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The (Ir)relevance of Transaction Costs in Climate Policy Instrument Choice

An analysis of ex post transaction costs of different market and non-market based climate policy instruments in the European Union and the United States

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Abstract

This paper provides an assessment of the relevance of ex post transaction costs in the choice of climate policy instruments in the European Union (focusing mainly on the example of Germany) and the US. It offers a review of all publicly available empirical ex post transaction cost studies of climate policy instruments decomposed by the main private and public sector cost factors, and offers hypotheses on how these factors may scale depending on instrument design and other contextual factors. The key finding is that the hypothesis of ex post transaction costs asymmetries across instruments being large and thus playing a pivotal role in climate policy instrument choice can be rejected for the evaluated schemes. Both total and relative ex post transaction costs can be considered low. This conjecture differs from the experience in other areas of environmental policy instruments, where high total transaction costs are therefore considered as important factors in the overall assessment of optimal environmental policy choice. Against this background, the main claim of this paper is that in climate policy instrument choice ex post transaction cost considerations play a minor role for large countries that feature similar institutional characteristics as Germany and the US. Rather, the focus should be on efficiency properties of instruments in incentivizing abatement, as well as equity and political economy considerations (and any other societally relevant objectives). In order to inform transaction cost considerations in climate policy instrument choice in countries adopting new climate policies, more data would be desirable in order to enable more robust estimates of design- and context-specific transaction cost scaling factors.

Keywords

Transaction costs, climate policy instruments, emissions trading schemes, carbon tax, OECD, EU Emissions Trading Scheme

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1. Introduction

In recent years a significant body of literature has evolved analyzing the optimal design of climate policy instruments for correcting market failures such as the external costs of emitting harmful greenhouse gases (GHG) or R&D underinvestment due to technology spill-overs (e.g. Fischer & Newell, 2008; Goulder & Parry, 2008; IPCC, 2007). As demonstrated e.g. by Stavins (1995) and Coggan, Whitten & Bennett (2010) the design, implementation and enforcement of such policy instruments involves transaction costs that might affect optimal policy choice. In other words, the ranking of policy instruments regarding the economic cost effectiveness criterion might change when transaction costs are included in the analysis (Ofei-Mensah & Bennett, 2013). Transaction costs are defined as "*the ex ante costs of establishing environmental policy in all of its aspects, and the ex post costs of administering, monitoring and enforcing the policy once established*" (Krutilla & Krause, 2010). This article provides an assessment of the relevance of *ex post* transaction costs in the choice of climate policy instruments such as permit trading, taxes and standards by reviewing and comparing the available data on their transaction cost performance. This is done by reviewing all publicly available empirical *ex post* transaction cost studies of climate policy instruments decomposed by the main private and public sector cost factors. In order to inform transaction cost considerations in climate policy instrument choice in countries adopting or changing their instruments, hypotheses are offered on how these can be expected to scale depending on various instrument design or regulatory context factors such as the number of regulated entities or the level of abatement. Different design options (e.g. the point of regulation and the design of trading schemes) and their influence on transaction costs are discussed. Where quantitative data are not available, qualitative considerations and hypothetical plausibility deductions supplement the analysis. *Ex ante* transaction costs are not considered as hardly any empirical data are available, even though these costs may be important factors influencing the implementation of climate change policies.

The key finding is that the hypothesis of *ex post* transaction costs asymmetries being large and thus playing a pivotal role in climate policy instrument choice can be rejected for the evaluated instruments. Both total and relative *ex post* transaction costs incurred by the public and private sector can be considered low. In Germany, for instance, they range from € 28 to € 81 million[1] annually, for different instruments on which relatively robust empirical evidence is available. This represents between 6 and 18 ct€ per regulated ton of CO_2 in the year 2011, or less than 2% of the certificate price in the EU ETS (European Union Emission Trading System) in 2011 (€ 10), or less than 1% of German net renewable electricity subsidies in 2011 (€ 12 billion) (Frontier Economics, 2012).

The paper is structured as follows: Section 2 reviews the literature on transaction costs in climate policy instrument choice. Section 3 introduces the transaction cost definition and the methodical approach of this study. Section 4 offers results and a discussion of transaction costs for (i) trading schemes, (ii) taxing schemes, (iii) a comparison of different points of regulation for tax or trading schemes, (iv) technology standards and (v) performance standards. Section 5 concludes.

2. Literature Review

Stavins (1995) was the first to demonstrate analytically that in the context of environmental policy instruments transaction costs of trading permits can significantly affect the efficiency of an emissions trading scheme. He finds that if transaction costs drive a significant cost wedge between a firms option to abate internally or to trade permits on the market, this can significantly reduce the efficiency gains of harmonizing marginal abatement costs across regulated entities. This hypothesis has been confirmed empirically for several environmental emissions trading schemes in the United States: Kerr & Maré (1998) found that transaction costs of permit trading reduced cost effectiveness by around 10 - 20% in the US lead phase-down scheme. Gangadharan (2000) showed that the presence of trading transaction costs reduces the probability of trading by about 32% in California's Regional Clean Air Incentives Market (RECLAIM) scheme. Hahn & Hester (1989) demonstrated that high transaction costs decreased trading activity in the Fox River scheme in Wisconsin. In this context it must be noted that transaction costs have two distinctly different effects. Transaction costs cause direct costs (such as costs of monitoring, reporting and verification of emissions (MRV), brokerage and trading costs) that can be readily measured and are subject of this study. Transaction costs can furthermore distort optimal abatement and lead to welfare losses (Stavins, 1995). The latter effect is further discussed below but its quantification is not subject to this study.

Still, empirical studies of transaction costs in the environmental policy literature remain patchy because most authors have limited their transaction cost definition and analysis to the narrow neoclassical definition of *the cost of using the market mechanism,* which usually has been operationalized as the brokerage costs of trading permits. This narrow definition entails two problems. First, it covers only a small portion of total transaction costs. Second, it does not allow for comparison of transaction costs to other policy instruments where no trading takes place, such as taxes or standards. Against this background, McCann, Colby, Easter, Kasterine & Kuperan (2005) and Krutilla & Krause (2010) developed a more inclusive taxonomy (see below) which provides a useful framework for a comparative analysis of environmental policy instruments. This broader perspective has been adopted by several empirical studies in recent years which revealed that in trading schemes, brokerage costs are small compared to other transaction cost components such as the costs of MRV (Brockmann et al., 2012; Jaraite, Convery & Di Maria, 2010; Loeschel et al., 2010, 2011; VBW, 2013).

There are three other recent studies that investigated the relative transaction cost performance of different climate policy instruments. Betz et al. (2010) compare total costs (abatement costs plus transaction costs) of achieving a certain reduction target by means of uniform firm coverage under an ETS to a scheme where small emitters opt out of the ETS, and instead are covered under a performance standard. They find that with modest emission reduction targets, shifting from uniform to partial coverage leads to overall cost savings. Yet if the level of ambition for CO_2 abatement increases, overall savings decrease due to the relatively lower abatement efficiency of the performance standard.

Ofei-Mensah & Bennett (2013) compare three climate policy instruments in the Australian transport energy sector: Two standards providing enhanced consumer information (the Fuel Label Program and the voluntary Fuel Efficiency Program) and a hypothetical Tradable Permit and Fee System. They identify strong asymmetries in transaction costs per ton CO_2-eq abated between the standards (about $ 2,5/tCO_2-eq abated) and the Tradable Permit system (about $ 7,2/tCO_2-eq). However, this conclusion critically hinges on their chosen cost metric of costs per ton abated and the assumption that the trading system would yield only 6,7MtCO_2-eq total cumulated emission reductions over a 15 year period. If more cumulative abatement would occur in this time span – e.g. 67MtCO_2-eq– transaction costs of trading according to this metric would drop accordingly (in the example, to $ 0,72/tCO_2-eq abated).

A recent synthesis article by Mundaca et al. (2013), describes transaction costs as "an important factor in public policy" in the context of climate policy instruments. The article reviews transaction costs in the context of energy efficiency technologies, renewable energy technologies, offset carbon markets and the EU ETS. However, transaction costs are not presented in a consistent metric across instruments making it difficult to draw broader conclusions. Clearly, transaction costs have been high or even exorbitant for some projects of the Kyoto-mechanisms (Michaelowa & Jotzo (2005) or Antinori & Sathaye (2007).

This paper complements the existing literature by reviewing and comparing all publicly available empirical studies of climate instrument transaction costs. The aim is to discern whether transaction costs critically affect the efficiency ranking of different climate policy instruments.

3. Definitions and Approach

This study adopts the following transaction cost definition (Krutilla & Krause, 2010; based on McCann et al., 2005): "*Transaction costs are the ex ante costs of establishing environmental policy in all of its aspects, and the ex post costs of administering, monitoring and enforcing the policy once established*." In other words: *"[transaction costs are all costs of the policy] excluding abatement costs*". This definition covers all phases and aspects of costs that are needed to carry out a meaningful comparison across different policy instruments (see Table 1 for a general list of public sector transaction costs, and see Tables 2 and 3 in the following sections for a list of private and public sector transaction costs in the case of emissions trading, further differentiating the general *ex post* transaction cost components as indicated in Table 1).[2]

Transaction Costs
(i) Research and information
(ii) Enactment or litigation
(iii) Design and implementation
(iv) Support and administration
(v) Contracting
(vi) Monitoring/detection
(vii) Prosecution/enforcement

Table 1: Transaction costs associated with public policies. A detailed explanation of the cost factors is provided in endnote.[3] Source: McCann et al. 2005.

The focus of this paper is on *ex post* transaction costs (in the following abbreviated TC) because for climate policy instruments only very limited data exists on the *ex ante* stages (categories (i)-(iii) in Table 1).[4] In the short term, if significant these *ex ante* costs may have an influence on firms' decisions in the sense of possibly leading to some investment hiatus. However in the long run *ex post* costs will usually be the dominant factor (Betz, 2010).[5]

Other studies that have investigated the TC of policy instruments use concepts such as information costs, search costs, administrative costs, compliance costs, legal expenses, monitoring costs and enforcement costs (Destatis, 2011; Dr. Röver & Partner KG, 2006; Margaree Consultants Inc., 1998; Kossoy & Guigon, 2012; LECG, 2003; Sandford, Godwin & Hardwick, 1989, p.154; McMahon, Chan &

Chaitkin, 2000; Smulders & Vollebergh, 2001; VBW, 2013). The definition adopted here enables to integrate all of these cost components.

As pointed out by Mundaca et al. (2013) and Macher and Richman (2008) the existing empirical literature on transaction costs for policy instruments lacks a well-developed and comprehensive theoretical foundation. While this paper does not attempt to systematically fill this gap, it aims at contributing to the development of transaction cost theory by introducing conceptual distinctions of how *ex post* transaction cost dimensions for climate policy instruments can scale with respect to differing regulatory contexts and policy designs. The need for such specification of the scaling of TC arises when attempting to compare estimates of TC across empirical cases in order to draw conclusions regarding their anticipated TC when introduced in other regulatory contexts, that is, to inform the choice and set-up of novel climate policy instruments.

Building on the empirical studies reviewed in this paper as well as on conceptual considerations, we suggest distinguishing the following dimensions along which TC components of climate policy instruments can scale:

1. **The number of regulated entities**
   Firms' MRV costs make up a large share of total TC. The more entities are being regulated, the higher the total private sector TC will tend to be.
2. **The size of regulated entities**
   Small installations tend to have smaller total TC than large installations. However the TC per t/CO_2 regulated tend to be smaller for large installations due to economies of scale.
3. **The number of regulated appliances (in case of an appliance standard)**
   For homogenous mass produced goods (such as household appliances or vehicles) certification must be carried out for each model of appliance (e.g. fridge or dryer) that is offered in a certain market. The TC therefore scale with the amount of different models of appliances that are regulated (but not with the total amount of appliances sold).
4. **The level of abatement**
   Identifying abatement options on the firm level and establishing an abatement cost curve is costly. Increasing levels of abatement therefore should thus raise TC.
5. **The volume of market transactions (in case of trading mechanisms)**
   Trading TC include the brokerage costs which must be paid for each traded unit (e.g. CO_2 certificate), and thus vary with the number of traded certificates.
6. **Other instrument or transaction costs-specific factors**
   For example, the magnitude of transaction costs from legal disputes will partly depend on how firms estimate the costs of the lawsuit versus the chances of winning the case. The more favorable this ratio, the more likely legal action will tend to be. Other factors such as the costs of lawsuits, corporate laws, and legal norms can also play a role for this specific cost factor.
7. **Time**
   Time as a TC scaling factor cuts both across the basic TC categories and the previous scaling dimensions: Cost factors (and the amount to which they scale) are likely to change over time. Due to e.g. learning-by-doing and technological change effects one might generally expect that absolute TC would fall over time (e.g. trading costs in the US SO_2 allowance market fell 98% over time (Joskow & Schmalensee, 1998; LECG, 2003)).

