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In or Out: Efficient inclusion of installations in an Emissions Trading Scheme?

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TABLE OF CONTENTS

Abstract

Regulators around the world are currently considering national emissions trading systems (ETS) as cost-effective instruments to reduce greenhouse gas emissions. In the process, they are confronted with numerous design issues. The coverage of installations in an ETS is one such issue. While “blanket coverage” that includes all industrial emitters of greenhouse gases in an economy has some intuitive appeal, and seems equitable, it does not take into full account all the costs related to the extent of coverage. This paper shows that an alternative approach of “efficient coverage” can achieve the same emission reduction outcome at lower social cost. The approach is based on maximising the benefits from inclusion of installations in an ETS at the same time as taking all relevant transaction costs into account. A broad definition of transaction costs is used, which covers the regulatory costs to the government as well as regulatory costs imposed on covered installations. We find that particularly for relatively modest emissions reduction targets the cost savings of an “efficient coverage” compared to a “blanket coverage” are significant.

Key words: Emissions Trading Scheme, Environmental Policy, Installation Coverage, Transaction costs.

JEL: Q50, Q58, H23

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1. Introduction

Emissions trading schemes (ETS) are designer markets, and as such they suffer from various shortcomings as a result of particular design choices. In 2005 the European Union (EU) initiated the first phase of an ETS for Carbon Dioxide (CO2) emissions, covering some 11,000 installations and approximately 45% of the total CO2 emissions of the then 25 (now 27) EU member states. Coverage in the EU ETS is based on the size of the installed capacity. For example, any combustion installation is mandated to participate in the ETS if it operates a rated thermal input greater than 20 megawatts (CEC 2003). The EU ETS covers emissions from large energy intensive industries such as cement manufacturing and steel mills, but also includes some small emitters such as hospitals and prisons.

The inclusion of combustion installations with a threshold of 20MW has lead to a broad coverage of small installations as this threshold includes active, as well as reserve capacity. A plot of a Lorenz Curve shows that 50 % of the installations covered under the EU ETS emitted less than 1.4 % of the total emissions in 2005 (see Figure 1). Given the relatively high transaction costs of participating in the ETS, this very uneven distribution of emissions suggests that the costs of operating an ETS with a near blanket coverage may be too high in comparison to the benefits from such a broad coverage. Reducing the number of covered emitters by, for example by directly targeting emissions as a cut off criterion, replacing the current criterion based on installed generating capacity, may produce superior outcomes as it would exclude installations with reserve capacities. In response to these concerns, the EU Commission has recently introduced an additional emission threshold of 10,000 tons of CO_2 equivalent per year (excluding emissions from biomass). This will allow combustion installations falling below this threshold not to be mandated to participate in the EU ETS from 2013 onwards, as long as their rated thermal input does not exceed 25 megawatts (CEC 2009). In order to limit possibilities to circumvent regulation, by for example building installations that will keep the emissions just below the threshold, the Commission foresees that all installations with emissions below the threshold that choose to stay out of the ETS will be regulated through other policies and regulation that will deliver a commensurate reduction in CO_2 emissions.

In light of this experience, the extent of coverage in an ETS seems to be a major design issue for regulators in many countries—including Australia, Japan, Canada, and the United States—that are currently considering implementing an ETS for reduction of CO_2 emissions. The regulators need to decide which emitting entities should be included in the system and become directly liable to surrender permits. Greenhouse gases (GHG) are emitted in an economy from many heterogeneous sources and sectors ranging from agriculture, to electricity generators, to oil refineries and steel mills. Regulators often assume that including more sources will automatically increase efficiency gains from trading due to the heterogeneity of abatement cost structures.[1] This may be true for the relatively large emitters, for whom the transaction costs pertaining to their participation in an ETS (e.g. cost of monitoring, reporting and verification) are relatively small compared to the benefits realised as a result of being able to trade emission allowances (Schleich and Betz, 2004). However, a blanket, or near blanket coverage means that a large number of small emitters will be mandated to participate in an ETS as was the case in the EU (Figure 1). For many of these emitters the costs of inclusion in an ETS are high, and

[1] A good overview of the mechanism of emissions trading can be found in Tietenberg (1985) and his bibliography on tradable permits http://www.colby.edu/personal/t/thtieten/trade.html.

given the small share of emissions coming from those emitters, there are little benefits to be realised from them being able to participate in the trading system.

