow prices of land and organic fertilizer nitrogen respectively. The system consisting of the first-order conditions and the land constraint implicitly defines the optimal land allocation $A_j^*(\eta, \mathbf{w}, \mathbf{p}, M, A)$ over all crops j as function of the prices, the availability of organic nitrogen and the total acreage of arable land.

Short-run demand for synthetic fertilizer nitrogen for crop j follows from Hotelling's lemma applied to $\Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j)$:

$$-\frac{\partial \Pi_j(\eta, \mathbf{w}, p_j, M_j, A_j)}{\partial \eta} = N_j^*(\eta, \mathbf{w}, p_j, M_j, A_j) \qquad \forall\, j$$

The average nitrogen application rate for each crop j, $\bar{r}_j$, is then given by

$$\bar{r}_j = \frac{N_j^*(\eta, \mathbf{w}, p_j, M_j, A_j)}{A_j} \qquad \forall j$$

If the production function for each crop has constant returns to scale with respect to land, then the optimal average application rate is only a function of the input and crop prices and the organic nitrogen allocated to crop j, i.e., $\bar{r}_j(\eta, \mathbf{w}, p_j, m_j)$.

Because we use aggregate yearly county-level data to estimate the input demand functions, we are assuming that the sign of the price elasticities which are consistent with profit-maximizing behaviour at farm-level will also apply in the aggregate. Furthermore, we use the average organic nitrogen application rate over all crops, i.e., $\bar{m}$ instead of m_j, because we lack the necessary information on the distribution of organic nitrogen across different crops.

B Estimating emissions from land and N use in previous years

For each year of cultivation (1998/99, 2000/01, 2002/03, 2004/05, 2006/07 and 2008/09), we used the country-specific emission factors used by the Swedish Environmental Protection Agency in the national GHG inventory reports to find a first estimate for total direct N_2O emissions. The emission factors are 0.8% N_2O-N of the synthetic N added to soil and 2.5% N_2O-N of added N from animal manure. The background emission factor for mineral soils is 0.5 kg N_2O-N per hectare per year (SEPA, 2011). For 2008/09, the estimate of total direct emissions using these emissions factors is $8.0 \cdot 10^3$ tonnes N_2O. The total estimate was found by multiplying total sales of synthetic nitrogen and our estimates of total additions of organic N by their respective emission factors and adding these to the background emission factor multiplied by the area of agricultural soils. In the official national inventory figures on direct N_2O emissions from agricultural soils, a number of additional adjustments are made related to, inter alia, nitrogen lost as ammonia (SEPA, 2011).

For the same six years of cultivation, we also derived an estimate of residual nitrogen to be used with the emission function. We used data on the average application rate for each crop category and assumed functions for crop N uptake, in line with equation (3). For ley and other crops, we used yield functions for representative crops[20] of the form

[20]For ley, we chose a function for fourth round ley which produced yield estimates consistent with official average yield data for ley in SCB (2009). The category of other crops is dominated in terms of acreage by oil seed crops but also includes a significant share of sugar beet and potato. The choice of a representative crop is not obvious, as the production function should reflect the average nitrogen uptake for such a broad category of crops. We chose a less concave yield function representing spring rapeseed. The high value of the parameter

$f_j(r) = a_j + b_j r + c_j r^2 + d_j r^3$ from the Swedish Board of Agriculture (Jordbruksverket, 2011). Parameter estimates for the average yield functions are found in Table 6, which also includes the assumed shares of nitrogen in crop yields for ley and other crops taken from Claesson & Steineck (1991). For cereals, we instead used a function that directly represents the relationship between N fertilization and the nitrogen in spring barley grain yield[21].

Based on the estimates of residual nitrogen for each crop type, we found an emission rate per hectare for each crop type from the emission function in equation (2). This should be an underestimate of the average emissions rate due to the convexity of the emission function. Table 7 shows results from the calculations for the cultivation year 2008/09. The average emission rate over all agricultural land is estimated as 1.6 kg N_2O-N per hectare and the estimate of total direct N_2O emissions over all cultivated cropland is $8.1 \cdot 10^3$ tonnes N_2O.

Our total estimates are in line with official emission figures using both the emission factor and the emissions function approach. The official figure on direct N_2O emissions from agricultural soils was, for example, 7.8 10^3 tonnes N_2O (SEPA, 2011) for 2009, compared with our estimates for the cultivation year 2008/09 of $8.0 \cdot 10^3$ tonnes N_2O with the emission factors and $8.1 \cdot 10^3$ tonnes N_2O with the emission function. The magnitude of our total estimates therefore seems reasonable for both the emission factor and the emission function approach.

C Simulations - estimating emissions from counterfactual land and N use

For our first estimate of the impact of the N tax on N_2O emissions, the counterfactual national sales of nitrogen (i.e., the national sales without a tax on synthetic fertilizer nitrogen) in year t without the N tax for each year was calculated according to

$$N_t^c = N_t \Big(\frac{\eta_t - \tau}{\eta_t} \Big)^{\hat{\gamma}_\eta}.$$

Assuming that the use of nitrogen from animal manure is completely inelastic to the price of synthetic N, we estimated the difference in N_2O emissions by multiplying the estimated difference in synthetic N sales with the emission factor for synthetic fertilizer nitrogen of

indicating rapeseed yield at a zero level of N fertilization was reduced and calibrated to be consistent with our own estimate of the average N uptake for the mix of crops included in the category of other crops.

[21] The function is taken from Johnsson *et al.* (2006) and is estimated for spring barley grown in southern Götaland, and matches well our own estimates of the average N uptake for cereals.

0.8% N_2O-N.

For our second estimate of the impact of the N tax, we calculated the counterfactual nitrogen application rate for crop type j in year t, $\bar{r}^c_{j,t}$, from the actual national average nitrogen application rate in year t, $\bar{r}_{j,t}$, as

$$\bar{r}^c_{j,t} = \bar{r}_{j,t}\Big(\frac{\eta_t - \tau}{\eta_t}\Big)^{\hat{\beta}_{j,\eta}}.$$

Similarly, we calculated the counterfactual land allocations in year t, $A^c_{j,t}$, from the actual national land allocations in year t, $A_{j,t}$, as

$$A^c_{j,t} = A_{j,t}\Big(\frac{\eta_{t-1} - \tau}{\eta_{t-1}}\Big)^{\hat{\alpha}_{j,\eta}}.$$

Because of lack of data, we assumed that the area of land for grazing is inelastic to the price of synthetic nitrogen. For the acreage of fallow and unfertilized land, we imposed a correction for the interdependency in land allocation by matching a decrease in cultivated acreage by an increase in unfertilized and fallow land of the same size and vice versa, such that the total area of agricultural land is unchanged in the counterfactual.

For the Monte Carlo analysis, we checked the sensitivity of our main outcome variable - the difference in N_2O emissions with and without the N tax - to variations in individual parameters. From this check, we chose to randomly vary the emission parameters ψ and ρ and the N price elasticities of the N application rates and land allocations. We also randomly varied the concavity parameters c_j in the production function for the three different crop categories because the value of this parameter is relatively uncertain and impacts the change in crop N uptake in the counterfactual.

Distributional assumptions used in the Monte Carlo simulation are presented in Table 8. We assumed that the price elasticities are normally distributed and used the point estimtes and standard errors from the econometric results as estimates of the mean and standard deviation, respectively. We truncated the maximum value of the price elasticities for the application rates at 0 because a positive value counters economic intuition. The distribution moments for the emission parameters ψ and ρ were chosen in discussion with an expert in soil N_2O emissions. For the concavity parameters, we assumed minimum and maximum values, which generated estimates of crop nitrogen uptake in a range consistent with available data.

Because parameter values are independently and randomly drawn for each run, the underlying assumption behind our Monte Carlo analysis is that the parameters are independently distributed. This is presumably not true, e.g., for the price elasticities, and should be kept in mind when evaluating the results. Furthermore, there are, of course, additional uncertainties regarding the structural model assumptions that are not accounted for in the Monte Carlo analysis. However, the Monte Carlo simulation should still be a useful illustration of some of the uncertainty relevant to our analysis.

Table 6: Assumed representative yield functions for cereals, ley and other crops.

	Cereals	Ley	Other crops
representative crop	*spring barley, south*	*4th round ley*	*spring rapeseed*
a	46	2950	1600
b	0.535	25.4	19
c	-0.0129	-0.021	-0.07
d	0	0	0.0001
k - share of N in yield	100%	2.5%	3.5%

Yield as a function of the N application rate in the form $f_j(r) = a_j + b_j r + c_j r^2 + d_j r^3$. The yield for cereals is shown in terms of nitrogen in grain yield.

Table 7: Estimation of total N_2O emission from agricultural land for the cultivation year 2008/09 based on the emission function approach.

	Cereals	Ley	Other crops	Other fields	All fields
average N application [kg N ha^{-1}]	*123*	*164*	*128*		
synthetic fertilizer N [kg N ha^{-1}]	*92*	*69*	*86*		51
organic N [kg N ha^{-1}]	*31*	*95*	*42*		32
mineralized N addition [kg N ha^{-1}], r	*105*	*113*	*105*		
yield [kg ha^{-1}] for average r	n/a	5552	2472		
crop N uptake [kg N ha^{-1}]	88	139	87		
residual N [kg N ha^{-1}], z	35	25	41	0	20
emission rate [kg N_2O-N ha^{-1}], e	1.8	1.7	1.9	1.5	1.7
acreage [10^6 ha], A	*1.0*	*0.6*	*0.3*	*1.2*	3.0
Total emissions [10^6 kg N_2O], E	2.8	1.5	0.9	2.8	8.1

Data from Statistics Sweden are in italics.
Yield estimates are based on data on average mineralized N additions, which include the nitrogen in synthetic fertilizer as well as the (estimated) mineralized share of added organic nitrogen. The nitrogen surplus is estimated as the difference between the total of applied synthetic and organic nitrogen and crop N uptake.
Other fields refers to fallow land, unfertilized soils and soils used for grazing.
We assume that the nitrogen surplus rate is zero for fallow land, unfertilized soils and soils used for grazing. Estimates of total emissions still include background emissions from these soils at the level of 1.5 $(\kappa + \psi)$ kg N_2O-N per hectare.
The estimate of the average nitrogen surplus in Swedish agricultural land is 20 kg N per hectare. This is low compared with official estimates of 32 kg N per hectare for 2009 (SCB, 2011). However, the official figures on Swedish soil N balances are not directly comparable to our estimates since they include additional sources of N such as atmospheric deposition and nitrogen fixing crops.

Table 8: Assumed distributions of parameters for Monte Carlo simulation.

Parameter	Distribution	Mean	St.dev	Min	Max	Mode
ψ	normal	0.08	0.03			
ρ	triangular			0	0.06	0.04
β_{w_1} - cereals	normal	-0.165	0.107		0	
β_{w_1} - ley	normal	-0.021	0.135		0	
β_{w_1} - other crops	normal	-0.446	0.214		0	
α_{w_1} - cereals	normal	-0.046	0.146			
α_{w_1} - ley	normal	0.054	0.089			
α_{w_1} - other crops	normal	0.966	0.344			
c - cereals	triangular			-0.01	0	-0.0013
c - ley	triangular			-0.1	0	- 0.021
c - other crops	triangular			-0.1	0	-0.052

c refers to the parameter in the relevant yield function of the form $f_j(r) = a_j + b_j r + c_j r^2 + d_j r^3$.

Paper III

On Refunding of Emission Taxes and Technology Diffusion*

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Abstract

We analyze diffusion of an abatement technology under a standard emission tax compared to an emission tax which is refunded in proportion to output market share. The results indicate that refunding can speed up diffusion if firms do not strategically influence the size of the refund. If they do, it is ambiguous whether diffusion is slower or faster than under a non-refunded emission tax. Moreover, it is ambiguous whether refunding continues over time to provide larger incentives for technological upgrading than a non-refunded emission tax, since the effects of refunding dissipate as the overall industry becomes cleaner.

Keywords: emission tax, refund, abatement technology, technology diffusion, imperfect competition

JEL Classification: H23, O33, O38, Q52

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1 Introduction

From a welfare point of view, the optimal rate of adoption of environmentally friendly technologies should balance the investment costs against the benefits of adoption in terms of reduced environmental damages and lower abatement costs. Nevertheless, the interplay of technology and environmental market failures implies that markets often underinvest in new technology. It is unlikely that environmental policy alone creates sufficient incentives for technological change - strengthening the case for second-best policies.

In theory, a strong and stable price of emissions implemented through an emission tax should induce both investment in R&D and a "cost-effective" allocation among firms of the burden of achieving given levels of environmental protection. In reality, however, introducing such an emission tax may prove politically infeasible since regulated firms will often argue that they will lose international competitiveness. As well as job losses if firms relocate or close, an additional concern is the relocation of pollution, or so-called emission leakage in the case of transboundary pollution such as greenhouse gas emissions.

One potential way of making emission taxes more politically feasible is to refund the tax revenues to the regulated industry (Hagem *et al.*, 2012; Aidt, 2010; Fredriksson & Sterner, 2005). One method for such refunding is to refund the revenues in proportion to the output market share. This is the approach that Swedish policy makers used in 1992 when introducing a charge on NO_x emissions from large combustion plants. The policy was explicitly intended to affect technology adoption. The refunding scheme enabled the introduction of an emission charge sufficiently high to induce abatement (Sterner & Höglund-Isaksson, 2006). This tax and refunding scheme, sometimes referred to as refunded emission payment (REP), has been extensively studied in the theoretical literature concerning the incentives for emission abatement and production and how it compares to optimal policy; see e.g., Fischer (2011), Cato (2010), Montero (2008), Sterner & Höglund-Isaksson (2006) and Gersbach & Requate (2004)[1]. From the empirical side, Sterner & Turnheim (2009) study the effects of the

[1] Gersbach & Requate (2004) and Sterner & Höglund-Isaksson (2006) analyze the incentives for abatement and production provided by an output based refunding scheme in markets characterized by imperfect and perfect competition, respectively. Cato (2010) studies the effects of refunding on market structure, showing that a refunding system might have to be complemented with an entry license to ensure that the system does not encourage too much market entry. Fischer (2011) studies the performance of refunding schemes when firms can strategically influence the size of the refund; since firms know that part of any emissions rents they create will be returned to them, refunding discourages large firms from abating emissions and subsidizes high emitters to a greater extent. Notably, Montero (2008) studies the use of refunds for inducing firms to reveal their private valuation of common pool resources. He propose a mechanism that builds upon a conventional uniform-price

Swedish refunded charge on NO_x emissions. Their results indicate that the charge had a very substantial role in explaining the sharp decrease in NO_x emission intensities; not only did the best plants make rapid progress in emission reductions, but there was also considerable catching up, such that today the majority of plants have lowered their emission intensities much more relative to the cleanest plants.

In this paper, we model the pattern of adoption of environmentally friendly technologies under a "standard" emission tax (hereinafter, emission tax) and an emission tax for which the revenues are returned to the aggregate of taxed firms in proportion to output (hereinafter, refunded tax). We consider the case of exogenous refunding, where firms take the size of the refund as given, vis-a-vis endogenous refunding, where firms recognize that a share of their emissions tax payments will be returned to them[2]. To the best of our knowledge, despite a growing body of literature analyzing the incentives for technological diffusion provided by different environmental policy instruments (see for instance van Soest (2005) and Coria (2009)), this is the first study investigating the effects of refunding an emission tax.

Like Coria (2009), our setting makes use of the framework by Reinganum (1981), who considers an industry composed of symmetric firms that engage in Cournot competition in the output market. When a technology that reduces the cost of compliance with an emission tax appears, each firm must decide when to adopt it, based in part upon the discounted cost of implementing it and in part upon the behavior of the rival firms. If a firm adopts a technology before its rivals, it can expect to make substantial profits at the expense of the other firms, since the cost advantage allows it to increase its output market share. On the other hand, the discounted sum of purchase price and adjustment costs may decline if the adjustment period lengthens, as various quasi-fixed factors become adjustable. Therefore, although waiting costs more in terms of forgone profits, it may save money on purchasing the new technology. Reinganum (1981) showed that diffusion, as opposed to immediate adoption, occurred purely due to strategic behavior in the output market, since adoptions that yield lower incremental benefits are deferred until they are justified by lower adoption costs.

Our results indicate that exogenous refunding of an emission tax based on output re-

sealed-bid auction. Part of the auction revenues are returned to firms, not as lump sum transfers but in a way that firms would have incentives to bid truthfully.

[2]Fischer (2011) refers to exogenous refunding as "fixed subsidy", and to an emission tax with an endogenous output-based rebate as the "refunded tax".

inforces the mechanism described by Reinganum (1981). Hence, technology diffuses faster into an imperfectly competitive industry if the regulator refunds the emission tax revenues but the firms do not recognize the impact of adoption on the average emission intensity. The intuition behind this result is straightforward: if the refund is based on output, adopters receive a net refund as the system rewards those firms that are cleaner than average. However, the incremental effect of the refund over taxes decreases as more and more firms adopt because of the lower overall pollution intensity and thus lower refund.

The paper is organized as follows. Section 2 introduces the model of technological diffusion. Section 3 and 4 analyze the adoption incentives provided by emission taxes with and without refunding, respectively. Section 5 analyzes technological catching up under the two policies. Section 6 presents numerical simulations and section 7 concludes.

2 The model

Assume an imperfectly competitive and stationary industry, where n firms choose their level of production simultaneously and compete in quantities. The inverse demand function is given by

$$P(Q) = a - bQ,$$

where $Q = \sum_{i=1}^{n} q^i$ and $a, b > 0$. The production technology exhibits constant returns to scale such that total variable costs are given by

$$C^i = c_0 q^i.$$

Production also generates emissions of a homogenous pollutant and emissions from firm i. e^i, are proportional to output q^i according to

$$e^i = \varepsilon_0 q^i.$$

To control emissions, the regulator has implemented a tax σ that each firm must pay for each unit of emission.

At date $t = 0$, an innovation in emissions abatement technology is announced. The new technology reduces the emission intensity from ε_0 to ε_1, i.e. $\varepsilon_1 < \varepsilon_0$, and also changes the marginal cost of production from c_0 to c_1[3]. Firms must now decide when to adopt the new

[3] As noted by Fischer (2011), this characterization is suitable for end-of-pipe technologies which scrub a certain

technology, taking into account the effect of the competitors' adoption on pre- and post-adoption profit flows. Note that $c_0 + \sigma\varepsilon_0 > c_1 + \sigma\varepsilon_1$ by assumption to ensure that the rate of profit flow (quasi-rent) is higher with the new technology. Moreover, we assume that no future technical advance is anticipated.

Let $\pi_0(m_1)$ be the rate of (Cournot-Nash) profit flow for firm i when m_1 out of n firms have adopted the cleaner technology and firm i has not. Next, let $\pi_1(m_1)$ be the rate of profit flow for firm i when m_1 firms have adopted the cleaner technology and firm i is among them. We assume that both $\pi_0(m_1)$ and $\pi_1(m_1)$ are known with certainty for all m_1.

Further, the following assumptions are made.

(1i) $\pi_0(m_1 - 1) \geq 0$ and $\pi_1(m_1) \geq 0$

(1ii) $\pi_1(m_1 - 1) - \pi_0(m_1 - 2) > \pi_1(m_1) - \pi_0(m_1 - 1) > 0$ for all $m_1 \leq n$.

Assumption (1ii) states that the increase in the profit rate from adopting as the $(m_1 - 1)$th firm should be higher than the increase in profit rate from adopting as the m_1th firm. This is to say, a firm that adopts earlier has a larger "relative" cost advantage than if it adopts later due to the strategic interaction in the output market.

Let τ_i denote firm i's date of adoption and let $p_1(\tau_i)$ be the present value of the investment cost for the new technology, including both purchase price and adjustment costs. We assume that $p_1(t)$ is a differentiable convex function with $p_1'(0) \leq \pi_0(0) - \pi_1(1)$ (2i), $\lim_{t\longrightarrow\infty} p_1'(t) > 0$ (2ii) and $p_1''(t) > re^{-rt}(\pi_1(1) - \pi_0(0))$ (2iii). Assumption (2i) ensures that immediate adoption is too costly, while assumption 2(ii) ensures that the costs of adoption decrease over time, but do not decrease indefinitely. This implies that there is an efficient scale of adjustment beyond which adoption costs increase again. Moreover, assumption 2(iii) ensures that the objective function defining the optimal timing of adoption is locally concave on the choice of adoption dates.