These dimensions can interact. Consider a limiting case: When, over time, the level of abatement is becoming so ambitious that emissions are approaching zero, less and less entities will remain to be regulated, and absolute trading volumes will tend to fall. TC will probably rise with respect to abatement cost information, but decline with respect to the other factors.

In addition there are fixed costs at the inception of the policy instrument that occur only once (such as administrative, informational and capital costs of setting up novel compliance structures), as opposed to annual running costs of a scheme over the long term (Jaraite et al., 2010). Finally, for clarity it is useful to distinguish TC components and scaling factors at the firm and at the aggregate system level (indicating total TC). This paper focuses on the latter.

The methodical approach of this paper is to assemble all publicly available *ex post* transaction cost data on existing climate policy instruments, and to use evidence from other policy instruments that are plausible analogies. The latter approach is taken only where TC data on climate policy instruments is scarce or non-available (e.g. CO_2 tax). Data is obtained from case studies, consultant reports, government reports, interviews, and own calculations (Appendix 1 provides a list of the publications that were reviewed for this study). Where possible, data are broken down into the *ex post* cost components as defined above. Overall, data are limited but available for the EU ETS, the US SO_2 ETS, US RECLAIM, the UK excise duties on hydrocarbon fuels and US Residential Appliance Standard. Where different studies report differing values for a scheme, we focus on the higher estimates to ensure that our aggregation is on the conservative (i.e. high-cost) side. However, by indicating ranges we also display the low-cost estimates. For policy instruments where neither data nor useful analogies exist (e.g. technology standards for large point sources) costs are assessed qualitatively.

The use of the metric for reporting TC depends on the quality of data and the purpose these are intended to serve. If the aim is to inform the comparison and choice of climate policy instruments in countries that consider adopting novel policies, there is no single metric that could convey all of the desired information in a satisfying manner. This is because as argued above cost components scale with respect to context- and policy-specific dimensions. Quantitative scaling factors cannot (yet) be reliably estimated based on the very few existing data points. This can be illustrated with regard to two prominent TC metrics used in the literature.

First, reporting *TC per regulated GHG units* (such as €ct/tonCO_{2eq}) is an elegant way of expressing TC in a given ETS relative to the visible price of an emission allowance. It is most often used for comparing MRV costs across firms in order to identify MRV TC scaling effects with respect to firm size (e.g. Jaraite et al., 2010; Loeschel et al., 2011). We make use of this metric when discussing these dimensions, but using this metric more generally for a comparison of TC across policy instruments and regions faces the challenge of masking potentially significant context-specific factors such as the mix of large and small regulated entities in a scheme.

Second, reporting TC per unit of GHG abated only makes sense when considering cases with comparable abatement levels. Consider the following example: In the first trading period of the EU ETS (2005-2007) little abatement occurred (Ellerman et al., 2010 p. 191), while the EU ETS incurred full TC. Thus, TC per abated emissions were relatively high. If the cap had been much tighter, TC per abated t/CO_2 would have been much lower, basically without changing the absolute TC as these are very likely to scale only to a limited extent with regard to abatement level (see Section 4.1). In this case and using this cost metric, in the one case very high and in the other very low TC would be reported, while absolute TC would have remained roughly constant. Without further qualification, such information seems not to be very helpful and potentially misleading for comparing costs across instruments and regions.

To sum up, if multiple data points on TC components in different contexts were available - and they might become available in the future -, this could inform numeric estimates of the different scaling factors. This is currently not the case. This study copes with this challenge by reporting total TC per country, using Germany and the United States as examples, for specific (climate) policy instruments and reporting in detail the magnitude of different cost components and factors potentially scaling these if transferred to other contexts. Also, some sensitivity analyses of TC with respect to program design are conducted (up- versus downstream point of regulation for the EU ETS). The main limitations of this approach are the same as for the other metrics, i.e. results are not directly comparable to other countries, and policy instruments are not directly comparable e.g. due to differences in the coverage and level of abatement they induce. Nevertheless, bearing this caveat in mind the existing data indicate that total TC of different instruments in Germany and US are low in relation to other macroeconomic figures, and relative costs of different instruments range within similar orders of magnitude.

4. Instrument Comparison

This section briefly introduces each considered climate policy instrument, followed by an analysis and discussion of its TC data components. An exception is Section 4.3, which compares different points of regulation (up- or downstream) for GHG trading and taxation schemes. The discussion of each cost component is followed by a hypothetical plausibility analysis of its dynamics with respect to its scalability.

4.1 Emissions Trading

Emissions trading is a *quantity instrument* because the maximum amount of permissible emissions for the regulated entities is set by the regulator (Tietenberg, 2006). Emission permits are distributed either by free allocation or auctioning, and can be traded among polluting entities. Firms engage in abatement efforts and permit trading until the equilibrium permit price emerges. Under ideal conditions the instrument is cost-effective because marginal abatement costs (MAC) are indicated by the market permit price and equalized across all participating firms.

The EU ETS is by far the largest application of emissions trading worldwide, and a number of TC studies of the EU ETS have been conducted, in particular in Germany and Ireland. They are usually based upon questionnaires handed out to companies, or on expert estimations.

These studies exclusively focus on private sector TC. The analysis in this paper adds public sector costs. Due to the comparatively good data availability on both private and public transaction costs in Germany, we take this country as a case study for EU ETS data, with a comparison to Irish data showing that private sector figures seem to be robust. Where several figures are available for the same cost component, the higher estimate was always adopted. Public agency costs in Germany are considered to be on the higher end (on a transaction costs per regulated entities basis) in EU comparison (Seidel, 2011). We have therefore used the German public sector transaction costs as a basis for the calculation of EU wide public sector transaction costs.[6]

Private Sector Costs

The EU ETS transaction costs components of the private sector are displayed in Table 2.

Private Sector Ex-Post Transaction Costs
(i) Assembling information on cost effective abatement at the facility-level
(ii) Monitoring, reporting and verification (MRV)
(iii) Application for free allocation (unless permits are auctioned)
(iv) Legal Expenses
(v) Trading permits

Table 2: Classification of private sector ex post transaction costs in the EU ETS.

Assembling information on cost effective abatement at the facility-level: Before firms can engage in abatement they need to assemble information over their abatement cost schedule. This involves financial and technical analysis, an analysis on the impacts for production and product quality (Hein & Blok, 1995). This requires additional personnel expenditure and advice from experts, including calculations and risk assessments for payback periods. It can be expected that these costs rise with an increasing abatement ambition because it becomes more difficult to find new abatement options once easy abatement options have been exploited. However with only € 0,9 million annual costs regulated for the EU ETS in Germany (Loeschel et al., 2011) these costs are currently very small and seen negligible compared to other cost components.[7]

MRV (monitoring, reporting and verification) of emissions is crucial to all market-based schemes since regulators require reliable emissions data, and firms have an incentive to underreport and over-emit. In the EU ETS, firms must report their emissions annually and have these reports verified by an external verifier, who must be accredited by a government agency (DEHSt, 2012a). Firms' unit MRV costs differ strongly with the size of regulated installations. This non-linearity of firms' MRV costs (see Figure 1) is due to economies of scale in the MRV process featuring relatively high facility-level fixed costs and has been well documented by several studies (Betz, 2005; CEC, 2008; Frasch, 2007; Heindl, 2012; Jaraite et al., 2010; Loeschel et al., 2010, 2011; Schleich & Betz, 2004). It has sometimes been argued that this may lead to a competitive disadvantage for small emitters (Jaraite et al., 2010). Heindl (2012) suggests that this effect may theoretically lead to a larger optimal firm-size and might even induce increasing market power in the permit market. However, given the low orders of magnitude involved, the empirical effect is very likely to be practically irrelevant.[8] Yet, TC of €1,51 per t/CO_2, as reported by Jaraite et al. (2010) for Ireland may lead to overproportional hardships for certain firms.

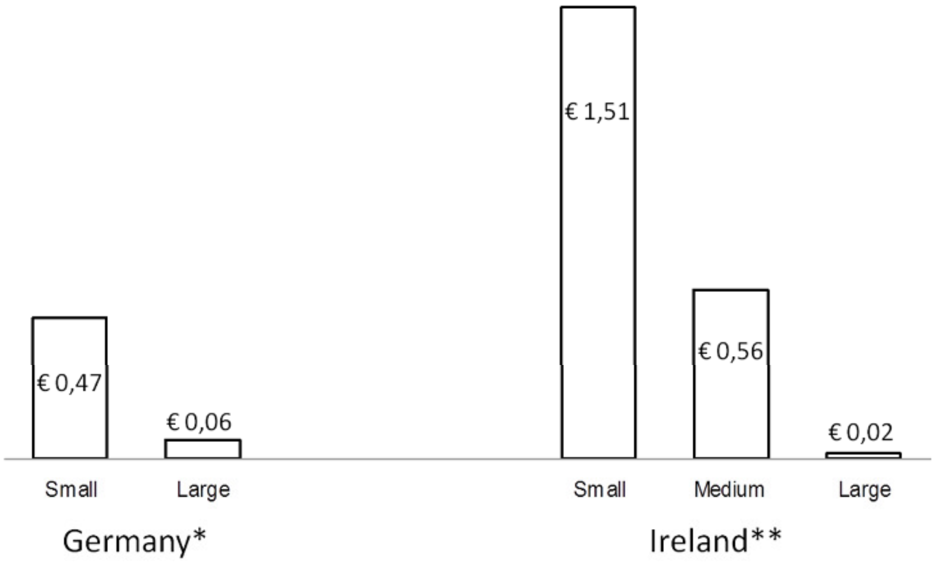


*Figure 1: EU ETS annual MRV running costs for small and large installations in € / tCO_2 regulated. *Median; **Average costs. Sources: Jaraite et al., (2010); Loeschel et al., (2011).*

In the period 2009-2011, average annual private sector MRV costs in Germany were reported between 2 €ct/tCO_2 (Loeschel et al., 2011) and 9 €ct/tCO_2 emitted (Destatis, 2011). In terms of absolute average per-entity costs this translates into roughly € 2.500 p.a. for small installations (annual emissions below 25.000 tCO_2) and roughly € 8.600 for large installations (annual emissions above 25.000 tCO_2).[9] For Germany, total MRV TC estimates range from € 11 to € 40 million p.a. As illustrated in Figure 2, MRV constitutes the largest share of private sector TCs (61% - 81% of total private costs[10]; Destatis, 2011; Loeschel et al., 2011; own calculation). Aggregate MRV costs scale with the number of regulated facilities and the mix of large and small sized entities (small/large emitters have lower/higher absolute but higher/lower per unit GHG costs) (Jaraite et al., 2010; Loeschel et al., 2011; Frasch, 2007).These costs are independent from the abatement level because MRV processes have to be carried out for all emitted emissions independent from the level of abatement. As for other components it can be expected that cost fall over time due to learning effects and improved technology.

Concerning the *allocation of permits,* firms covered by the EU ETS have received the largest amount of certificates for free in the first two trading periods (2005-2012). In the third trading period (2013-2020) free certificates are handed out based on firms' export shares and relative efficiency (benchmarking).[11] Brockmann et al. (2012) found that for the third trading period the median TC of applying for free allocation of certificates amounted to a one-time expenditure of € 25.000 for personnel and external costs per firm at the beginning of the trading period. Extrapolating to all covered firms and distributed over 8 years, this amounts to roughly € 2,5 million total annual costs for all firms in the case of Germany.[12] The dynamics of this cost component depends on the amount of firms that qualify for free allocation and therefore on the eligibility criteria for free certificates. The larger the number of eligible entities, the larger the aggregate TC from this component.

Since the rules for granting free certificates are not clearly defined, firms exercise their right to legally dispute allocations in court.[13] A survey by VBW (2013) finds that these figures translate into € 5.400 litigation costs per installation in Germany, or € 10,7 million overall p.a. [14] The scalability of this component is not obvious. It can however be expected that legal action scales with the success rate in relation to the costs of litigation. Yet, also other factors such as legal traditions, corporate laws and political acceptance of regulations by companies may have an influence.