The current paper has two main objectives. One is to propose a conceptual framework to describe the problem of efficient coverage of emitting installations in an emission trading scheme by explicitly taking transaction costs into account. The other is to conduct an empirical analysis of the theoretical proposition for an efficient coverage level in an emissions trading scheme. The empirical analysis is based on 2005 data from the EU ETS. We compare the existing "blanket system", a system which includes all emitters of CO2 above a certain capacity threshold, with a coverage that maximises net social benefits from the ETS – called here "efficient coverage".

The literature on the efficiency of tradable permit systems starting with Crocker (1966), Dales (1968), and followed by Montgomery (1972) has typically abstracted from any transaction costs. The inclusion of transaction costs in tradable permit markets has been formally introduced by Stavins (1995), where it is theoretically demonstrated that initial allocation affects the final equilibrium if marginal transaction costs are non-constant. Montero (1997) shows that even for constant marginal transaction costs, the initial allocation may affect the final equilibrium. More recently Cason and Gangadharan (2003) have supported Stavins' findings experimentally. Other empirical studies have estimated the transaction costs of trading schemes (Foster and Hahn 1995; Dwyer 1992). However, the theoretical models (Stavins 1995) and most empirical studies only take into account the costs of trading emissions permits (which we call here 'trading costs') as the only transaction costs. This may be traced back to Coase's definition of transaction costs as "costs (…) in carrying out market transactions" (1960, p. 15). In addition, empirical studies which do estimate a broader range of transaction costs usually do not measure transaction costs in a consistent and meaningful way, which makes it difficult to compare cost estimates (McCann et al. 2005). We subscribe to a broader definition of transaction costs that subsumes all costs, including the costs of monitoring emissions on the part of the regulated entity, as well as the administration costs to the regulator, as well as the trading costs. In this paper transaction costs are defined as all those costs to society as a whole that are related to an emissions trading scheme, and that can not be classified as cost of abatement. This is in line with Betz (2003) and Jaraite et al. (2009), who estimate the transaction costs for the EU ETS using this broader approach. The concept goes back to Commons (1934) and Stigler (1972), the latter comparing transaction costs to frictions in the physical world.[2]

The number, size or sector of the installations covered in an ETS has only been treated in theoretical literature if it had implications on the market structure and influenced negatively the performance of the ETS market (Hahn, 1984). Empirical research on coverage was conducted for the European Commission (Graus and Voogt, 2007) or other governments (Commonwealth of Australia, 2008) and by Hargrave (2000) but was not nested within an appropriate theoretical framework. The latter study compares the total number of covered installations for an upstream approach with a downstream approach for the US.[3] The focus of the present paper is on the downstream approach, since transaction costs are more relevant in this case. In addition, the data for

[2] "The world without transaction costs turns out to be as strange as the physical world would be with zero friction" (Stigler 1972, p. 12)

[3] A *downstream* approach requires fossil fuel users to acquire emission allowances. An *upstream* approach requires permits to be acquired by fuel producers. In theory, both systems lead to the same efficient outcome, since prices are perfectly passed through and firms react in the same way to a price signal and a quantitative constraint. However, in practice there may be differences depending on, for example, how costs are passed through (e.g. asymmetric versus symmetric pass-through).

the empirical study reported in this paper come from the EU ETS, where installations are covered based on a downstream approach.

The paper investigates the hypothesis that cost-effectiveness claims of emissions trading systems with broad coverage may often exaggerate the benefits of including installations because the transaction costs of coverage are neglected. We find that an installation should only be covered as long as the marginal benefits of doing so exceed the marginal costs, including all transactions costs. Therefore blanket coverage of installations in an ETS will in most cases be inferior to an efficient coverage. The empirical analysis supports the theoretical findings, showing that the efficient level of coverage varies with the level of emission reduction targets. For relatively small emission reduction levels the difference in total social cost between blanket coverage and efficient coverage is highly significant, but that difference diminishes when reduction targets become more ambitious.

2. Model

Consider a social planner whose objective is to achieve an exogenously set cap (C) for a uniformly-mixed flow pollutant, such as CO_2, at a minimum cost. The planner has two policy instruments to choose from, with an aim to maximise net social benefits (B). The first alternative is that all installations emitting CO_2 with a generating capacity beyond a given threshold be covered by an ETS (policy 1); the second alternative is that some installations are covered by an ETS, and others by a uniform emissions standard (e.g. the standard defines emissions per output for each sector and is based on Best

Available Technology) (policy 2).[4] Both policies vary in their transaction costs which include regulatory costs as well as trading costs for the ETS. Under the ETS, deterrent sanctions provide for full compliance such that the emissions e_i of any installation i, equal the amount of allowances surrendered, a_i. The objective function consists of the aggregate net social benefits depending on the policy alternative chosen, subject to meeting the given cap, C, so that: $\sum_{i=1}^{n} a_{0i} = \sum_{i=1}^{n} e_i \leq C$, where a_{0i} denotes the quantity of initially allocated allowances to installation i (i.e grandfathered allowances) and n is the number of installations under blanket coverage. The quantity of CO_2 emissions allowed with this initial allocation is equivalent to those proscribed by an emissions standard.