Further, we define $V^i(\tau_1, ..., \tau_{i-1}, \tau_i, \tau_{i+1}, ..., \tau_n)$ to be the present value of firm i's profits net of any investment costs for the new technology when firm k adopts at τ_k, $k = 1, .., n$. Given an ordering of adoption dates $\tau_1 \leq \tau_2 \leq ... \leq \tau_n$, we can write the present value of firm i's profits as

proportion of emissions. It is also a good representation of a technology that improves fuel efficiency, which means that it reduces emissions per unit of electricity or useful heat of pollutants, which are highly correlated with fuel use (such as CO_2 and SO_2).

$$V^i(\tau_1, ..., \tau_{i-1}, \tau_i, \tau_{i+1}, ..., \tau_n) = \sum_{m_1=0}^{i-1} \int_{\tau_{m_1}}^{\tau_{m_1+1}} \pi_0(m_1)e^{-rt}dt + \sum_{m_1=i}^{n} \int_{\tau_{m_1}}^{\tau_{m_1+1}} \pi_1(m_1)e^{-rt}dt - p_1(\tau_i),$$

where $\tau_0 = 0$ and $\tau_{n+1} = \infty$.

Maximization of V^i given the ordering $\tau_1 \leq \tau_2 \leq ... \leq \tau_n$ (and thus the restriction $\tau_{i-1} \leq \tau_i^* \leq \tau_{i+1}$) gives each firm i an optimal date of adoption, τ_i^*, and is implicitly defined by

$$\frac{\partial V^i}{\partial \tau_i} = (\pi_0(i-1) - \pi_1(i))\, e^{-r\tau_i^*} - p_1'(\tau_i^*) = 0. \tag{1}$$

This first-order condition says that it is optimal to adopt the new technology on the date when the present value of the cost of waiting to adopt (the increase in profit rate due to adoption) is equal to the present value of the benefit of waiting to adopt (the decrease in investment cost). We define $\Delta\pi_i = \pi_1(i) - \pi_0(i-1)$ and (1) can then be written

$$\frac{\partial V^i}{\partial \tau_i} = -\Delta\pi_i e^{-r\tau_i^*} - p_1'(\tau_i^*) = 0,$$

$i = 1, ..., n$. Furthermore, V^i is strictly concave at τ_i^* for all i. As shown by Reinganum (1981), there are $n!$ sequences in which the adoption date defined by (1) is a Nash equilibrium (demonstration in Appendix A). This result holds regardless of firms being homogenous when the adoption decision is made at time 0.

To further encourage adoption of new abatement technologies, the regulator has considered refunding the emission tax revenues to the firms in proportion to market share. In the following sections, we characterize one of the $n!$ sequences of adoption, analyzing the impact of refunding on the optimal date of adoption. That is, we analyze the difference in adoption profits $\Delta\pi_i$ between a standard emission tax and an emission tax refunded in proportion to output. A higher $\Delta\pi_i$ implies faster adoption (a lower τ_i^*) because of the concavity of $V^i(\tau_i^*)$ and vice versa.

3 Adoption incentives under an emission tax

If we have m_1 adopters of the new technology and rank the firms according to their order in the adoption sequence (taking it as given), we can write the profit rate maximization problem for the adopters as

$$\pi^j = \max_{q^j} \left[P(Q) - c_1 - \sigma\varepsilon_1 \right] q^j,$$

for $j = 1, 2, ..., m_1 - 1, m_1$,

and the profit maximization problem for the $n - m_1$ non-adopters as

$$\pi^j = \max_{q^j} \left[P(Q) - c_0 - \sigma\varepsilon_0 \right] q^j,$$

for $j = m_1 + 1, m_1 + 2, ..., n - 1, n$.

The first order conditions for the adopters and non-adopters respectively are

$$P(Q) + P'(Q)q^j = c_1 + \sigma\varepsilon_1,$$

$j = 1, 2, ..., m_1 - 1, m_1$,

$$P(Q) + P'(Q)q^j = c_0 + \sigma\varepsilon_0,$$

$j = m_1 + 1, m_1 + 2, ..., n - 1, n$.

Thus, both types of firms set marginal revenue equal to marginal costs inclusive of the tax payment for the emissions embodied in an additional unit of output. Because marginal cost is lower for the adopters, they produce more than non-adopters.

We define the profit-maximizing level of production for the m_1 adopters under an emission tax to be q_1^T and the profit-maximizing level of production for the $n - m_1$ non-adopters to be q_0^T. We further assume that $q_0^T > 0$ [4]. Now, if we let $\zeta_0^T = c_0 + \sigma\varepsilon_0$ denote marginal costs inclusive of emission tax payments under an emission tax before adoption of the new technology and let $\zeta_1^T = c_1 + \sigma\varepsilon_1$ denote marginal costs after adoption, the equilibrium output levels under an emission tax for adopters and non-adopters, respectively, are

$$q_1^T(m_1) = \frac{a - \zeta_1^T + [n - m_1] \left[\zeta_0^T - \zeta_1^T \right]}{b\,[n+1]},$$

$$q_0^T(m_1) = \frac{a - \zeta_0^T - m_1 \left[\zeta_0^T - \zeta_1^T \right]}{b\,[n+1]},$$

for which $q_1^T(m_1) > q_0^T(m_1) > 0$ and $q_1^T(m_1) - q_1^T(m_1 - 1) = q_0^T(m_1) - q_0^T(m_1 - 1) < 0 \vee m_1 \leq n$.

[4]From the equilibrium output level for technology 0 given below, it is clear that this assumption is satisfied for all $m_1 \leq n - 1$ if $a - n\,[c_0 + \sigma\varepsilon_0] + [n-1]\,[c_1 + \sigma\varepsilon_1] > 0$

Furthermore, $q_1^T(m_1) > q_0^T(m_1 - 1) \vee m_1$. That is, adoption allows firms to increase their output. Moreover, it allows adopters to increase their market share since, due to strategic behavior in the output market, non-adopters reduce their output to offset the effect of an increased supply on the market price.

Under an emission tax with m_1 adopters of the new technology, the equilibrium profit rate for adopters of the new technology is

$$\pi_1^T(m_1) = b\left[q_1^T(m_1)\right]^2,$$

and the equilibrium profit rate for the non-adopters

$$\pi_0^T(m_1) = b\left[q_0^T(m_1)\right]^2,$$

see Appendix B for derivation of equilibrium profits and output.

We can now find an expression for the increase in profit rate due to adoption for the firm that is the ith to adopt, under an emission tax.

$$\Delta\pi_i^T = b\left[\left[q_1^T(i)\right]^2 - \left[q_0^T(i-1)\right]^2\right]. \tag{2}$$

$\Delta\pi_i^T$ is positive but decreasing in i (in accordance with assumption 1ii and demonstrated in Appendix A.1).

4 Adoption incentives under a refunded tax

Under an emission tax which is refunded to the regulated firms in proportion to output market share, the profit rate maximization problem for the m_1 firms which have adopted the new technology is

$$\pi^j = \max_{q^j}\left[\left[P(Q) - c_1 - \sigma\varepsilon_1\right]q^j + \sigma E\frac{q^j}{Q}\right],$$

for $j = 1, 2, ..., m_1 - 1, m_1$,

and the profit maximization problem for the $n - m_1$ non-adopters

$$\pi^j = \max_{q^j}\left[P(Q) - c_0 - \sigma\varepsilon_0\right]q^j + \sigma E\frac{q^j}{Q},$$

$j = m_1 + 1, m_1 + 2, ..., n-1, n$, with aggregate emissions (E) and aggregate output (Q) given by:

$$E = \sum_{i=1}^{n} e^i$$
$$Q = \sum_{i=1}^{n} q^i$$

and the average emission intensity $\bar{\varepsilon}$ given by:

$$\bar{\varepsilon} = \frac{E}{Q}. \tag{3}$$

4.1 Exogenous Refunded Tax

With reference to the Swedish NO_x charge, we first focus on the case where the number of firms in the industry is large enough so that each firm considers its own impact on the average emission intensity (and therefore also the size of the refund) as neglible[5].

The first order conditions for the adopters and non-adopters respectively are then

$$P(Q) + P'(Q)q^j = c_1 + \sigma \left[\varepsilon_1 - \bar{\varepsilon}\right], \tag{4}$$

for $j = 1, 2, ..., m_1 - 1, m_1$,

$$P(Q) + P'(Q)q^j = c_0 + \sigma \left[\varepsilon_0 - \bar{\varepsilon}\right], \tag{5}$$

for $j = m_1 + 1, m_1 + 2, ..., n-1, n$.

Thus both types of firms set marginal revenue equal to marginal costs inclusive of the emission tax minus the marginal refund. The marginal refund is given by the emission tax rate times the average emission intensity and works as an implicit output subsidy. Thus, just as under an emission tax, adopters produce more than non-adopters because of lower marginal cost. However, output will be higher for both adopters and non-adopters under a refunded tax because of the refund.

We define the profit-maximizing level of production for adopters under an emission tax with exogenous refunding to be q_1^X and the profit-maximizing level of production for non-

[5] In the case of the Swedish NO_x charge, market power in the market for refunding is not a major concern. Although participants include large producers in industries that may not be perfectly competitive, in 2000 no plant had more than roughly 2% of the rebate market (Sterner & Höglund-Isaksson, 2006), since the tax-refund program includes several industries. Thus, by applying the program broadly, Sweden avoids the market-share issues that could arise with sector-specific programs (see Fischer 2011).

adopters to be q_0^X. If $q_0^T > 0$, the equilibrium output levels under an exogenously refunded tax for adopters and non-adopters, respectively, are

$$q_1^X(m_1) = q_1^T(m_1) + \frac{\sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]}, \tag{6}$$

$$q_0^X(m_1) = q_0^T(m_1) + \frac{\sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]}, \tag{7}$$

where $\bar{\varepsilon}^X(m_1) = \frac{m_1\varepsilon_1 q_1^X + [n-m_1]\varepsilon_0 q_0^X}{m_1 q_1^X + [n-m_1]q_0^X} > 0$. Because the average emissions intensity decreases with the number of firms adopting the new technology[6], the difference in output with and without a refund decreases as m_1 increases. Equilibrium profit rates under a refunded tax with m_1 adopters of the new technology are

$$\pi_1^X(m_1) = b\left[q_1^X(m_1)\right]^2,$$
$$\pi_0^X(m_1) = b\left[q_0^X(m_1)\right]^2,$$

see Appendix B for derivation of equilibrium profits and output.

We can now find an expression for the increase in profit rate due to adoption for the firm, which is the ith to adopt, under an exogenous refunded tax.

$$\Delta\pi_i^X = b\left[\left[q_1^X(i)\right]^2 - \left[q_0^X(i-1)\right]^2\right].$$

Substituting in (6), we have that

$$\Delta\pi_i^X = b\left[\left[q_1^T(i) + \frac{\sigma\bar{\varepsilon}^X(i)}{b\,[n+1]}\right]^2 - \left[q_0^T(i-1) + \frac{\sigma\bar{\varepsilon}^X(i-1)}{b\,[n+1]}\right]^2\right]. \tag{8}$$

Since each firm considers its own impact on the average emission intensity as negligible, $\bar{\varepsilon}^X(i) = \bar{\varepsilon}^X(i-1)$ from the perspective of the firm, and hence (8) simplifies to

$$\Delta\pi_i^X = \Delta\pi_i^T + \frac{2\sigma\bar{\varepsilon}^X(i)}{[n+1]}\left[q_1^T(i) - q_0^T(i-1)\right].$$

The difference in the increase in profit rate from adoption under a standard emission tax

[6]Let $s_1(m_1)$ to denote the market share of an individual adopter with m_1 adopters in the industry. The average emission intensity can be represented as $\bar{\varepsilon}(m_1) = \varepsilon_0 - m_1 s_1(m_1)\delta$, where $\delta = \varepsilon_0 - \varepsilon_1$. Note that $\bar{\varepsilon}(m_1) < \bar{\varepsilon}(m_1 - 1)$ if $[m_1-1]s_1(m_1-1) < m_1 s_1(m_1)$. That is to say, the average emission intensity decreases with adoption if the total output share of adopters increases with adoption.

compared to an exogenous refunded tax is then given by

$$\Delta\pi_i^X - \Delta\pi_i^T = 2\frac{n\left[\zeta_0^T - \zeta_1^T\right]}{b\left[n+1\right]^2}\sigma\bar{\varepsilon}^X(i), \tag{9}$$

since $q_1^T(i) - q_0^T(i-1) = \frac{n\left[\zeta_0^T - \zeta_1^T\right]}{b[n+1]} > 0$.

Under these assumptions, the following proposition holds:

Proposition 1 *A technology that reduces the emission intensity of production diffuses faster under an exogenously refunded than under a non-refunded emission tax.*

We see from (9) that, for the same tax per unit of emissions, σ, $\Delta\pi_i^X > \Delta\pi_i^T$. That is, the diffusion of the new technology is faster under the exogenous refunded tax. However, since the average emission intensity and the refund decreases as the technology diffuses into the industry, it is optimal for the late adopters to wait longer to adopt relative to the early adopters so that investment cost goes down further with time. The additional impact of the refund over taxes therefore diminishes for the firms later in the adoption sequence.

4.2 Endogenous Refunded Tax

So far we have assumed that each firm considers its own impact on the average emission intensity and thus the size of the refund as negligible. However, since firms in the present framework have market power in the output market and emissions are proportional to output, it is appropriate to also consider the case where firms have market power in the market for refunding. If firms take into account their influence on the size of the refund, the first order condition for the adopters are

$$P(Q) + P'(Q)q^j = c_1 + \sigma\left[\varepsilon_1 - \bar{\varepsilon}\right]\left[1 - \frac{q^j}{Q}\right], \tag{10}$$

for $j = 1, 2, ..., m_1 - 1, m_1$,
and for non-adopters

$$P(Q) + P'(Q)q^j = c_0 + \sigma\left[\varepsilon_0 - \bar{\varepsilon}\right]\left[1 - \frac{q^j}{Q}\right], \tag{11}$$

for $j = m_1 + 1, m_1 + 2, ..., n - 1, n$.

Let q_1^D and q_0^D be the profit-maximizing level of production for adopters and non-adopters, respectively, under endogenous refunding. Defining $Q^X(m_1) = m_1 q_1^X + [n - m_1]\, q_0^X$, $Q^D(m_1) =$

$m_1 q_1^D + [n - m_1]\, q_0^D$ and $\bar{\varepsilon}^D(m_1) = \frac{m_1 \varepsilon_1 q_1^D + [n-m_1]\varepsilon_0 q_0^D}{Q^D} > 0$, it can be shown from the equilibrium conditions in (4) and (5), and (10) and (11) (see Appendix C), that

$$Q^D(m_1) - Q^X(m_1) = \frac{n\sigma}{b\,[n+1]} \left[\bar{\varepsilon}^D(m_1) - \bar{\varepsilon}^X(m_1)\right],$$

i.e., total output under endogenous and exogenous refunding is the same only if the average emissions intensities $\bar{\varepsilon}^D(m_1)$ and $\bar{\varepsilon}^X(m_1)$ are the same. Thus, comparing the FOCs that define the profit-maximizing level of production for adopters and non-adopters under exogenous and endogenous refunding (i.e., equations (4)-(10) for adopters and (5)-(11) for non-adopters), we can say that, for equivalent average emission intensity, $q_1^X > q_1^D \; \forall m_1 < n$ and $q_0^X < q_0^D \; \forall m_1 > 0$. Hence, more production is shifted toward non-adopters under endogenous refunding compared to exogenous refunding for equivalent average emission intensities (see also, Fischer 2011, pp 223). Furthermore, $q_1^X(n) = q_1^D(n)$ and $q_0^X(0) = q_0^D(0)$ since the net tax is zero when the firms are homogenous.

As shown in Appendix B, equilibrium profit rates under an endogenous refunded tax with m_1 adopters of the new technology are

$$\pi_1^D(m_1) = b\left[1 - \frac{\sigma}{bQ^D(m_1)}\left[\varepsilon_1 - \bar{\varepsilon}^D(m_1)\right]\right]\left[q_1^D(m_1)\right]^2,$$
$$\pi_0^D(m_1) = b\left[1 - \frac{\sigma}{bQ^D(m_1)}\left[\varepsilon_0 - \bar{\varepsilon}^D(m_1)\right]\right]\left[q_0^D(m_1)\right]^2.$$

The increase in profit rate due to adoption for the firm that is the ith to adopt, under a refunded tax with firm influence on the size of the refund, is then given by

$$\Delta\pi_i^D = b\left[\left[q_1^D(i)\right]^2 - \left[q_0^D(i-1)\right]^2\right] + \sigma\left[\frac{\left[\varepsilon_0 - \bar{\varepsilon}^D(i-1)\right]}{Q^D(i-1)}\left[q_0^D(i-1)\right]^2 - \frac{\left[\varepsilon_1 - \bar{\varepsilon}^D(i)\right]}{Q^D(i)}\left[q_1^D(i)\right]^2\right]. \tag{12}$$

By using equation (3), and that $\varepsilon_0 = \varepsilon_1 + \delta$ with $\delta > 0$, we can write:

$$\varepsilon_0 - \bar{\varepsilon}^D(m_1) = m_1 s_1^D(m_1)\delta, \tag{13}$$
$$\varepsilon_1 - \bar{\varepsilon}^D(m_1) = -\left[n - m_1\right] s_0^D(m_1)\delta,$$

where $s_1^D(m_1)$ and $s_0^D(m_1)$ represent the market shares of an individual adopter and non-adopter, respectively, with m_1 adopters in the industry.

Substituting (13) into (12) yields:

$$\Delta\pi_i^D = b\left[\left[q_1^D(i)\right]^2 - \left[q_0^D(i-1)\right]^2\right] \qquad (14)$$
$$+ \sigma\delta\left[[i-1]\, s_0^D(i-1)s_1^D(i-1)q_0^D(i-1) + [n-i]\, s_0^D(i)s_1^D(i)q_1^D(i)\right].$$

Unfortunately, equations (14) cannot be easily compared to (2) or (8) since output levels and emission intensities are endogenous. Nevertheless, to be able to say something about the impact of firms' strategically influencing the size of the refund and the adoption decision, we follow the approach in Fischer (2011) and compare adoption incentives between exogenous and endogenous refunding for an equivalent average emission intensity. That is to say, we compare adoption profits under exogenous vs. endogenous refunding for the firms which are the first and last to adopt. This yields:

$$\Delta\pi_1^D - \Delta\pi_1^X = b\left[\left[q_1^D(1)\right]^2 - \left[q_1^X(1)\right]^2\right] + \sigma\delta\,[n-1]\, s_0^D(1)s_1^D(1)q_1^D(1), \qquad (15)$$

and

$$\Delta\pi_n^D - \Delta\pi_n^X = b\left[\left[q_0^X(n-1)\right]^2 - \left[q_0^D(n-1)\right]^2\right] \qquad (16)$$
$$+ \sigma\delta\,[n-1]\, s_0^D(n-1)s_1^D(n-1)q_0^D(n-1).$$

Note that the first term (in brackets) on the right hand side of equations (15) and (16) is negative, while the second term is positive. Therefore, equations (15) and (16) indicate that it is ambiguous whether adoption will be slower under endogenous than under exogenous refunding because of the existence of two counteracting effects, an "output" effect and a "refunding" effect. Firstly, as stated before, for equivalent average emissions intensities, production is shifted toward non-adopters under endogenous refunding. Consequently, this production shifting lowers the benefit of adoption under endogenous versus exogenous refunding for the firms which are the first and last to adopt. Secondly, the magnitude of the refund has an effect. Because production is shifted toward non-adopters, the average emission intensity is larger under endogenous refunding, and so is the refund, which increases the benefits of adoption under endogenous versus exogenous refunding.

Overall, we expect the first effect to dominate. Moreover, the larger the number of firms in the industry, the smaller should be the difference between exogenous and endogenous refunding, because the strategic interaction between firms in the output market is reduced in such a case. Note also that the second effect depends critically on the effect of adoption on emissions per unit of output, i.e., the larger is δ, the larger the increase in emissions from shifting production toward non-adopters, and the larger is the second effect. These observations lead to the following proposition.