Permit *trading* is central to an ETS as it leads to equalization of marginal abatement costs across all firms. In his seminal study on transaction costs Stavins (1995) shows that transaction costs of permit trading which he defines as the "direct financial costs of brokerage services" can lead to inefficient allocation of abatement across firms. From a cost-effectiveness perspective trading costs drive a wedge between firms' MAC. Transaction costs reduce beneficial permit exchanges and corresponding adjustments in the allocation of abatement, thus reducing the cost effectiveness of the policy instrument (Stavins, 1995). In the EU ETS in 2011 the trading volume amounted to 9,7 billion certificates (Kossoy & Guigonet, 2012).[15] Trading TC range from 0,1 - 10 €ct/tCO_2, with costs apparently having fallen towards the lower end of this range over time (Convery & Redmond, 2007; EEX, 2012; Hacker, 2012; Kossoy & Guigonet, 2012; Weber, 2012). In the EU ETS overall trading costs amounted to roughly € 46 million in 2011, and about € 11 million p.a. for Germany (see Appendix 2 for calculation). These trading costs are small and there is no empirical indication that they discourage trading (Jaraite et al., 2010). Negative cost effectiveness effects as was the central concern of Stavins (1995) are therefore likely to be minor for the case of the EU ETS. Trading TC scale with the volume of trading because brokerage costs arise for each transaction. In the EU ETS it can be observed that the majority of trades are carried out by market intermediaries seeking arbitrage opportunities, and not by regulated firms (Weber, 2012). It is therefore likely that price fluctuations (that give rise to profit opportunities for market intermediaries) determine the trading levels rather than the amount of regulated GHG or the amount of abated emissions, even though in a larger trading system overall higher aggregate trading levels would be expected than in smaller schemes. It can furthermore be expected that trading costs fall over time, as illustrated by the comparison with other environmental permit trading schemes below.

Public Sector Costs

The EU ETS TC components to the public sector are displayed in Table 3.

Public Sector Ex-Post Transaction Costs
(i) Compliance agency
(ii) Registry
(iii) Auctioning

Table 3: Classification of public sector ex post TC in the EU ETS.

The *compliance agency* is the central authority to administer an ETS. It oversees the MRV process by conducting sample checks, allocates free certificates, appoints exchanges to auction certificates, provides information, and engages in litigation. The total annual costs for these tasks amounted to roughly € 15 million or roughly € 7.700 per regulated entity for Germany in the period 2005 - 2011 (CEC, 2012a; DEHSt, 2008; Dr. Röver & Partner KG, 2006; Seidel, 2011).[16] The compliance agency TC scale with several factors, of which the number of regulated entities is probably the most decisive. Furthermore there are initial start-up costs such as setting up administrative processes and communicating and coordinating these with regulated entities. It is likely that the cost of the compliance agency also scales with the quality of its performance since e.g. rigorous verification will be more costly than lax controls. In terms of per-entity costs, these may decline with increasing number of regulated entities due to economies of scale; an example on the public sector side may be the emission registry operation and maintenance.

The *public registry* is the electronic database for reconciling certificate accounts and the emission data of all regulated entities and is usually run by the compliance agency. Costs related to the registry are mainly software and servicing costs. In Germany they averaged about € 0,76 million per year for the first and second trading period (DEHSt, 2011a). This is largely a fixed component because the costs are determined by licensing and maintenance costs of the software. It however also contains a variable component since it is likely that the costs of the software scales with the number of accounts. Due to improved technology, costs may fall over time. However costs may also rise due to higher regulation requirements and safeguards against fraud (European Voice, 2013).

Auctioning is the alternative to free allocation of certificates. The costs charged by exchanges to carry out the auctions on behalf of governments are about 0,3 ct per certificate (Point Carbon, 2011). A 10% share of auctioned permits in Germany in 2011 amounted to roughly € 0,13 million, TC while a 50% share (as implemented from 2013 onwards) would amount to TC of € 0,7 million per year (DEHSt, 2012b). These costs clearly scale with the amount of auctioned certificates. As in the case of trading, these costs may fall over time.

Offsets for Kyoto Mechanisms

In the EU ETS firms can also use credits from the Kyoto offset mechanisms (Clean Development Mechanism (CDM), and Joint Implementation (JI)) for compliance. These credits usually in total cheaper (due to lower abatement costs) than EU ETS allowances but tend to feature higher TC (due to a costly project cycle). In the second trading period 302 million Kyoto offset credits have been used for compliance in Germany (CEC, 2014; DEHStd, 2014).[17] Based on data by Krey (2005), Antinori and Sathaye (2007) and Michaelowa and Jotzo (2005) we find a range of possible TC resulting from the use of offset credits for Germany between €15 - 32 million p.a.[18] It must however be noted that these TC do not affect the companies in the EU ETS directly since the TC are already incorporated in the offset price, which is usually lower than costs of European Union Allowances (EUAs). Nevertheless, these costs hamper the efficiency of the EU ETS link to the CDM via the distortive effect identified by Stavins (1995).

Due to the various cost components of the CDM process cycle[19] a separate analysis on the dynamics of these cost components should be subject to further research but is outside the scope of this study. For the EU ETS the total TC incurred by offset credits clearly scale with the number of credits used for compliance.

Summary, Discussion, and Comparison to other Environmental Permit Trading Schemes

Figure 2 aggregates *ex post* TC estimates of the EU ETS in Germany. Total annual costs in the one year trading period 2010-2011 amount to € 81 million in Germany, or 18 €ct/tCO_2 being regulated.[20] Private sector costs are 68 - 80% of total TC, while the public sector costs are 20 - 32%.[21]

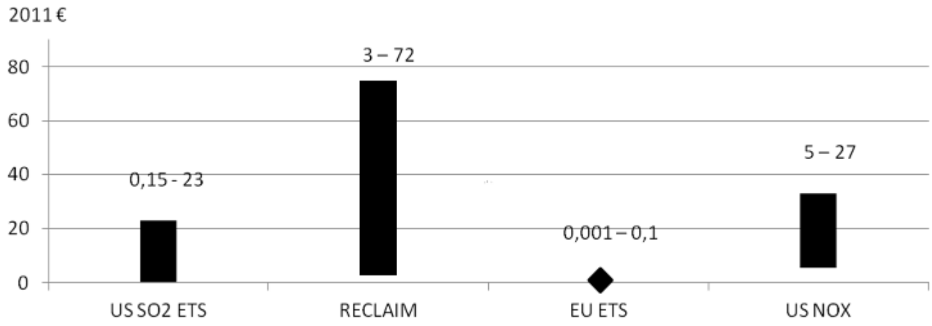
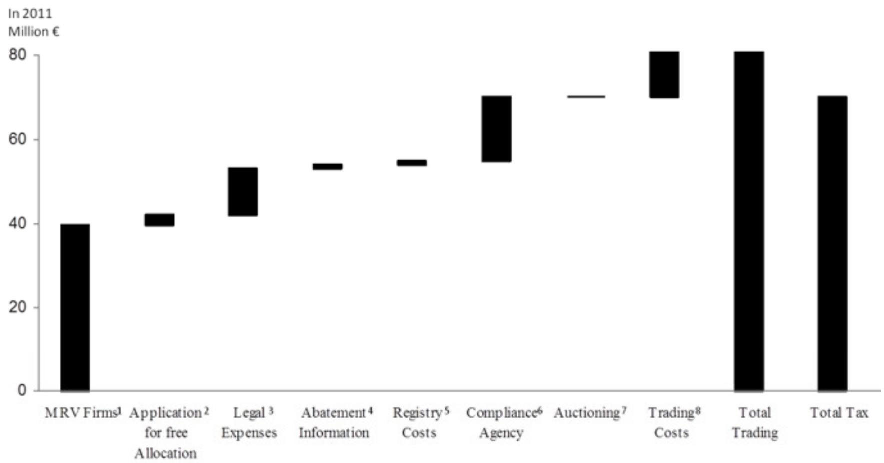


Figure 2: Year 2011 EU ETS and hypothetical carbon tax TC for different cost components for Germany based on data from the first and second trading period in total costs. The bars on the left side indicate the cost components using conservative (higher) cost data, the table on the right indicates cost ranges. CDM TC are excluded to focus on the direct costs of the EU ETS.

Data sources:

1. Destatis, 2011[22]; 2. Brockmann et al. 2012; Destatis, 2011; 3. VBW, 2012; 4. Loeschel et al., 2011; 5. DEHSta, 2011; 6. Dr. Röver & Partner KG, 2006; DEHStb, 2011; DEHSt, 2008; CECa, 2012; 7. EEX, 2012; Point Carbon, 2011; 8. Kossoy & Guigonet al., 2012; Weber, 2012, EEX 2012. If numbers differed across the studies for the same cost components we always used the highest estimate in order to ensure robustness of results.

Several studies have analyzed the effects of trading costs of trading schemes targeting other environmental pollutants (Joskow & Schmalensee, 1998; LECG, 2003; Stavins, 1995). These schemes feature much higher trading costs, especially during their inception. The RECLAIM scheme featured costs from 3 - 72€ per traded unit (tNOx and tSOx), the US NO_x budget trading scheme featured trading TC of € 5-27 per unit (LECG, 2003) and the US SO_2 scheme are today in the range of € 0,15, but started at around € 23 (Joskow & Schmalensee, 1998; LECG, 2003) (Figure 3). Due to learning effects by market participants and brokers and furthermore due to improved technology the costs dropped more than 98% over time. The claim of inefficient abatement and resulting welfare losses due to TC of permit trading (Stavins, 1995) thus seems to be more relevant for these schemes (e.g. Gangadharan, 2000), and especially at the time of their inception, than for the EU ETS where it seems sensible to assume that the relevance of this effect is negligible.



	Total costs in 2011 million €
MRV Firms	11 - 40
Application for Allocation	2,4 - 2,6
Legal Expenses	10,7
Abatement Information	0,9
Registry Costs	0,8
Compliance Agency	15
Auctioning	0,13
Trading costs	11
Total Trading	51 - 81
Total Tax	41 - 71

Figure 3: Ranges of ETS per unit (permit) trading costs comparison for private sector.
Sources: Joskow & Schmalensee, 1998; Margaree Consultants, 1998; LECG, 2003; EEX, 2012; BlueNext, 2012; Convery & Redmond, 2007; Hacker, 2012; Weber, 2012.

The comparison of public sector TC reveals that US environmental permit trading schemes are leaner than the EU ETS. While the EU ETS, and the NO_x budget ETS in the US have running costs of € 3.900 and € 7.700 per regulated entity, respectively, the US SO_2 scheme is far less costly with only € 780 per regulated entity[23] (Dr. Röver & Partner KG, 2006; DEHStb, 2011; DEHSt, 2008; LECG, 2003; McLean, 1997). For the US SO_2 scheme the total firms' MRV costs amounted to approximately € 100 million and the trading costs to approximately € 3,5 million. Together with the € 1,6 million from public sector costs this amounts to total TC of € 105 million (EPA, 2001; LECG, 2003; Joskow & Schmalensee, 1998).[24,25]

All of these metrics suggest that the TC of the EU ETS are not prohibitively high in absolute terms in a macro-economic perspective. In any case, for policy instrument choice it is the relative costs of instruments that matter. This point is addressed in the conclusions.

4.2 Tax

Emissions taxation is a *price instrument* as the emission tax is set by the regulator, while the emission quantity is determined by firms' market behavior. Like a permit trading scheme, a carbon tax can achieve the equalization of marginal abatement costs across all regulated entities. Under standard assumptions of competitive markets trading and taxation schemes are symmetric. Asymmetries of instrument performance have been discussed for conditions of uncertainty (Weitzman, 1974; Hepburn, 2006) or incentive structures for subnational jurisdictions in a federal system (Shobe & Burtraw, 2012). This study investigates how taxing and trading compare with regard to TC performance.

A TC comparison of tax and trading schemes needs to be very careful not to compare "apples with oranges" as is sometimes done in the public debate, where real world "second best" schemes (such as the EU ETS) are compared to theoretical "first best" taxation schemes. In particular, specifications regarding the point of regulation and of rent distribution need to be treated in a conceptually equivalent manner. A downstream trading scheme with a large portion of grandfathered certificates (such as the EU ETS) should be compared to a downstream taxing scheme with an equivalent amount of tax exemptions. Even though the incentives for abatement may be different, the two options (grandfathering and exemptions) should be regarded as analogous responses to identical political economy considerations.[26,27]

Firms' *MRV costs* would be the same in both schemes for the firms that are subject to MRV since data requirements are identical.[28] Drawing on data from the previous section, a downstream taxing scheme for Germany with tax exemptions equivalent to free allocation in the EU ETS would cause TC of approximately € 41 - 71 million (see Figure 2). This is approximately € 10 million less than a trading scheme, which appears negligible for cost effectiveness considerations of policy instrument choice in a macroeconomic perspective.