The aggregate net social benefit from implementing policy 1 (denoted by $B^1(n)$) of having a blanket coverage of installations in an ETS as opposed to regulating all installations by an emissions standard is defined as the difference of the sum of total regulatory costs (comprising of abatement cost and transactions cost, further explained below) across n individual installations under the standard, TCR_i^{ST}, and the sum of those costs across the same n installations covered by the ETS, TCR_i^{ETS}:

$$B^1(n) = \sum_{i=1}^{n} \left(TCR_i^{ST} - TCR_i^{ETS} \right). \tag{1}$$

The aggregate net social benefit from implementing policy 2 ($B^2(m)$), where some installations are covered by the ETS, and the rest are regulated by an emission standard is given by:

[4] Another alternative is that the installations not covered in an ETS are left completely unregulated. This possibility has been considered, but the results are not reported here. Some discussion on this is offered in the ultimate section of this paper.

$$B^2(m) = \max_m \sum_{i=1}^{n} TCR_i^{ST} - (\sum_{i=1}^{m} TCR_i^{ETS} + \sum_{i=m+1}^{n} TCR_i^{ST}) = \max_m \sum_{i=1}^{m} (TCR_i^{ST} - TCR_i^{ETS}) \qquad (2)$$

where m is the number of installations efficiently covered in an ETS, such that $1 \leq m \leq n$, and $i = 1,\ldots, m$. The aggregate net social benefits will be maximised where the marginal benefit of adding the m^{th} installation to the ETS is equal to the associated marginal cost of adding that m^{th} installation.

In the case of regulation by an emissions standard, the total cost of regulation (TCR_i^{ST}) for an individual installation i comprises of the total cost of abatement (an integral under the marginal abatement cost curve, (MAC_i,) and the regulatory cost (RC_i^{ST}) associated with the emissions standard, including all transactions costs:

$$TCR_i^{ST} = \int_{e_i^{ST}}^{e_i^{NR}} MAC_i(e_i)\,de_i + RC_i^{ST} \qquad (3)$$

where e_i^{ST} is the level of allowable CO_2 emissions for installation i under an emissions standard, and e_i^{NR} is the level of CO_2 emissions for installation i when there are no regulations on CO_2 emissions. For installations covered by an ETS, the total cost of participation in the ETS, TCR_i^{ETS} comprises of the following:

$$TCR_i^{ETS} = \int_{e_i^{ETS}}^{e_i^{NR}} MAC_i(e_i)\,de_i + p_i \cdot \left(e_i^{ETS} - a_{0i}^{ETS}\right) + RC_i^{ETS}\ , \qquad (4)$$

where MAC_i is the marginal cost of abatement for installation i, e_i^{ETS} represent the emissions of an installation covered in an ETS; p_i is the equilibrium market price for allowances (adjusted for trading cost (t_i), as described below); a_{0i}^{ETS} is the initial allocation of emission allowances to installation i under an ETS; and RC_i^{ETS} are the other regulatory cost (administration, monitoring, verification).[5]

In competitive emissions trading markets the equilibrium price for allowances p^* is the same for all installations. However, the price that an individual installation faces, p_i will vary according to the installation specific trading costs t_i associated with the exchange of the allowances. The trading costs for buyers of allowances are added to the price paid for allowances, and for sellers they are subtracted from the price received. This can be expressed by:

$$p_i = p^* \pm t_i \,, \qquad (5)$$

3. Data

Several sources of data were used to conduct an empirical analysis along the lines of the conceptual framework proposed above. Installation-level data on verified emissions and allowance allocations for 2005 were available for the EU ETS. The Community Independent Transaction Log (CITL) has published data on allowance allocations and verified emissions for 9,847 installations in 23 EU member states. The installations were grouped in eight industrial sectors: Cement and lime, Ceramics, Combustion (any facility with installed capacity of more than 20 megawatts of rated thermal input), Glass, Iron and steelworks, Pulp and paper, Refineries, and a sector of other installations.

[5] In order to be comparable the allowable level of emissions under the emissions standard has been set equal to the initial allocation of allowance under the ETS; $e_i^{ST} = a_{0i}^{ETS}$.