Proposition 2 *A technology that reduces the emission intensity of production tends to diffuse more slowly under an endogenously vs. an exogenously refunded emission tax the more concentrated the industry is.*

Next, we compare the adoption incentives under an endogenous refunded tax and an emission tax for the firm which is first to adopt and the firm which is the last, nth, firm to adopt. This yields:

$$\Delta\pi_1^D - \Delta\pi_1^T = \left[\Delta\pi_1^D - \Delta\pi_1^X\right] + 2\frac{n\left[\zeta_0^T - \zeta_1^T\right]}{b\left[n+1\right]^2}\sigma\varepsilon_0,$$

and

$$\Delta\pi_n^D - \Delta\pi_n^T = \left[\Delta\pi_n^D - \Delta\pi_n^X\right] + 2\frac{n\left[\zeta_0^T - \zeta_1^T\right]}{b\left[n+1\right]^2}\sigma\varepsilon_1.$$

The difference in profit increase for endogenous refunding versus a non-refunded emission tax is given by the sum of the difference between endogenous and exogenous refunding (the first term), and the difference between exogenous refunding and an emission tax (the second term). As discussed previously, the first term is on the net likely to be negative while the second is positive. Hence, compared to exogenous refunding, it is clear that taxes are less likely to induce a faster diffusion than endogenous refunding. Nevertheless, it is still the case that the "output" effect should dominate the "refunding" effect if the number of firms in the industry is small, to the extent that diffusion is likely to be slower under endogenous refunding. These observations lead to the following proposition.

Proposition 3 *A technology that reduces the emission intensity of production tends to diffuse more slowly under an endogenously refunded vs. a non-refunded emission tax the more concentrated the industry is.*

5 Incentives for continuous technological upgrading

In the previous sections, we showed under what conditions exogenous refunding helps to speed up the path of technology adoption. However, this positive effect of refunding dissipates as the average emission intensity of the industry decreases. In order to analyze to what extent refunding provides continuous increased incentives for technological upgrading, we consider the case when further technological advance occurs at some point in the future. This new technology, which we will call technology 2, unexpectedly arrives at some time t_2 after k^T and k^X firms would have already adopted technology 1 under an emission tax and an exogenous refunded tax, respectively. As shown in the previous sections, since the exogenous refund induces a faster adoption than the emission tax, $k^X \geq k^T$.

We study the difference in adoption incentives for the new technology provided by these instruments for three groups: (1) the *laggards* - those firms that would not have adopted technology 1 at t_2 either under the emission tax or the refunded tax (i.e., $n - k^X$ firms), (2) the *intermediates* - those firms that would have adopted technology 1 at t_2 under refunding, but would not have adopted under an emission tax (i.e., $k^X - k^T$ firms), and finally, (3) the *early adopters* - those firms that would have adopted technology 1 at t_2 under both schemes (i.e., k^T firms). If refunding provides a continuous and larger incentive to technological upgrading than taxes, we should expect the difference in the increase in profit rate from adoption with and without refunding to be positive for all groups. Moreover, if refunding produces a "catching up" effect - understood as an increased incentive for firms dirtier than average to adopt new technologies, we should expect the difference in profit increase for the *laggards* to be unambiguously positive.

Technology 2 is characterized by a marginal production cost c_2 and emission intensity ε_2, with $\varepsilon_2 < \varepsilon_1 < \varepsilon_0$. Let $\zeta_2^T = c_2 + \sigma\varepsilon_2$. By assumption, we have that $\zeta_0^T > \zeta_1^T > \zeta_2^T$. Now let m_1 be the number of adopters of technology 1 and m_2 be the number of adopters of technology 2. At time t_2, we thus have $m_1 = k$ and $m_2 = 0$. Further, let $\pi_2(m_1, m_2)$ be the profit rate for firm j when m_1 firms have adopted technology 1, m_2 firms have adopted technology 2, and firm j is among the adopters of technology 2. We define $\pi_1(m_1, m_2)$ and $\pi_0(m_1, m_2)$ accordingly. The firm which has not adopted technology 1 at time t_2 now has the choice between two technologies. However, for simplicity, we assume that $p_2(t)$, the present value cost at time t_2 of investing in technology 2 at t, is not larger than the cost of investing in technology 1 at t, i.e., $p_2(t) \leq p_1(t)e^{rt_2}$ for $t \geq t_2$. This implies that it will never be profitable to adopt technology 1 once technology 2 has appeared[7].

The lower marginal costs imply higher profit rates with technology 2 compared to both technology 1 and technology 0. The increase in profit rates from adoption of technology 2 will thus be higher for a firm which produces with technology 0 than for a firm which has already adopted technology 1. I.e., the following conditions apply: $\pi_2(m_1, m_2) > \pi_1(m_1, m_2) > \pi_0(m_1, m_2)$ as well as $\pi_2(m_1, m_2 + 1) - \pi_0(m_1, m_2) > \pi_2(m_1 - 1, m_2 + 1) - \pi_1(m_1, m_2)$ for all m_1, m_2 for which $m_1 + m_2 < n$. Furthermore, we assume that $p_2(t)$ (defined for $t \geq t_2$) is a differentiable convex function for which $p_2'(t_2) \leq \pi_0(k, 0) - \pi_2(k, 1)$, $\lim_{t \longrightarrow \infty} p_2'(t) > 0$ and $p_2''(t) > re^{-rt}(\pi_2(k, 1) - \pi_0(k, 0))$. Lastly, we define $\Delta\pi_{02,j} = \pi_2(k, j) - \pi_0(k, j - 1)$ and $\Delta\pi_{12,j} = \pi_2(n - j, j) - \pi_1(n - j + 1, j - 1)$.

We can now determine the optimal adoption dates for technology 2 for the three groups of firms from first order conditions similar to (1) (see more details on the results in this section in Appendix D). The $n - k$ firms which produce with technology 0 at t_2 will first find it profitable to adopt technology 2 at τ_j^*, implicitly defined by

$$\Delta\pi_{02,j}\, e^{-r\left[\tau_j^* - t_2\right]} - p_2'(\tau_j^*) = 0$$

for $j = 1, 2, ..., m_1 - 1, n - k,$, and the k firms which produce with technology 1 at t_2 will adopt technology 2 at τ_j^*, implicitly defined by

$$\Delta\pi_{12,j}\, e^{-r\left[\tau_j^* - t_2\right]} - p_2'(\tau_j^*) = 0$$

for $j = n - k + 1, n - k + 2, ..., n - 1, n$.

To analyze the schedule of adoption dates for technology 2, we again need to analyze the difference in the increase in profit rate from adoption with and without refunding for each position in the adoption sequence.

Under an emission tax, equilibrium output and profit levels for the three technologies are[8]

$$q_0^T(m_1, m_2) = \frac{a - \zeta_0^T - m_1\left[\zeta_0^T - \zeta_1^T\right] - m_2\left[\zeta_0^T - \zeta_2^T\right]}{b\,[n + 1]},$$

[7]This is not a necessary condition for technology 2 to always be preferred. What is required is that the net present value of adopting technology 2 at some point in time after t_2 is always greater than the net present value of adopting technology 1.

[8]As seen from the expression below, $q_0^T > 0$ if $a - \zeta_0 - m_1[\zeta_0 - \zeta_1] - m_2[\zeta_0 - \zeta_2] > 0$

$$q_1^T(m_1, m_2) = \frac{a - \zeta_1^T - [n - m_1 - m_2]\left[\zeta_1^T - \zeta_0^T\right] - m_2\left[\zeta_1^T - \zeta_2^T\right]}{b\,[n+1]},$$

$$q_2^T(m_1, m_2) = \frac{a - \zeta_2^T - [n - m_1 - m_2]\left[\zeta_2^T - \zeta_0^T\right] - m_1\left[\zeta_2^T - \zeta_1^T\right]}{b\,[n+1]},$$

and

$$\pi_0^T(m_1, m_2) = b\left[q_0^T(m_1, m_2)\right]^2,$$

$$\pi_1^T(m_1, m_2) = b\left[q_1^T(m_1, m)\right]^2,$$

$$\pi_2^T(m_1, m_2) = b\left[q_2^T(m_1, m_2)\right]^2.$$

Under the exogenously refunded tax, the expressions corresponding to the case with two technologies are

$$q_0^X(m_1, m_2) = q_0^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]},$$

$$q_1^X(m_1, m_2) = q_1^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]},$$

$$q_2^X(m_1, m_2) = q_2^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]},$$

and

$$\pi_0^X(m_1, m_2) = b\left[q_0^X(m_1, m_2)\right]^2 = b\left[q_0^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]}\right]^2,$$

$$\pi_1^X(m_1, m_2) = b\left[q_1^X(m_1, m_2)\right]^2 = b\left[q_1^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]}\right]^2,$$

$$\pi_2^X(m_1, m_2) = b\left[q_2^X(m_1, m_2)\right]^2 = b\left[q_2^T(m_1, m_2) + \frac{\sigma\bar{\varepsilon}^X(m_1, m_2)}{b\,[n+1]}\right]^2.$$

where

$$\bar{\varepsilon}^X(m_1, m_2) = \frac{m_2\varepsilon_2 q_2^X(m_1, m_2) + m_1\varepsilon_1 q_1^X(m_1, m_2) + [n - m_1 - m_2]\,\varepsilon_0 q_0^X(m_1, m_2)}{m_2 q_2^X(m_1, m_2) + m_1 q_1^X(m_1, m_2) + [n - m_1 - m_2]\, q_0^X(m_1, m_2)}.$$

Laggards

Let us first analyze the difference in the increase in profit rate from adoption of technology 2 with and without refunding for the *laggards* which would not have adopted technology 1 by t_2 under either policy and are therefore producing with technology 0. Because

$\bar{\varepsilon}^R(k^R, j) = \bar{\varepsilon}^R(k^R, j-1)$ from the perspective of the firm under exogenous refunding, the difference in the profit rate increase from adoption of technology 2 under the exogenous refunded tax compared to the emission tax is

$$\Delta\pi^X_{02,j} - \Delta\pi^T_{02,j} = 2\frac{n\left[\zeta^T_0 - \zeta^T_2\right]}{b\left[n+1\right]^2}\left[\left[k^T - k^X\right]\left[\zeta^T_0 - \zeta^T_1\right] + \sigma\bar{\varepsilon}^X(k^X, j)\right], \tag{17}$$

for $j = 1, 2, ..., n - k^X - 1, n - k^X$.

If $k^X = k^T$, we have from (17) that, analogous to the result with two technologies, $\Delta\pi^X_{02,j} > \Delta\pi^T_{02,j}$. Hence, the adopters of technology 2 switching from technology 0 would invest earlier under the refunded tax than under an emission tax. However, if $k^X > k^T$, the sign of this expression is ambiguous. A negative sign would indicate faster diffusion of technology 2 under an emission tax than a refunded tax. The explanation for this possible outcome is simple: switching to technology 2 from technology 0 under the emission tax can be more profitable if there are fewer competitors with technology 1 and instead more competitors with technology 0. The sign of (17) is negative and adoption is faster under an emission tax if the difference in profit increase coming from higher output under an emission tax is larger than the profit increase coming from the refund.

Intermediates

Let us now examine the difference in profits for the *intermediates*, which only exist if the number of firms which would have adopted technology 1 by t_2 under the emission tax is lower than the number of adopters of technology 1 at t_2 under the exogenous refunded tax, i.e., $k^T < k^X$. The jth adopter, for which $j \in \left[n - k^X + 1, n - k^T\right]$, would switch from technology 0 under an emission tax, and from technology 1 under a refunded tax. In the eyes of the firms, $\bar{\varepsilon}^X(n-j, j) = \bar{\varepsilon}^X(n-j+1, j-1)$. Therefore, the difference between the policies in adoption time is determined by the following:

$$\begin{aligned}\Delta\pi^X_{12,j} - \Delta\pi^T_{02,j} = {} & \frac{n\left[\zeta^T_0 - \zeta^T_1\right]}{b\left[n+1\right]^2}\left[2k^T\left[\zeta^T_0 - \zeta^T_2\right] - \left[n-2j\right]\left[\zeta^T_0 + \zeta^T_1 - 2\zeta^T_2\right] - 2\left[a + \zeta^T_2\right]\right] \\ & + 2\frac{n[\zeta^T_1 - \zeta^T_2]}{b\left[n+1\right]^2}\sigma\bar{\varepsilon}^X(n-j, j),\end{aligned} \tag{18}$$

for $j \in \left[n - k^X + 1, n - k^T\right]$.

The sign of (18) is ambiguous, so the *intermediates* would adopt either earlier or later under an exogenous refunded tax compared to a standard emission tax.

Early adopters

Finally, let us analyze the incentives to adopt technology 2 under the emission tax and the refunded tax for those firms that would have adopted technology 1 by t_2 under both policies, i.e., the *early adopters*. When the first of the firms with technology 1 invests in technology 2, there is no longer any firm using technology 0. This means that there are again only two production technologies in the market and that results are comparable to the ones in section 4.1. Because $\bar{\varepsilon}^X(n-j,j) = \bar{\varepsilon}^X(n-j+1,j-1)$ from the perspective of the firm, the difference in profit rate increase is given by:

$$\Delta\pi^X_{12,j} - \Delta\pi^T_{12,j} = \frac{2n\left[\zeta^T_1 - \zeta^T_2\right]}{b\left[n+1\right]^2}\sigma\bar{\varepsilon}^X(n-j,j) \tag{19}$$

for $j = n-k^T+1, n-k^T+2, ..., n-1, n$.

(19) is positive. This indicates that the jth adopter of technology 2, which would switch from technology 1 under both policies, invests earlier under the refunded tax than under a standard emission tax.

In sum, our results indicate that, although exogenous refunding provides continuous incentives for technological upgrading, these incentives are not unambiguously larger than those provided by an emission tax. This is particularly the case for firms that are dirtier than average (the so- called *laggards*). In relative terms, the gains of investing in a new technology, in terms of increased output and refunding, dissipates as the overall industry becomes cleaner. The previous findings can be summarized in the following Proposition.

Proposition 4 *A technology that reduces the emission intensity of production does not unambiguously diffuse faster under an exogenously refunded tax than under a non-refunded emission tax among those firms that are dirtier than average.*

6 Numerical illustrations

In the following, we present simulations on the diffusion patterns and welfare effects under a standard emission tax as well as exogenous and endogenous refunding.

6.1 Diffusion

To illustrate the diffusion patterns under the policies and how the patterns are affected by the degree of market concentration, we present numerical simulations for an industry composed of 5 and 15 firms, respectively. For the simulations, we assume the following function for the present value of the investment cost

$$p_1(t) = K_1 e^{-[\theta + r]t}$$

where $\theta > 0$ captures drivers such as learning and technological progress which lead to decreasing investment costs over time (here assumed exogenous and generating a constant rate of decrease in costs). We assume $\theta = 3\%$, $r = 6\%$ and $K_1 = 20$ and for the remaining parameters $a = 10$, $b = 1$, $\varepsilon_0 = 1$, $\varepsilon_1 = 0.5$, $c_0 = c_1 = 1$ and $\sigma = 1$.

Figures 1 and 2 illustrate the adoption times for each firm in the sequence. We see from Figure 1 with $n = 5$ firms that, for this set of parameters, the exogenous refunded tax induces a faster diffusion than the non-refunded emission tax, just as discussed in section 4.1. However, with endogenous refunding, the firms would adopt later than under exogenous refunding, as well as later than they would under a non-refunded emission tax. Figure 3 illustrates the contribution from the "output" and "refunding" effects to the difference between endogenous and exogenous refunding. As discussed in section 4.2, the output effect dominates the refunding effect, such that, on net, the difference is negative for each adopter in the adoption sequence. We also see that, even though the additional difference between exogenous refunding and a non-refunded tax is positive, the net difference between endogenous refunding and an emission tax is still negative.

With $n = 15$ firms in Figure 2, diffusion takes longer since gains from adoption are lower. Here, also, the exogenous refunded tax induces faster diffusion than the non-refunded emission tax. However, with endogenous refunding, the first firm would adopt at a point in time very close to but later than the adoption time under the emission tax, while the last firm would adopt earlier than under an emission tax and at a point in time very close to the adoption time under the exogenous refunded tax. With $n = 15$ firms, differences in adoption times are, however, relatively small. This illustrates that, as the number of firms increases, the diffusion pattern under a refunded tax also approaches the pattern under a standard emission tax.

In Figure 4, the difference in profit increase between endogenous and exogenous refunding is disaggregated into "output" and "refunding" effects with 15 firms in the industry. It is still true that the output effect dominates the refunding effect such that diffusion is slower under endogenous versus exogenous refunding for each firm in the sequence. However, the relatively larger difference in profit increase between exogenous refunding and an emission tax implies that, on net, endogenous refunding induces faster adoption than an emission tax for all but the first firm in the adoption sequence, as also noted from Figure 2. Figure 4 also illustrates that, for $n = 15$, the outcome under endogenous refunding is well approximated by the outcome under exogenous refunding for firms later in the adoption sequence.

6.2 Welfare

The policies have different effects on welfare because of the different patterns of adoption. Faster diffusion of the cost-reducing technology raises consumer surplus and lowers environmental damages in present value terms for the whole diffusion period but also raises total investment costs. Welfare effects are also different under the policies because, even with the same number of adopters at a certain point in time, equilibrium output and aggregate emissions differ[9].

With m_1 adopters of the new technology, consumer surplus is given by

$$CS(m_1) = \frac{b}{2}[Q(m_1)]^2$$

We assume that the emitted substance is a flow pollutant which causes damages only in the current period and has a constant value of marginal damage from emissions $\delta = 1$. Total environmental damages D at a point in time with m_1 adopters of the new technology is then given by

$$D(m_1) = \delta E(m_1)$$

Net tax revenues are $TR(m_1) = \sigma E(m_1)$ under the emission tax. In contrast, $TR(m_1) = 0$ under the refunded tax since all the tax revenues are refunded back to the firms and included in firm profits.

[9]The welfare comparison is made between outcomes under the two policies for the same level of the emission tax. Because of the refund, equilibrium output and aggregate emissions differ. From a welfare comparison perspective, one could argue that the relevant comparison is between different tax rates under the two policies which induce the same level of aggregate emissions. However, for this model, there is no emission tax level other than zero that induces the same level of emissions under the two policies before diffusion has started and after the technology has completely diffused into the industry.

The welfare rate (or instantaneous welfare), $w(m_1)$, excluding investment costs, with m_1 adopters of the new technology is the sum of consumer surplus, firm profits and tax revenues minus environmental damages and is given by

$$w(m_1) = CS(m_1) + [n - m_1]\,\pi_0(m_1) + m_1\pi_1(m_1) + TR(m_1) - D(m_1)$$

Total discounted welfare W net of investment costs can now be written

$$W = \sum_{m_1=0}^{n} \left[\int_{\tau^*_{m_1}}^{\tau^*_{m_1+1}} w(m_1)e^{-rt}dt \right] - \sum_{m_1=1}^{n} p(\tau^*_{m_1})$$

with $\tau^*_0 = 0$ and $\tau^*_{n+1} = \infty$.

Tables 1 and 2 show the welfare levels over time under an emission tax, an exogenous refunded tax and an endogenous refunded tax with 5 and 15 firms, respectively. Adoption times are the same as in the previous section, with $\tau^*_{m_1}$ determining the start of the m_1th period with current value of the welfare rate $w(m_1)$ at each point in time. We see from table 1 that, for $n = 5$, discounted welfare is similar under exogenous and endogenous refunding. It appears that the benefit of reaching higher welfare rates with the faster diffusion in the first case is matched by the benefit of lower investment cost with slower diffusion in the second.

Discounted welfare is lower under the emission tax compared to both refunding situations. This is not driven by differences in how early clean production is traded for lower investment cost but by a difference in the level of welfare. This welfare difference comes from the fact that we have assumed that consumer surplus is quadratic in aggregate production while environmental damage is linear in aggregate emissions. This leads to higher welfare rates with a refund since more production is valued more highly than less emissions at the margin. Had we assumed that environmental damages were quadratic in aggregate emissions, the opposite would have been true, i.e., welfare rates would have been higher without refunding.