A different approach to determine the TC of a CO_2 tax is to compare it to other taxes that require similar administrative efforts by the private and public sector. Since a CO_2 tax would be designed like a levy on fuel according to its CO_2 content, excise duties on hydrocarbon fuels are a good proxy. Data from a study on administrative and compliance costs of excises on hydrocarbon oils are available for the United Kingdom (Sandford et al., 1989 p.155).[29] The tax covers light oil, road vehicle fuel, fuel oil and gas oil.[30] The major findings from this study are that the majority of costs are the MRV costs of installations and the reading of meters (Sandford et al., 1989 p.156). For the years 1986-1987, public sector costs amounted to € 14,2 million and private sector costs were € 31,7 million for the entire UK.[31] The TC for taxation schemes feature the same cost components (except of auctioning and trading) as an

ETS. Thus, the same considerations concerning the scaling of these cost factors with respect to different context and design variables apply (see section 4.1).

4.3 Carbon Pricing Transaction Costs: Upstream versus Downstream Point of Regulation

The point of regulation is the location in the fossil fuel processing chain where the regulator carries out MRV and requires the submission of emission permits. Options can be distinguished according to up-, mid- and downstream regulation. For upstream schemes, the point of regulation is at the exploitation or import of the fossil fuel.[32] All other downstream consumers would not be subject to any MRV and trading, thus eliminating all TC for downstream firms. The upstream carbon price can be expected to be devolved to the downstream level, as is standard procedure with vehicle fuel excise duties. This approach would eliminate the inefficiency of non-linear MRV costs for small companies as discussed in Section 4.1 and illustrated in Figure 1. By contrast, midstream regulation is carried out at the processing or storing level, and downstream regulation is exercised at firm (as in the EU ETS) or even final consumer level (Flachsland et al., 2011).[33]

The TC savings resulting from adopting an upstream scheme compared to a downstream scheme depend on two factors. First, on the difference of TC per emitted t/CO_2 between small and large entities, and second on the ratio between small and large entities.[34,35]

Downstream			Upstream			
	TC per t CO_2	TC total in million		TC per t CO_2	TC total low million	TC total high million
Large firms	€ 0,02	€ 9	Producers and Importers	€ 0,01[1]	€ 2,7[2]	€ 10[3]
Small firms	€ 0,5	€ 3	Small firms		No costs	
Total		€ 12			€ 2,7	€ 10
Savings: € 2 - € 9,3 million						

Table 4: Potential saving from switching to EU ETS upstream regulation in Germany [36]
1)Very large installation have TC of about € 0,01 (Loeschel et al., 2011)
2) For the low estimate we have calculated the MRV costs for a large refinery (1,8 million tCO_2 emissions; MRV costs of € 18.000 per year)[37]*, and used this number as the average TC for the 150 importers and producers.*
3) This calculation is based on the TC for large installations in Ireland (we assume them to have similar TC as large installations in Germany).These costs amount to are € 66.000 per year (Jaraite et al., 2010).

These calculations show that switching the point of regulation from downstream to upstream could bring savings in the range of € 2- 9 million p.a.for Germany. Especially small firms that currently feature high relative TC would benefit from these savings. However, the savings are limited in absolute terms because most emissions (over 99%) in Germany come from large sources that already feature small TC today. The scope for total TC reductions in a macroeconomic perspective is again limited. At least as long as other sectors (i.e. heating and transport) are not included in the ETS, the benefits of switching to upstream regulation in the EU ETS thus appear to be limited.[38] The cost savings roughly calculated here also need to be pitched against the full *ex ante* transaction (opportunity) costs of negotiating and implementing such a change. However, the calculations suggest that ETS that are newly set up should opt for upstream coverage from their very inception at least from an *ex post* transaction cost perspective.[39]

4.4 Technology Standard

Technology standards are the most traditional instrument of environmental policy making. One major reason is that they are quite easy to design and monitor (Sterner, 2003 p. 80), i.e. *ex ante* and *ex post* TC are often considered to be low. The drawback of this instrument is that an equalization of marginal abatement costs across facilities is not possible. Strongly differing marginal abatement costs among firms leads to low overall cost effectiveness of the instrument. This has led to a relative decline in the adoption of technology standards compared to market-based instruments in recent years (Aldy & Stavins, 2011).

From a TC perspective some forms of technology standards for point sources have the advantage that only little MRV is necessary. Since, in many instances, it is not worth the effort to remove technologies once installed, random and sporadic monitoring is sufficient to enforce compliance and the public sector effort and data requirements are low (Hepburn, 2006).[40] Other forms of technology standards, such as the requirement to inspect domestic heating systems for CO_2-emissions in Germany MRV must be carried out regularly.[41] Such forms of technology standards therefore might incur substantial MRV TC, but no data are available. Data for a standard for the regulation of large point sources (e.g. the planned CO_2 standards for power plants in the US (EPA, 2013) or the regulation of SO_2 via scrubbers in Germany) are not available. For the purpose of this study, no general quantitative estimate for MRV costs of technology standards for large point sources could be identified or can be plausibly determined by analogy. It seems safe to assume that they will be lower than those for any other policy instrument considered in this study.

4.5 Performance Standards

A performance standard limits the amount of emissions for a certain unit of output (e.g. CO_2 per ton of cement, CO_2 per driven km, kWh per liter refrigerator capacity). In contrast to a technology standard it has the advantage that firms are free to choose emission reduction options in the production process, rather than being obliged to fulfill specific technical requirements. It furthermore has the advantage that the standard can be easily adapted to technological progress (e.g. Japan's Top Runner Program).[42] Non-tradable performance standards have the disadvantage that a harmonization of marginal abatement costs across firms is not possible.[43] From a TC perspective a performance standard for heterogeneous point sources of pollution entities (e.g. cement factories or steel mills) does not entail significant cost reductions relative to an emission trading system as analogically all emissions have to be monitored and verified, and trading costs arise as well (in the case of *white certificate* trading). The costs of monitoring are reduced if a performance standard is set for homogenous non-point source pollution goods that are subject to mass production (e.g. vehicles or household appliances), and no regular inspection is required. Each model of appliance (e.g. fridge or dryer) that is offered in a certain market must be certified only once. Three different performance standards are briefly discussed below.

The *U.S. Appliance Standard* covers a variety of appliances such as refrigerators, space heating, water heating and other electrical equipment. For compliance, the products must perform better than a maximum limit of energy consumption per unit of the relevant output. Standard setting, test procedures, certification and enforcement are carried out by the U.S. Department of Energy.

The calculation of annual TC from different sources is summarized in Table 5.

Total costs 2011 € Million*	Source
4,1	Levin et al. 1994
5,0	McMahon et al. 2000
9,3	Meyers et al. 2003
10,1	McMahon et al. 2000
11,6	Meyers et al. 2003
13,9	Gillingham et al. 2004
17,4	Gillingham et al. 2004

Table 5: Annual public sector TC of the U.S. Appliance Standard in € 2011.
** As cost data is adjusted for inflation and converted into € (see endnote 1) it is not identical to original data. The original data can be found in Appendix 1.*
Sources: Gillingham, Newell & Palmer, 2004; Levin, Hirst, Koomey, McMahon & Sanstad, 1994; Meyers, McMahon, McNeil & Liu, 2003; McMahon, Chan & Chaitkin, 2000

The total annual public sector TC of the U.S. Appliance Standard range from € 4,1 to 17,4 million in 2011.[44,45] These TC are substantially lower than for the other climate policy instruments reviewed in this paper, indicating that the regulation via appliance standards appears relatively cheap. This advantage needs to be pitched against the abatement opportunities related to this instrument (which are likely to be limited) and the inferior overall cost effectiveness performance of technological standards related to incomplete marginal abatement cost harmonization, the salience of which will increase with the rising stringency of a program. This point is addressed in Section 5. The TC of an appliance standard are likely to scale mainly with the number of different appliances covered since MRV (i.e. certification) must be carried out for each model. Furthermore there are also ongoing fixed TC on the government and the firms' side. If regular inspection of a device is required (e.g. in case of home heating systems in Germany), the number of appliances as well as the inspection procedure will scale the total TC of that instrument.

A *hypothetical performance standard for large point sources in Germany*, covering the sectors regulated under the EU ETS, would cause TC of roughly € 41 - € 71 million p.a. The rationale behind this calculation is that the costs of MRV, the registry and the public agency will be roughly identical to those of an ETS or tax.[46]

The TC for the *EU Emission Vehicle Standard*[47] and the *US Corporate Average Fuel Economy (CAFE) program*[48] are likely to be low but no data are available. The scaling properties of TC are likely to be analogous to an appliance standard. Each model (but not every single vehicle) has to be certified prior to sales but regular inspections of each vehicle are not necessary. Furthermore there are also ongoing fixed TC on the government and the firms' side.

5. Conclusions

Table 6 and Figure 4 summarize the reviewed TC data of different (climate) policy instruments. They show that *ex post* transaction costs of policy instruments in large industrialized countries are relatively low in macroeconomic terms. For example, the TC for the EU ETS in Germany are less than 1% of the subsidies for renewable electricity in Germany which amounted to € 12 billion in 2011 (Frontier Economics, 2012). Furthermore the comparison shows that TC do not strongly differ across instruments.

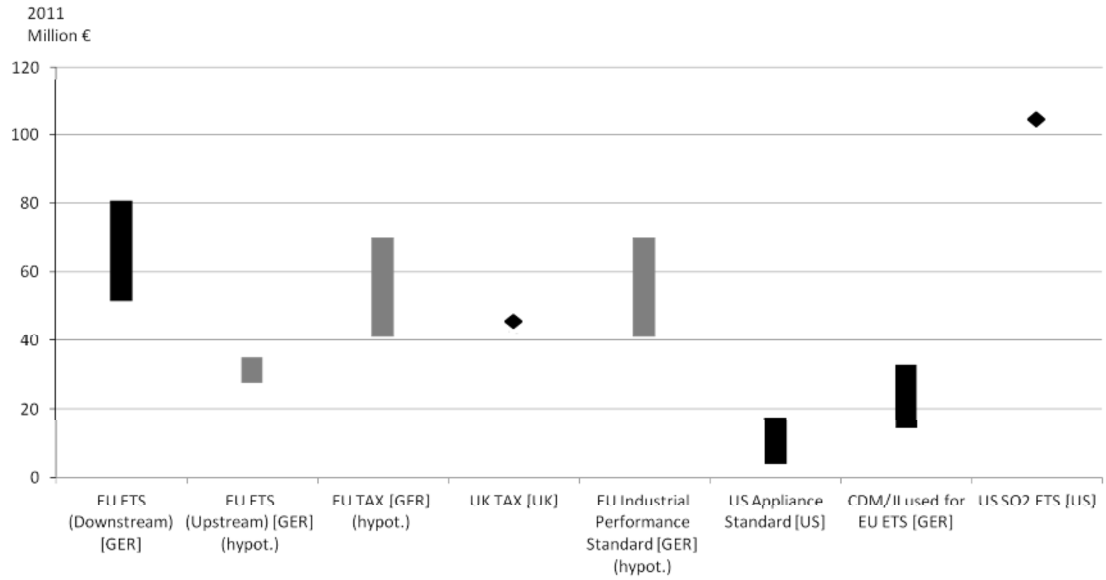


Figure 4: Aggregated total TC for policy instruments. Where data was available from multiple sources, ranges are indicated. The bars in grey indicate that the calculations are based on hypothetical schemes. Bars in black indicate that the data originates from empirical studies on schemes that are in operation.
Data: See Section 4.

Scheme	Country	Pollutant	Total TC in 2011 Million €	Regulation Method
EU ETS (Downstream)	Germany	CO_2	51 - 81	Downstream; Point sources
EU ETS (Upstream) (hypot.)	Germany	CO_2	28 - 35	Upstream; Point sources
EU TAX (hypot.)	Germany	CO_2	41 - 71	Downstream; Point sources
UK TAX	UK	CO_2	46	Upstream; Point sources
EU Industrial Performance Standard (hypot.)	Germany	CO_2	41 - 71	Downstream; Point sources
US Appliance Standard	US	CO_2	4 - 17	Downstream; Non-point sources
CDM/JI used for EU ETS*	CDM/JI Host Countries / Germany	CO_2	15 – 33*	Certification; Point sources
US SO_2 ETS	US	SO_2	105	Downstream / CEMS; Point sources

Table 6: Aggregated TC for all considered instruments. If data was available from multiple sources, ranges are indicated. The column "regulation method" indicates how MRV is carried out.
**) These are the TC of offset credits used for compliance for the EU ETS in Germany.*
Data: See Section 4.