3.1 Sectoral CO_2 Abatement Costs

Data on the cost of abating CO_2 emissions were difficult to obtain, due to limited amount of information available in the literature, and the lack of transparency on industry abatement costs. After a careful review of the existing literature, two bottom-up studies that report cost estimates were identified as a source of data most relevant for our purposes. These were De Beer et al., 2001, and Hendricks et al., 2001. These studies provide estimates on costs per tonne of CO_2 emissions reduction for specific technologies and for related emissions reduction potentials as percentages of total CO_2 emissions for most of the industrial sectors represented in the EU ETS. However, the data reported in these studies contain numerous data points where abatement of CO_2 emissions can be achieved at a net benefit to the installation. This means that rather than being a costly activity, some reduction measures are seen as being economically beneficial to those installations—a phenomenon often referred to in the literature as 'no regret potential' (e.g. Brechet and Jouvet, 2009). This is a well known problem and the reasons for this potential being unused are widely discussed in the literature (Stern 2006). Studies have demonstrated that barriers such as transaction costs may prevent cost-effective technologies and practices from being adopted (Joskow and Marron 1992).

By taking into account transactions costs to overcome the barriers which prevent the implementation of those 'no-regrets' measures – such as, for example, information barriers – the costs for those measures may well become positive and explain why those measures have not been implemented yet.[6] While no accurate estimates for those

[6] Substituting clinker – which is very CO2 intensive to produce - with the waste product of fly ash from blast furnaces e.g. of steel industry is an example of a 'no-regrets' option. However, there may be barriers which prevent this measure from being implemented in practice which are costly, i.e. involve high transaction costs (i) technical barriers such as the quality assurance of the new blend, and builders must be

transaction costs that might be preventing adoption of these 'no regret' abatement options have been reported in the literature, the simple reality is that emitters are not observed to be widely adopting them. To represent this situation in our empirical model, the following method was applied and is in line with the approach taken by others (e.g. Hyman et al 2002). The estimation included an intercept shifter, which effectively raised the estimated abatement cost curves, so that even the least expensive abatement options still incur some, albeit small cost.[7] Using this method marginal abatement cost curves were estimated for six industrial sectors identified in the CITL. These sectors were: Cement and lime, Combustion, Glass, Iron and steelworks, Pulp and paper, and Oil refineries.

3.2 Transactions cost data

Data on transaction costs is generally difficult to obtain since they can not be easily observed and measured. In order to estimate the transaction costs in a comprehensive way we used the definition and classification developed by Betz (2003) and compile data from various sources (see Table 1).[8] While starting-up transaction costs are not insignificant, the on-going transaction costs are the dominant costs in the long run. Therefore our focus was on the average on-going transaction costs for the covered

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educated on these of high fly ash cement as well as reassured as to its quality (ii) market barriers such as market resistance to high fly ash blended cement which need to overcome (CDM Executive Board 2004).

[7] Running our model with the negative cost estimates would result in a market permit price of zero or below. This will imply that no trade occurs and firms will just internalise the benefit of their savings. Introducing an emissions trading scheme would not make any sense as regulatory costs would occur and no benefits from trade would be achieved.

[8] A strict classification of transaction costs in necessary in order to make them comparable. The survey conducted by the European Commission in participating member states revealed a wide range of annual administration and monitoring cost estimates; from €2,100 – €5,000 in Sweden, to €8,700 – €21,500 in the Netherlands, and €12,500 to more than €20,000 in Germany (Grauss and Voogt, 2007). Part of the cost differentials may to some degree be the result of a vague definition on what is subsumed under transaction costs.

companies, excluding the set-up costs in early years. Transaction costs also depend on the size of the company and its emissions level. Therefore we differentiated between three classes: small emitters (emissions below 20 kt CO2e), medium emitters (20 – 1,700 kt CO2e) and large emitters (emissions above 1,700 kt CO2e) based on a study by Jaraite et al (2009).[9] Data from the following studies were included in our estimation of transaction costs:

Administration costs on the part of the government are based on a study for the EU ETS in Germany using the figures for the Emissions Trading Authority in the German budget that are charged as a levy on installations (Ewringmann et al, 2005). [10]

Monitoring, verification and reporting costs for the EU ETS are based on Jaraite et al (2009), which used surveys and interviews of regulated companies in Ireland.

Strategy and risk management and accounting costs are mainly based on personal interviews (Betz 2003) in a German case study.

Trading costs are derived from ECX exchange, and costs for over-the-counter (OTC) trading from Convery and Redmond (2007).