Table 2 shows the welfare outcomes with $n = 15$ firms. Comparing outcomes for the same policy with $n = 5$ and $n = 15$, consumer surplus and emissions are higher and firm profits lower for all three policy situations. With more competition in the industry, the benefits of adoption for an individual firm are significantly lower and therefore diffusion is slower than with only five firms in the industry. Both welfare rates and adoption times are so similar with 15 firms that there is, in practice, no difference in the value of discounted welfare across policies.

Note that, when it comes to output, our simulations indicate that non-adopters do produce slightly more and adopters slightly less with endogenous refunding compared to the case with exogenous refunding in line with Fischer (2011) and as discussed in section 4.2. However, at the aggregate level, output does not differ significantly between the two refunding situations, which explains why consumer surplus is almost the same in both cases.

Figures 5 and 6 illustrates how the level of aggregate emissions develops over time under the different policies. The figures show that the difference in the level of aggregate emissions between the policies is smaller after the new technology has been completely diffused into the industry. This is driven by the fact that the additional output under the refunded tax is then produced with lower emission intensity, and also because the difference in aggregate output is smaller between the two policies.

7 Conclusions

The main conclusion is that a refunded emission tax speeds up diffusion in an imperfectly competitive industry relative to a non-refunded emission tax if firms do not strategically influence the size of the refund. If they do, diffusion is, in contrast, likely to be slower than under a non-refunded emission tax if the industry is highly concentrated.

It is straightforward to see that, as the number of firms increases and the equilibrium comes closer to the outcome under perfect competition, the difference in diffusion patterns with and without refunding goes to zero. However, our findings are only valid in the context of an output market that is not perfectly competitive. If there is perfect competition, diffusion is not an equilibrium since, in that case, adoption would yield the same incremental benefits to all firms.

These results should apply to end-of-pipe technologies that convert a certain proportion of emissions. For energy production, the findings also should be valid for fuel efficiency improving technologies when it comes to pollutants such as CO_2 and SO_2. The implications of refunding for other types of abatement technologies is a potential area for future research.

Our results are based on the assumption that firms do not anticipate the appearance of a more efficient technology farther into the future. Allowing for such anticipation should delay optimal adoption times but, for this to alter the main comparative results, the effect of refunding would have to interact with the anticipation effect.

We have focused only on the incentives to technological diffusion provided by output-based refunding. Refunding might also be based on investments in abatement technologies, like the Norwegian NO_x fund from which emission fee revenues are refunded in proportion to abatement expenditure (see Hagem *et al.*, 2012). Such a case is outside the scope of our study, and further research is needed to understand the incentives provided by that type of scheme.

The welfare implications of the differences in diffusion patterns under our particular assumptions appear to be small. There could be a more relevant difference from a welfare point of view if faster diffusion also speeds up learning and endogenously lowers investment costs. There should also be larger benefits to faster diffusion if emissions are a stock pollutant.

The fact that the rate of technology adoption is influenced by (exogenous) refunding is potentially good news for a regulator, who has political constraints on the level of the tax to be imposed, but wants to promote faster uptake of existing abatement technologies as a way to speed up the pace of emission reductions.

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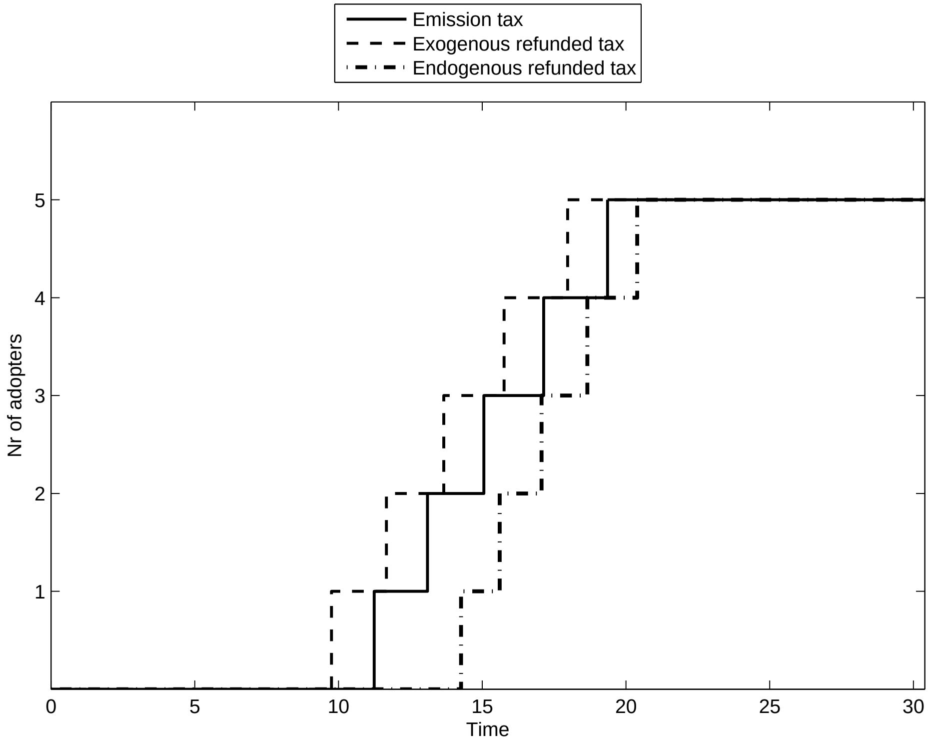

Figure 1: Diffusion with 5 firms in the industry

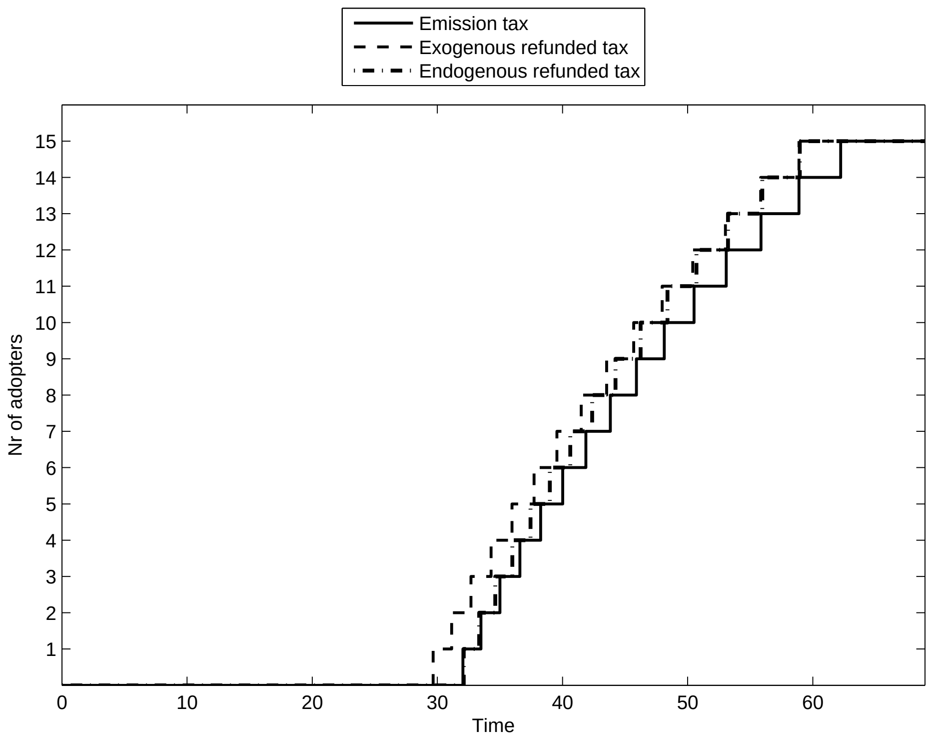

Figure 2: Diffusion with 15 firms in the industry

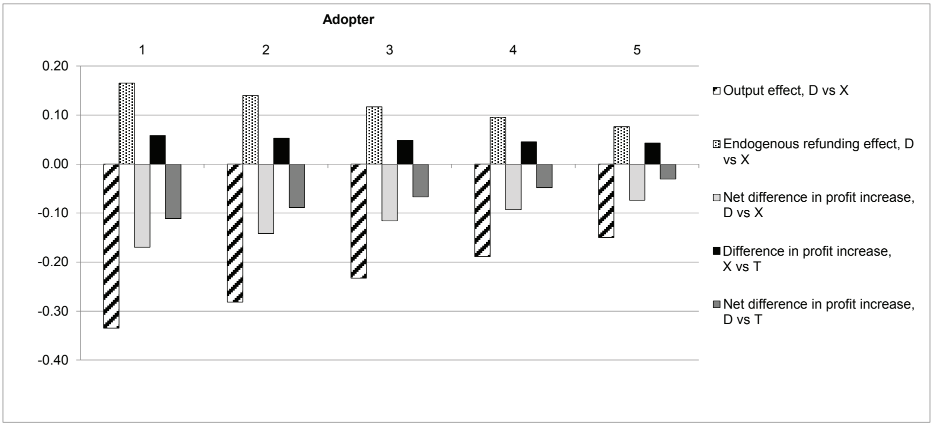

Figure 3: Output and endogenous refunding effects explaining net differences in profit increase from adoption with 5 firms in the industry. T refers to emission tax, X to exogenous refunded tax and D to endogenous refunded tax.

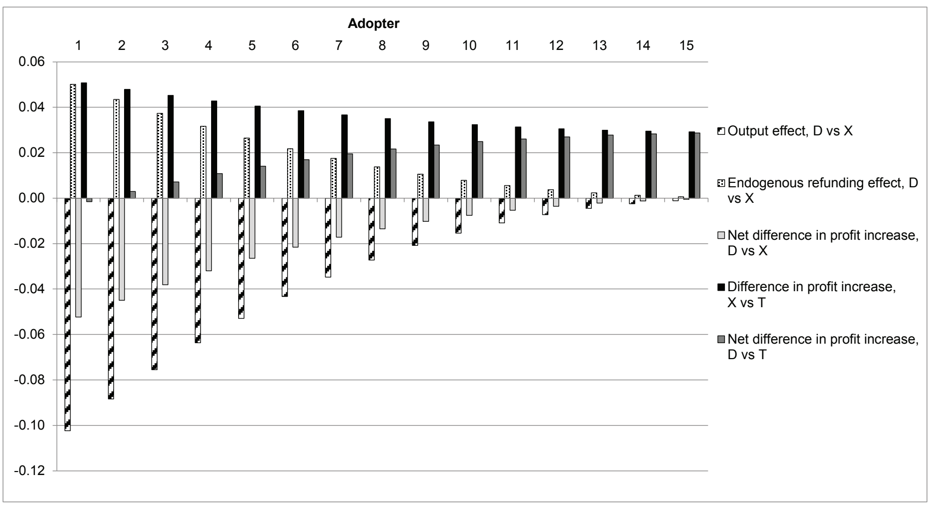

Figure 4: Output and endogenous refunding effects explaining net differences in profit increase from adoption with 15 firms in the industry. T refers to emission tax, X to exogenous refunded tax and D to endogenous refunded tax.

Table 1: Welfare estimates with 5 firms in the industry.

Period/number of adopters	0	1	2	3	4	5	Total all 6 periods
Emission tax							
Start of period/date of adoption	0.0	11.2	13.1	15.1	17.1	19.4	
Present value investment cost	0.0	7.3	6.2	5.2	4.3	3.5	
Current value consumer surplus rate	22.2	22.8	23.3	23.9	24.5	25.1	
Current value firms' profit rate	8.9	9.3	9.6	9.9	10.0	10.0	
Current value tax revenue rate	6.7	5.9	5.2	4.5	4.0	3.5	
Current value environmental damage rate	6.7	5.9	5.2	4.5	4.0	3.5	
Current value welfare rate (excl investment)	31.1	32.1	33.0	33.8	34.5	35.1	
Present value total period welfare	254.4	21.4	21.7	21.7	21.4	179.7	**520**
Exogenous refunded tax							
Start of period/date of adoption	0.0	9.8	11.7	13.7	15.8	18.0	
Present value investment cost	0.0	8.3	7.0	5.8	4.8	4.0	
Current value consumer surplus rate	28.1	28.0	27.9	27.9	28.0	28.1	
Current value firms' profit rate	11.3	11.4	11.4	11.4	11.4	11.3	
Current value tax revenue rate	0.0	0.0	0.0	0.0	0.0	0.0	
Current value environmental damage rate	7.5	6.5	5.7	4.9	4.3	3.8	
Current value welfare rate (excl investment)	31.9	32.8	33.7	34.4	35.1	35.6	
Present value total period welfare	235.5	24.6	24.4	24.0	23.4	198.0	**530**
Endogenous refunded tax							
Start of period/date of adoption	0.0	14.3	15.6	17.1	18.7	20.4	
Present value investment cost	0.0	5.5	4.9	4.3	3.7	3.2	
Current value consumer surplus rate	28.1	28.0	27.9	27.9	28.0	28.1	
Current value firms' profit rate	11.3	11.4	11.4	11.4	11.4	11.3	
Current value tax revenue rate	0.0	0.0	0.0	0.0	0.0	0.0	
Current value environmental damage rate	7.5	6.6	5.7	5.0	4.3	3.7	
Current value welfare rate (excl investment)	31.9	32.8	33.6	34.3	35.0	35.6	
Present value total period welfare	305.5	12.4	13.5	14.4	15.1	171.5	**532**

Table 2: Welfare estimates with 15 firms in the industry.

Period/number of adopters	0	1	2	3	4	5	10	15	Total all 16 periods
Emission tax									
Start of period/date of adoption	0.0	32.0	33.5	35.0	36.6	38.3	48.1	62.2	
Present value investment cost	0.0	1.1	1.0	0.9	0.7	0.6	0.3	0.1	
Current value consumer surplus rate	28.1	28.4	28.6	28.8	29.1	29.3	30.5	31.8	
Current value firms' profit rate	3.8	4.0	4.2	4.4	4.6	4.7	4.9	4.2	
Current value tax revenue rate	7.5	7.0	6.6	6.2	5.9	5.5	4.4	4.0	
Current value environmental damage rate	7.5	7.0	6.6	6.2	5.9	5.5	4.4	4.0	
Current value welfare rate (excl investment)	31.9	32.4	32.8	33.3	33.7	34.1	35.4	36.0	
Present value total period welfare	453.5	5.5	5.4	5.3	5.2	5.1	4.1	14.3	**531**
Exogenous refunded tax									
Start of period/date of adoption	0.0	29.7	31.1	32.7	34.3	36.0	45.7	59.0	
Present value investment cost	0.0	1.4	1.2	1.1	0.9	0.8	0.3	0.1	
Current value consumer surplus rate	35.6	35.4	35.2	35.0	34.9	34.8	34.8	35.6	
Current value firms' profit rate	4.7	5.0	5.1	5.3	5.4	5.5	5.5	4.7	
Current value tax revenue rate	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
Current value environmental damage rate	8.4	7.9	7.4	6.9	6.5	6.1	4.7	4.2	
Current value welfare rate (excl investment)	31.9	32.4	32.9	33.4	33.8	34.2	35.6	36.1	
Present value total period welfare	442.1	6.4	6.3	6.1	6.0	5.8	4.6	17.4	**531**
Endogenous refunded tax									
Start of period/date of adoption	0.0	32.1	33.3	34.6	36.0	37.4	46.2	59.0	
Present value investment cost	0.0	1.1	1.0	0.9	0.8	0.7	0.3	0.1	
Current value consumer surplus rate	35.6	35.4	35.2	35.1	35.0	34.9	34.8	35.6	
Current value firms' profit rate	4.7	4.9	5.1	5.2	5.3	5.4	5.4	4.7	
Current value tax revenue rate	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
Current value environmental damage rate	8.4	7.9	7.4	7.0	6.6	6.2	4.8	4.2	
Current value welfare rate (excl investment)	31.9	32.4	32.9	33.3	33.7	34.1	35.5	36.1	
Present value total period welfare	454.3	4.5	4.5	4.6	4.6	4.6	4.2	17.4	**531**

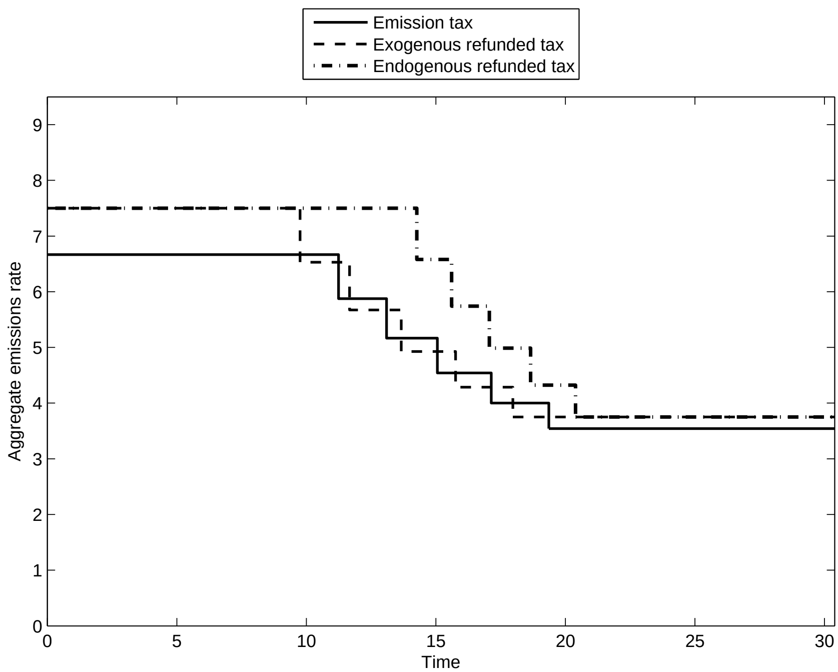

Figure 5: Aggregate emissions over time with 5 firms in the industry.

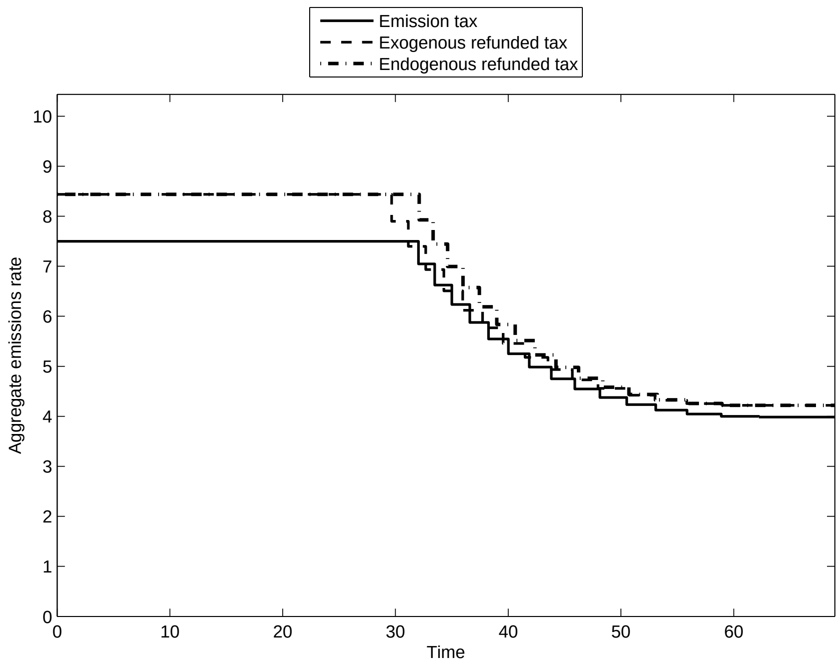

Figure 6: Aggregate emissions over time with 15 firms in the industry.

Appendices

A Demonstration of Nash equilibrium - Reinganum (1981)

(1i) $\pi_0(m-1) \geq 0$ and $\pi_1(m) \geq 0$

(1ii) $\pi_1(m-1) - \pi_0(m-2) > \pi_1(m) - \pi_0(m-1) > 0$ for all $m < n$.

(2i) $p'(0) \leq \pi_0(0) - \pi_1(1)$

(2ii) $\lim_{t \longrightarrow \infty} p'(t) > 0$

(2iii) $p''(t) > re^{-rt} (\pi_1(1) - \pi_0(0))$

Demonstration very similar to Reinganum (1981).

Proposition Given a weak ordering of adoption dates $\tau_1 \leq \tau_2 \leq ... \leq \tau_n$, each firm has a unique optimal adoption date τ_i^* such that $0 \leq \tau_1^* < \tau_2^* < .. < \tau_n^* < \infty$.