Compared to the potential orders of magnitude of macroeconomic inefficiencies from non-optimal policy instruments, especially in the longer run the magnitude of TC differences as indicated in Table 6 appear to be very small or negligible. For example, using a numerical dynamic general equilibrium model Kalkuhl et al. (2011) calculate that exclusively relying on a feed-in tariff instead of an optimal carbon tax can increase the costs of climate policy by 0.8% of total consumption in terms of balanced-growth equivalents (see also Fischer & Newell, 2008).

Figure 5 offers a framework to discuss TC in relation to different abatement levels for discerning whether TC are relevant for the total cost effectiveness ranking of climate policy instruments (for a similar line of argument, see Betz et al. 2010). It depicts two scenarios for the regulation of GHG in one country:[49] For two abatement levels, low and high, the sums of total system-level abatement costs and total TC for a technology standard and a market based instrument are compared.

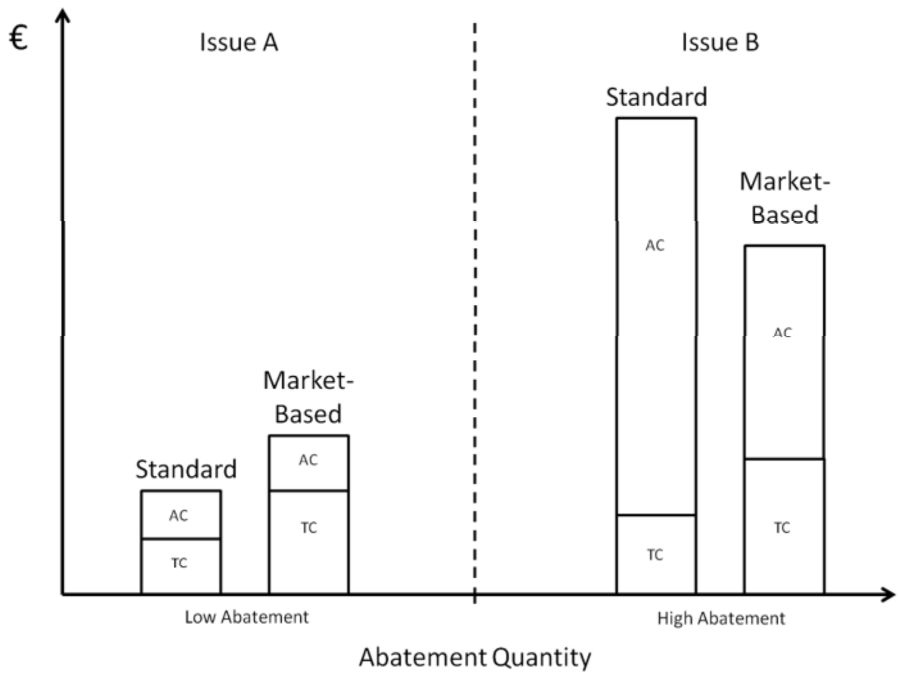


Figure 5: Optimal policy instrument choice for different issues A (low level of abatement) and B (high level of abatement) when both the cost effectiveness (defined as total abatement costs (AC)) and TC properties of instruments are taken into account.

** Total abatement costs are assumed to be lower for market based instruments due to the equalization of MAC.*

Standards tend to feature lower TC than market-based instruments, but are inferior regarding MAC harmonization across regulated entities. The magnitude of this adverse cost effectiveness property will increase with the overall quantity of abatement, thus increasing relative total abatement costs of the instrument with rising program stringency. Thus, standards may be superior instruments at low levels of abatement (Issue A), but for ambitious abatement programs (Issue B) market-based instruments can be expected to perform better due to lower total abatement costs. It can be argued that climate policy in large countries in general is a type B policy issue because ambitious emission reductions will be required in the long-term. Given the small TC differences across instruments observed in this study, it can be concluded that it is the relative abatement costs of instruments, or any other societally relevant evaluation criteria, that are of practical relevance. However several limitations to this finding have to be considered:

First, as discussed in section 3 and 4, TC scale with respect to different country- and design-specific factors. For example, TC may be substantially higher in other jurisdictions with different institutional structures, governance capacities and market infrastructures. This may be true for other OECD countries but especially for emerging economies and developing countries. Research on schemes such as CDM and Reducing Emissions from Deforestation and Degradation (REDD) have shown that TC can be large and are therefore likely to distort optimal abatement (Antinori and Sathaye, 2007; Bottcher et al., 2009; Fichtner and Rentz, 2003; Michaelowa and Jotzo, 2005). Also, in very small jurisdictions with few abatement such as cities, options differences in TC may outweigh the advantage of MAC harmonization, indicating an Issue A situation. To better inform the choice and set-up of novel climate policy instruments with respect to relative TC, more empirical TC data from applications in different contexts would be required to be able to robustly estimate the magnitude of these scaling effects.

Second, TC in this study all relate to average *ex post* costs over a longer time period. Short term changes of regulation such as new MRV requirements may affect investment behavior and might imply financial hardship for certain firms in the short term which can create costs at the system-level. Furthermore other *ex ante* costs such as setting up administrative processes and communicating and coordinating these with regulated entities, but especially the societal costs of political bargaining may be important factors limiting development of climate change policies. These *ex ante* costs should be subject to further research.

Third, it has been pointed out that economies of scale resulting in differing MRV costs for firms may lead to a competitive disadvantage for small emitters, but the effect seems to be small in case of the EU ETS (see Figure 1, and Endnote 8 in Section 4.1).

Despite these caveats, current evidence indicates that for large jurisdictions in OECD countries such as Germany and the US *ex post* transaction cost considerations play a minor role in climate policy instrument choice. In these contexts, the focus of policy-choice considerations should rather be on cost effectiveness properties of instruments in incentivizing abatement, and other societally relevant objectives (e.g. equity or political economy considerations).

This conjecture differs from the experience in other areas of environmental policy such as pollution trading in the Minnesota river basin or for the US lead phase-out trading program, where very high *ex post* transaction costs have been observed and were thus considered as important factors in the overall assessment of optimal environmental policy choice.[50]

Clearly, some caution is warranted given that the underlying data is somewhat meager and patchy, except for the EU ETS. The relevant question here is whether better data would alter the general findings of this study. As the discussion has attempted to demonstrate, it seems unlikely that for large countries with similar institutional settings as the US and Germany new data will change the picture dramatically and make *ex post* transaction costs a substantial factor for climate policy instrument choice in a macroeconomic perspective.

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Appendix 1

Instrument	Study	used	not used	Short data overview
EU ETS	Bergmann et al. 2005		x	€ 53.000 to develop a auctioning procedures; Auctioning costs at Chicago Board of Trade (CBOT) for SO_2 auction cost is independent from auctioned amount around $10.000.
	BlueNext 2012		x	Administrative and technical fees: Admission fees €7.500; trading license (Global Vision) €/pa 4.000; annual fees €/pa 1.000 – 3.500 Trading fees: Transactions from market orders €/t 0,01 – 0,02; transactions from OTC orders €/t 0,008 – 0,015; delivery €/trade Auction: Tick 0,01 €/t; Spot EUA: Price Tick 0,01 €/t; Futures EUA: Price Tick 0,01 €/t
	Brockmann et al. 2012	x		16 firms in Germany applied for the opt-out close; Application for free certificates caused TC of € 25.000 (median) per firm.
	Convery and Redmond 2007	x		Broker cost € 0,05 - 0,1 t/CO_2
	DEHSt 2008	x		11 Million annual running costs for the entire DEHSt
	DEHSt 2011a	x		Registry costs: Costs first trading period (2005-2007): license agreement: € 143.520; source code: € 35.880; technical assistance: € 41.860; software maintenance agreement: € 304.980; hosting register: € 704.460; total; € 1.230.700; Costs second trading period (2008-2010): Nat. adjustment CR-software: € 813.900; additional contract go-live ITL: € 34.558; Generic web-service: € 64.736; Generic web-service; Software maintenance agreement 2008 – 2010: € 642.265; Hosting register 2008 – 2010: €1.012.102; Total: € 2.567.561;
	Destatis 2011	x		Determination of emissions, reporting and verification; € 38.566.000 p.a.; Composing monitoring plan; € 768.000 p.a.; Declaration operator change, changes to operating procedures of installation, decommissioning: €40.000 p.a.; application for certificates, verification of application: €2.423.000 p.a.; Adaptation of monitoring concept: €224.000 p.a.;
	Dr. Röver & Partner KG 2006	x		IT Costs (2004 - 2007): € 10.246.000 Total DEHSt Costs (2004 - 2007): € 44.400.000
	EEX 2012	x		Spot market: Secondary trading: € 0,006; primary auction: € 0,003 Future market: Secondary trading: € 0,0028; primary auction: € 0,0028
	Ellerman et al. 2010	x		Total EU early Implementation Costs (2004 - 2007) € 360.000.000 Total EU MRV Costs (2005 - 2007): € 462.000.000
	Frasch 2007	x		Transaction costs per t / CO_2: Major utility: € 0,27; Public service utility: € 0,79; Lime works: € 0,14
	Hacker 2012	x		Trading transaction costs of € 0,1 t / CO_2
	Heindl 2012	x		Overall annual transaction costs for all German firms regulated under the EU ETS are estimated at 8.7 EUR Total TC mean: €12.223; MRV TC mean: €8.433; trading TC mean: €4.659; abatement information costs mean: €4.193
	Jaraite et al. 2010	x		MRV Costs per t/CO_2: Avg.: € 0,04; large: € 0,02; medium: € 0,56; small:

				€ 1,51
	Loeschel et al. 2010	x		Small emitter (< 25.000 t/CO_2) median TC: €1,79 per t/CO2 Large emitter (≥ 25.000 t/CO_2) median TC: €0,36 per t/CO_2
	Loeschel et al. 2011	x		MRV costs (Median): < 25.000 tCO_2 p.a.: € 0,47; > 25.000 tCO_2 p.a.: € 0,06; (n=87) Trading TC (Median): < 25.000 tCO_2 p.a.: € 0,1; > 25.000 tCO_2 p.a.: € 0,02; (n=46) Abatement information costs (Median): < 25.000 tCO_2 p.a.: € 0,14; > 25.000 tCO_2 p.a.: € 0,02; (n=23) Total TC (Median): < 25.000 tCO_2 p.a.: € 0,64; > 25.000 tCO_2 p.a.: € 0,08; Total TC (avg.): € 0,029
	UBA 2009	x		First trading period: 806 appeals against 1600 allocation decisions; 602 appeals against cost decisions; Second trading period: 373 appeals against allocation decisions
	VBW 2012	x		Costs for legal prosecution per installation: Germany: €5.354, Spain: €963, France: €62 Maintenance of monitoring schemes per installation: Germany: €8.676, Spain: €11.057, France: €10.120 Emission monitoring per installation: Germany: €8.362, Spain: €8.033, France: €6.890 Emissions registry per installation: Germany: €2.340, Spain: €5.7773, France: €3.619 Total running costs per installation: Germany: €24.733, Spain: €25.830, France: €20.690 Total firms running TC p.a.: € 56,9 million (Germany); € 34,1 million (Spain); € 26 million (France)
	Weber 2012	x		see appendix 2
UK ETS	National Audit Office 2004 & Murtishaw 2004		x	Brokers have charged a rather low fee of 2% pe r transaction (National Audit Office, 2004). The UK ETS scheme has reported prices of emissions allowances that varied from a peak of $22.5 to $4.5 per tonne of carbon dioxide or $82.5 to $16.5 per tCeq. A 2% fee would amount to a charge of $ 0,33 – 1,65 per t Ceq. for brokerage services alone.
Australian Fuel Efficiency Program	Ofei-Mensah & Bennett 2013	x		Transaction costs for the Fuel Label Program ranged between $6.7 and $9.8 million. For the Fuel Efficiency Program, public agency transaction costs are $2.5-4.0 million or 43-44% of the total costs, while private agency transaction costs of $3.2-5.5 million represent 56-57% of the total transaction costs of the program.
Australian Fuel Label Program	Ofei-Mensah & Bennett 2013	x		Fuel Efficiency Program, the present values of transaction costs results ranged from $5.7 to $9.5 million. Public agency transaction costs for the Fuel Label Program are $4.2-6.0 million, under the four scenarios, representing 61-64% of the total transaction costs of the program. Private agency transaction costs of $2.4-3.8 million constitute 36-39% of the total costs.
Australian Tradable Permit and Fee System	Ofei-Mensah & Bennett 2013	x		For the Tradable Permit and Fee System transaction costs ranged between $37.1 and $78.7 million . Public agency transaction costs for the Tradable Permit and Fee System are $8.8-17.9 million, which translate to 23-24% of the total costs. Private agency transaction costs are $28.3-60.8 million or 76-77% of the total transaction costs of the program.
US SO_2 Trading	Doucet and Strauss 1994	x		In US SO_2, transaction costs of trading were $10 per allowance in the early stages