In summary, ongoing regulatory costs in the EU ETS before – excluding trading costs - amount to approximately €21k for small, €35k for medium, and €82k for large installations.

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[9] We divide the verified emissions in the respective category in Table 1 (Jaraite et al, 2009) by the applicable number of companies and by 6. Dividing by 6 will give us the annual emissions per installation (verified emissions were given for 2005-2007 and emissions were given on company level and average number of installation per company was assumed to be 2).

[10] In Germany the costs for the administration of the German Emissions Trading Authority are financed through a fee on each allocated allowance.

Obtaining data on the cost of administration and monitoring for installations covered by an emissions standard also presented a challenge. No literature sources were available to obtain an estimate on the transaction costs for a uniform emissions standard. Therefore these costs were derived from the ETS figures. We were comparing the elements in each transaction costs category (see Table 1) and decided if those costs would also occur under an emissions standard. We were e.g. assuming that the costs for monitoring the emissions and verifying the data are comparable. The costs related to trading were however excluded, such as costs for risk management or brokerage fees or the costs for managing a registry. Annual transaction costs of approximately €14k (small), €24k (medium) and €68k (large) were estimated for companies covered by the emissions standard.

4. Method

4.1 Abatement Cost Structure and Functional Form

According to the cost-effectiveness criterion, the marginal abatement costs (MACs) should be equated between installations covered in an ETS (Montgomery, 1972). This condition holds only when the MAC functions of the installations are convex and increasing across the full range of abatement. Several functional forms that satisfy these conditions and that suitably represent the abatement of CO_2 emissions have been identified by Böhringer *et al.* (2004). The identified functions exhibit the desirable property of having a value of zero at a given baseline (unregulated) emission level. One of the considered functions is the exponential function:

$$MAC_i = e^{\beta_i A_i} - 1, \qquad (7)$$

where A_i denotes the quantity of emissions abated by installation i, and β_i is a parameter to be estimated. This function was used for the empirical analysis in this paper, due to it satisfying many of the desirable properties for representing the marginal cost of abatement in this context. The total abatement cost (TAC_i) is given by the integral of the MAC_i function:

$$TAC_i = \frac{1}{\beta_i}\left(e^{\beta_i A_i} - 1\right) - A_i, \tag{8}$$

where the constant of integration *(c)* was eliminated by recognising that when the abatement level is zero, the TAC must also be zero.

4.2 Generating Abatement Cost Function Estimates

To simulate the heterogeneity between installations within an individual sector, it was assumed that each of the six considered sectors is composed of four installations, each of which corresponds to a quartile of the recorded CO_2 emissions for a given sector. The rationale for looking at quartiles was to have a manageable number of installations while maintaining some representation of the emissions structure of the sector as a whole. This process effectively amounts to classifying installations into representative groups of small, medium, large, and very large emitters within an industrial sector. The data on quartiles of CO_2 emissions for the seven industry sectors from EU ETS are presented in Table 2.

Given the functional form of the marginal abatement cost curve specified in equation (7) and the installation specific baseline CO_2 emissions values, it was possible to estimate the installation specific values for the parameter of the MAC function, β. The parameters were estimated using ordinary least squares so that the MAC function was fitted through the abatement cost data by minimising the sum of the squares. This was done by specifying an objective function corresponding to the sum of squares, and minimising it by varying the values for the parameters for each of the four representative installations in each industrial sector (see Table 3).

4.3 Simulating alternative coverage scenarios

Once the parameters of the marginal abatement cost functions for the representative installations in the industrial sectors were estimated, the costs and benefits of alternative levels of coverage under the two policies outlined in the theory section were derived. For policy 1 the aggregate benefit function was generated by calculating the total regulatory cost of blanket coverage of all installations in an ETS. For the simulation of policy 2, installations were transferred from being covered in an ETS to being covered by an emissions standard, with those installations that contribute the least marginal benefits from being covered in the ETS being transferred first. This process ensured that the resulting aggregate benefit function was concave in m (the efficient number of installations covered in an ETS). In order to determine the effect of the varying emissions reduction target (denoted by C in Eq.1 above) on the number of installations efficiently covered in an ETS, ten alternative aggregate CO_2 reduction targets were simulated. These went from 1% to 10% reduction in the annual aggregate CO_2 emissions. For example, a 3% aggregate annual CO_2 emission reduction amounts to 15% reduction over five years. An optimisation algorithm (EXCEL Solver) was then used to solve for the optimal number of installations covered in an ETS, along the lines of Equation 1.