Proof From assumption 1 and 2iii, V^i is strictly concave in τ_i for $\tau_i \in (\tau_{i-1}, \tau_{i+1})$, so first-order conditions are necessary and sufficient for finding an optimal date of adoption τ_i^*.

Furthermore, by assumption 2i $\frac{\partial V^1}{\partial \tau_1}_{\tau_1=0} = \pi_0(0) - \pi_1(1) - p'(0) \geq 0$ and thus $\tau_1^* \geq 0$. By assumption 2ii $\lim_{t \longrightarrow \infty} p'(t) > 0$, it also follows that $\lim_{\tau_n \longrightarrow \infty} \frac{\partial V^n}{\partial \tau_n} = -p'(\tau_n) < 0$ which implies that $\tau_n^* < \infty$.

We also need to show that $\tau_i^* \in (\tau_{i-1}^*, \tau_{i+1}^*)$. If we evaluate $\frac{\partial V^i}{\partial \tau_i}$ at $\tau_i = \tau_{i-1}^*$, we get

$$\begin{aligned} \frac{\partial V^i}{\partial \tau_i}_{\tau_i=\tau_{i-1}^*} &= (\pi_0(i-1) - \pi_1(i))\, e^{-r\tau_{i-1}^*} - p'(\tau_{i-1}^*) \\ &= (\pi_0(i-1) - \pi_1(i))\, e^{-r\tau_{i-1}^*} - (\pi_0(i-2) - \pi_1(i-1))\, e^{-r\tau_{i-1}^*} \end{aligned}$$

which is strictly positive by assumption 1ii.

Similarly, we evaluate $\frac{\partial V^i}{\partial \tau_i}$ at $\tau_i = \tau_{i+1}^*$

$$\begin{aligned} \frac{\partial V^i}{\partial \tau_i}_{\tau_i=\tau_{i+1}^*} &= (\pi_0(i-1) - \pi_1(i))\, e^{-r\tau_{i+1}^*} - p'(\tau_{i+1}^*) \\ &= (\pi_0(i-1) - \pi_1(i))\, e^{-r\tau_{i-1}^*} - (\pi_0(i) - \pi_1(i+1))\, e^{-r\tau_{i-1}^*} \end{aligned}$$

which is strictly negative by assumption 1ii.

Since V^i is strictly concave in τ_i and $\frac{\partial V^i}{\partial \tau_i}_{\tau_i=\tau_i^*} = 0$, the unique maximum is achieved at

$\tau_i^* \in (\tau_{i-1}^*, \tau_{i+1}^*)$. *Q.E.D.*

We also need to demonstrate that $\tau^* = (\tau_1^*, \tau_2^*, .., \tau_n^*)$ is a Nash equilibrium.

Proposition $\tau^* = (\tau_1^*, \tau_2^*, .., \tau_n^*)$ as defined by (1) is a Nash equilibrium in adoption dates.

Proof If $\tau^* = (\tau_1^*, \tau_2^*, .., \tau_n^*)$ is a Nash equilibrium it must be that, given $\tau_1^*, \tau_2^*, .., \tau_{i-1}^*, \tau_{i+1}^*, ..., \tau_n^*$, i will prefer τ_i^* to any other date T. First, suppose i chooses a $T \in \left[\tau_{k-1}^*, \tau_k^*\right]$ where $k < i$

$$\begin{aligned} V^i(\tau_1^*, \tau_2^*, .., \tau_{i-1}^*, \tau_{i+1}^*, ..., \tau_n^*, T) = & \sum_{m=0}^{k-2} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_0(m) e^{-rt} dt + \int_{\tau_{k-1}^*}^{T} \pi_0(k-1) e^{-rt} dt \\ & + \int_{T}^{\tau_k^*} \pi_1(k) e^{-rt} dt + \sum_{m=k}^{i-2} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_1(m+1) e^{-rt} dt \\ & + \int_{\tau_{i-1}^*}^{\tau_{i+1}^*} \pi_1(i) e^{-rt} dt + \sum_{m=i+1}^{n} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_1(m) e^{-rt} dt - p(T) \end{aligned}$$

Maximizing with respect to T gives

$$\left(\pi_0(k-1) - \pi_1(k)\right) e^{-rT^*} - p'(T^*) = 0$$

That is $T^* = \tau_k^*$. That is, in each interval $\left[\tau_{k-1}^*, \tau_k^*\right]$, with $k < i$, V^i reaches its maximum at the right boundary τ_k^*.

Next, suppose i chooses a $T \in \left[\tau_k^*, \tau_{k+1}^*\right]$ where $k > i$

$$\begin{aligned} V^i(\tau_1^*, \tau_2^*, .., \tau_{i-1}^*, \tau_{i+1}^*, ..., \tau_n^*, T) = & \sum_{m=0}^{i-2} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_0(m) e^{-rt} dt + \int_{\tau_{i-1}^*}^{\tau_{i+1}^*} \pi_0(i-1) e^{-rt} dt \\ & + \sum_{m=i}^{k-2} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_0(m) e^{-rt} dt + \int_{\tau_k^*}^{T} \pi_0(k-1) e^{-rt} dt \\ & + \int_{T}^{\tau_{k+1}^*} \pi_1(k) e^{-rt} dt + \sum_{m=k+1}^{n} \int_{\tau_m^*}^{\tau_{m+1}^*} \pi_1(m) e^{-rt} dt - p(T) \end{aligned}$$

Maximizing with respect to T gives

$$(\pi_0(k-1) - \pi_1(k))\, e^{-rT^*} - p'(T^*) = 0$$

That is, $T^* = \tau_k^*$. That is, in each interval $\left[\tau_k^*, \tau_{k+1}^*\right]$, with $k > i$, V^i reaches its maximum at the left boundary τ_k^*. Thus, the maximum of V^i must be in $\left[\tau_{i-1}^*, \tau_{i+1}^*\right]$. We already know from the previous demonstration that the maximum on that interval is τ_i^*. We have thus demonstrated that, given $\tau_1^*, \tau_2^*, .., \tau_{i-1}^*, \tau_{i+1}^*, ..., \tau_n^*$, i will prefer $T = \tau_i^*$ to all other $T \in [0, \infty)$. $\tau^* = (\tau_1^*, \tau_2^*, .., \tau_n^*)$ is therefore a Nash equilibrium. *Q.E.D*

A.1 Assumption 1ii)

For the existence of a Nash equilibrium, we need to check that assumption 1ii holds under the different policies.

A.1.1 Emission tax

Let us consider first the case of taxes. Let $\zeta_1^T = c_1 + \sigma\varepsilon_1$, $\zeta_0^T = c_0 + \sigma\varepsilon_0$ and $\rho = \frac{1}{b[n+1]^2}$. Then,

$$\pi_1(m_1 - 1) = \rho \left[a - [n - m_1 + 2]\, \zeta_1^T + [n - m_1 + 1]\, \zeta_0^T\right]^2,$$
$$\pi_0(m_1 - 2) = \rho \left[a + [m_1 - 2]\, \zeta_1^T - [m_1 - 1]\, \zeta_0^T\right]^2,$$

and thus $\Delta\pi_{m_1-1}^T = \pi_1(m_1 - 1) - \pi_0(m_1 - 2)$ is equal to:

$$\Delta\pi_{m_1-1}^T = \rho \left[\zeta_1^T + \zeta_0^T\right] \left[n^2 \left[\zeta_1^T + \zeta_0^T\right] - 2n \left[a + [m_1 - 2]\, \zeta_1^T + [m_1 - 1]\, \zeta_0^T\right]\right].$$

By analogy, $\Delta\pi_{m_1}^T = \pi_1(m_1) - \pi_0(m_1 - 1)$ is equal to:,

$$\Delta\pi_{m_1}^T = \rho \left[\zeta_1^T + \zeta_0^T\right] \left[n^2 \left[\zeta_1^T + \zeta_0^T\right] - 2n \left[a + [m_1 - 1]\, \zeta_1^T + m_1\zeta_0^T\right]\right].$$

and hence:

$$\Delta\pi_{m_1-1}^T - \Delta\pi_{m_1}^T = 2n\rho \left[\zeta_1^T + \zeta_0^T\right]^2 > 0 \ \vee \vee m_1 \geq 2.$$

That is, assumption 1ii holds under the emission tax.

A.1.2 Exogenous refunded tax

Since under the exogenously refunded tax $\bar{\varepsilon}^X(m_1) = \bar{\varepsilon}^X(m_1-1)$, $\Delta\pi^X_{m_1-1} - \Delta\pi^X_{m_1}$ can be represented as:

$$\Delta\pi^X_{m_1-1} - \Delta\pi^X_{m_1} = \Delta\pi^T_{m_1-1} - \Delta\pi^T_{m_1} + \tag{20}$$
$$\frac{2\sigma\bar{\varepsilon}^X(m_1)}{b\left[n+1\right]}\left[\left[q^T_1(m_1-1) - q^T_0(m_1-2)\right] - \left[q^T_1(m_1) - q^T_0(m_1-1)\right]\right], \tag{21}$$

Since $q^T_1(m_1) - q^T_0(m_1-1) = \frac{n\left[\zeta^T_1 - \zeta^T_0\right]}{b[n+1]}$, equation (20) simplifies to:

$$\Delta\pi^X_{m_1-1} - \Delta\pi^X_{m_1} = 2n\rho\left[\zeta^T_1 + \zeta^T_0\right]^2 > 0 \ \vee m_1 \geq 2.$$

A.1.3 Endogenous refunded tax

$$\begin{aligned}\Delta\pi^D_{m_1-1} - \Delta\pi^D_{m_1} = b&\left[1 - \frac{\sigma}{bQ^D(m_1-1)}\left[\varepsilon_1 - \bar{\varepsilon}^D(m_1-1)\right]\right]\left[q^D_1(m_1-1)\right]^2\\ - b&\left[1 - \frac{\sigma}{bQ^D(m_1-2)}\left[\varepsilon_0 - \bar{\varepsilon}^D(m_1-2)\right]\right]\left[q^D_0(m_1-2)\right]^2\\ - b&\left[1 - \frac{\sigma}{bQ^D(m_1)}\left[\varepsilon_1 - \bar{\varepsilon}^D(m_1)\right]\right]\left[q^D_1(m_1)\right]^2\\ + b&\left[1 - \frac{\sigma}{bQ^D(m_1-1)}\left[\varepsilon_0 - \bar{\varepsilon}^D(m_1-1)\right]\right]\left[q^D_0(m_1-1)\right]^2.\end{aligned}$$

Since $\varepsilon_0 - \bar{\varepsilon}^D(m_1) = m_1 s^D_1(m)\delta$, and $\varepsilon_1 - \bar{\varepsilon}^D(m_1) = -\left[n-m_1\right]s^D_0(m_1)\delta$, this equation can be represented as:

$$\begin{aligned}\Delta\pi^D_{m_1-1} - \Delta\pi^D_{m_1} = b&\left[1 + \frac{\sigma\delta\left[n-m_1+1\right]s^D_0(m_1-1)}{bQ^D(m_1-1)}\right]\left[q^D_1(m_1-1)\right]^2\\ - b&\left[1 - \frac{\sigma\delta\left[m_1-2\right]s^D_1(m_1-2)}{bQ^D(m_1-2)}\right]\left[q^D_0(m_1-2)\right]^2\\ - b&\left[1 + \frac{\sigma\delta\left[n-m_1\right]s^D_0(m_1)}{bQ^D(m_1)}\right]\left[q^D_1(m_1)\right]^2\\ + b&\left[1 - \frac{\sigma\delta\left[m_1-1\right]s^D_1(m_1-1)}{bQ^D(m_1-1)}\right]\left[q^D_0(m_1-1)\right]^2.\end{aligned}$$

$\Delta\pi^D_{m_1-1} - \Delta\pi^D_{m_1} > 0 \vee m_1 \geq 2$ if and only if:

$$
\begin{aligned}
& b\left[\left[q_1^D(m_1-1)\right]^2 - \left[q_1^D(m_1)\right]^2\right] \\
& + \sigma\delta\,[n-m_1]\left[s_0^D(m_1-1)s_1^D(m_1-1)q_1^D(m_1-1) - s_0^D(m_1)s_1^D(m_1)q_1^D(m_1)\right] \\
& + \sigma\delta s_0^D(m_1-1)s_1^D(m_1-1)q_1^D(m_1-1) \\
& > \\
& b\left[\left[q_0^D(m_1-2)\right]^2 - \left[q_0^D(m_1-1)\right]^2\right] \\
& + \sigma\delta\,[m_1-1]\left[s_0^D(m_1-1)s_1^D(m_1-1)q_0^D(m_1-1) - s_0^D(m_1-2)s_1^D(m_1-2)q_0^D(m_1-2)\right] \\
& + \sigma\delta s_0^D(m_1-2)s_1^D(m_1-2)q_0^D(m_1-2).
\end{aligned}
$$

We have that $s_0^D(m_1-1)s_1^D(m_1-1)q_j^D(m_1-1) \simeq s_0^D(m_1)s_1^D(m_1)q_j^D(m_1) \vee m_1 \neq 1, n$ & $j \in \{0,1\}$[10] and thus this expression simplifies to:

$$
\begin{aligned}
& b\left[\left[q_1^D(m_1-1)\right]^2 - \left[q_0^D(m_1-2)\right]^2\right] + \sigma\delta s_0^D(m_1-1)s_1^D(m_1-1)q_1^D(m_1-1) \\
& > \\
& b\left[\left[q_1^D(m_1)\right]^2 - \left[q_0^D(m_1-1)\right]^2\right] + \sigma\delta s_0^D(m_1-2)s_1^D(m_1-2)q_0^D(m_1-2).
\end{aligned}
$$

Finally, if $q_1^D(m_1) > q_0^D(m_1-1) \vee m_1 \geq 2$, we expect this condition to be satisfied.

B Cournot equilbrium with two technologies

We leave the completely analogous derivation of the equiilbrium for three technologies to the interested reader.

B.1 Emission tax

Let $\zeta_0^T = c_0 + \sigma\varepsilon_0$ denote marginal costs inclusive of emission tax payments under an emission tax before adoption of the new technology and let $\zeta_1^T = c_1 + \sigma\varepsilon_1$ denote marginal costs after adoption. If we have m_1 adopters of the new technology and rank the firms according to their order in the adoption sequence (taking it as given), we can write the profit rate maximization problem for the adopters as

[10]Note that $s_1^D(0) = 0$, and $s_0^D(n) = 0$.

$$\pi^j = \max_{q^j} \left[P(Q) - \zeta_1^T \right] q^j,$$

for $j \leq m_1$,

and the profit maximization problem for the $n - m_1$ non-adopters as

$$\pi^j = \max_{q^j} \left[P(Q) - \zeta_0^T \right] q^j,$$

for $j > m_1$

Substituting in $P(Q) = a - b \sum_{i=1}^{n} q^i$, the first order condition for the adopters is

$$q^j = \frac{a - b \sum_{i \neq j} q^i - \zeta_1^T}{2b},$$

and the first order condition for the non-adopters

$$q^j = \frac{a - b \sum_{i \neq k} q^i - \zeta_0^T}{2b}.$$

Since the m_1 adopters are symmetric they will all have the same profit-maximizing level of production. We denote this profit-maximizing level q_1^T. Similarly, the level of production is the same for all $n - m_1$ non-adopters and we denote this profit-maximizing level q_0^T. We thus have the following equilibrium conditions

$$q_1^T = \frac{a - b \left[[m_1 - 1] \, q_1^T + [n - m_1] \, q_0^T \right] - \zeta_1^T}{2b},$$

$$q_0^T = \frac{a - b \left[m_1 q_1^T + [n - m_1 - 1] \, q_0^T \right] - \zeta_0^T}{2b}.$$

Solving for q_1^T and q_0^T, we find the levels of equilibrium output for adopters and non-adopters, respectively

$$q_1^T(m) = \frac{a - \zeta_1^T + [n - m_1] \left[\zeta_0^T - \zeta_1^T \right]}{b \, [n + 1]},$$

$$q_0^T(m) = \frac{a - \zeta_0^T - m_1 \left[\zeta_0^T - \zeta_1^T \right]}{b \, [n + 1]}, \tag{22}$$

yielding equilibrium price

$$P^T(m) = \frac{a + m_1 \zeta_1^T + [n - m_1] \, \zeta_0^T}{[n + 1]},$$

and equilibrium profits for adopters and non-adopters, respectively,

$$\pi_1^T(m_1) = \left[P^T(m_1) - \zeta_1^T\right] q_1^T(m_1) = \left[\frac{a + m_1\zeta_1^T + [n - m_1]\,\zeta_0^T}{[n+1]} - \zeta_1\right] q_1^T(m_1) = b\left[q_1^T(m_1)\right]^2.$$

$$\pi_0^T(m_1) = \left[P^T(m_1) - \zeta_0^T\right] q_0^T(m_1) = \left[\frac{a + m_1\zeta_1^T + [n - m_1]\,\zeta_0^T}{[n+1]} - \zeta_0\right] q_0^T(m_1) = b\left[q_0^T(m_1)\right]^2.$$

For interior solutions with $q_0^T(m_1) > 0$ for all $m_1 < n$, we see from (22) that this requires $a - \zeta_0 - [n-1]\,[\zeta_0 - \zeta_1] > 0$ to be true.

B.2 Exogenously refunded tax

Similarly, under an exogenously refunded tax, the profit maximization problems for the adopters and non-adopters, respectively, are

$$\pi^j = \max_{q^j}\left[[P(Q) - c_1 - \sigma\varepsilon_1]\,q^j + \sigma E\frac{q^j}{Q}\right] = \max_{q^j}\left[P(Q) - \zeta_1^T + \sigma\bar{\varepsilon}\right] q^j, \tag{23}$$

for $j = 1, 2, ..., m_1 - 1, m_1$,

$$\pi^j = \max_{q^j}\,[P(Q) - c_0 - \sigma\varepsilon_0]\,q^j + \sigma E\frac{q^j}{Q} = \max_{q^j}\left[P(Q) - \zeta_0^T + \sigma\bar{\varepsilon}\right] q^j, \tag{24}$$

for $j = m_1 + 1, m_1 + 2, ..., n - 1, n$.

with aggregate emissions (E) and aggregate output (Q) are given by:

$$E = \sum_{i=1}^{n} \varepsilon^i q^i.$$
$$Q = \sum_{i=1}^{n} q^i.$$

First-order conditions for the adopters and non-adopters are

$$q^j = \frac{a - b\sum_{i\neq j} q^i - \zeta_1^T + \sigma\bar{\varepsilon}}{2b},$$

for $j = 1, 2, ..., m_1 - 1, m_1$, and

$$q^j = \frac{a - b\sum_{i\neq j} q^i - \zeta_0^T + \sigma\bar{\varepsilon}}{2b},$$

for $j = m_1 + 1, m_1 + 2, ..., n-1, n$.