	EPA 2001	x		Initial capital cost for CEMs for coal-fired electric power plants: \$700,000; Annual operating costs for CEMs: \$50,000; On an annualized basis that spreads the capital costs over a capital recovery period, the cost of operating a CEM is approximately \$125,000 each year. This amount is equivalent to about \$0.16 per kilowatt of installed capacity. During the first 5 years EPA spent \$44 million for implementation of the Acid Rain Program; EPA allocated an additional \$18.9 million to state and local governments to implement the program.
	Joskow and Schmalensee 1998	x		Commissions per allowance: \$20 in August 1994; \$3.50 in mid-1994 (avg.), \$2.00 in late 1995 (avg.), \$1.50 in September 1996 (avg.)
	LECG 2003	x		SO_2 Allowance Trading: Administrative costs to participants: 0,5% to 1.5% transaction cost (up to 10% on small trades); + 1% "slippage"; + planning and research; Administrative costs to governments: Issues allowances, records trades, audits emissions records. Estimate \$38 million 1990-1995.
	Margaree Consultants 1998	x		SO_2 trading: Brokerage fee of \$1.00 per allowance. The price of an allowance has fluctuated between \$100 and \$200, so the transaction cost is less than 1%
	McLean et al. 1997	x		US Title IV SO_2 Administration Costs: Program Development and Support: \$19.9 million; Data System Development: \$9.9 million; Program Operations: \$10.8 million; Other: \$3.5 million; The SO_2 allowance trading annual administrative costs in 2003: \$460.722; Emissions reporting costs to EPA in 2003: \$1.034.035
	Sovacool et al. 2011		x	Transaction costs accounted for 7 to 25 percent of the value of traded credits
US RECLAIM	EPA 2001	x		Sources installed continuous emission monitors (CEMS), which cost \$100,000 to \$150,000 each, on every boiler emitting 10 tons annually or more. The District projected that the one-time costs of installing monitoring equipment would be approximately \$13 million, with negligible annual operating costs. The District projected that annual savings in compliance costs relative to traditional forms of regulation would be an average of \$58 million annually for each of the next 10 years.
	Foster and Hahn 1995	x		RECLAIM: Brokerage fees are levied in proportion to the value of the trade at a rate that can vary between 4 and 25 %. Administrative fees to the regulatory body can amount to \$2,900 per trade. Additional administrative fees are levied for certifying ERCs (\$1,700), banking them (\$900), and reissuing them in smaller units (\$900). In addition, firms can incur between \$7,500 and \$15,000 for the preparation of the substantial supporting documentation required to certify an emissions reduction. A typical value for the joint financial transaction costs of both partners required to bring about an exchange would be \$25,000, with \$10,000 as a lower bound.
	Gangadharan et al. 2000		x	The presence of transaction costs reduces the probability of trading by about 32%
	LECG 2003	x		Administrative costs to participants: 3% to 10% transaction cost + \$200 /contract cost + planning and research
	Margaree Consultants 1998	x		RECALIM RTC: 3.5% or \$35 per ton transacted plus a flat \$150 per order placed. The total fee includes a \$50 fee charged by SCAQMD to register each trade. The SCAQMD collects the registration fee each time, so it receives at least \$100 for each trade through a broker. The total cost to complete a trade is \$300 plus \$70 per ton. An average trade consists of a little over 1,000 tons and the weighted average price has been just

				over $1,000 per ton. Thus the transaction cost averages about 7% of the price.
NOx Budget Program	LECG 2003	x		NOX Budget: 500 sources, start-up $ 5.5 million US, operations $2.4 million/yr NOX trading program: Transaction costs in the newer NOX trading program are in the range of 1% to 5% of the value of allowances being transacted (US$600 presently)
Point of Regulation	Hargrave 2000		x	Estimations on numbers of regulated entities under upstream regulation
	Jaraite et al. 2010	x		See EU ETS
	Loeschel et al. 2011	x		See EU ETS
TAX	DEA 2000		x	Administration costs €4 million p.a. (1996-2000); 6,7 million annually administrative costs the first years after introduction of the Green Tax Package. Since then, the costs have been lower.
	DEHSt 2011	x		see EU ETS
	Destatis 2012	x		see EU ETS
	Dr. Röver & Partner KG 2006	x		see EU ETS
	Loeschel et al. 2011	x		see EU ETS
	OECD 2001		x	The UK Landfill tax raises £305 million and administrative costs are around £2 million per year. Denmark CO_2 tax is estimated that additional administrative costs are around 1-2% of the tax revenue levied on business.
	Sandford et al. 1989	x		Oil industry: 74 duty-paying companies, of whom 12 contributed 95 %t of the revenue. Compliance costs of oil excises UK: £17.600.000; Administrative costs of oil excises UK: £7.500.000; Total TC of oil excises UK: £25.100.000; Total revenue of oil excises UK: £75.000.000.000; Share of compliance costs of oil excises UK 0,23%; Share of administrative costs of oil 0,10%; Total TC of excises in the UK 0,34%; Total operating costs(% of total revenue)of different types of taxes: Income tax: 4,93%; VAT: 4,72%; Corporation tax: 2,74% Petroleum revenue tax: 0,56%; Excise duties (hydrocarbon oils; tobacco, alcoholic drinks): 0,45%; Minor taxes (stamp duty, cars, betting and gambling): 2,33%
	Smulders Vollebergh 2001		x	Energy Excises Applying to Households, Industry, and the Electricity Sector in the Nordic Countries, 1990; for tables please see publication.
	Vaillancourt et al. 2008		x	Compliance with personal income taxes ranged from between $2.9 billion and $5.5 billion
	VBW 2013	x		see EU ETS
FIT	Breitschopf et al. 2009		x	Top down calculation based on Destatis data: € 28 million p.a.; Bottom up calculation: € 22 – 40 million p.a.
	del Rio & Gual 2007		x	The trans action costs of the Spanish FIT can be deemed low. The support system was very simple and a large administrative structure was not required because governmental planning and control efforts were minimised.
	Destatis 2006		x	fully displayed in Breitschopf et al., 2009

	Langniss 2003		x	German EEG: Total annual billing costs of RE generators: € 8 million/year. Authors assume similar costs (€ 12 million/year) for grid operators. Total annual direct transaction costs around € 20 million/year in 2001. Equivalent to 1.3% of the total EEG remuneration.
Quota	Battjes et al. 2000 in Oikonomou et al. 2008		x	Dutch tradable green certificates scheme: Ex-Ante Study: Projections for 2010: Transaction costs: 0,11 € ct/kWh Projections for 2020: 0,08 € ct/kWh
	Bergek & Jacobsson 2010		x	Swedish tradable green certificates (TGC): Approximately 1.9 billion SEK were transferred to electricity suppliers to cover their transaction costs (2003-2008). Transaction costs decreased and reached 185 million SEK in 2008.
	Kaberger et al. 2004		x	Swedish tradable green certificates (TGC): 18% of the average consumer cost (for a green certificate) covers the costs and profits of the electricity suppliers for managing the purchasing and trading of the certificates. These transactions costs represent only a part of the total transaction costs of the system. A possibly even greater cost occurs among the many small companies producing electricity from the wind- hydro and biomass fuelled powerplants.
	Langniss 2003		x	Texas RPS: Transaction costs of 2,9 % of the traded value. Total transaction costs of USD 1.7 million/year.
	Smith et al. 2000		x	Massachusetts Renewable Portfolio Standard: Restricted Unbundling: Start-up costs: $200,000 Fixed ongoing costs: $3,000,000/year ariable ongoing costs: $1.5/MWh in 2003 decreasing to $0.5/MWh by 2012 Massachusetts RECs: Start-up costs: $500,000 Fixed ongoing costs: $1,600,000/year Variable ongoing costs: $0.5/MWh in 2003 decreasing to $0.2/MWh by 2012 Full Certificates: Start-up costs: $175,000 Fixed ongoing costs: $840,000/year Variable ongoing costs: $0.5/MWh in 2003 decreasing to $0.2/MWh by 2012
	Sovacool 2011		x	U.S. Renewable Energy Credits (RECs): Transaction costs have accounted for about 10 percent of the value of credits traded
	van der Linden et al. 2005		x	U.S. Renewable Energy Credits (RECs): The largest and most expensive tracking system in the US covers the New England region (tracking all generation, not just renewable generation), and required $200.000 of up-front capital and recovery of annual operating costs of $ 900.000 to $ 2.400.000 over five years (equating to a transaction fee that is as high as $0,0176/MWh). The other tracking systems in operation in the US required substantially less funding.
Quota / FIT	Skytte et al. 2003		x	Case study responses from 12 different EU Member States: Transaction costs in the planning phase range from approx. 2% - 25% of total investment costs (see publication for more figures). In accordance with former studies transaction costs can constitute 10% of the traded value (at the beginning of the trading program) down to 2% of the traded value (in m ature and highly liquid markets).

Standards	Ericsson 2006		x	The administrative costs related to the Green Tax Package for public authorities were prior to the introduction estimated to 6,7 million €/yr (50 million DKK/yr), of which the VA scheme accounted for 4,1 million €/yr (30 million DKK). However, in total 4,1 million €/yr were set were set aside in the government budget for adadministrating the Green Tax Package for 1996-2000 (Office of the Auditor General, 1998). The actual administrative costs related to the VA scheme have decreased considerably over the years, from the 4 million € (30 million DKK) for 1996 to about 0,4 million € (3 million DKK) for 2005. DEA, the CCTA and the Energy Complaints Board. The administrative costs have continuously been reduced over the years, in particular those for the DEA. In 1996 the DEA's administrative costs for the VA scheme amounted to roughly €2,7 million and occupied up to 31 employees (Office of the Auditor General, 1998). The DEA's administrative costs for 2006 are timated to roughly €270.000, occupying 3 employees.
	Gillingham et al. 2004	x		EPA spends around $50 million annually on administering all Energy Star programs
	Hein & Blok 1995		x	Transaction costs mainly consist of information costs (2-6% of the investment); decision making (1-2%); monitoring (<1%); Transaction costs of energy efficiency improvement measures are estimated to be between 3% and 8% of the investment.
	Levin et al. 1995	x		Cumulative expenditures by the federal government for the appliance-efficiency program total about $50 million from 1979 to 1993
	McMahon et al. 2000	x		Governmental administrative costs for the entire appliance, lighting, and equipment standards program represent an additional $5-10 million annually for developing test procedures, analyzing, and implementing regulations.
	Meyers et al. 2003	x		The amount of taxpayer funds used to support DOE 's residential appliance standards program over the past 20 years is in the range of US$200 –US$250 million.
	Mundaca 2007a		x	The scale of TC was estimated to be around 10% and 30% of total investments costs for the lighting and insulation segments, respectively.
Kyoto Mech-anisms	Antinori and Sathaye 2007	x		Transaction costs for projects that used the LBNL framework (Data Set 1) range from $0,03 per tonne of carbon dioxide for large projects to $4,05 per tonne of carbon dioxide for smaller ones, with a weighted average of $0,36 per tonne of carbon dioxide for all projects.
	Chadwick 2006		x	Case study of Ghana LPG CDM Project: Up-front costs: Project documents $75.000; Approvals and registration $55.000; Methodology development $200.000; Total up-front costs $330,000; (Costs without methodology) $130.000; Annual recurring costs and income: Monitoring and verification ($8.000) per year; 2700 tCO_2e CERs $54.000 per year at $20/$tCO_2e$; Net cash flow at $20/tC02e $40.000 per year
	Fichtner and Rentz 2003		x	The findings show that transaction costs of AIJ projects range between 7% and more than 100% of production costs with 80% of projects lying between 14% and 89%.
	Krey 2004	x		Total transaction costs range from 0.07 to 0.47 $US/t CO_2.
	Michaelowa and Jotzo 2005	x		Transaction costs of CDM projects under the prototype carbon fund (PCF) range from €0,19-€0,71 per t CO_2 For estimates of transaction costs for different project components please see table in paper