5. Simulation Results

The results from the simulation runs are reported in Table 4. There are several observations that can be made from these results broadly supporting the theoretical proposition that an efficient level of coverage in an ETS is likely to be more cost-effective than a blanket coverage of all installations.

At each level of emissions reduction targets, the efficient coverage of installations in an ETS (policy 2) had a lower total cost of achieving the specified reduction target, compared to the blanket coverage (policy 1). The results also show the relationship between the efficient level of coverage of installations in an ETS, and the stringency of the aggregate emissions reduction target. As the reduction target becomes more stringent it becomes efficient to cover more installations in an ETS. This can be attributed to increases in efficiency gains from trading based on the differences in aggregated abatement costs under an ETS compared to the emissions standard, when the stringency of reduction targets is increasing. The implication is that the benefits accruing to each installation covered in an ETS—which originate in the heterogeneity of abatement costs—increase for all installations with ever more stringent reduction targets. As the cost of administration and monitoring remain unchanged regardless of the stringency of the target, the growing benefits of having an ETS outweighs these costs, and hence the aggregate net benefit from including installations in an ETS are quite high.

In general, we observe consistently large benefits from inclusion in the ETS for the Oil refining, Iron and steelworks and Cement sectors for each of the emission reduction targets. This translates into a pattern of near complete coverage for these sectors in stringent reduction scenarios, a likely consequence of high baseline emissions per installation and a high heterogeneity in abatement costs. In those sectors the emissions of small and medium emitters are high compared to the other sectors like

Combustion, Glass and Pulp and paper. Those sectors demonstrated a tendency that they would be excluded under an "efficient coverage" approach from the ETS at low emission reduction targets. Only for the largest combustion and glass installations the situation changed and they would be covered with an increasing stringency of the target. This pattern may be attributed to the large proportion of installations within those sectors with small baseline emissions. In this sense, low baseline emissions overshadow any abatement cost heterogeneity between installations, the consequence being that the benefits from inclusion are too low to outweigh the costs of coverage.

6. Summary, Conclusion and Policy implications

The question of how to design an emissions trading scheme in relation to the extent of coverage of installations is one of the key design issues that regulators across the world will have to address as they set up tradable allowance systems for greenhouse gas emissions. This paper provides conceptual and empirical insights on this issue.

From a conceptual perspective, it is important to identify the key elements of the criteria for efficient extent of coverage of installations in an ETS. Not surprisingly, these key elements turn out to be the benefits and the costs, both in total and at the margin, that can be attributed to the number of installations covered in an ETS. The more challenging task that this paper undertook was to represent benefits from regulating installations with blanket coverage in an ETS, and compare them to an alternative policy where those installations not covered by an ETS are regulated by an emissions standard. In addition, the costs of implementing alternative policies were broken down to cost of abatement and transactions cost including those of trading as well as those of administration, monitoring and other compliance cost. This kind of conceptual representation enabled the derivation of efficiency conditions for the efficient extent of coverage of installations in an ETS. Our theoretical analysis showed that an installation should be covered by the ETS as long as the marginal benefits of doing so exceed the marginal cost. Regulators following this criterion would ensure maximum net social benefit.

The empirical work presented in the paper supported these analytical findings and showed that blanket coverage of installations in an ETS is inferior to the coverage according to the efficiency criterion. The process of conducting the empirical analysis presented several challenges. Of these, some included issues related to the limited availability of data, such as abatement costs and potential as well as transaction costs data. Another challenge was the choice of marginal abatement cost functional form.[11] Most of those limitations were addressed by conducting sensitivity analyses, thus testing the robustness of our results.

The empirical study enabled us to further understand the relationships between the different variables (e.g. size of companies, transaction costs, overall emission reductions, heterogeneity in abatement costs). They confirmed the hypothesis that blanket coverage of installations by the ETS is not likely to be a cost-effective policy. In all cases blanket coverage was a more costly option compared to the efficient extent of coverage of installations. Dependent on the desired level of emission reduction, the efficient coverage of installations varies. In particular, for relatively small emission reduction targets (e.g. 3%) the difference in costs between blanket coverage and efficient coverage was rather notable, and only 3 of the 24 installations were efficiently covered by the ETS. For more

[11] We approached this challenge by being mindful of the desirable properties to be exhibited by the function, its tractability and computational limitations imposed by the choice of the functional form, as well as the possibility for interpretation of the parameters of the function.

ambitious reduction targets, the cost difference between the two options is diminished, and the number of installations efficiently covered in the ETS increased.