Substituting in the profit-maximizing levels q_1^X for the m_1 adopters and q_0^X for the $n - m_1$ non-adopters, we can write

$$q_1^X(m_1) = \frac{a - \zeta_1^T + [n - m_1]\left[\zeta_0^T - \zeta_1^T\right] + \sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]} = q_1^T(m_1) + \frac{\sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]},$$
$$q_0^X(m_1) = \frac{a - \zeta_0^T - m_1\left[\zeta_0^T - \zeta_1^T\right] + \sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]} = q_0^T(m_1) + \frac{\sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]}.$$

and the average emission intensity is $\bar{\varepsilon}^X(m_1) = \frac{m_1\varepsilon_1 q_1^X + [n-m_1]\varepsilon_0 q_0^X}{m_1 q_1^X + [n-m_1] q_0^X} > 0$.

$$P^X(m) = \frac{a + m_1\zeta_1^T + [n - m_1]\,\zeta_0^T - \sigma n\bar{\varepsilon}^X(m_1)}{[n+1]}$$

Equilibrium profits for adopters and non-adopters, respectively,

$$\pi_1^X(m_1) = \left[P^X(m_1) - \zeta_1^T + \sigma\bar{\varepsilon}^X(m_1)\right] q_1^X(m_1) = \left[\frac{a - \zeta_1^T + [n - m_1]\left[\zeta_0^T - \zeta_1^T\right] + \sigma\bar{\varepsilon}^X(m_1)}{[n+1]}\right] q_1^X(m_1),$$
$$= b\left[q_1^X(m_1)\right]^2,$$
$$\pi_0^X(m_1) = \left[P^X(m_1) - \zeta_0^T + \sigma\bar{\varepsilon}^X(m_1)\right] q_0^X(m_1) = \left[\frac{a - \zeta_0^T + m_1\left[\zeta_1^T - \zeta_0^T\right] + \sigma\bar{\varepsilon}^X(m_1)}{[n+1]}\right] q_0^x(m_1),$$
$$= b\left[q_0^X(m_1)\right]^2.$$

B.3 Endogenously refunded tax

Under an endogenously refunded tax, the profit maximization problems for the adopters and non-adopters are the same as the ones for the exogenously refunded tax found in (23) and (24), respectively. However, when the firm recognizes that it can influence the size of the refund, the first-order conditions are

$$q^j = \frac{a - b\sum_{i\neq j} q^i - c_1 - \sigma\left[\varepsilon_1 - \bar{\varepsilon}\right]}{\left[2b - \sigma\left[\varepsilon_1 - \bar{\varepsilon}\right]\frac{1}{Q}\right]},$$

for $j = 1, 2, ..., m_1 - 1, m_1$,

$$q^j = \frac{a - b\sum_{i\neq j} q^i - c_0 - \sigma\left[\varepsilon_0 - \bar{\varepsilon}\right]}{\left[2b - \sigma\left[\varepsilon_0 - \bar{\varepsilon}\right]\frac{1}{Q}\right]},$$

for $j = m_1 + 1, m_1 + 2, ..., n-1, n$.

Substituting in the profit-maximizing levels q_1^D for the m_1 adopters and q_0^D for the $n - m_1$ non-adopters, and suppressing the argument of m_1 for clarity, we can write

$$q_1^D = \frac{\left[1 - \frac{\sigma}{bQ^D}\left[\varepsilon_0 - \bar{\varepsilon}^D\right]\right]\left[a - \zeta_1^T + \sigma\bar{\varepsilon}^D\right] + \left[n - m_1\right]\left[\zeta_0^T - \zeta_1^T\right]}{\phi}, \tag{25}$$

$$q_0^D = \frac{\left[1 - \frac{\sigma}{bQ^D}\left[\varepsilon_1 - \bar{\varepsilon}^D\right]\right]\left[a - \zeta_0^T + \sigma\bar{\varepsilon}^D\right] - m_1\left[\zeta_0^T - \zeta_1^T\right]}{\phi}, \tag{26}$$

where

$$\phi = b\left[n+1\right] + \frac{1}{b}\left[\frac{\sigma}{Q^D}\right]^2\left[\varepsilon_1 - \bar{\varepsilon}^D\right]\left[\varepsilon_0 - \bar{\varepsilon}^D\right] - \frac{\sigma}{Q^D}\left[\left[\varepsilon_1 - \bar{\varepsilon}^D\right]\left[n - m_1 + 1\right] + \left[\varepsilon_0 - \bar{\varepsilon}^D\right]\left[m_1 + 1\right]\right] > 0 \quad \forall\, m.$$

Substituting (25) and (26) into $P^D = a - bQ^D$ with $Q^D = m_1 q_1^D + [n - m_1]\, q_0^D$, we get

$$\begin{aligned} \pi_1^D &= \left[P^D - \zeta_1^T + \sigma\bar{\varepsilon}^D\right] q_1^D, \\ &= b\left[1 - \frac{\sigma}{bQ^D}\left[\varepsilon_1 - \bar{\varepsilon}^D\right]\right]\left[q_1^D\right]^2. \end{aligned}$$

$$\begin{aligned} \pi_0^D &= \left[P^D - \zeta_0^T + \sigma\bar{\varepsilon}^D\right] q_0^D, \\ &= b\left[1 - \frac{\sigma}{bQ^D}\left[\varepsilon_0 - \bar{\varepsilon}^D\right]\right]\left[q_0^D\right]^2. \end{aligned}$$

C Comparison of endogenous versus exogenous refunding

Rewriting the equilibrium conditions (10) for the m_1 adopters as

$$a - bQ^D(m_1) - bq_1^D(m_1) = c_1 + \sigma\left[\varepsilon_1 - \bar{\varepsilon}^D(m_1)\right]\left[1 - \frac{q_1^D(m_1)}{Q^D}\right],$$

and (11) for the $n - m_1$ adopters as

$$a - bQ^D(m_1) - bq_0^D(m_1) = c_0 + \sigma\left[\varepsilon_0 - \bar{\varepsilon}^D(m_1)\right]\left[1 - \frac{q_0^D(m_1)}{Q^D}\right],$$

we can sum over all n conditions to get

$$m_1\left[a - bQ^D(m_1) - bq_1^D(m_1)\right] + [n-m_1]\left[a - bQ^D(m_1) - bq_0^D(m_1)\right]$$
$$= m_1\left[c_1 + \sigma\left[\varepsilon_1 - \bar{\varepsilon}^D(m_1)\right]\left[1 - \frac{q_1^D(m_1)}{Q^D}\right]\right] + [n-m_1]\left[c_0 + \sigma\left[\varepsilon_0 - \bar{\varepsilon}^D(m_1)\right]\left[1 - \frac{q_0^D(m_1)}{Q^D}\right]\right]$$

This simplifies to

$$\begin{aligned} na - [n+1]\, bQ^D(m_1) &= m_1\zeta_1^T + [n-m_1]\,\zeta_0^T - n\sigma\bar{\varepsilon}^D(m_1) \\ &\quad - \sigma\left[m_1\varepsilon_1\frac{q_1^D(m_1)}{Q^D} + [n-m_1]\,\varepsilon_0\frac{q_0^D(m_1)}{Q^D}\right] \\ &\quad + \sigma\bar{\varepsilon}^D(m_1)\left[m_1\frac{q_1^D(m_1)}{Q^D} + [n-m_1]\,\frac{q_0^D(m_1)}{Q^D}\right] \end{aligned}$$

yielding

$$Q^D(m_1) = \frac{na - m_1\zeta_1^T - [n-m_1]\,\zeta_0^T + n\sigma\bar{\varepsilon}^D(m_1)}{b\,[n+1]}.$$

Similarly, using the n equilibrium conditions in (4) and (5), we get

$$Q^X(m_1) = \frac{na - m_1\zeta_1^T - [n-m_1]\,\zeta_0^T + n\sigma\bar{\varepsilon}^X(m_1)}{b\,[n+1]}.$$

Hence,

$$Q^D(m_1) - Q^X(m_1) = \frac{n\sigma\left[\bar{\varepsilon}^D(m_1) - \bar{\varepsilon}^X(m_1)\right]}{b\,[n+1]},$$

The first-order conditions under policy $k \in \{T, X, D\}$ and technology $j \in \{0, 1\}$ can also be written

$$a - bQ^k - bq_j^k = \psi_j^k.$$

where ψ_j^k denotes the marginal cost inclusive of the costs of the emissions policy. We drop the argument of m_1 for clarity. We can then write

$$q_j^k = \frac{a - \psi_j^k}{b} - Q^k,$$

with

$$\psi_j^T = \zeta_j^T,$$
$$\psi_j^X = c_j + \sigma\left[\varepsilon_j - \bar{\varepsilon}^R\right],$$
$$\psi_j^D = c_j + \sigma\left[\varepsilon_j - \bar{\varepsilon}^R\right]\left[1 - \frac{q_j^D}{Q^D}\right].$$

Comparing equilibrium quantities under exogenous and endogenous refunding for adopters, we can write

$$q_1^X - q_1^D = \frac{\psi_1^D - \psi_1^X}{b} + Q^D - Q^X,$$
$$= \frac{c_1 + \sigma\left[\varepsilon_1 - \bar{\varepsilon}^D\right]\left[1 - \frac{q_1^D}{Q^D}\right] - \left[c_1 + \sigma\left[\varepsilon_1 - \bar{\varepsilon}^X\right]\right]}{b} + \frac{n\sigma}{b\left[n+1\right]}\left[\bar{\varepsilon}^D - \bar{\varepsilon}^X\right],$$
$$= \sigma\frac{\left[\bar{\varepsilon}^D - \varepsilon_1\right] q_1^D\left[n+1\right] - \left[\bar{\varepsilon}^D - \bar{\varepsilon}^X\right] Q^D}{b\left[n+1\right] Q^D} > 0,$$

since $\left[\bar{\varepsilon}^D\left(m_1\right) - \varepsilon_1\right] > \left[\bar{\varepsilon}^D\left(m_1\right) - \bar{\varepsilon}^X\left(m_1\right)\right]$ and $q_1^D\left(m_1\right)\left[n+1\right] > Q^D\left(m_1\right)$ for $0 < m_1 < n$.

Furthermore, for non-adopters, we can write

$$q_0^X - q_0^D = \frac{\psi_0^D - \psi_0^X}{b} + Q^D - Q^X,$$
$$= \frac{\left[c_0 + \sigma\left[\varepsilon_0 - \bar{\varepsilon}^D\right]\left[1 - \frac{q_0^D}{Q^D}\right]\right] - \left[c_0 + \sigma\left[\varepsilon_0 - \bar{\varepsilon}^X\right]\right]}{b} + \frac{n\sigma}{b\left[n+1\right]}\left[\bar{\varepsilon}^D - \bar{\varepsilon}^X\right],$$
$$= \frac{-\sigma\left[\varepsilon_0 - \bar{\varepsilon}^D\right] q_0^D\left[n+1\right] + \sigma\left[\bar{\varepsilon}^X - \bar{\varepsilon}^D\right] Q^D}{b\left[n+1\right] Q^D}.$$
$$= \frac{-\sigma m_1 \delta s_1^D\left[n+1\right] q_0^D - \sigma\left[s_1^X - s_1^D\right] m_1 \delta Q^D}{b\left[n+1\right] Q^D}.$$

which implies that $q_0^X < q_0^D \vee s_1^X \geq s_1^D$.

D Optimal adoption times with three technologies

Given an ordering of adoption dates $\tau_1 \leq \tau_2 \leq ... \leq \tau_j \leq ... \leq \tau_n$ for technology 2, where the first $n-k$ adopters switch from technology 0 and the following k adopters switch from technology 1, we can write the present value of adopting technology 2 for firm j at τ_j as

$$V_2^j(\tau_1, ..., \tau_{j-1}, \tau_j, \tau_{j+1}, ..., \tau_n) = \sum_{m_2=0}^{j-1} \int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_0(k, m_2) e^{-r[\ t - t_2]} dt$$
$$+ \sum_{m_2=j}^{n-k} \int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_2(k, m_2) e^{-r[\ t - t_2]}\ dt$$
$$+ \sum_{m_2=n-k+1}^{n} \int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_2(n - m_2, m_2) e^{-r[\ t - t_2]}\ dt - p_2(\tau_j)$$

for $j=1,2,...,n-k$ and

$$V_2^j(\tau_1,...,\tau_{j-1},\tau_j,\tau_{j+1},...,\tau_n) = \sum_{m_2=0}^{n-k-1}\int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_1(k,m_2)e^{-r[\ t-t_2]}\,dt$$
$$+\sum_{m_2=n-k}^{j-1}\int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_1(n-m_2,m_2)e^{-r[\ t-t_2]}\,dt$$
$$+\sum_{m_2=j}^{n}\int_{\tau_{m_2}}^{\tau_{m_2+1}} \pi_2(n-m_2,m_2)e^{-r[\ t-t_2]}\,dt - p_2(\tau_j)$$

for $j=n-k+1,n-k+2,...,n-1,n$ and where $\tau_0=t_2$ and $\tau_{n+1}=\infty$.

From the assumptions that $\pi_2(m_1,m_2) > \pi_1(m_1,m_2) > \pi_0(m_1,m_2) \geq 0$, $\pi_2(m_1,m_2+1)-\pi_0(m_1,m_2) > \pi_2(m_1-1,m_2+1)-\pi_1(m_1,m_2)$ for all m_1,m_2 for which $m_1+m_2<n$ and $p_2''(t) > re^{-rt}\left(\pi_2(k,1)-\pi_0(k,0)\right)$, V_2^j is strictly concave in τ_j for $\tau_j\in\left[\tau_{j-1},\tau_{j+1}\right]$, which implies that first-order conditions are necessary and sufficient conditions for determining τ_j^*. The first-order conditions are

$$\frac{\partial V_2^j}{\partial \tau_j} = \left[\pi_0(k,j-1)-\pi_2(k,j)\right]e^{-r\left[\ \tau_j^*-t_2\right]} - p_2'(\tau_j^*) = 0 \tag{27}$$

for $j=1,2,...,n-k$ and

$$\frac{\partial V_2^j}{\partial \tau_j} = \left[\pi_1(n-j+1,j-1)-\pi_2(n-j,j)\right]e^{-r\left[\ \tau_j^*-t_2\right]} - p_2'(\tau_j^*) = 0 \tag{28}$$

for $j=n-k+1,...,n-1,n$.

We now define $\Delta\pi_{02,j}=\pi_2(k,j)-\pi_0(k,j-1)$ and $\Delta\pi_{12,j}=\pi_2(n-j,j)-\pi_1(n-j+1,j-1)$ and we can then write (27) and (28) as

$$\frac{\partial V_2^j}{\partial \tau_j} = -\Delta\pi_{02,j}e^{-r\left[\ \tau_j^*-t_2\right]} - p_2'(\tau_j^*) = 0$$

for $j=1,2,...,n-k$ and

$$\frac{\partial V_2^j}{\partial \tau_j} = -\Delta\pi_{12,j}e^{-r\left[\ \tau_j^*-t_2\right]} - p_2'(\tau_j^*) = 0$$

for $j=n-k+1,...,n-1,n$.

From the assumptions that $p_2(t) < p_1(t)e^{rt_2}$ and $c_2+\sigma\varepsilon_2 < c_1+\sigma\varepsilon_1$, we know that firms which have not adopted technology 1 by date t_2 will not have an incentive to adopt it after technology 2 has appeared at t_2. Therefore, at t_2, the firms only face the decision of when to adopt technology 2.

For the *laggards*, the increase in profit rate under the emission tax, $\Delta\pi^T_{02,j}$, for the firm which is the jth to adopt technology 2 and switches from technology 0 under a refunded tax, is given by:

$$\Delta\pi^T_{02,j} = \pi^T_2(k^T, j) - \pi^T_0(k^T, j-1) = b\left[q^T_2(k^T, j)\right]^2 - b\left[q^T_0(k^T, j-1)\right]^2$$

for $j = 1, 2., ..., n - k^T - 1, n - k^T$,

and the increase in profit rate under the exogenously refunded tax, $\Delta\pi^X_{02,j}$, by:

$$\begin{aligned}\Delta\pi^X_{02,j} &= \pi^X_2(k^X, j) - \pi^X_0(k^X, j-1) \\ &= b\left[q^T_2(k^X, j) + \frac{\sigma\bar{\varepsilon}^X(k^X, j)}{b\,[n+1]}\right]^2 - b\left[q^T_0(k^X, j-1) + \frac{\sigma\bar{\varepsilon}^X(k^X, j-1)}{b\,[n+1]}\right]^2\end{aligned}$$

for $j = 1, 2, ..., n - k^X - 1, n - k^X$.

The difference in the increase in profit rate from adoption of technology 2 under the exogenous refunded tax compared to the emission tax for the jth adopter of technology 2, which would switch from technology 0 under both policies, is thus equal to:

$$\begin{aligned}\Delta\pi^X_{02,j} - \Delta\pi^T_{02,j} = {} & b\left[q^T_2(k^X, j) + \frac{\sigma\bar{\varepsilon}^X(k^X, j)}{b\,[n+1]}\right]^2 - b\left[q^T_0(k^X, j-1) + \frac{\sigma\bar{\varepsilon}^X(k^X, j-1)}{b\,[n+1]}\right]^2 \\ & - \left[b\left[q^T_2(k^T, j)\right]^2 - b\left[q^T_0(k^T, j-1)\right]^2\right],\end{aligned}$$

for $j = 1, 2, ..., n - k^X - 1, n - k^X$. If, as before, we assume that one firm switching from technology 0 to technology 2 considers its own impact on the refund as negligible, i.e., $\bar{\varepsilon}^X(k^X, j) = \bar{\varepsilon}^X(k^X, j-1)$, and use that

$$q^T_2(k^X, j) - q^T_2(k^T, j) = q^T_0(k^X, j-1) - q^T_0(k^T, j-1) = \frac{[k^T - k^X][\zeta^T_0 - \zeta^T_2]}{b[n+1]},$$

and

$$q^T_2(k^X, j) - q^T_0(k^X, j-1) = q^T_2(k^T, j) - q^T_0(k^T, j-1) = \frac{n\,[\zeta^T_0 - \zeta^T_2]}{b\,[n+1]},$$

the difference in profit rate increase for the *laggards* simplifies to:

$$\Delta\pi^X_{02,j} - \Delta\pi^T_{02,j} = 2\frac{n\,[\zeta^T_0 - \zeta^T_2]}{b\,[n+1]^2}\left[\left[k^T - k^X\right]\left[\zeta^T_0 - \zeta^T_1\right] + \sigma\bar{\varepsilon}^X(k^X, j)\right].$$

The *intermediates* exist only if the number of firms which would have adopted technology 1 by t_2 under the emission tax is lower than the number of adopters of technology 1 at t_2 under the exogenous refunded tax, i.e., $k^T < k^X$. The jth adopter, for which $j \in \left[n - k^X + 1, n - k^T\right]$, would switch from technology 0 under an emission tax, and from technology 1 under a refunded tax. The difference in adoption time between the policies is then determined by the following difference:

$$\begin{aligned}\Delta\pi^X_{12,j} - \Delta\pi^T_{02,j} &= b\left[q^T_2(n-j,j) + \frac{\sigma\bar{\varepsilon}^X(n-j,j)}{b\,[n+1]}\right]^2 \\ &- b\left[q^T_1(n-j+1,j-1) + \frac{\sigma\bar{\varepsilon}^X(n-j+1,j-1)}{b\,[n+1]}\right]^2 \\ &- \left[b\left[q^T_2(k^T,j)\right]^2 - b\left[q^T_0(k^T,j-1)\right]^2\right]\end{aligned}$$

Since $\bar{\varepsilon}^X(n-j,j) = \bar{\varepsilon}^X(n-j+1,j-1)$ from the perspective of the firm, we have

$$\begin{aligned}q^T_2(n-j,j) - q^T_1(n-j+1,j-1) &= \frac{n\left[\zeta^T_1 - \zeta^T_2\right]}{b\,[n+1]} \\ q^T_2(n-j,j) + q^T_1(n-j+1,j-1) &= \frac{2a - [n-2\,[j+1]]\,\zeta^T_2 + [n-2j]\,\zeta^T_1}{b\,[n+1]} \\ q^T_2(k^T,j) + q^T_0(k^T,j-1) &= \frac{2a + [n-2j]\,\left[\zeta^T_0 - \zeta^T_2\right] - 2k^T\left[\zeta^T_0 - \zeta^T_1\right] - 2\zeta^T_2}{b\,[n+1]}\end{aligned}$$

, so that we can simplify the difference in profit rate increase to

$$\begin{aligned}\Delta\pi^X_{12,j} - \Delta\pi^T_{02,j} = &\frac{n\left[\zeta^T_0 - \zeta^T_1\right]}{b\,[n+1]^2}\left[2k^T\left[\zeta^T_0 - \zeta^T_2\right] - [n-2j]\left[\zeta^T_0 + \zeta^T_1 - 2\zeta^T_2\right] - 2\left[a + \zeta^T_2\right]\right] \\ &+ 2\frac{n[\zeta^T_1 - \zeta^T_2]}{b\,[n+1]^2}\sigma\bar{\varepsilon}^X(n-j,j)\end{aligned}$$

For the *early adopters* the increase in profit rate from adoption of technology 2 under an emission tax is

$$\begin{aligned}\Delta\pi^T_{12,j} &= \pi^T_2(n-j,j) - \pi^T_1(n-j+1,j-1) \\ &= b\left[q^T_2(n-j,j)^2 - \left[q^T_1(n-j+1,j-1)\right]^2\right]\end{aligned}$$

for $j = n - k^T + 1, n - k^T + 2, ..., n-1, n,$

and under an exogenous refunded tax:

$$\begin{aligned}\Delta\pi^X_{12,j} &= \pi^X_2(n-j,j) - \pi^X_1(n-j+1,j-1)\\ &= b\left[q^T_2(n-j,j) + \frac{\sigma\bar{\varepsilon}^X(n-j,j)}{b\,[n+1]}\right]^2\\ &- b\left[q^T_1(n-j+1,j-1) + \frac{\sigma\bar{\varepsilon}^X(n-j+1,j-1)}{b\,[n+1]}\right]^2\end{aligned}$$

for $j = n-k^X+1, n-k^X+2, ..., n-1, n$. Since $\bar{\varepsilon}^X(n-j,j) = \bar{\varepsilon}^X(n-j+1,j-1)$, the difference in profit rate increase is given by:

$$\Delta\pi^X_{12,j} - \Delta\pi^T_{12,j} = 2b\left[q^T_2(n-j,j) - q^T_1\,(n-j+1,j-1)\right]\frac{\sigma\bar{\varepsilon}^X(n-j,j)}{b\,[n+1]}$$

for $j = n-k^T+1, n-k^T+2, ..., n-1, n$.