	Michaelowa et al. 2003		x	micro CDM projects transaction costs can reach several hundred Euro per tCO_2 equivalent
	Mundaca & Rodhe 2004		x	Estimated transaction costs per CDM wind-energy project: CDM phase Estimated TC (US$) Negotiation: $55.000 Approval: $20.000 Baseline: $35.000 Validation: $25.000 Contingency Ex-post implementation: $15.000 Monitoring, verification and certification: $25.000
	Sovacool 2011		x	Traders and brokers procuring credits and matching buyers and sellers get a commission of 3–8 percent of the value of the credit (with an industry average of 5 percent).
	Williams 2009		x	Estimated Transaction Costs of CDM Projects: Preparation of PIN/PDD: 100.000–150.000Rs ($2.000-3.200) + 5% success fee (of the CER revenue) Validation: 300.000–400.000Rs ($6.200- 8.300) Verification: 200.000–300.000Rs ($4.200-6.300)(yearly) Certification cost: 2% deduction of CER + $0.10 per CER – for adaptation fund and administration expenses of the CDM executive board Identification of buyers and sale of CERs: 1–2% of CER volume CER Registration Costs; Average tons of CO_2 Reduction per Year: 15.000 or less : $5 15.000 to 50.000: $10 50.001 to 100.000: $15 100.001 to 200.000: $20 >200.000: $30
Canda Offset Scheme	Marbek Resource Consultants 2004		x	Average transaction costs per ton vary over a wide range with design choices and project types, from as much as $19/ton for independent agriculture projects in a scenario with a high degree of precision and complexity, to as little as $0.05/ton for landfill gas projects in a scenario with more simplified approaches to quantification, verification and other elements
Non Climate	Falconer and Whitby 2000		x	Total annual administrative cost varies greatly. In Belgium, an estimated 20.000 ECU were spent annually on the organic aid scheme, compared to annual costs of 600.000 – 1.500.000 ECU on the French arable conversion scheme, and 900.000 – 1.400.000 ECU for the livestock extensification scheme.
	Fang et al. 2005		x	Estimates of transaction costs showed that the total costs of the trading projects were increased by at least 35% after transaction costs were taken into account.
	Hahn and Hester 1989		x	A second factor influencing program performance is how administrative requirements for trading affect transaction costs. When these requirements are especially complex or impose major informational burdens on firms, transaction costs increase and trading decreases.
	Kerr and Maré 1998		x	Loss of cost effectiveness through transaction costs is around 10 to 20%
	McCann and Easter 1999		x	The tax on phosphate fertilizers had the lowest transaction costs ($0.94 million), followed by educational programs on best management practices ($3.11 million), the requirement for conservation tillage on all cropped land ($7.85 million), and expansion of a permanent

				conservation easement program ($9.37 million).
	McCann and Easter 2000		x	The Transaction costs for the National Resource Conservation Service(NRCS) are in the range of 38% of overall conservation costs
	Mettepenningen et al. 2009		x	Private transaction costs incurred by participants in European agri-environmental schemes are around 25% of the compensation payment.
	Pannell et al. 2013		x	Based on McCann et al.'s (2005) typology of transaction costs associated with public policies, transaction costs amounted to 68% of the Australian Government's funds.
	Sovacool et al. 2011		x	Individual Transferrable Quotas (ITQs) Fisheries New Zealand: Transaction costs account for 1 to 3 percent of the value of all trade

Table 7: evaluated studies

Appendix 2

Calculation of Trading Transaction Costs:

Table 8:

Total transaction volume	9.700.000.000
EUA total Volume	7.900.000.000
EUA Futures	7.000.000.000
EUA Options	790.000.000
CER & ERU total Volume	1.800.000.000
CER & ERU Futures	1.656.000.000

Table 8: Trading volume of different certificates. Data: Kossoy & Guigon (2012).

Table 9:

	Share	Volume	TC per t/CO_2	Total TC
Exchange-based screen trades	49%	4.628.540.000	0,004	€ 18.514.160
OTC (without intermediaries)	34%	3.211.640.000	0,002	€ 6.423.280
OTC (with intermediaries)	5%	472.300.000	0,032	€ 15.113.600
Bilateral trades (without intermediaries)	11%	1.039.060.000	0,001	€ 1.039.060
Bilateral trades (with intermediaries)	1%	94.460.000	0,03	€ 2.833.800
Total transaction volume of futues and options for EURs, CERs and ERUs	100%	9.446.000.000		**€ 43.923.900**

Table 9: Trading TC for futures trades. Data: Kossoy & Guigon (2012); Weber (2012); own calculation.

Table 10:

	Share	Volume	TC per t/CO_2	Total TC
Primery auction	37%	92.900.000	0,003	€ 278.700
Exchange-based screen trades	1%	2.540.000	0,004	€ 10.160
OTC (without intermediaries)	32%	80.836.531	0,002	€ 161.673
OTC (with intermediaries)	16%	40.418.265	0,032	€ 1.293.384
Bilateral trades (without intermediaries)	10%	24.872.779	0,001	€ 24.873
Bilateral trades (with intermediaries)	5%	12.436.389	0,03	€ 373.092
Total Spot and Options for EURs, CERs and ERUs	100%	254.003.964		**€ 2.141.882**

Table 10: Trading TC for spot trades. Data: Kossoy & Guigon (2012); Weber (2012); own calculation.

Table 11:

Total trading transaction costs EU	**€ 46.065.782**
Total trading transaction costs Germany	**€ 10.595.130**

Table 11: Total trading TC Germany and the EU. Data: Kossoy & Guigon (2012); Weber (2012); own calculation.

Appendix 3

Table 12:

	Total TC Low	Total TC High
MRV Firms	48.826.314 €	183.978.104 €
Application for allocation	*10.839.425 €*	*11.757.063 €*
Legal Expenses	1.435.817 €	50.049.441 €
Abatement Information	3.996.012 €	3.996.012 €
Registry Costs	3.324.792 €	3.324.792 €
Compliance Agency	68.096.474 €	68.096.474 €
Auctioning	623.580 €	623.580 €
Trading costs	46.618.571 €	46.618.571 €
Total Trading	**183.760.985 €**	**368.444.037 €**

Table 12: Extrapolation of Germany’s transaction costs to the EU

Appendix 4

Table 13:

Type of Offset	2008	2009	2010	2011	2012	2008-2012	2008-2012 annual average
CER	23.721.741	25.999.308	33.374.387	41.123.292	45.117.050	169.335.778	33.867.156
ERU	0	670.990	4.194.506	33.231.950	94.760.110	132.857.556	26.571.511
Total	23.721.741	26.670.298	37.568.893	74.355.242	139.877.160	302.193.334	60.438.667

Table 13: CER & and ERU credits used in the second trading period (2008-2012) in Germany

Table 14:

Sources	Average TC per t CO2 abated in 2011 €	Total TC for offset credits in 2011 €
Krey 2004	0,24*	14.505.265
Antinori and Sathaye 2007	0,29	17.622.882
Michaelowa and Jotzo 2005	0,54*	32.410.235

Table 14: Calculation of transaction costs from CER & and ERU credits is based on CDM TC.

**)These two studies only indicate ranges but no weighted average. For this calculation we have used the average of the ranges.*

Endnotes

[1] Throughout this paper all € values have been adjusted for inflation to the year 2011 using Fxtop.com inflation calculator. All original $ values have been adjusted for inflation to the year 2011 using www.usinflationcalculator.com, and converted into € with the average exchange rate of January 2011 of € / US$ 1,3398 (www.oanda.com).
[2] Degla (2012) offers a useful summary of the various transaction costs definitions applied in the literature.
[3] McCann et al. (2005) provide the following definition of these cost components: (i) research, information gathering, and analysis associated with defining the problem; (ii) enactment of enabling legislation, including lobbying and public participation costs, or the costs of changing laws through the courts or modifying existing regulations; (iii) design and implementation of the policy, which may include costs of regulatory delay; (iv) support and administration of the ongoing program; (v) contracting costs, which may include additional information costs, bargaining costs, and decision costs, which are relevant when a market has been set up for a pollutant, or natural resource; (vi) monitoring/detection, which may include both the monitoring of the environmental outcome, or the level of compliance with the regulation, tax/subsidy scheme, or private contract, as well as the development of monitoring technologies; and (viii) prosecution/inducement/conflict resolution costs incurred if lack of compliance is found.
[4] The only available *ex ante* cost data are on firms' early implementation costs (or one time fixed costs) for the EU ETS in Ireland (Jaraite et al., 2010).
[5] Early implementation costs in Ireland are on average 4 €ct/t CO_2 emitted (3 €ct for large emitters and 51 €ct for small ones). These *ex ante* costs are equivalent to the firms' annual MRV costs (4 €ct/t CO_2). In the long run early implementation costs therefore play a minor role (Jaraite et al., 2010).
[6] There are potential TC economies of scale in the public sector due to fixed costs such as emission registry software and updating regulations for MRV procedures. For example, the public sector transaction costs per regulated entity in Lichtenstein are likely to be higher than in Germany. This effect is not taken into account in our calculation.
[7] Comparing TC for gathering information on cost effective abatement across policy instruments is not straightforward because not all instruments face these costs in the same way. For ETS, trading and performance standards they must be borne by firms and are accounted as a share of the abatement cost from the firms' perspective. It could therefore also be argued that they are part of the abatement costs since no abatement could occur without prior finding abatement options. For technology standards (except performance standards) these costs are borne by the regulator who decides on the abatement action. In a situation where minimal scheme-wide marginal abatement costs (MAC) shall be achieved the regulator would ideally set a customized standard for each firm according to its MAC. If the regulator would be equivalently capable of identifying abatement options, the costs would be identical to the firms search costs. In this case they would clearly be categorized as public sector transaction costs. However in practice the regulator does not engage in such actions because he lacks the company-specific information. Instead the regulator sets a uniform standard all firms must fulfill. This causes inefficiencies which lead to higher MAC. It can be expected that the extra search costs on firm level are in sum much lower than the gain of lower MAC, see e.g. Tietenberg (1985).
[8] Consider the following simple calculation: Using the figures provided by Heindl (2012), a firm that emits 1 million tons of CO_2 each year faces average (total) transaction costs of roughly € 0,03 per ton of CO_2 (or 30.000€ per year). A firm that emits 10.000 tCO_2 each year faces average (total) transaction costs of about 0,76€ per ton of CO_2 (or 7.600€ per year). The negligible absolute orders of magnitudes involved in these calculations strongly suggest the practical irrelevance of this effect for changes in competitiveness of companies.
[9] This calculation is based on firms' MRV cost data provided by Loeschel et al. (2011). The high-cost estimates (used in Figure 2) could not be used since no differentiation between small and large firms is made in the studies by VBW (2013) and Destatis (2012).
[10] The second figure provided by Loeschel et al. (2011) does not include firms' costs for application for free allocation and legal expenses.
[11] Firms must apply for these certificates and justify reasons for their eligibility. In the third trading period the amount of freely distributed certificates to industrial emitters will decrease from a maximum of 80% in the beginning to 30% by 2020. Only firms that can prove to be affected by carbon leakage can continue to receive up to 100% of their certificates for free. Apart from a few transitional exemptions, the power sector has to obtain all emission allowances in auctions (IETA, 2012; UBA, 2013). It is estimated that over the entire third trading period 50% of the allocated amount (cap) will be auctioned (DEHSt, 2012b).

[12] This extrapolation may be an underestimation of costs since large emitters may have higher application costs that are not reflected in the median. However the data from Destatis (2011) indicate total annual application costs of € 2,4 million, which suggests that the figures are robust.

[13] In the first trading period (2005-2007) 806 appeals have been lodged against 1600 allocation decisions and 604 appeals against cost decisions in Germany (UBA, 2009). In the second trading period there were 373 appeals against allocation decisions (UBA, 2009).

[14] It has been stated that the enthusiasm for legal dispute is a special German phenomenon (Seidel, 2011; VBW, 2013). Litigation costs in Spain and France were substantially lower with only € 960 and € 60 per installation or € 0,9 million and € 60 thousand total p.a. costs, respectively (VBW, 2013). Unfortunately no data are available for other instruments and other countries. It is quite possible that the reported costs would be eclipsed by litigation costs in the US, where traditionally courts are more involved in the interpretation of government regulations.

[15] This includes trades by firms who need to balance their emission accounts for compliance, sellers of offset certificates for CDM and JI, and market intermediaries such as banks and trading houses who have speculative motives, but also help to increase the market liquidity. These market intermediaries are responsible for the largest amount of the trading volume and therefore also bear the largest share of trading transaction costs (Weber, 2012). Different certificates are eligible for EU ETS compliance (i.e. European Union Allowances (EUAs), Certified Emission Reduction units (CERs) and Emission Reduction Units (ERUs)) and can be traded on exchanges (49% of all trading activity), over the counter (39%) or in bilateral trade (12%).