Our estimates are based on annual reductions reflecting the fact that our transaction costs and marginal abatement costs estimates are calculated on an annual basis. Annual reduction targets in the EU ETS have been very modest (3% per year for a five-year period, see Betz, Rogge and Schleich 2006) in Phase II (2008-2012) and for Phase III a linear annual factor of 1.74% is foreseen (CEC 2009).

These results also support the phasing-in of sectors over time, when targets get more stringent as was suggested by the New Zealand's scheme which is phasing in sectors over time in their ETS (New Zealand Government 2008). We recommend starting the ETS with sectors with relatively high emission baselines (emissions per installation) and a high heterogeneity in abatement costs. Over time when targets are getting more stringent we suggest to phase-in those sectors with relatively lower emission baselines and more homogeneous abatement costs. Our results are also in line with the latest changes to the EU ETS directive as described earlier, which will introduce an emissions threshold from 2013 onwards in order to exclude small emitters.

The potential to reduce transaction costs further in order to make a "blanket coverage" more efficient seems unlikely as the companies already have an incentive to meet the requirements such as reporting at lowest costs. Lowering the costs further may compromise the data quality or increase the risk of regulatory failure (e.g. by not detecting underreporting).

Another scenario, not reported here, of leaving the installations not covered by an ETS completely unregulated was also simulated. The results of that scenario are consistent with the ones presented here, and confirm the main findings. However, such a policy could lead to competitive distortions between covered and non covered installation in one sectors which are not acceptable by European law. In addition, leaving small emitters unregulated could lead to perverse incentives which would reduce economic efficiency. Companies could, for example, build a number of small installations instead of one big installation in order to circumvent any regulation. Therefore the EU ETS Directive foresees that the small installations which are not covered by the ETS are subject to equivalent measures (CEC2009). However, the equivalence of the other measures is not defined and is difficult to ensure given the volatility of the permit price.[12] Recent proposals for trading schemes in the US (America's Climate Security Act, ACSA 2008) and Australia (Commonwealth of Australia 2008) are proposing an upstream approach, which would counteract any perverse incentives or competitive distortions. Providing guidance on the cut-off criteria between downstream or upstream coverage should be on the agenda for future research. .

[12] Anecdotal evidence reveals that some small companies in the EU ETS were favouring paying the penalty for each of tonne of CO2e emitted if this would prevent any further engagement with the regulation. However, since the penalty involved a make good provision of permits later on and there is the risk of loosing the production license, this was not an option.

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Figure 1: Share of verified emissions 2005 compared to share of number of installations (Lorenz Curve) (Community Independent Transaction Log (CITL) data)