Using similar tricks as before, this becomes:

$$\Delta\pi^X_{12,j} - \Delta\pi^T_{12,j} = \frac{2n\left[\zeta^T_1 - \zeta^T_2\right]}{b\,[n+1]^2}\sigma\bar{\varepsilon}^X(n-j,j)$$

for $j = n-k^T+1, n-k^T+2, ..., n$.

Paper IV

Diffusion of NO_x abatement technologies in Sweden*

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Abstract

This paper studies how different NO_x abatement technologies have diffused under the Swedish system of refunded emissions charges and analyzes the determinants of the time to adoption. The policy, under which the charge revenues are refunded back to the regulated firms in proportion to energy output, was explicitly designed to affect investment in NO_x-reducing technologies. The results indicate that paying a higher net NO_x charge increases the likelihood of adoption, but only for end-of-pipe post-combustion technologies. We also find some indication that market power considerations in the heat and power industry reduce the incentives to abate emissions through investment in post-combustion technologies. Adoption of post-combustion technologies and the efficiency improving technology of flue gas condensation is also more likely in the heat and power and waste incineration sectors, which is possibly explained by a large degree of public ownership in these sectors.

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

JEL Classification: H23, O33, O38, Q52

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1 Introduction

The long-run impact of emission regulations is mainly determined by the incentives they provide for innovation and diffusion of more environmentally benign technologies. In Sweden, a charge on NO_x emissions from large combustion plants was introduced in 1992, as a complement to the existing system of individual emission standards (SEPA, 2003). The regulation, under which the charge revenues are refunded back to the regulated firms in proportion to energy output, was explicitly designed to affect technology investment (Sterner & Turnheim, 2009). Judging from the significant reductions in emission intensities achieved since the introduction of the policy, this objective would appear to have been reached.

However, changes in emission intensities is the combined result of upfront investments in abatement technology, fuel switching, and improved knowledge of how to optimize the combustion process (Höglund-Isaksson & Sterner, 2009). Sterner & Turnheim (2009) sought to separate reductions in emission intensities at the regulated Swedish plants into contributions from technology diffusion versus innovation and found both factors very important. In this paper, we focus on one of these factors: the diffusion process for NO_x-reducing technologies.

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

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

Popp (2010) used the framework in Karshenas & Stoneman (1993) to analyze adoption of NO_x control technologies at coal-fired power plants in the US. He found that expectations of future technological advances slow down the diffusion of combustion modification technologies. Due to the differences in regulation across states, he could also identify environmental regulations as the dominating determinants behind adoption of both combustion as well as post-combustion technologies.

A recent strand of theoretical literature (e.g. Amir *et al.*, 2008; Baker *et al.*, 2008; Bauman *et al.*, 2008; Calel, 2011) have also highlighted the fact that different types of abatement technologies have different impacts on the marginal costs of abatement and production. This in turn implies that adoption incentives provided by environmental regulations differ across different types of abatement technologies and that the drivers behind the diffusion of these technologies are likely to be different.

A distinction is often made between end-of-pipe technologies that are add-on measures that curb emissions after their formation and clean technologies that reduce resource use and/or pollution at the source (Frondel *et al.*, 2007). Empirically, the drivers for investments in end-of-pipe versus clean technologies have been explored by Frondel *et al.* (2007) and Hammar & Löfgren (2010). The results of Frondel *et al.* (2007) suggest that regulatory measures and the stringency of environmental policy are positively correlated with investment in end-of-pipe technologies while investment in clean technologies seem to be motivated by market forces and the potential for cost savings. Hammar & Löfgren (2010) found that the price of energy is an important determinant for investments in end-of-pipe technologies while internal learning by doing as measured by expenditures on green R&D increase the probability of investment in clean technologies. Their results also suggested that the two types of technologies are complementary.

Other studies (e.g. Millock & Nauges, 2006; del Río González, 2005; Pizer *et al.*, 2001) identify cost savings, industry sector, plant size and financial strength such as self-financing capacity, profits and resources at the parent company as important determinants for adoption of clean technologies.

This study contributes to the literature on the drivers of technology diffusion by comparing the determinants of the time to adoption for three types of environmental technologies; first, technologies which reduce the formation of NO_x at combustion; second, end-of-pipe NO_x technologies; and lastly, technologies which improve energy efficiency. We do this for plants regulated under a particular type of earmarked tax system which redistributes the emission tax revenues from dirty to clean plants. Moreover, unlike many of the previous studies analyzing diffusion of abatement technologies, which focus on diffusion within one industry, we compare environmental technology adoption across different industry sectors.

We analyze the factors that affect the time and decision to invest in NO_x abatement technologies by applying a proportional hazard model. The factors analyzed best explain adoption of end-of-pipe post-combustion technologies. The results suggest that paying a higher NO_x charge net of the refund increases the likelihood of adoption of this type of technology. We also find indications of economies of scale and that market power considerations in the heat and power industry reduces the incentives to invest in post-combustion technologies. Adoption of post-combustion technologies and the efficiency improving technology of flue gas condensation is also more likely in the heat and power and waste incineration sectors.

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

2 NO_x policies

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

2.1 The Swedish NO_x charge

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

Because NO_x emissions vary significantly with temperature and other combustion parameters (Sterner & Höglund-Isaksson, 2006), continuous measurement of the flue gas was required to implement the charge. The installation of the measuring equipment was judged too costly for smaller plants and the charge therefore only imposed on larger boilers. In order not to distort competition between larger plants and smaller units not subjected to the charge, a scheme was designed to refund the charges back to the regulated plants in proportion to energy output.

Energy output within the NO_x charge system is measured in terms of so-called useful energy, which can be either in the form of electricity, steam or hot water depending on end-use[1]. Regulated entities belong to the heat and power sector (between 1992 and 2009 on average 52% of total useful energy production in the system), the pulp and paper industry (on average 23% of useful energy production), the waste incineration sector (15%) and the chemical (5.5%), wood (3.1%), food (1.7%) and metal (1.0%) industries. Initially the charge only covered boilers and gas turbines with a yearly production of useful energy of at least 50 GWh, but in 1996 the threshold was lowered to 40 GWh and in 1997 further lowered to 25 GWh per year (Höglund-Isaksson & Sterner, 2009).

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

The practical procedure in the NO_x system is as follows. At the end of January each year, the firms declare emissions and production of useful energy at each production unit to the Swedish Environmental Protection Agency (SEPA). At the end of June, SEPA publishes the net charges to be paid at each facility. For firms paying a positive net charge, payment is due by October 1, while firms receiving a net refund receive their money, at the latest, two months later (SEPA, 2004).

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

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

2.2 Individual emission standards

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

In 2003, the Swedish Environmental Protection Agency (SEPA) conducted an evaluation of the effect of the emission standards compared to the NO_x charge during the period 1997-2001[4], finding that emission intensities for boilers not subject to emission restrictions were higher than for boilers with restrictions. Emission intensities also remained unchanged for boilers without restrictions during those years. In contrast, emission intensities were 11% lower in 2001 compared to 1997 for boilers with restrictions. Relevant to note is that boilers without emission standards often belonged to smaller plants and that fewer boilers in the wood and pulp and paper industry were subject to restrictions, while restrictions were more common for boilers in the waste incineration and heat and power sector. Because emissions were generally much below the quantitative restrictions, the conclusion from SEPA (2003) is that, for boilers in the heat and power and waste incineration sectors, the NO_x charge was more effective than the restrictions in reducing NO_x emissions. Figure 1 illustrates that the emission standards do not appear to have been the binding factor limiting emissions in 2001 for any of the boilers which were part of the NO_x charge system in both 1997 and 2001 and which were subject to an emission standard in terms of mg NO_x per MJ of fuel at the time when SEPA audited the plant.

Since the SEPA (2003) evaluation, there has been no comprehensive survey of the emission standards and how they have developed over time. SEPA has supplied us with data on emission standards in place in 2012 for 42 out of 50 firms in the NO_x system, randomly selected for an interview survey for the SEPA (2012) report. The majority of the quantitative restrictions were in terms of mg NO_2 per MJ of fuel[5]. Figure 2 illustrates the emission stan-

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

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

[5] Out of 81 different forms of quantitative restrictions for the boilers at these firms, 52 were in terms of mg NO_2 per MJ of fuel, 25 in terms of mg NO_2 per m^3 of flue gas or ton of NO_2 per year and 4 in terms of mg NO_2

dards for firms in this subsample with an emission standard in equivalent units of mg NO_x per MJ of fuel. In most cases, we do not know in which year the standard came into effect and for the comparison with actual emissions we therefore illustrate emissions as an average over the period 1992-2009. Nevertheless, from Figure 2 it appears as if on average the standard has not been binding over the period. However, we cannot rule out the possibility that the standard was binding at some point in time.

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

2.3 Incentives provided by NO_x charge and standards

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

per ton of pulp or paper. One heat and power plant and a production line at one waste incineration plant instead had technology standards mandating SNCR (or equivalent) and SCR, respectively.

$\frac{e_i \varphi_{i,k}}{q_i}$.

In its most intuitive form the cost minimization problem can be written

$$\min_{q_i, \varepsilon_i} C_{i,k}(q_i, e_i) + \sigma e_i - \sigma E \frac{q_i}{Q} \tag{1}$$

subject to

$$q_i \geq \overline{q}_i$$
$$\frac{e_i \varphi_{i,k}}{q_i} \leq \overline{\xi}_i.$$

The second term in (1) is the total NO_x charge payment for firm *i* and the third term is the size of the total refund which is negative and lowers compliance cost. Following Fischer (2011), we can also write the minimization problem in (1) on a more compact form as

$$\min_{q_i, \varepsilon_i} C_{i,k}(q_i, \varepsilon_i) + \sigma \left[\varepsilon_i - \overline{\varepsilon}\right] q_i \tag{2}$$

subject to

$$q_i \geq \overline{q}_i \tag{3}$$

$$\varepsilon_i \varphi_{i,k} \leq \overline{\xi}_i. \tag{4}$$

We will in the following refer to $\sigma [\varepsilon_i - \overline{\varepsilon}]$ as the net NO_x charge which is in units of SEK per unit of useful energy. The net NO_x charge is positive for a firm with an emission intensity higher than the average emission intensity $\overline{\varepsilon}$, i.e., $\varepsilon_i > \overline{\varepsilon}$, and negative for a firm with an emission intensity which is lower than average, i.e., $\varepsilon_i < \overline{\varepsilon}$. That is to say, the refunded NO_x charge serves to raise the average cost of energy production for the firms that are dirtier than average and lower the average cost for the firms that are cleaner than average.

The two first-order conditions (FOCs) for the cost-minimizing energy production, $q^*_{i,k}$, and emission intensity, $\varepsilon^*_{i,k}$, with technology *k* are

$$\frac{\partial C_{i,k}(q^*_{i,k}, \varepsilon^*_{i,k})}{\partial q_i} + \sigma \left[\varepsilon^*_{i,k} - \overline{\varepsilon}\right] \left[1 - \frac{q^*_{i,k}}{Q}\right] = \lambda_{\overline{q}_i} \tag{5}$$

$$-\frac{\partial C_{i,k}(q^*_{i,k}, \varepsilon^*_{i,k})}{\partial \varepsilon_i} \frac{1}{q^*_{i,k}} = \sigma \left[1 - \frac{q^*_{i,k}}{Q}\right] + \lambda_{\overline{\xi}_i} \frac{\varphi_{i,k}}{q^*_{i,k}} \tag{6}$$

with the complementary slackness conditions

$$\lambda_{\overline{q}_i} \geq 0, \; \lambda_{\overline{q}_i}[q^*_{i,k} - \overline{q}_i] = 0$$

$$\lambda_{\overline{\zeta}_i} \geq 0,\ \lambda_{\overline{\zeta}_i}[\varepsilon^*_{i,k}\varphi_{i,k} - \overline{\zeta}_i] = 0.$$

According to the FOCs, the firm should choose the useful energy production and emission intensity that makes marginal cost inclusive of the net NO_x charge equal to the shadow price of useful energy (condition (5)). At the same time, it should set the marginal abatement cost equal to the sum of the NO_x charge[6] and the shadow price on the emissions constraint (condition (6)).

It is quite natural to assume that constraint (3) is binding with a shadow price of useful energy which is larger than zero. In contrast, if a standard is so lax that constraint (4) is not binding or no standard exists then FOC (6) reduces to

$$-\frac{\partial C_{i,k}(q^*_{i,k}, \varepsilon^*_{i,k})}{\partial \varepsilon_i}\frac{1}{q^*_{i,k}} = \sigma\left[1 - \frac{q^*_{i,k}}{Q}\right]. \tag{7}$$

Comparing (6) and (7), we see that if the marginal cost is non-decreasing in the emission intensity, a firm with a binding individual emission standard (i.e., $\lambda_{\overline{\zeta}_i} > 0$) should choose a lower emission intensity than a firm with a comparable boiler without a binding emission standard (operating at the same level of efficiency and producing the same level of output). This is to say, a binding standard induces the firm to operate at a marginal cost of abatement which is higher than the NO_x charge.

Note that the annual gains, g_i, from adopting a new technology $k = 1$ when boiler i already has technology $k = 0$ installed can be represented as:

$$g_i = \left[C_{i,0}(q^*_{i,0}, \varepsilon^*_{i,0}) - C_{i,1}(q^*_{i,1}, \varepsilon^*_{i,1})\right] + \left[\sigma\left[\varepsilon^*_{i,0} - \bar{\varepsilon}\right] q^*_{i,0} - \sigma\left[\varepsilon^*_{i,1} - \bar{\varepsilon}\right] q^*_{i,1}\right]. \tag{8}$$

The first term on the right hand side of expression (8) represents the reduction in production and abatement costs due to the new technology while the second term represents the reduction in tax liabilities net of the refund. The extent to which refunding increases adoption gains depends on the average emissions intensity, which is endogenous to the adoption decisions taken by all firms in the industry.

[6]The adjustment $\left[1 - \frac{q^*_{i,k}}{Q}\right]$ in (5) and (6) reflects the fact that a firm with a larger share in total useful energy $\frac{q^*_{i,k}}{Q}$ pays a lower effective charge on emissions than a firm with a smaller share. In (5), it also implies that an above average emitter pays a lower net NO_x charge and a below average emitter gets a lower marginal net subsidy with a larger market share (Fischer, 2011). See Fischer (2011) for more details on the incentives provided by the refunded charge. In practice, the average boiler share in total useful energy is 0.3% with a maximum of 4.1%. At firm level, the share is on average 2.1%, with a maximum of 11.7%, suggesting that the market share distortion is perhaps relevant only for the largest heat and power producers.

3 NO_x abatement technologies

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

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

Post-combustion technologies, on the other hand, are end-of-pipe solutions that reduce NO_x in the flue gases after the combustion stage, either through catalytic or non-catalytic reduction of NO_x compounds. Selective catalytic reduction (SCR) uses ammonia or urea to reduce NO_x into water and molecular nitrogen (N_2) on catalytic beds at lower temperatures. SCR is highly efficient in reducing NO_x emissions but is a large and costly installation. Selective non-catalytic reduction (SNCR) on the other hand does not require catalysts and cooling of the flue gases and is therefore less costly but also less efficient (Höglund-Isaksson & Sterner, 2009). Emission reductions can be as high as 90% with SCR compared to 35% with SNCR (Linn, 2008).

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

One important determinant of adoption is naturally investment cost. The cost of installing combustion technologies are highly variable across different boilers. Costs depend on size, purification requirements, system of injection, type of chemicals used and the complexity of the control system. According to Linn (2008), the total installation cost of a low NO_x burner or a overfire air injector is in the US roughly 10 million USD, although it varies with boiler characteristics.

Investment costs for the post-combustion technologies SCR and SNCR are also boiler and plant specific and vary with boiler capacity, among other things (SEPA, 2012). Linn (2008) also notes that the cost of retrofitting SCR and SNCR varies with plant characteristics but quotes cost estimates for the US of 40 million USD for SCR as opposed to about 20 USD for SNCR over the lifetime of the unit.

Moreover, some technologies are not commercially available below certain size thresholds (Sterner & Turnheim, 2009). Technology adoption also depends on access to information and the degree of involvement in R&D and innovation activities (Sterner & Turnheim, 2009), which would seem to support the existence of learning effects.

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

4 Model of the investment decision

We use the framework in Karshenas & Stoneman (1993) and consider a situation in which a firm has the choice to install a new technology in a boiler i which is included in the refunded NO_x charge system. The cost of doing the installation at time t is $I(Z_i(t), L_i(t), S_i(t), t)$ where $Z_i(t)$ is a vector of boiler-specific characteristics which may affect investment costs. $L_i(t)$ is

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

a vector of the number boilers at the plant and firm that unit i belongs to and that may give rise to internal learning effects which decrease investment costs. $S_i(t)$ is the stock of boilers already installed with the new technology in the industry of unit i which could affect investment costs if there are external learning effects.

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

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

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

$$V_i(t) > 0. \tag{9}$$

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

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

Differentiating with respect to t we get

$$y_i(t) = -g(R_i(t), Z_i(t), L_i(t), S_i(t), t) + rI(Z_i(t), L_i(t), S_i(t), t) - \frac{dI(Z_i(t), L_i(t), S_i(t), t)}{dt} \leq 0. \tag{10}$$

According to (10), it is not profitable to wait longer to adopt at time t if the profit gains in period t is larger or equal to the cost of adoption in period t given by the sum of the annuity of the investment cost and the decrease in investment cost over time.

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

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

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

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

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

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

$$h_i(t) = \lambda_0(t)\exp(X'_{it}\beta),$$

where $\lambda_0(t)$ is the so called baseline hazard and X_{it} is composed of the vectors $R_i(t)$, $Z_i(t)$, $L_i(t)$ and $S_i(t)$ which are likely to affect whether the arbitrage condition in (10) is fulfilled. A variable which negatively affects $y_i(t)$ should increase the hazard rate and vice versa.

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

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

We now specify that $g_{i,k}(t)$ is a function of the previously discussed explanatory variables

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

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

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

$$h_{i,k}(t) = h_k(R_i(t), D_i(t), Z_i(t), L_{i,k}(t), S_{i,k}(t), t). \tag{11}$$

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

5 Data and explanatory variables

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

Our sample consists of 556 boilers for which the information required to estimate at least one of the three econometric models is available[10]. Descriptive statistics are presented in

[10] The number of boilers paying the NO_x charge has varied over the years because of the change of the production threshold in 1996 and 1997 but also because of entrances of new boilers in other years and the option to produce below the threshold even though emissions are monitored. 669 boilers paid the charge in at least one year between 1992 and 2009 with 182 boilers paying the charge in 1992 and 427 boilers in 2009. Out of the 182 boilers in 1992, 19 boilers had at least one of the technologies installed in 1992; 7 had only a post-combustion tech-

Table 1. As pointed out in section 2, the number of regulated boilers under the NO_x scheme increased in 1996 and 1997. Moreover, new boilers have become part of the scheme over the period under analysis. Because the boilers which entered later into the system may be different from the early entrants in their propensity to adopt, we estimate the empirical model for the full sample and for the subsample of boilers that have been part of the NO_x scheme (and our panel) since 1992 (See Table 6 in the appendix) and compare the results.