[16] In Germany regional authorities are involved in the MRV process in addition to the federal authority. Based on personal communication by the German national ETS agency (DEHSt) (Seidel, 2011) it is conservatively assumed that two full time staff are responsible for each of the 16 German states (Länder). This amounts to 32 full time staff per year. With an annual salary of € 60.000 and 70% additional other expenditures (assumed same share as in the DEHSt), this amounts to € 3,3 million per year. Furthermore 57 personnel at DG Climate Action of the EU Commission are responsible for administering the EU ETS (CEC, 2012). Assuming € 90.000 average annual salary plus 70% other expenditures (assumed same share as in the DEHSt), the total transaction costs at DG Climate Action amount to a € 8,6 million. To calculate the German share in DG Climate Action costs we assume 23% of costs in proportion to German emissions in the EU ETS, which amounts to roughly €2 million.

[17] In the second trading period (2008-2012) 1.058 million offset credits have been used for compliance in the EU ETS.

[18] Krey (2005) found an ex post transaction cost range of 6-42 €ct/tCO2 abated (=per CDM credit) for CDM projects in India. Antinori and Sathaye (2007) report average TC of 29 €ct/tCO2, and Michaelowa and Jotzo (2005) identify a range of 22-85 €ct/tCO2 per CDM credit. In all case studies, small projects tend to have higher TC per t/CO2 than large projects. These data were used to calculate a range of possible TC resulting from the use of CDM for Germany which amounts to €15 - 32 million p.a. For the higher bound 54€ct/tCO2 per abated ton was used which is the average of the range in the study by Michaelowa and Jotzo (2005). In this case we do not use the highest TC estimate provided by Michaelowa and Jotzo (2005) which would have been 85 €ct/tCO2. The reason is that these data come from a small and very early (2002) CDM project. Since CDM TC are likely to have fallen over time and since it is unlikely that countries would import only CDM credits from small (TC intensive) projects, the average and not the highest TC estimate was used (see also Appendix 4).

[19] Costs for composing the project design document (PDD), baseline setting, monitoring plan, documentation, environmental impact assessment, validation and registration (Krey, 2005),

[20] A back-of-the-envelope calculation applying these data to the EU yields total TC of € 180 - 370 million p.a. total for the EU (see Appendix 3). Ellerman et al. (2010) found firms MRV costs of €462 million for the entire EU ETS for the first trading period (2005-2007). This amounts to annual costs of €154 million. These figures are based on the Irish survey of Jaraite et al. (2011). This number is similar to our calculations, which estimate firm MRV costs ranging from €49 - 184 for the entire EU ETS (based on the German MRV data of 2011 as reported by (Loeschel et al. 2011) and (Destatis 2011).

[21] Data from the hypothetical Tradable Permit and Fee System in Australia show a remarkably similar ratio of public sector versus private sector TC 76 - 77% (private sector) vs. to 23 - 24% (public sector) (Ofei-Mensah & Bennett, 2013).

[22] This data has been obtained using the Standard Cost Model methodology (Bundesregierung, 2006).

[23] McLean (1997) reports annual running costs of only € 1,6 million total ($ 1,5 million in 1997) for the US SO2 trading scheme. The apparent reason for low TC for the compliance agency in the US SO_2 trading scheme is that all regulated entities are equipped with continuous emission monitoring systems (CEMS), enabling the US Environmental Protection Agency (EPA) to remotely monitor emissions and keeping the effort and costs low (Ellerman, 2000 p. 249). This illustrates that technology also can have a substantial influence on TC.

[24] In 1999 18,7 million emission allowances have been traded (LECG, 2003) with assumed trading TC of € 0,15 (or $ 0,2) per trade (Joskow & Schmalensee, 1998).
[25] Firms' MRV costs are substantially higher for the US SO_2 scheme than in the EU ETS. In 2001 there were 2.100 regulated entities with average MRV costs of € 47.000 (or $ 50.000) (EPA, 2001). Also initial capital costs were large. Ellerman (2000) estimated them at approximately $ 700.000 per generating unit (at 371 MWe per unit).
[26] We assume that due to political economy considerations (i.e. the rent distribution interests of firms) a taxing scheme would also be designed in a downstream manner.
[27] A first best taxation scheme with no tax exemptions and upstream regulation should be compared to an upstream trading scheme with 100% auctioned certificates. Given the current structure of the EU ETS, we compare it with a hypothetical downstream taxation scheme with inframarginal tax exemptions analogous to free certificate allocation and as suggested by Jotzo and Pezzey (2012). It is assumed that eligibility for exemptions would depend on the same parameters that qualify firms for free certificates (i.e. historic emissions, export share and efficiency).
[28] We assume that the number of regulated firms is identical in both schemes. Under the assumption that firms exempted from the tax would not be subject to MRV, the respective firms' and aggregate MRV costs would be lower than in a trading scheme. However, as suggested by Jotzo and Pezzey (2013) a carbon tax system can be designed symmetrically to an ETS with free allocation if inframarginal tax exemptions are implemented. In that case, MRV would be required at least for all firms not receiving 100% free allocation.
[29] The study the private and public sector costs were estimated via interviews, questionnaires and analogies from other excise taxes.
[30] MRV is carried out upstream with 74 companies (12 contributed 95% revenue) regulated.
[31] The original £ values have been adjusted for inflation to the year 2011 using Fxtop.com inflation calculator and converted into € with the average exchange rate of January 2011.
[32] Fossil products that are not finally combusted to emit their carbon content into the atmosphere need to be recompensed at a later stage in the production chain, which should be technically feasible.
[33] The carbon that is bound in oil, coal and natural gas –and which is discharged into the atmosphere when the fuel is combusted– can be readily determined at each step in the process chain (IPCCb, 2007). Our discussion omits other gases and sectors (e.g. agriculture) because this would substantially increase complexity.
[34] In this context 'large entities' refers to the producers and importers of fossil fuels.
[35] In Germany there are 900 large installations (>25.000 t/CO_2) that make up almost 99% of the emissions, and 1.100 small installations (<25.000 t/CO_2) (CITL, 2013). Large installations have average annual MRV and trading costs of roughly € 10.000 and small installations of roughly € 2.700 (Loeschel et al., 2011; own calculation). The firms' MRV costs data used for the calculation are based on Loeschel et al. (2011). The high-cost estimates (used in Figure 2) could not be used since no differentiation between small and large firms is made in the studies by VBW (2013) and Destatis (2012). For the calculation in Table 4 we assumed 150 importers and producers of fossil fuels that would have to bear MRV costs of 1€ct/tCO_2 (BAFA, 2014; Loeschel et al., 2011). The number of firms that import mineral oil products into Germany amounts to 132 (BAFA, 2014). The number of companies that import and produce gas and coal in Germany is very small since the sector is characterized by a high concentration. The Verein der Kohlenimporteure (2014) and BAFA (2014) estimated the number between 10 and 20 firms (no public information is available). For the calculation we therefore used the number of 150 firms that would be subject to MRV in an upstream scheme.
[36] The study by Loeschel et al. (2011) does not state the average transaction costs for small (emissions below 25.000 tCO_2) and large (emissions above 25.000 tCO_2) installations. It however states the median for both sizes and the average transaction costs for different installation sizes (Figure 15 in Loeschel et al., 2011), and total average transaction costs. Using these data points combined with the distribution of emissions of large and small firms in Germany (derived from CITL, 2013) an approximate computation was carried out to derive the average values for small and large installations used in this calculation.
[37] Since the average transaction costs for large installations amounts to roughly € 10.000 (which also includes many medium sized installations), and transaction costs scale with the number of emissions (Jaraite et al., 2010) we believe the transaction costs of € 18.000 for importers and producers to be a valid basis for the low cost estimate.
[38] Moreover, upstream regulation would also reduce the costs of compliance agencies. No data is available, but the mere fact that compliance agencies in Germany would not have to deal with the current approx. 2.000 installations, but only with roughly 150 producers, importers and refiners of fossil fuels would very likely reduce the administrative burden. In the calculation in Table 6 in section 5, we have assumed that the cost of the

compliance agency would be half of what they are in a downstream scheme (i.e. € 7,7 million). The other cost components are assumed to be equivalent to an upstream scheme.

[39] There is anecdotal evidence that downstream regulation creates additional incentives to carry out abatement measures than would otherwise be the case. On site MRV may induce behavioral changes following the management principle of “what gets measured, gets managed.” A possible explanation could be that the awareness that is created through measurement spurs this incentive (Seidel, 2013). Further research is necessary to explore the empirical validity of this relationship.

[40] For example, if coal fired power plants were to be banned by a technology standard; it should be rather easy to monitor whether or not a plant is still operating.

[41] An example from a non-climate related area is the requirement to inspect catalytic converters for cars regularly in order to ensure their proper functioning.

[42] The most efficient products in a class set the standard, which all other products have to comply with within a certain period (Kimura, 2010).

[43] Tradable performance standards lead to an equalization of marginal abatement costs, at least under idealized theoretical conditions. In such a scheme, firms that have lower marginal abatement costs can abate more and sell their awarded white certificates to firms with higher marginal abatement costs. These firms can then use these certificates for compliance.

[44] Or $ 5,5 - $ 23,3 million in 2011 dollars

[45] The program covers 82% of home energy use, 67% of commercial building energy use, and roughly 50% of industrial energy, which amounts to approximately 2,5 billion tCO_2 regulated under the standard (EIA, 2008; Energy.gov, 2012).

[46] In theory, industrial CO_2 performance standards can be adopted to limit the CO_2 emissions per unit of output in the sectors currently covered by the EU ETS. This would require a fixed limit of CO_2 emissions for each output product such as cement, paper, aluminum and electricity that would have to be set by the regulator. As the benchmarking experience within the EU ETS has shown, further subgroups of product specification and regional differences might be required to account for special conditions (Ecofys, 2009). It must be noted that a performance standard for the EU ETS sectors without tradable white certificates would imply inferior efficiency performance, as the equalization of marginal abatement costs among firms would not be possible.

[47] The EU Emission Standard for vehicles was introduced by Directive 98/69 (1998) and specifies the maximum limit of CO_2 emission per driven kilometer for different vehicles sizes. A fleet-average CO2 emission target of 130 g/km for passenger cars must be reached by vehicle manufacturers by 2015 using vehicle technology (CEC, 2012). Emission limits are set according to the mass of vehicle, using a limit value curve. The curve is set in such a way that a fleet average of 130 grams of CO_2 per kilometer is achieved by 2015. The limit value curve means that heavier cars are allowed higher emissions than lighter cars while preserving the overall fleet average. Only the fleet average is regulated, so manufacturers are still able to produce vehicles with emissions above the limit value curve provided these are balanced by vehicles below the curve (CEC, 2012). The scheme allows car manufacturers to equalize marginal abatement costs by pooling together, in effect enabling implicit white certificate trading. Manufacturers can group together to form a pool which can act jointly in meeting the emissions target. In forming a pool, manufacturers must respect the rules of competition law and the information that they exchange should be limited to average specific emissions of CO_2, their specific emissions targets, and their total number of vehicles registered (CEC, 2012). Furthermore, approved "eco-innovations" can count for up to 7 g CO_2/km towards the target (CEC, 2012). Under the test procedure used for vehicle type approval, certain innovative technologies cannot demonstrate their CO_2-reducing effects when being type approved. Manufacturers can be granted emission credits equivalent to a maximum emissions saving of 7g/km per year for their fleet if they equip vehicles with innovative technologies, based on independently verified data (CEC, 2012). TC arise in the process of monitoring vehicles sales, specifying vehicle-type emissions in the New European Driving Cycle (NEDC), the approval of eco-innovations and the administration of compliance procedures. Most compliance processes of this policy are administrative in nature, without the need for special requirements, causing probably only low TC. The core processes of monitoring compliance with the instrument are straightforward since the required data such as auto sales and car specific CO_2 emissions are readily available. The decision to grant eco innovation credits may be more difficult since definitive guidelines specifying the validity of innovations cannot exist given the fundamental uncertainty over innovations, which may give ground for dispute between the regulator and firms. Also, the decision on a CO_2 fleet target and the pathway to achieve this goal involves modeling and strategic decision-making, which may tie up managerial resources.

[48] The US CAFE program was introduced with the Energy Policy and Conservation Act (EPCA) of 1975 to decrease fuel consumption and the dependence on imported oil (Fischer, 2008). The standard mandates vehicle

manufacturers to average a certain amount of miles per gallon. This amount was tightened over time. TC for the scheme are small since auto-makers only must report vehicle sales and fuel economy to the regulator who collects and verifies this information.

[49] The number of regulated emission and the number of regulated entitiesare assumed to be identical in both scenarios.

[50] For examples please see e.g. Fang, 2005; Falconer & Whitby, 2000; Kerr & Maré, 1998; McCann & Easter, 2000; Mettepenningen, Verspecht & van Huylenbroeck, 2009; Pannell et al., 2013; Coggan et al., 2010.