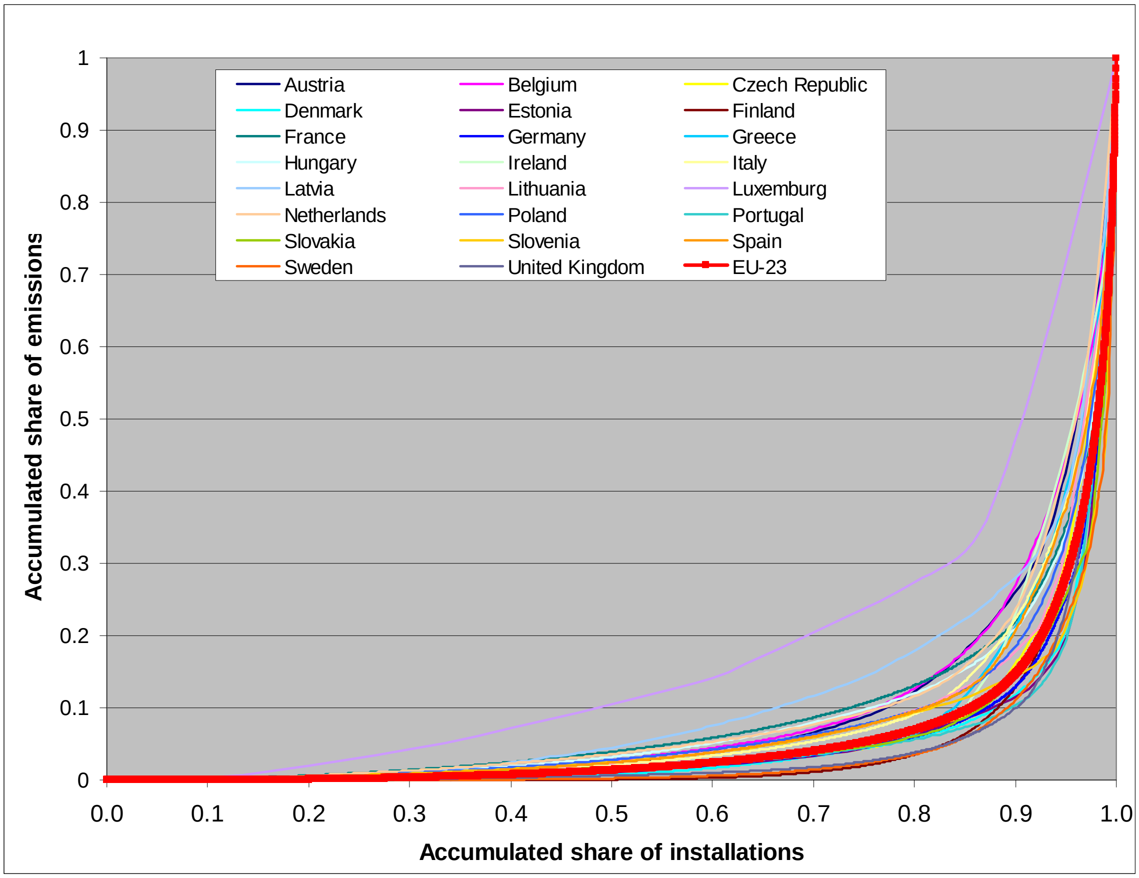

Table 1. Transaction cost estimates for ETS and standard (Divers sources see notes)

		Small emitters ≤20 kt		Medium emitters 20 – 1,700 kt		Large emitters ≥ 1,700 kt	
	Transaction costs category	ETS	Standard	ETS	Standard	ETS	Standard
Cost pertaining to the Government[a] (thousand Euros)	Administrative costs including operating the registry, conducting oversight, managing new entrants, allocation, sanctioning etc.	4	2	4	2	4	2
Cost pertaining to the regulated entities (thousand Euros)							
	Strategy and risk management and accounting[b]	5		9		12	
	Monitoring, Reporting and Verification[c]	12	12	22	22	66	66
Total Regulatory costs (thousand Euros per emitter)		**21**	**14**	**35**	**24**	**82**	**68**
Trading Costs[d] (Euros / EUA)		0.025		0.025			0.006
Fixed membership fee (thousand Euros per installation)							2.5

Note:

a) Ewringmann et al 2005 for ETS and it is assumed that half of that costs would occur under a standard
b) Betz 2003
c) Jaraite et al. 2009
d) EEX data for large and very large (http://www.ecx.eu/index.php/ECX-EUA-Futures-Fees-Margins) and OTC based on Convery and Redmond (2007)

Table 2. Representative small, medium, large, and very large emitters from each covered sector, with emission levels reported in kilo tonnes (kt) per annum (*Source:* Own manipulation of data from the Community Independent Transaction Log (CITL))

	Small emitter	**Medium emitter**	**Large emitter**	**Very Large emitter**
Sector	thousand tons of CO_2 emissions			
Cement and lime	48.54	218.62	544.79	2864.43
Combustion	4.62	14.85	52.66	12,497.63
Glass	15.24	34.47	72.84	592.75
Iron and steelworks	25.64	57.06	144.64	11534.47
Oil refining	157.69	574.11	1520.57	6266.75
Pulp and paper	6.59	18.40	43.22	421.19

Table 3. Estimates of the parameters of the marginal abatement cost functions for the representative installations (own estimates)

	Small emitter	**Medium emitter**	**Large emitter**	**Very Large emitter**
Sector	β	β	β	β
Cement	0.842	0.187	0.075	0.014
Combustion	2.590	0.806	0.227	0.001
Iron and steelworks	1.649	0.741	0.292	0.004
Glass	1.646	0.728	0.344	0.042
Oil refining	0.0768	0.0211	0.0080	0.0019
Pulp and paper	2.853	1.022	0.435	0.045

Table 4. Estimated total cost (including cost of abatement and cost of compliance) under each of the simulated policies and four emission reduction targets (own estimates)

Reduction target	**Total Cost Policy 1**	**Total Cost Policy 2**	**% saving**	**Number of installations efficiently covered in an ETS**
% per year	million € /year	million €/year	% per year	Number of installations out of 24
1	1.0	0.8	24.9	0
2	1.1	0.9	16.5	0
3	1.4	1.2	13.5	3
4	1.7	1.6	9.2	5
5	2.2	2.1	6.1	7
6	3.0	2.8	4.0	9
7	3.9	3.8	2.6	10
8	5.1	5.0	1.7	12
9	6.7	6.6	1.2	14
10	8.6	8.5	0.9	14