5.1 Dependent variables - technology adoption

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

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

5.2 Explanatory variables

Net NO_x charge liabilities

From expression (8) it is clear that the NO_x charge might affect the incentives to invest in a

nology installed, 3 had only a combustion technology installed, 5 had only condensation technology installed and 4 boilers had a combination. Since we are using lagged variables to explain adoption and do not have data before the start of the program in 1992, we cannot include these early adopters in our sample. These boilers are however included in the sense that they might install another technology in one of the following years. Concerning other boilers excluded from the sample, the boilers not included but paying the charge in at least one year have on average significantly lower production of useful energy and a significantly lower share of boilers with any of the technologies installed in any year between 1992 and 2009. These are not surprising results seeing that a boiler for which information is missing for the estimations is likely to have produced below the threshold and not been part of the charge and refunding scheme for most of the period. The profitability of installing the technologies at such boilers which can strategically produce just below the threshold should reasonably be low. Due to these observable differences, our results are likely not representative for boilers producing around the production threshold.

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

new technology due to reductions in production and abatement costs and net NO_x charge liabilities. Unfortunately, we cannot identify the effect of the stringency of the NO_x charge because there has only been one change to its level (in 2010) since its implementation in 1992. Nevertheless, to account for the effect on production and abatement costs we control for a series of rank and learning effects that are described later in this section. On the other hand, as pointed out before, the net reduction of NO_x charge liabilities is endogenous to the adoption decisions taken by all firms in the industry. Thus, to account for the effects of the NO_x charge liabilities we include the variable *Net NOx charge level$_{t-1}$*, which is the net NO_x charge paid at the individual boiler in the previous year denoted $\sigma\left[\varepsilon_i - \bar{\varepsilon}\right]$ in the cost function in (2). This proxy is based on the behavioural assumption that the upfront cost of the NO_x charge is a determinant of the adoption decision. Such an assumption finds some support in Sterner & Turnheim (2009) who suggest that the refunding systems reporting and yearly publication of individual emission intensities and net payments probably acts as an additional incentive for firms to compete for emission-reducing measures.

The intuition behind our proxy is that that being a net payer within the refunding system stimulates adoption of emission-reducing technologies due to a greater expected reduction in NO_x payments.

As discussed in section 2.3, a confounding factor is the emission standards issued for some of the boilers by county authorities. Unfortunately, we only have information about the emission standards for a small randomly selected subsample of 42 firms. Anecdotal evidence tells us that some county authorities apply more stringent standards than others. To try to capture some of the difference in regulatory stringency across regions, we use the county location of the boilers for which we know the emission standard to construct county average standards. We use these averages to divide the counties into two halves: one with relatively more stringent standards, and one with relatively lax standards and estimate our model separately for the two groups.

Technology substitutes or complements

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

Biofuel use

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

Entry effects

We include the indicator variables *Entrant 1996-1997* and *Entrant 1998-2009*, which are equal to one if the boiler entered the NO_x charge system in those years, and zero otherwise. Because *Capacity* is already controlling for differences in size between earlier and later entrants, *Entrant 1996-1997* and *Entrant 1998-2009* should be capturing any other potential effect of

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

[13]For CO_2 emission factors see SEPA (2009). The carbon tax is based on the carbon content of the fuel; a number of deductions and exemptions from the carbon tax have been introduced in different sectors, and this also varies according to the type of generation in the case of the heat and power sector. Additionally, not all the plants are part of the EU ETS system and the overlapping process with the carbon tax has added other tax exemptions for the plants within the EU ETS. The EU ETS CO_2 price employed to compute the CO_2 fuel costs is the yearly average spot price. We also tried including the sector-specific prices of CO_2 separately as an explanatory variable but this was not significant. Furthermore, considering potential reverse causality for the use of biofuels, results remained largely unchanged when lagging the relative biofuel to fossil fuel cost.

late entry into the system on the profitability of adopting cleaner technologies, such as lower investment cost or the redistribution of charge revenues that might have occurred when smaller and dirtier than average units entered and increased charge revenues in 1996 and 1997.

Rank effects

With respect to rank effects, i.e., inherent characteristics of the firm and boiler which may affect adoption, we consider industry sector and boiler capacity. Firms in different industry sectors face different economic conditions and levels of competition which may affect the propensity and ability to adopt new technologies. The NO_x charge may also affect the heat and power industry differently than the other sectors since useful energy as it is defined is the end product of the heat and power sector but mainly an intermediate input for the other industries.

There are also some indications from SEPA (2003) that the stringency of quantitative restrictions may vary between industry sectors, with the heat and power sector and waste incineration possibly being subject to more stringent regulation. We include the dummy variables *Pulp-paper sector* and *Waste incineration* and, due to the relatively small sample of boilers in the remaining industries, a common dummy, *Other sectors*, for the food, chemical, wood and metal industries. We use heat and power as our reference sector, because this is the dominant sector in the NO_x system.

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

Stock effects or external learning

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

Internal learning

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

6 Econometric results

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

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

All the tables present the estimated coefficients and therefore their sign is indicative of whether an explanatory variable speeds up or retard the adoption decision. Given the nonlinearity of the hazard function and to ease interpretation of the magnitude of these coefficients throughout the text, the effect of a covariate on the conditional probability of adopting is calculated as $exp(\beta)$, i.e., as the hazard ratio. In that manner, exponentiated coefficients larger than one imply that the covariate increases the hazard of adoption, whereas values lower than one mean that it decreases the hazard. For example, a hazard ratio of 1.02 indicates that a one unit increase in the explanatory variable increases the hazard of adopting the technology by 2%. This effect is interpreted as a proportional shift of the hazard rate relative to the baseline, all other things equal. Likewise, if a hazard ratio is equal to 3, this would imply that boilers in the analysis group (e.g., belonging to a particular sector) are three times more likely to adopt compared to the reference group. In this case, it is usual to say that the likelihood of adoption increases by a factor of 3.

6.1 Adoption of combustion technologies

The estimated proportional hazard models explaining the diffusion pattern of combustion technologies are in Table 2. Overall, we identify few of the drivers behind the adoption of this group of technologies.

The net charge liabilites in terms of *Net NOx charge level*$_{t\text{-}1}$ does not influence the hazard of adoption. Although we found a positive coefficient of the net charge for the pooled sample, the heat and power sector, and the counties with stringent emission intensity standards, the effect of the net NO_x charge is not statistically significantly different from zero at conventional levels across all subsamples.

The results in general show a low explanatory power also for the other covariates across subsamples. Few of the variables are statistically significant. In the pooled regression, the only variable that plays a role in the adoption decision is *Entrant 1998-2009*. The hazard of adopting the technology for those boilers that entered the program in that period is 66% higher than the hazard of the boilers that entered in 1992-1995. This variable is also statistically significant and positive for the heat and power sector, but statistically insignificant for the rest of the subsamples. Entry effects may have been more important for the heat and power sector since NO_x-reducing measures may have greater priority in that sector. For these potentially relatively new boilers, the cost of installation may be lower, leading them to adopt soon after entry into the system.

The conditional probability of adopting combustion technologies seems to be independent of installing the most expensive type of technology in the previous year, *Post-comb.*$_{t\text{-}1}$. Having flue gas condensation installed appears to influence the likelihood of adoption in the heat and power sector. A boiler in the heat and power sector with flue gas condensation already installed has a 60% higher hazard of adopting a combustion technology. This result is mainly present when the combustion technology is flue gas recirculation, indicating complementarity of these two particular types of technologies.

Although there seems not to be any statistical difference in the sector dummies for the pooled model and the counties with stringent standards, boilers belonging to the pulp and paper sector within the counties with lax standards were less likely to adopt than boilers in the heat and power sector. This result indicates that boilers in the pulp and paper industry are relatively less prone to adoption unless induced to invest in NO_x-reducing measures by stringent emission standards.

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

As expected, internal learning increases adoption; however these effects are only statistically significant for *Plant comb.*$_{t-1}$ in the non-heat and power sectors regression and in counties estimated to have lax emission intensity standards. For instance, in the counties with lax standards, having one additional boiler within the same plant already equipped with a combustion technology increases the likelihood of adoption of another boiler in that plant by 50%. Neither stock effects nor external learning were observed in our estimation of combustion technologies.

Our results for the general model are qualitatively similar when we estimate the regressions using a Weibull parametric model. An important difference with the estimates in the Cox model is that *Sector flue gas.* $_{t-1}$ is statistically significant at 1% level, suggesting the presence of stock effects. Moreover, there seem to be differences across sectors in the hazard of adopting combustion technologies.

Statistical testing indicates that we cannot reject the hypothesis that the Weibull shape parameter is equal to one at the 1%, 5%, and 10% significance level, implying that the baseline hazard is constant over time. This result is also consistent with Popp (2010)'s argument that unmeasured learning effects are small because the technologies have been well known for a long time. Our results also remain largely unchanged when we study the subsample of boilers who have been part of the NO_x scheme since 1992 (see Appendix Table 6).

6.2 Adoption of post-combustion technologies

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

It appears that the significant coefficient for the net NO_x charge is driven by adoption in the counties estimated to have stringent standards, possibly explained by the net charge level being more negatively correlated with the likelihood of being subject to a binding standard in the pooled sample. Nevertheless, since we cannot fully control for the individual stringency of the standard nor the year of implementation, we cannot rule out that a boiler being more dirty than average may also be more likely to become subjected to an emission standard.

Whether the boiler has combustion or flue gas condensation technology already installed has no statistically significant effect in the pooled and sector subsamples. However, there are some differences in the effect of these technologies when classifying counties by stringency of standards. The effect of a combustion technology installed in the previous year, *Comb.*$_{t\text{-}1}$, is positive and statistically significant at the 5% level in counties with lax standards. In those counties, a combustion technology already installed increases the hazard of adopting by a factor of 2.1. Although combustion technologies seem to be independent of the adoption decision of post-combustion technologies in counties with stringent standards, in counties with lax standards the low cost investments of NO_x reducing technologies appear to have been first exhausted before plants moved to more expensive abatement investments.

In the case of flue gas condensation technology, *Flue gas cond.*$_{t\text{-}1}$ is statistically significant in counties with stringent standards. Installing this technology in the previous year raises the hazard of adopting post-combustion technology by a factor of 3.6. Therefore, complementarities between technologies seem to be associated more with the stringency of the emission standards: post-combustion and combustion technology in counties with lax standards, and post-combustion and flue gas condensation technology in counties with stringent standards. Unfortunately, we do not have the information on individual emission standards at the boiler level to properly control for this.

As expected, boiler capacity, *Capacity* has a significant and positive effect on the conditional probability of adoption in most of the subsamples. A 10 MW increase in the boiler capacity increases the hazard by 2.2%-4.6%, which is consistent with the economies of scale related to SCR and SNCR technologies.

A boiler belonging to the *Waste incineration* sector is more likely to be equipped with a post-combustion technology than a boiler in the heat and power industry (see columns 1 and 5), while boilers in *Other sectors* such as the wood, metal food, and chemical sectors are significantly less likely to be equipped with a post-combustion technology (see columns 1 and 4). That waste incineration boilers are more likely to be equipped with post-combustion technologies could possibly be explained by the ownership structure of this sector. Waste incineration boilers often belong to public utilities which may have motives other than pure profitability for investing in emission reducing technologies.

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

Consistent with expectations, boilers in the heat and power sector and in counties with stringent standards that entered in the period 1996-1997 were less likely to adopt than boilers entering in 1992-1995. Internal learning seems to have been a relevant factor explaining the conditional likelihood of adoption. The variable *Plant post-comb.*$_{t\text{-}1}$ was positive and statistically significant in most of the subsamples, while *Firm post-comb.*$_{t\text{-}1}$ was negative and statistically significant in only two of them. The highest internal learning effect of *Plant post-comb.*$_{t\text{-}1}$ is present in the heat and power sector. The decreased hazard induced by *Firm post-comb.*$_{t\text{-}1}$

may indicate that internal learning is counteracted by financial constraints. Alternatively, as shown by Fischer (2011), firms with market power that are cleaner than average have an incentive to abate less since the market share adjustment in (6) implies that a larger firm has a lower marginal cost of emissions.

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

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

6.3 Adoption of flue gas condensation technology

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

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

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

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

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

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

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

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

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

7 Conclusions

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

We primarily find results on drivers behind the adoption of post-combustion technologies. Because the NO_x charge scheme, on the net, only taxes the firms which are dirtier than average, we tested the hypothesis that the net charge liabilities stimulates adoption. However, the net NO_x charge per unit of energy did not seem to encourage adoption of combustion or flue gas condensation technologies. The net NO_x charge only plays a role in stimulating adoption of the most expensive technologies: post-combustion installations. Because these types of technologies can be characterized as end-of-pipe solutions which allow firms to choose emissions independently from output to a much larger extent than the other technologies, this result is possibly simply driven by larger potential gains for boilers with initially higher emission levels.

Adoption of post-combustion technologies is also more likely in the waste incineration sector and adoption of the efficiency improving technology of flue gas condensation more likely in both the waste incineration and heat and power sectors. This can possibly be explained by a comparatively large degree of public ownership in these two sectors.

The capacity of the boiler increases the likelihood of adoption of post-combustion technologies in line with the expected economies of scale. There are also indications of an internal learning effect such that more boilers already installed with post-combustion technologies at the plant increases the likelihood of adoption. In contrast, in the heat and power industry, more boilers at the next step of aggregation, the firm, makes adoption less likely. A potential explanation is that firms with a large share of the useful energy output in the refunding system has reduced incentives to abate emissions because of the negative effect of abatement on the size of the refund.

The Swedish NO_x charge and refunding scheme is complex, as are the causalities and timing involved, and this topic would benefit from a more detailed future study of the dynamics of plant regulations if more data becomes available. Disentangling the incentives for investment in different types of environmental technologies provided by this scheme is also a potential area for future research.

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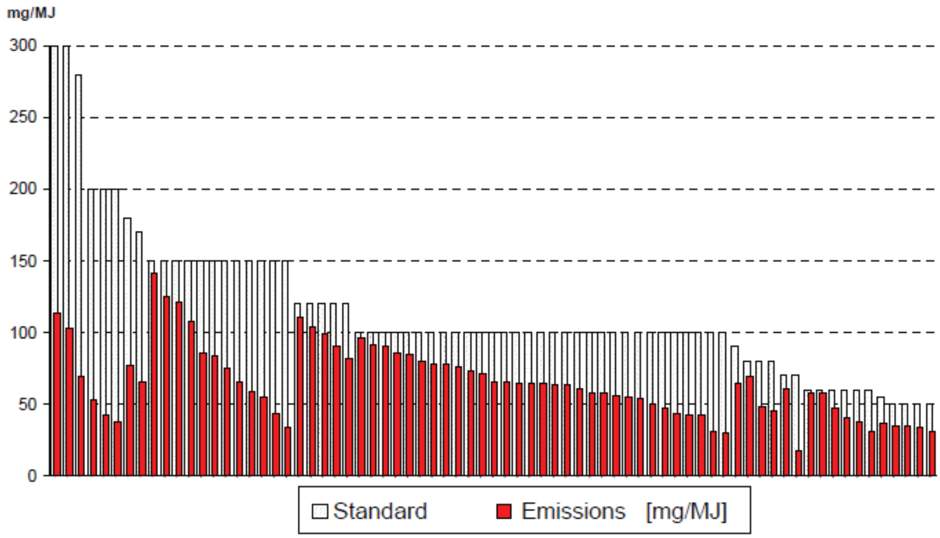

Figure 1: Emission standards and actual emissions (in terms of mg NO_x per MJ of fuel) in 2001 for boilers which were in the NO_x charge system in both 1997 and 2001 (SEPA, 2003).

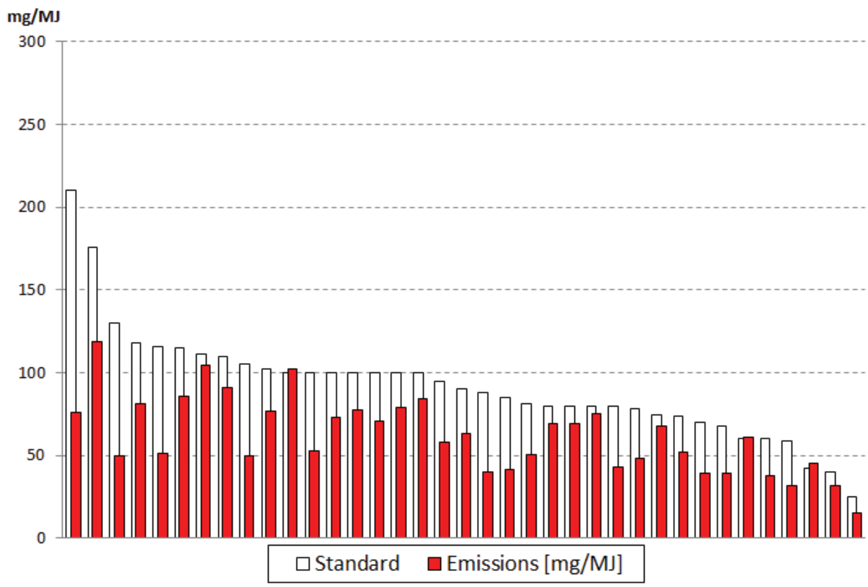

Figure 2: Emission standards and actual emissions (average over 1992-2009) for boilers which were randomly sampled for the SEPA(2012) report and also subject to emission standards in terms of mg NO_x per MJ of fuel. Data supplied by SEPA. Averages at plant level.

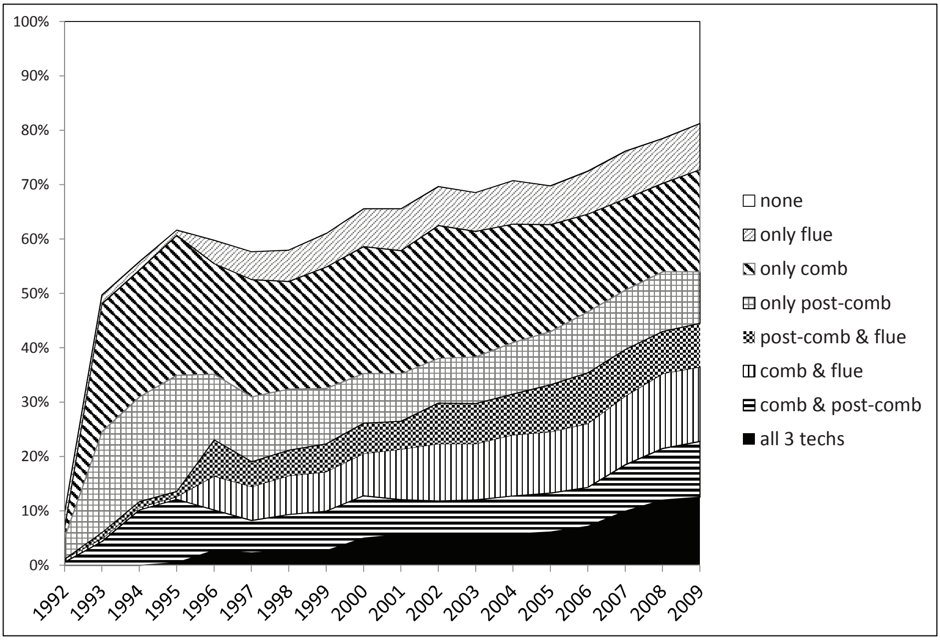

Figure 3: Diffusion of post-combustion, combustion and flue gas condensation technology among the boilers in the joint sample for all three proportional hazard models.

Table 1: Descriptive statistics.

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

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

Table 2: Adoption of combustion technologies.

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

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

Table 3: Adoption of postcombustion technologies.

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

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

Table 4: Adoption of flue gas condensation technology.

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

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

Appendix

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

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

Notes: This table shows the coefficients of the parametric proportional hazard model with Weibull distribution for (1) post-combustion technologies, (2) combustion technologies, and (3) flue gas condensation technology for the pooled sample 1992-2009. The dependent variable is an indicator variable equal to one if the boiler has one of the technologies installed described above and zero otherwise. Continuous variables are in units of 10. Sector dummies are not shown to save space. Standard errors, in parentheses, are robust to heteroskedasticity and arbitrary correlation within firm-level clusters. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

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

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

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

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